Conférence Européenne des Directeurs des Routes
Conference of European Directors of Roads

Assessment methodologies and mitigation measures for the impacts of road projects on soils – ROADSOIL

Report on the framework for the assessment of the impact from road projects on soil functioning

Deliverable D1.1

04 April 2023

Norwegian Institute for Bioeconomy Research NIBIO
Swiss Federal Institute for Forest, Snow and Landscape Research WSL
Swedish University for Agricultural Sciences SLU

CEDR Call 2019: Soils

Assessment methodologies and mitigation measures for the impacts of road projects on soils – ROADSOIL

D1.1 Report on the framework for the assessment of the impact from road projects on soil functioning

Due date of deliverable: 30/11/2022

Actual submission date: 04/04/2023

Start date of project: 01/03/2021 End date of project: 28/02/2023

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Executive summary

Within the EU28, 189 km^2 of cultivated and natural land has been sealed for the expansion of the transport network in the period 2012–2018. Whenever land is sealed, previously present soil functions such as crop production, water regulation and climate regulation are lost or greatly impaired. Apart from the area fully converted during road projects, additional areas are used temporarily during the construction phase, such as for storage and/or transport. Temporal changes of land use involve removing soil and/or adding material (soil or other) and loading of soil during storage and traffic. These acts can severely reduce the extent to which a soil can fulfil its functions if soil functional characteristics are altered beyond critical limits. The loss of soil functions is offsetting the capacity of soils to deliver ecosystem services.

Conceptually, soil threats, soil functions, and ecosystem services are closely related. The ROADSOIL project outlined suitable frameworks for the protection of soil functioning in road projects based on a literature review on the concepts of soil functions, (soil) ecosystem services and soil threats. Two approaches were developed, one for the planning phase and one for the construction phase. The framework for the planning phase focusses on the soils that are part of proposed road trajectories. This framework anticipates a balanced supply and demand of soil functions at an (intra)regional scale. The framework developed for the construction phase focusses on the local sites that are temporary used in the construction phase (among others, for storage, buildings or as work roads). This second framework aims to reducing the likelihood and impact of soil threats that would degrade soil functioning.

Central in the protection of soils in road projects is the soil's initial state, i.e. the soil characteristics prior to any road-related activities take place. This state is to be used in the planning phase for evaluation of the impact of different road corridors on the level of soil functioning, for identification of areas for compensating and mitigating measures, and for monitoring the effectiveness of compensating and mitigating measures. Investigation of the soil's initial state should include biological, chemical and physical soil characteristics, preferably at different soil depths. The selection of indicators may differ depending on the soil functions in focus and resources available. For some of these soil functional characteristics, thresholds values exist that indicate the limit at which soil functions are greatly impaired, whereas for others, such thresholds values are not yet established or of a wide range due to differences in terms of soil types, land use and climatic zones.

Figure 1. Nordskovvej Silkeborg. Photo by Jakob Stigsen Andersen, Midtjyllandsavis

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1. Introduction

Within the EU28, 189 km^2 of agricultural and natural land has been taken for the expansion of the transport network in the period 2012–2018. Most of the area, namely 54% or 102 km^2 was under arable or permanent crop production[1]. While these converted areas now fulfil the function of supporting transportation, previous soil functions such as primary production, water regulation and climate regulation are lost or greatly impaired. Apart from the areas fully converted, additional soils will have been temporarily used, for example for storage, buildings or as work roads. Such temporary changes of the use of land involve a risk of soil degradation through (un)intentional changes of soil components, for example the loss of organic matter or porosity, and these changes can thereby threaten the extent to which a soil can fulfil its functions after road construction is finished. While many land use changes take place at a small scale, its impact is cumulative and acts on a large scale [1].

The loss of soil functioning is a global problem, across soil types and land uses. In 2015, 33% of land globally was considered moderately to highly degraded due to erosion, salinization, compaction, acidification and chemical pollution [2]. The loss of soil functioning is strongly associated with global issues such as the loss of biodiversity and climate change [3], but is equally relevant at smaller scales. Degraded soil can impact communities', families' and business' finances and/or wellbeing, for example due to reduced quantity or quality of harvests, flooding events or contaminated water. On all accounts, the importance of protecting soils from further degradation processes has been gaining awareness for decades [4–7], and is amplified by the high pressure on land [8]. This has, among others, resulted in the EU Soil Strategy [9] for the protection, restoration and sustainably use of soils.

Figure 2. The pressure on land is high as society has many uses for the land. Societies impact soil functioning through soil sealing of soil and degrading remaining soils biologically, chemically and/or physically. Photo: Silvia Tobias.

[1] Other areas were by pastures and mosaic farmland (55 km^2), forest and woodland shrub (22 km^2), natural grasslands, heathland, wetlands and open space (together 11 km^2), answering to 29, 12, and 6 % of the conversion, respectively. Source: CORINE Land Cover Accounting Layers.

The transport sector has a large responsibility to reduce the pressure on soil resources. The protection of soils in road projects is critical in three phases: i) the planning phase, ii) the construction phase, and iii) the maintenance phase. The planning phase includes defining the corridor for the new road as well as assigning sites temporarily needed for the construction. Moreover, the planning phase allows early consideration of compensating and mitigating measures. The construction phase includes all executive activities related to the preparation and building of a road. This usually involves (a risk of) changing a soil's composition, for example through removing soil, adding material or through putting soil under mechanical stress during traffic. During construction, protecting soil functioning relates to the areas that remain, or are converted back to, unsealed soils (natural or cultivated). Here, it is critical to maintain or develop preferred soil biological, chemical and physical functional characteristics. The construction phase is also the phase in which many mitigating measures are implemented. Monitoring of mitigation measures is crucial both during implementation and post-implementation.

This report presents frameworks for the protection of soil functioning during planning and construction of roads based on a review and synthesis of the concepts of soil threats, soil functions and ecosystem services (Chapter 2 and Appendixes 1–2). Moreover, earlier proposed spatial planning assessment tools were reviewed (Chapter 3). These reviews were used to select the most suitable approaches for the protection of soils in the context of road projects. Two frameworks are outlined (Chapter 4 and 5). The frameworks conceptualise the protection of soil functioning for i) the potential road trajectories in the road planning phase and for ii) the temporary construction sites in the road construction phase. The first framework (Chapter 4) addresses the impact on soil in the planning phase, which relates to the positioning of the road corridor through the landscape. This conversion of land to road impacts the largest volume of soil. The second framework (Chapter 5) relates to the impact of the building phase on soil that is temporarily used. This is where soil is exposed to potential threats, which may permanently reduce the soil's performance. For each framework, we conclude with a summary, and data and knowledge gaps. The effect of mitigating measures on the protection of soil functioning is beyond the scope of this report, but covered by the CEDR Call 2019 Soils reports D4.1, 4.2 Comprehensive Literature and Best-practice Review for Avoiding, Mitigating and Compensating for Impacts on Soil [10] and D6.1 Guidelines for soil handling in infrastructure projects from planning processes to construction work in field / D6.2 Material for education in soil properties relevant for entrepreneurs and drivers of machines used in road construction [11].

2. A review of state-of-the-art concepts used in assessments of soil functioning

Soil is a natural dynamic system with living and non-living components that interact with both the overlying atmosphere and the underlying strata. These interactions promote many biological, chemical and physical processes vital to life on Earth. Clusters of soil processes, called soil functions, are, for example, largely responsible for the regulation and purification of water, biomass production, recycling of nutrients and regulation of climate. Soil functions, in turn, determine largely the capacity of soils to deliver ecosystem services [12]: the quantifiable, functional outputs of complex natural processes [13].

Conceptually, (soil) ecosystem services, soil functions, soil processes and soil threats are closely linked (Figure 3). It is not uncommon for processes to be confused with functions, or for functions to be mixed with ecosystem services [14–17]. This confusion is partly explained by the interdisciplinary of the concepts; *boundary concepts* such as ‘soil function’ and ‘ecosystem service’ cannot be explained or related to solely within disciplines, nor without considering processes of and effects on other systems than the one studied. Boundary concepts are dynamic in nature and are generally defined differently between scientific disciplines or between science and society. While some argue that widely agreed upon definitions are required to really contribute to protecting the environment in decision making [15,18,19], others argue that different definitions can coexist [20].

A comprehensive review of the origin and recent developments of the concepts of soil functions (including soil quality assessments) and ecosystem services are included in Appendixes 1–2, respectively. Below, a summary is included to serve a basic understanding of the concepts for support of the succeeding chapters, followed by a review of the concept of soil threats and ways that road projects may aggravate threats to soil.

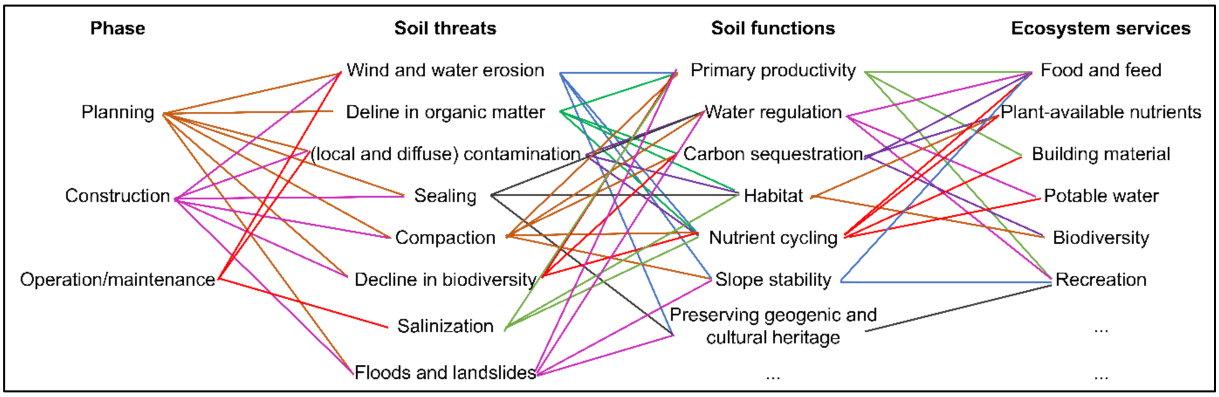


Figure 3. Soil threats, soil functions, and ecosystem services are closely related. Different phases of road projects can cause multiple threats, and thereby influence soil functions as well as ecosystem services. Adapted from Bünemann et al. (2018) [15], who developed it from the scheme presented by Kibblewhite, Ritz and Swift (2008) [13], modified by Brussaard (2012) [21]. From Ten Damme and Keller (2023) [22].

When the functional characteristics of soil change over time, the extent to which a soil fulfils different functions will change. Some changes of a soil’s characteristics become visible very slowly, for example due to soil formation processes, whereas other changes can occur within a much shorter timeframe, often due to human intervention through changes in the land use or management. Changes of a soils functional characteristics may improve, limit or degrade a soils ability to conduct certain processes. The biological, chemical and physical components of a soil influence namely which, and at which rate, processes take place. In other words: soil components act both as inputs and catalysts for soil processes.

The recognition of the importance of a well-functioning soil led to political commitment to protect soils. The Commission of the European Communities officially acknowledge that soil is increasingly exposed to biological, chemical and physical soil degrading processes. Eight major soil threats to European soils were identified (Figure 4): i) (wind and water) erosion, ii) decline in organic matter, iii) (local and diffuse) contamination, iv) sealing, v) compaction, vi) decline in biodiversity, vii) salinization, and viii) floods and landslides [23]. While many of these degrading processes occur naturally, they become threats when aggravated by human activities [24]. The processes, consequences and key drivers of each of these eight treats are described in Chapter 2.1. Moreover, it highlights for each threat how road projects can aggravate manifestation of the risks. Where possible, compensating and mitigating measures are briefly mentioned. Practical applications of the compensating and mitigating measures are covered by Geiges *et al.* [10] and by Haraldsen and Tobias [11].

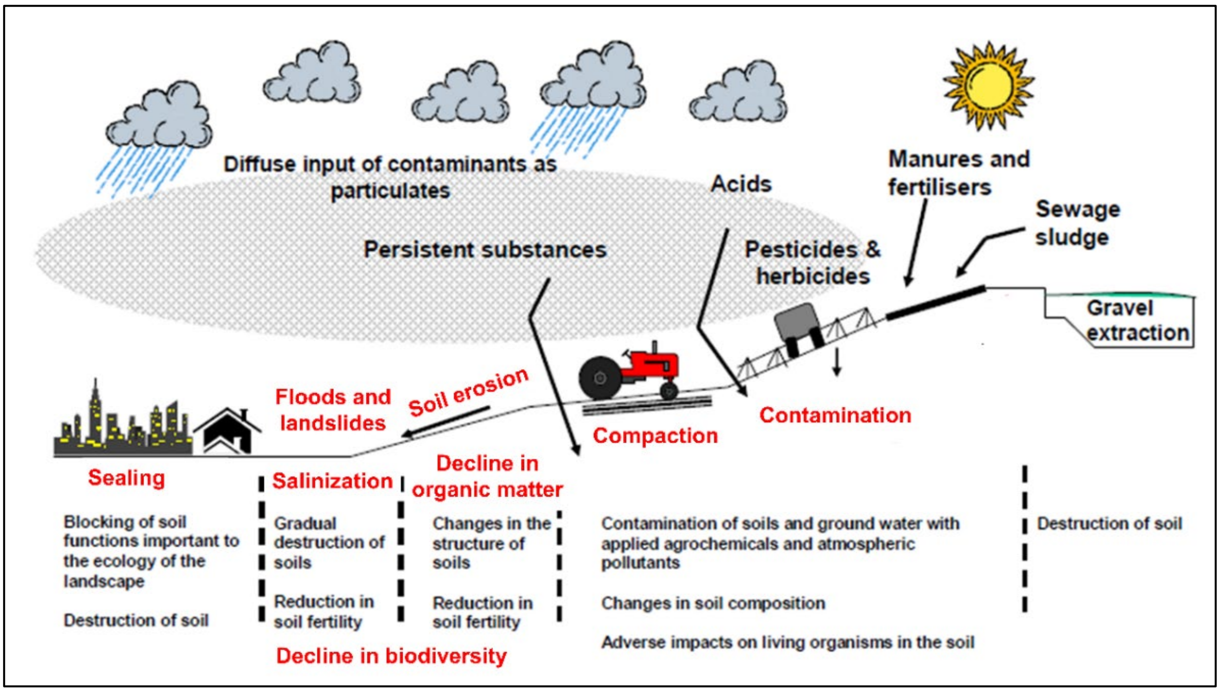


Figure 4. Major soil threats (in red) due to human activities. Adapted from Tóth [25] pp. 5–10

Soil threats are processes that change a soil's functional characteristics to the extent that soils' ability to fulfil crucial processes sustainably, now and in the future, is endangered. When a soil in one place is degraded, it may not always be possible to compensate the loss in another place, since soils have certain capacities to perform functions to begin with. Moreover, functions may be very relevant in one place but less beneficial somewhere else, such as can be the case for example for water and flood regulation. This complicates compensating the loss of soil functioning, and altogether this has called for the need to protect soils worldwide. Sustainable use of our environment, including soils, implies that the ability of ecosystems to supply the services enjoyed by society is not reduced [26,27]. However, it has been recognised that decisions, in particular those concerning the use of land, have often been taken without fully considering the impacts of these decision on the environment [3,28]. Another consequence of poor decisions on land use and management is that soils may not be used to provide the functions they would do best [29–33].

Blum argued that sustainable use of soil is only possible by carefully balancing soil functions temporal and/or spatial (local or regional) [6,30]. For this reason, Blum introduced the concept of soil functions (Appendix 1). Three ecological and three non-ecological soil functions were highlighted: i) the production of biomass, ii) the protection of humans and the environment, iii) containing a gene reservoir, iv) providing a physical basis for human activities, v) providing a source of raw materials, and vi) preserving geogenic and cultural heritage [30]. Soon, a seventh function was added by the European Commission in the proposed Soil Framework Directive [34]: the ability to store carbon. These seven functions were prioritised as they were considered being particularly vulnerable to soil threats [2] (Figure 5). This highlights that the concept of soil functions is not static. Rather, the concept is driven by pressing socio-environmental challenges such as climate change and biodiversity.

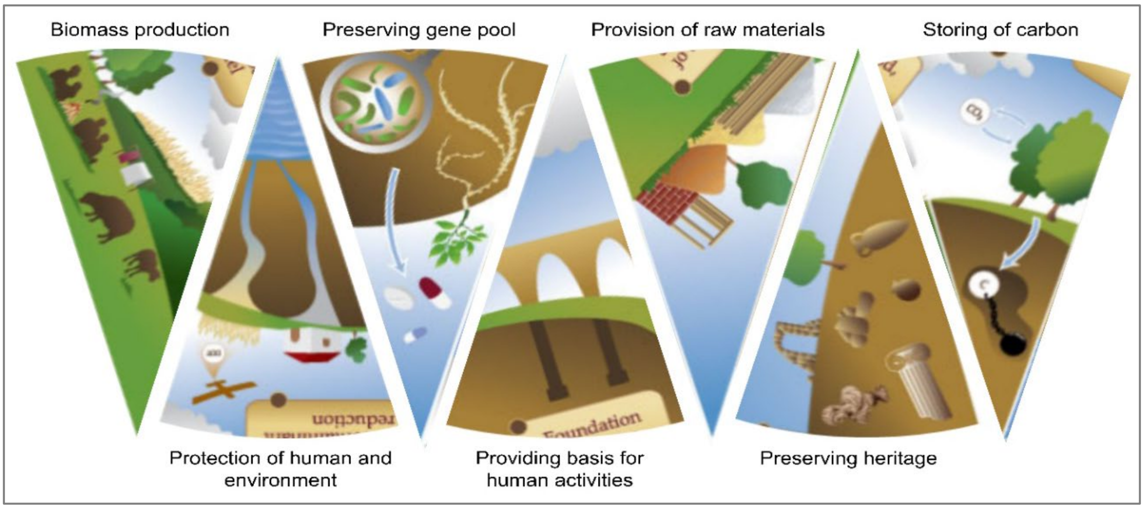


Figure 5. Seven widely prioritised soil functions, particularly vulnerable to soil threats. Adapted from [35].

It is generally considered that all soils are able to fulfil all functions, although it is known they do not do so to the same extent [31,36–38]. Namely, soils are naturally extremely heterogeneous in terms of the physical, biological and chemical interactions that take place [16,29], and soil extrinsic factors such as climate, topography and management differ spatially [15,19,39]. Some soils then have a higher capacity for carbon sequestration, while others may have a higher capacity for denitrification. Moreover, soil functions are often partially interdependent because they share some soil processes and characteristics [39]. This results in synergies and trade-offs between functions, which are likely to vary over time and space due to differences in pedoclimatic characteristics and management, which shape soil characteristics. For example, Coyle *et al.* showed that an improvement of drainage conditions can be expected to increase primary productivity, but at the expense of carbon sequestration [36]. The concept of soil functions thus emphasizes the multifunctionality of soils, which may be optimised, but cannot be maximised, because maximising a certain soil function will generally impact another function [31,39].

Due to the fact that soils fulfil multiple functions, (local) soil management requires a prioritisation of some soil functions over others. Prioritisation of soil functions strongly depends on the individuals awareness and preferences [37,40]. Five soil functions often prioritised in agriculture are i) nutrient storage, ii) water regulation, iii) productivity, iv) habitat, and v) carbon sequestration). On different land uses, other functions can be favoured. For example, Safaei *et al.* assessed soil functions in terms of soil stability, infiltration, and nutrient cycling for semi-arid forests and rangelands [41]. In peatlands, greenhouse-gas regulation may be prioritised. Assessments of soil functions are based on measurable soil properties, optionally supplemented by information on land use (Table 1, Appendix 1). Consequently, monitoring of soil functions is closely linked to soil quality assessments [15,28,31].

Table 1. Example of indicators used to estimate the level of soil functioning. Adapted from Jost et al. [32].

	Input parameters							
Soil function	SOC content (C)	pH	Soil texture	Bulk density	Horizon thickness	Stone content	Soil depth	Land use
Nutrient storage	x	x	x	x	x	x		
Water regulation	x		x	x	x	x		
Productivity	x	x	x	x	x	x	x	
Habitat	x	x	x					x
Carbon sequestration	x		x	x	x	x		x

Soil functions are, in principle, intrinsic to a soil, as they are bundles of soil biological, chemical and physical processes that take place regardless of human interest. Consequently, soil functions are difficult to quantify and value; it is the output of the processes that may or may not contribute to human welfare and wellbeing. The quantifiable outputs of soil processes are ecosystem services (Appendix 2). For example, *water infiltration* and *water retention* are soil processes contributing to the function *water regulation*. Society desires water drainage to a certain extent (to prevent flooding, to increase bearing capacity) but also needs a retention of water (to prevent plant-drought stress). In this case, ecosystem services are in terms of discharge or plant available water. Another soil function is primary productivity or biomass production, which includes processes such as nutrient uptake, rooting and water uptake, while the value depends on the plant species, its quantity and/or quality.

The concept of ecosystem services has, since decades, revolved around the wish to quantify the value of environmental services. This value has been expected to add weight on behalf of natural systems in decision-making, and thereby protect these natural systems [42–46]. In 1987, De Groot introduced a conceptual framework (Figure 6) highlighting that natural processes and components provide the goods and services that fulfil human needs, while, *vice versa*, human needs affect natural processes and components. The concept of ecosystem services was greatly popularised by the UN Millennium Ecosystem Assessment (MEA) [47]. Since the MEA, a great number of ecosystem services frameworks have been developed and reviews have been performed, refining the concept of ecosystem services. Many of these frameworks can, in their essence, be reduced to De Groots (1987) framework.

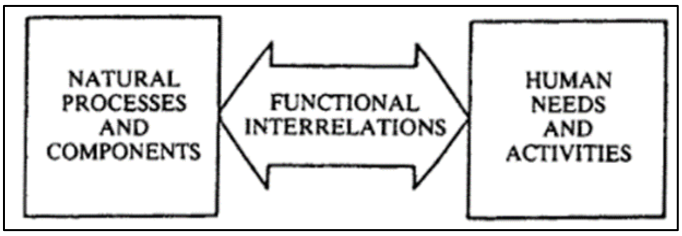


Figure 6. The conceptual framework introduced by De Groot [44], which devolved into the concept of ecosystem services.

Since the concept of ecosystem services addresses interdependencies between nature and human wellbeing [48,49], ecosystem services cannot be discussed without considering both nature and society. On the one hand, people influence the extent to which services can be produced through utilisation of services and management of land. On the other hand, the value of services varies depending on, among others, spatial and temporal variation of supply and demand. Accounting for ecosystem services requires i) identification or classification of services, ii) quantification of the services, and iii) valuation of services. Accounting of ecosystem services is further complicated by the fact that most services are enjoyed locally, whereas the supply is influenced by processes at a much larger scale. Decisions on land use (including management) may thus influence the services generated. Not all changes of soil will significantly affect the generation of an ecosystem service, but when a change does, it generally affects multiple services provided by that system [45]. As with soil functions, ecosystem services are highly interdependent: both synergies and trade-offs exist [50,51]. As a consequence of the multifunctionality, optimisation of the generation of a single ecosystem service will often cascade and affect the generation of other ecosystem services. This effect may not be restricted to the study area but possibly extend beyond the areas boundaries, meaning that local changes are felt at greater distance [51].

'Soil ecosystem services' is a distinction made after it was observed that frameworks and regulations did not generally promote the understanding of the critical role that soils have in the provision of ecosystem services [7,52–54]. Namely, frameworks addressing ecosystem services have often reduced soil processes to intermediate services, despite the knowledge that the capability of ecosystem services highly varies between soils [3,16,40,55].

2.1 Soil threats in road projects

Soil threats are biological, chemical and/or physical processes that lead to severe soil degradation, changing the functional characteristics of soil to the extent that soil functioning is hampered. Many of these degrading processes occur naturally, but they become threats if aggravated by human activities [24]. Political commitment to soil protection led the Commission of the European Communities to acknowledge that soil is increasingly exposed to threats. Eight major soil threats to European soils were identified (Figure 4): i) (wind and water) erosion, ii) decline in organic matter, iii) (local and diffuse) contamination, iv) sealing, v) compaction, vi) decline in biodiversity, vii) salinization, and viii) floods and landslides [23]. An overview of these threats and their impacts on soil functional characteristics is given in Table 2.

Table 2. Direct effects of the eight major soil threats on soil biological, chemical and physical characteristics.

	Soil functional characteristics		
Soil threat	*Biological*	*Chemical*	*Physical*
(wind and water) erosion	X		X
Decline in organic matter	X	X	X
(local and diffuse) contamination		X	X
Sealing	X	X	X
Compaction	X		X
Decline in biodiversity	X	X	
Salinization	X	X	X
Floods and landslides	X		X

Following the communication by the Commission of the European Communities [23], Tóth made a first attempt of mapping the risk of soil threats across Europe [25]. Moreover, the project ENVASSO (ENVironmental ASsessment of Soil for mOnitoring, 2006–2009) was launched with the aim "*to define and document a soil monitoring system for implementation in support of a European Soil Framework Directive, aimed at protecting the continent's soils*" [56]. The ENVASSO project identified sets of potential indicators for the purpose of soil monitoring for each of the eight above-mentioned soil threats plus for the soil threat 'desertification' ('*a cross-cutting issue associated with erosion, decline in organic matter, salinization and decline in biodiversity*'), taking into account the thematic and policy relevance and data availability of the indicators. Huber *et al.* highlighted the temporal relevance and the requirements in terms of frequency and scale of monitoring for a subset of the indicators [56].

The RECARE (Preventing and Remediating Degradation of Soils in Europe through Land Care) project (2013–2018) built upon the ENVASSO project. The RECARE project aimed to develop effective prevention and remediation strategies of degraded soils, based on the DPSIR (Driver(s), Pressure(s), State, Impact(s), Response(s)) framework, addressing each soil threat separately [24]. Priority indicators identified in the ENVASSO project were revised and amended. For example, the RECARE project differentiated between loss of organic matter for mineral and peat soils. Both ENVASSO and RECARE used 'data availability' as a primary criterion for selecting indicators.

Most recently, the European Topic Centre on Urban Land and Soil Systems described the rationale for a series of widely used soil quality indicators, and how they can be evaluated, for the assessment of soil threats across Europe [57]. Doing so, Baritz *et al.* emphasized that indicators and evaluation criteria are influenced by natural and societal differences across Europe, for example in terms of soil types, climatic zones and land use [57]. Therefore, the authors had to conclude that despite having a profound knowledge of indicators and monitoring, the definition and classification of indicators, the sampling, measuring and evaluations systems are still diverse. Thresholds were then also given for specific soil functions in specific local conditions. Consensus between countries regarding valid (regionalized) critical limits used as thresholds for specific soil functions has not been established.

The indicators recommended by Huber *et al.* [56], Stolte *et al.* [24] and Baritz *et al.* [57] can be used to assess and monitor the state of soil threats. They are, however, not indicators that reflect the likelihood of threats from actually manifesting, and do not identify areas with high risk. That said, Stolte *et al.* specifically addressed drivers of soil threats [24], and Tóth [25], Huber *et al.* [56] and Bartiz *et al.* [57] included short reviews of the mechanisms behind the threats to varying extents. Reducing the impact of soil threats requires pro-active minimisation of the likelihood of threats manifesting, more than a reactive monitoring of the extent.

2.1.1 Erosion by wind and water

Soil erosion is a natural process in which wind or water causes a redistribution or loss of soil particles. Soil formation rates are estimated at 1–4.5 t ha^{-1} annually and the calculated tolerable soil loss is estimated at 0.3–1.4 t ha^{-1} [57]. When soil erosion is aggravated by human activities, i.e. erosion exceeds accepted rates of soil formation, soil erosion is considered a soil threat [56]. Different types of erosion are distinguishable. Wind and water erosion are often prioritized soil threats [24], while 'landslides', an extreme form of erosion, is often presented as a threat itself [23]. Other types of erosion are tillage erosion and coastal erosion.

Generally, the eroded material is (fertile) topsoil material. Soil erosion then leads to a decrease in soil organic matter of the eroded site (through removal and exposure of it at greater depths), soil biodiversity and water holding capacity on-site, and thereby impacts various biological, chemical and physical soil processes. Among others, yield loss up to 4% per 10 cm lost soil are expected. Soil erosion by water enhances flood risks and increases the volume of floods. Eroded soil can clog up drainage systems but is often deposited on land elsewhere or ends up in water bodies downstream. Soil erosion thereby increases the risk of contamination because the eroded material can be contaminated. Moreover, eroded material itself can also be considered contamination, if alien to or unwanted in the deposited area.

Both water and wind erosion are highly influenced by climate, yet land and soil management can exacerbate erosive conditions. Key-drivers for water erosion are:

- Heavy rainfall (in particularly following dry periods);
- Slopes (particularly if long and steep);
- Non-vegetated soils;
- High water repellency;
- Poor soil drainage;
- Low aggregate stability.

Key-drivers for wind erosion are:

- Strong air movements;
- Non-vegetated soils;
- Low soil moisture;
- Light soil texture (such as sandy and silty soils)

The risk of soil erosion in road projects is to be considered in both the planning and construction phase. In the road planning phase, construction sites should preferably be allocated to areas with the lowest risk of erosion, such as to areas flat rather than steep, to areas with good rather than poor soil drainage, although the ease of implementing mitigating measures to compensate the risks could be taken into account. For example, the steepness of a slope may be decreased or existing drainage improved. The road trajectory will affect local water cycle as it is likely to result in a reduced area for water infiltration and an increased area for water runoff to the road verges. Waterways to discharge the water runoff from the road change local pressure gradients, and thereby field moisture conditions. Moreover, the drainage may accumulate in downstream regions and there increase flood peaks [58]. Soil erosion risk also need to be considered in relation to temporary storage of removed soil layers. Practical solutions are described by Haraldsen and Tobias [11].

It is critical to recognise that key-drivers listed above can be enhanced during road construction, for example through a reduction of soil vegetation and through formation of impermeable soils such as sealed or compacted. This may not only be the case within the construction area but may also be the case if fields are temporarily cut-off from for example annual crop rotations. Soil erosion mitigation measures are site-specific. Reducing the risk of soil erosion is primarily done by increasing soil (plant) cover, reducing management practices that promote soil erodibility and those that reduce soil water infiltration, by addressing the shape of the terrain, and by improving water discharge.

2.1.2 Decline in organic matter

Soil organic matter (SOM) is the organic fraction of soil without living soil flora and fauna, and exist mainly of carbon (48–58%), nitrogen, phosphorus, sulphur and hydrogen. A decline of soil organic matter manifests when the rate of decomposition exceeds the build-up of soil organic matter. It can also be lost through soil erosion. Soil organic matter is typically quantified as soil organic carbon. Soils are the second largest carbon pool on earth, after oceans.

The carbon concentration is generally highest in the topsoil, resulting in relatively large proportions of soil organic carbon in the top 20 cm, where 35–50% are global averages for different biomes. Soil layers below 30 cm depth have lower carbon concentrations but contribute to more considerably to global soil carbon stocks [59,60]. Organic matter is translocated from the upper to the deeper soil layers for example by anecic earthworms.

Soil organic matter is an important source for plant nutrients and influences soil structure, stability, water retention, cation exchange capacity and soil biodiversity. Loss of soil organic matter is a loss of carbon storage and can be a source of water contamination, particularly if lost from peatlands, which are soils with > 30% organic matter. Within Europe, peatlands represent a much larger share of soils in the north compared to the south, which relates well to differences in temperature and rainfall [61]. Peatland oxidation is an important source of CO_2 and N_2O emissions, and degraded peat soil are prone to severe drying of the topsoil, which can lead to water repellent conditions. While peatlands have the ability to slow down water discharge through water retention, risks of wind and water erosion are increased on degraded peatlands. Stakeholder consultations across 20 European countries by Vanino et al. showed that improving soil organic matter and conservation of peatlands is considered a priority soil challenge in northern, western and central Europe (acknowledged by over 30% of the respondents) [62].

It is widely accepted that adequate levels of soil organic matter improve soil functioning (2% soil organic carbon is typically used as a lower threshold), yet evidence for critical thresholds are sparse. Moreover, not only the quantity of soil organic matter is important; changes in the quality of soil organic matter may also affect various soil functions. The loss of organic matter is dependent on the turnover of carbon in the soil. Important drivers of a decline in soil organic matter are:

- Increasing aeration;
- Increasing temperature;
- Low levels of clay in the soil texture;
- Decreasing soil moisture;
- Topsoil removal;
- Conversion of grassland, forest and otherwise naturally vegetated land to arable land;
- Decreasing soil organic matter input;
- High application rates of nitrogen-containing fertilisers (leads to rapid mineralisation of organic matter).

The risk of loss of soil organic matter in road projects can be reduced by carefully considering the extent of soil disturbance, including topsoil removal, conversion of the use of grassland, forest or other naturally vegetated land, and the effects on organic matter inputs. Correctly storing and restoring of organic-matter rich soils, removed for construction, will limit the loss of soil organic carbon.

2.1.3 (local and diffuse) contamination

Soil contamination is the result of pollution, when substances added to soil potentially cause irreversible or non-irreversible damage. While there are over 700 emerging pollutions described in the European environment, which are largely taken into consideration as potential risks in aquatic environments, their potential risk in terrestrial systems is mostly unknown.

Generally, soil contamination is divided into topics based on the source of pollution, whether local/point or diffuse, and the type of pollutant, whether organic, inorganic or particulate. Most organic contaminants are going through a process of decay, but inorganic contaminants do not and stay present in soil. Polycyclic aromatic hydrocarbons (PAHs) are a particular major concern. Though PAHs are organic compounds, they have been found difficult to degrade.

Moreover, some breakdown products produced during degradation of PAHs have been found to be more toxic than the original PAH [63]. Worldwide, mineral oil and heavy metals are the major source of contamination. Microplastics are an emerging form of contaminant from diffuse polluting sources and can become critical within a couple of decades if pollution is not halted. In particular, tyre wear is expected to be an important source for microplastics in the environment [64,65], as 10–20% of a tyre's weight is worn off during traffic [66]. This causes among others emissions of PAHs, with varying toxicity dependent of the rubber formula [67]. Tyre wear particles are, however, rarely found in pure form, but are often mixed with road particles, meaning that the rubber particles from tyre wear are incorporated with metals worn off from car or pavement [68,69]. Wind, water and road runoff play a crucial role in the distribution of the particles.

The use of fertilizers, manure and agrochemicals have been important sources of cadmium, copper, mercury, palladium and zinc. The concentration of these heavy metals in soils have also been found to inversely correlate with the distance from roads.

When soil contaminations are present in soil above a certain level, partial or total loss of soil functions can occur. Soil contamination can decrease biological activity, primary productivity, the filter function of soil to protect water bodies and it can affect human health, for example through the uptake of contaminants by crops.

Natural sources of soil contamination are, among others, related to parent material (e.g. serpentine soils), volcanic emissions, fires and bioaccumulation. Arguably, deposition of eroded material is a form of contamination, which makes soil erosion a driver of soil contamination. The main drivers of soil contamination are, however, of anthropogenic nature:

- Production and processing of metals;
- Waste disposal;
- Combustion of fossil fuels;
- Production of (in)organic chemicals and fertilisers;
- Application of agrochemicals;
- Application of manure, particularly if containing veterinary drugs.

Roads contribute to soil pollution over a range of activities during the construction and operation. Among others[63,70–72]

- The production of asphalt leads to the contamination by PAHs;
- Accidental spilling of oils chemicals and other hazardous can occur;
- Contaminated soils can be brought to another site;
- Combustion of vehicles releases soot particles, nitrogen oxide, sulphur oxide and PAHs and may release fuel additives such as tetraethyl lead and methyl-tertiary butyl ether;
- Wear of tyres causes severe diffuse pollution;
- Wear of brake pads are a significant source of copper, zinc and lead;
- Management activities may require the use of chemicals for weed-control and de-icing salts, and;
- Roadside litter may be produced.

The risk of biological contamination by invasive alien plant species is highly related to roads [73]. Namely, in the construction phase, seeds can be transported from one site to another by the machines involved. Once opened for society, in particular long-distance driving can introduce seeds into new areas. Since invasive alien plant species outcompete naturally present plant species, a change of plant composition occurs, and this may affect soil biological characteristics. If the invasive plant species are controlled by the use of herbicides, risk of soil contamination are further increased.

Strategies to reducing the impact of soil contamination are minimising these sources of pollution. Regarding the pollution from road operation, the most effective measure is to reduce the number of vehicles. However, traffic flow characteristics also greatly affect the emission of pollutants, where free flowing traffic is less polluting due to more efficient engine operation and less erosion of brakes [70].

2.1.4 Soil sealing

Soil sealing is defined as the destruction or covering of soils by buildings, construction or layers of artificial material that is completely or partly impermeable, such as asphalt and concrete. Sealing of soil interrupts the exchange between the soil and the atmosphere. It decreases the availability of soils for plant growth as well as available soil habitat, greatly reduces the filter and storage functions of soils (of pollutants, water and elements as carbon), increases the risk of runoff, thereby triggering floods, and increases surface temperature. The impact of sealing soil on the performance of neighbouring sites is yet poorly documented, but some impacts are obvious: if impervious, water will runoff the sides of the sealed area instead of infiltrate the soil; if an agricultural field is split, the remaining fields will have more headland where soil structure is degraded due to turning machines [74].

Soil sealing only has an anthropogenic cause and is driven by the want of expansion of infrastructure and settlements. The threat of sealing can be reduced by carefully considering the need for expansion of the road network and by minimising the area needed for these expansions or by 'de-sealing' sealed areas. The area of land sealed for the expansion of the transport network is relatively small in relation to the total area sealed for expansion of economic sites and infrastructure (Table 3). The consequence of road projects on soil sealing reaches beyond the area needed for the road itself [72]. In particular, newly constructed motorways attract for example gas stations, rest stops, shopping centres and industrial estates. In other words, roads may lead to further urbanisation and sealing of soil.

Alternatively to letting an increase in infrastructure result in increased sealing, other sealed areas can be 'de-sealed' or dismantled [10]. However, de-sealing does not *per se* restore soil functioning. For example, Matthees *et al.* observed that even 10–20 years after road removal, soil organic matter and pH had not restored and thereby neither had nutrient availability and vegetation dynamics [75]. In this case, topsoil was mixed with subsoil during road removal.

Table 3. Land cover flow from different land uses (in columns) to sealed areas in the EU28 during 2012–2018. Source: CORINE Land Cover Accounting Layers [76].

Land cover flow type	Arable land & permanent crops	Pastures and mosaic farmland	Forests and transitional woodland shrub	Natural grassland, heathland, sclerophylous vegetation	Wetlands	Open space with little or no vegetation	Water bodies	Artificial surfaces
lcf2 Urban residential expansion	230	131	29	3	0	0	0	0
lcf3 Expansion of economic sites and infrastructure	1393	710	511	153	49	10	19	41
lcf32 Expansion of transport network	102	54	22	9	0	1	0	0
% lcf32 of lcf2+lcf3	**6**	**6**	**4**	**6**	**0**	**10**	**0**	**0**

2.1.5 Soil compaction

Soil compaction refers to a densification of soil and distortion of soil porosity when mechanically loaded from the surface; a rearrangement of soil particles that reduces the size, frequency and connectivity of existing pores and void spaces. Compacted soils are also characterised by a higher soil strength. Soils with a penetration resistance of more than 2 MPa tend to severely restrict root growth, while transport of water and air are greatly restricted if porosity is reduced to less than 10%. Although soil can be naturally compact, soil compaction is frequently a threat to forest and agricultural soils, triggered by the use of agricultural, forest or construction machines or by animal trampling. Due to the increase in wheel loads of machines, the risk of compaction has increased to deeper soil layers. While alleviating soil compaction in the upper part of the soil can be done through cultivation, subsoil compaction is much more persistent. Cultivation of the topsoil layer may, however, also form a plough pan under the tilled topsoil layer. This pan layer is extremely compact and strong and acts as a bottleneck for subsoil functioning.

Compaction particularly affects macropores (pores > 50 µm in diameter); the pores extremely important in the transport of water and air. This results in an increase in anoxic conditions and surface runoff. Soil compaction thereby adversely impacts habitat (for soil fauna and roots) and the increased soil strength reduces bioturbation. The loss of air volume also impacts climate regulation, for example as denitrification is enhanced, which can result in higher N_2O-emissions, and because cultivation of compacted soils requires more energy, which increases CO_2-emissions. Deep cultivation (to alleviate compaction) can destroy cultural artefacts stored in soil, and compaction can induce or accelerate erosion (from wind and water) and cause landslides. Typically, harvest at compacted soils suffer from a permanent yield penalty of about 5% [77], but losses over 10% and up to 38% have also been reported [78].

Compaction occurs within a couple of seconds when the stress, for example, from the load of a wheel, exceeds the strength of the soil. The stress exerted on the soil surface propagates into the soil. The magnitude of stress decreases with increasing soil depth but reaches deep into the subsoil the higher the wheel load. The risk of soil compaction generally increases for higher mechanical stresses and lower soil strength, or with:

- Higher soil moisture content during static (storage) or dynamic (traffic) loading;
- Higher average contact pressure;
- Higher wheel loads;
- Longer loading time.

In road projects, the primary contribution to controlling the risk of soil compaction is by evaluating the level of stress the soil is exposed to against the soil strength. Construction sites have to be prioritised on sites with a relatively high rather than low soil strength. Soil strength is greatly influenced by soil moisture content; the soil strength generally decreases with increasing soil moisture content [79]. It then follows that due to temporal variably soil moisture conditions, constructions activities may need to be postponed to protect the soil structure against compaction. This is particularly the case when soils are exposed to mechanical stresses from the (heavy) machines. There are some rules of thumb and decision support tools to assist in such situations.

Soil compaction is only prevented if stress does not exceed soil strength. As a rule of thumb, vertical stress at 50 cm depth should not exceed 50 kPa to prevent subsoil compaction [80]. Another guideline is limiting the average ground pressure (the wheel load divided by the area) to 80 kPa for wet soils and 200 kPa for dry soils [81]. Decision support tools such as Terranimo® [82], widely accessible via www.terranimo.world or www.terranimo.dk, estimate the risk of compaction based on an estimation of soil strength and induced stress.

A critical factor for soil strength is the soil moisture content, hence improving drainage condition can reduce the risk of compaction. Stress can significantly be reduced by distributing the load over a larger surface, such as by the use of large tyres, tracks, and reduction of tyre inflation pressure, and by reducing the wheel loads. At the time of composing this report, the Terranimo® decision support tool is based on agricultural machines and agricultural undercarriage systems. A different undercarriage system, such as metal tracks instead of rubber, or tyres of a different construction can affect the estimate of the risk of soil compaction.

Soil compaction as described above is one form of soil physical deformation from mechanic stresses during wheeling. The use of machines can also induce shear failure, cause soil kneading, and through vibrations from the engine rearrange soil particles [83]. Moreover, static loading can also result in consolidation, following bearing capacity theories.

2.1.6 Decline in biodiversity

A decline in soil biodiversity comprises a reduction in the quantity as well as variety of forms of life in soil. This includes diversity between species as well as within species. Soils contain at least 25% of all living organisms on Earth. It is widely agreed to differentiate between i) microbes, ii) soil microfauna (< 0.1 mm), iii) soil mesofauna (0.1–2 mm), and iv) soil macrofauna (> 2mm), yet it is expected that about 99% of all species have not been identified. The true consequences of a decline in soil biodiversity are still difficult to predict, but it is considered that soils with higher biodiversity are more resistant and resilient to disturbances. Direct effect on soil functioning may, to some extent, depend on the species that are eradicated.

Soil biota are directly and indirectly responsible for, among others, mineralisation of nutrients from soil organic matter, water regulation and formation and maintenance of soil structure. A decline in soil biodiversity does not occur independent of other soil threats.

Drivers of a decline in biodiversity are:

- Soil sealing;
- Decrease in soil organic matter
- Increased contamination
- Increased salinization
- Soil erosion
- Soil compaction

Therefore, the threat of a decline in soil biodiversity is primarily controlled by decreasing the risk of the abovementioned soil threats.

2.1.7 Salinization

Salinization is the accumulation of water-soluble salts (chlorides, sulphates, carbonates and bicarbonates of sodium, potassium, magnesium and calcium) in the soil, and is the term used to refer to three different soil states: i) saline soils with elevated salt concentrations, ii) alkaline soils with alkaline compositions (high pH, often due to a dominance of (bi)carbonate anions), and iii) sodic soils, also called alkali soils, with a disturbed cation ratio in favour of monovalent cations such as sodium and potassium over divalent cations.

Consequences of soil salinity vary with the form and stage of salinization. Too high concentrations of sodium cause soil structural degradation, as the bonds between soil particles weakens when soil is wetted. Salinization can impact the metabolism of soil organisms and reduce or even destroy plant growth, among others through osmotic inhibition and plant mineral nutrient imbalance. This can leave fertile soils barren, which, in turn, increases the risk of erosion through wind and water.

Salinization has natural drivers, in which case the process is referred to as primary salinization, among others:

- Physical or chemical weathering;
- Transport from saline geological deposits;
- Discharge of saline water from industries and mining activities;
- Rising water table.

Secondary salinization is a result of human activities, such as through:

- Inappropriate irrigation practices, in particular in areas with low precipitation and high evapotranspiration;
- Use of salt-rich irrigation water;
- Poor drainage conditions, such as coastal protection measures may block natural drains of water rich in salts and increases leakage to the groundwater system may cause the water table to rise;
- De-icing salts from roads.

Roads relate to the threat of soil salinization in particularly due to the use of de-icing salts such as sodium chloride (NaCl) – a widely used approach in wintertime in large parts of Europe. Various studies have shown that the soil chemistry of the roadside is greatly impacted by the use of sodium chloride, where sodium has been found to replace and thereby cause the leaching of calcium, lead, copper and zinc [84,85]. This change of soil chemistry can affect soil biodiversity and primary productivity, but it may also increase the risk of soil water erosion. Namely, while calcium has a stabilising effect on soil aggregates, sodium causes dispersion. De-icing salts impact the soil near the roadside most (3–6 m), but the salts have been found at greater distance from the roadside too, moved there by runoff water. This causes de-icing salts to also modify soil characteristics further downstream [86].

Suggested measures to reduce salinization on-site are improved irrigation techniques and improving drainage conditions. Instead of using salts for de-icing roads, sand can be used. However, the use of sand may contribute to the clogging of drainage systems.

2.1.8 Floods and landslides

Flooding refers to the overflowing of waterbodies, to the accumulation of and runoff from drainage water on the soil surface. A landslide is an extreme form of soil erosion where a mass of rock, debris, artificial fill or soil moves down a slope under the force of gravity, when the inherent resistance of the slope is exceeded by the shear forces acting on that slope. Both type of events are only more recently considered as soil threats for the soils they physically impact; The Commission of the European Communities mentioned the soil threats flood and landslides to highlight the impact of these events on other soil threats such as erosion and contamination, with sediments or other, while being aggravated by soil threats as compaction or sealing [23].

Events as flooding and landslide also have direct impacts on soil. Overflowing can be in the form of water ponding on the soil surface, which causes anoxic conditions in soil and may, among others, impact soil biodiversity, primary productivity and the regulation of greenhouse gases. Flooding events on sloped surfaces are erosion by water. Both may transport particulate and soluble contaminants. A landslide exposes a threat to soil both where it slid from and where it is deposited. The effect, particularly at the start, depends on the volume of soil that is lost. For example, whether only topsoil or all soil material slid.

Different types of landslides can be classified according to their ‘mode of failure’: i) flowing, ii) sliding and slumping, and iii) toppling and falling. The susceptibility of soil to any of the modes is related to four main factors: i) type of soil (rock, granular, or clay and silt contents); ii) strength of soil, iii) slope steepness, and iv) soil moisture content. However, the direct drivers of a landslide are site- and time-specific, and often the combination of interactions between the factors mentioned caused failure. Therefore, reducing the risk of landslides will also be site- and time-specific. General measures include, for example, stabilising slopes that have already (partly) failed, re-vegetating, increasing soil drainage, reducing external loading.

3. Assessment of soil functioning in spatial planning

The concepts of soil threats, soil functions and ecosystem services have been developed, discussed and refined over the past decades, but their practical implications, among which in spatial planning, is still limited. According to Drobnik *et al.,* only few attempts were made to include soil functioning in spatial planning frameworks [28], even though several approaches to map soil quality exist. The usability of the concept of ecosystem services has been explored too. For example, Bagstad *et al.* compared 17 tools that can assess, quantify, model, value and/or map multiple ecosystem services [87]. The authors concluded that depending on the distinct geographic and decision context, different approaches would be more appropriate. Moreover, the authors argued that if one tool is flexible and robust enough for the quantification (and valuation) of ecosystem services in diverse context, its use will still be dependent on the time required to run the tool. For a case-study area of approximately 12'000 km^2 (the San Pedro river Basin in Arizona), Bagstad *et al.* estimated a 10–800 person-hours are needed [87].

To arrive to a suitable framework for the assessment of soil functions in road projects, with the aim to reduce the impact of road projects on soils, we reviewed examples of research addressing soils in spatial planning. This, in combination with the reviews of soil threats, soil functions and soil ecosystem services as summarised in Chapter 2 and presented in Appendixes 1–2, are the foundation of the outlined frameworks presented in Chapter 4 and 5.

3.1 Soil-based frameworks for spatial planning

A number of frameworks for soil function assessment have been developed recently to facilitate spatial planning decisions (see also Geiges et al., 2021, p.15). They enable balancing trade-offs between soil functions and land-uses[88]. In principle, these frameworks assign values of soil function performance to specific areas, so that an "asset" of soil quality can be calculated for a region. Authorities can define a share of this soil quality asset that can be "consumed" for building purposes. Examples of soil function assessment frameworks are the Soil Quality InDicator (SQUID)[28], the BOKS (Bodenkonzept Stuttgart)[89], and operational frameworks as outlined by Fossey *et al.* [90] and by Choquet *et al.* [54]. Soil function assessment frameworks are based on unitary soil information from detailed soil maps or field campaigns. Therefore, in practice they are only used at the local level for municipal land-use planning, such as the BOKS. However, many of these frameworks are only mentioned in scientific literature, such as the SQUID, because the necessary soil data is missing for practical use.

The use of these soil function assessment frameworks for road projects is currently limited because roads span across multiple areas for which the necessary soil data are usually not available unitarily. However, these frameworks could help defining compensation measures within certain regions in the context of a road project. On the one hand, they can help quantifying the impact on the soil functions due to the new road (section), e.g. expressed in soil index points. On the other hand, they can be used to quantify the benefit of the compensation measures to restore/enhance (some of) the soil functions.

Drobnik *et al.* calculated and compared soil quality maps for ten municipalities in Switzerland [28], based on the BOKS (Bodenkonzept Stuttgart [89]) and the SQUID (Soil Quality InDicator [28]). Both indices were developed for spatial decision-making. The BOKS is based on the size of different parcels and scores for different soil functions: i) the suitability for the production of biomass, ii) regulation of the water cycle, iii) filtering and buffering capacity, iv) preserving geogenic and cultural heritage, v) contamination, and vi) level of sealing. In the SQUID approach, 23 soil-based ecosystem services are indexed, based on the quality of (10 different) soil functions and expert-assigned weights of the average contribution of a soil function to an ecosystem service, and then aggregated into a single indicator [28].

Both indices were then spatially clustered into cold- and hotspots, for both of which confidence levels of 90, 95 and 99% were calculated. Confidence levels < 90% were considered non-significant. These methods were applied to plot maps with a resolution of 20m^{x}20 m that did not include forest, water bodies and small gardens within settlements. Drobnik *et al.* observed a distinct difference between two indices: the BOKS showed low spatial soil quality variability, while the SQUID resulted in high spatial soil quality variability [28]. The authors therefore concluded that the BOKS lacks spatial differentiation for small-scale spatial planning, while the SQUID lacks the ability to provide a general impression. Or, the other way around, the BOKS worked well for obtaining an overall impression, while the SQUID worked well for small-scale spatial planning.

Fossey *et al.* [90] and Choquet *et al.* [54] introduced two different operational frameworks to consider soil ecosystem services in territorial planning. Fossey *et al.* focussed on agricultural systems and proposed to model and map the current and potential state of soil-ecosystem services provision per site in a given study area (Figure 7) [90]. The current state is based on the soils existing biological, chemical and physical characteristics and the current land use (crop- or grassland) and management. Next, soil-based ecosystem services are also modelled for all alternative pairs of soil and land use plus management, which gives the potential state. Comparing the two maps reveals gains or losses for a change of soil use or management, which then can serve as an aid in decision making.

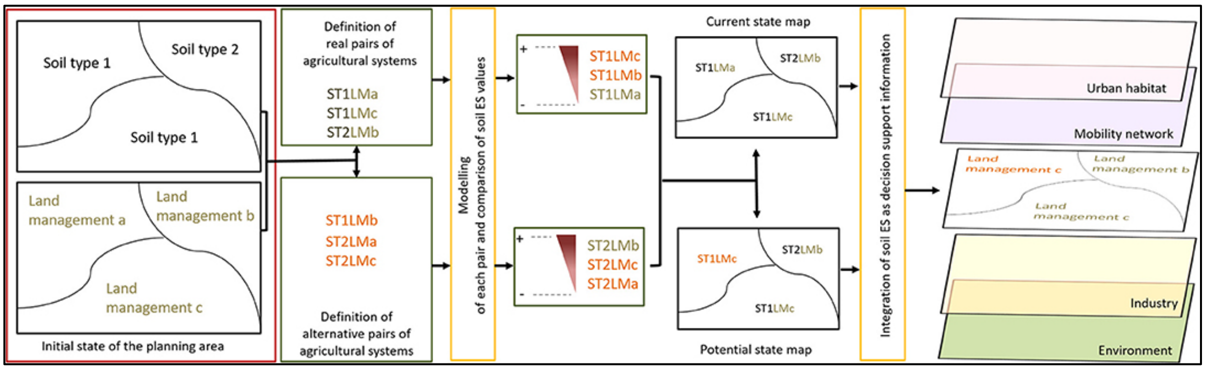


Figure 7. The operational model to account for (soil) ecosystem services in an agricultural system as proposed by Fossey et al. [90].

The BOKS and the SQUID are examples of empirical models, and these approaches are criticised by Greiner *et al.* [12] and by Choquet *et al.* [54]. They argued that empirical models established statistically or based on expert knowledge generally assess the capacity of soil to deliver a specific service more or less independently of land use and management, but lack the mechanistic representation of soil processes that include site-specific environmental factors, temporal and/or spatial variations in land-use and management. Mechanistic models may instead be used for a range of soil-supported services for which empirical models have not been established. Established models may be particularly suitable for use in the Environmental Impact Assessment (EIA) [91] to assess the impact of soil compaction (e.g. using Terranimo ® [82], Chapter 2.1.5) and soil erosion (e.g. using RUSLE [92]).

Choquet *et al.* compared the ability of empirical and process-based models to translate the spatial variation in soil properties of the Saclay plateau (France) into levels of (soil-supported) ecosystem services, thereby addressing the question whether or not empirical models represent soil functioning realistically [54]. According to Choquet *et al.,* this uncertainty roots in limited knowledge of the availability, spatial and temporal variability and/or reliability of underlying data [54]. However, the authors acknowledge that no single tool (neither empirical nor process-based) for modelling of ecosystem services covered a realistic representation of the variability and complexity of soil processes and properties.

Empirical modelling was then based on i) The Muencheberg Soil Quality Rating (M–SQR) expert-based tool [93] for the provision of plant biomass, ii) the SENSIB [94] for the provision of water, iii) MERLIN v2 [95] for the regulation of water quality, and iv) the 'carbon saturation approach' [96] for climate regulation. The process-based modelling was based on yields, water flow (at one meter-depth), non-leached nitrogen and storage of organic carbon in the top 30 cm, all simulated with the CERES-EGC crop model [97]. While the process-based modelling worked well for arable soils, empirical modelling seemed to have advanced enough to make some comparison across land-use such as needed in spatial planning. An important note from the authors is that the availability of empirical models for different pedoclimatic conditions is very uneven for different ecosystem services.

The BOKS, SQUID, and both the approaches of Choquet *et al.* [54] should allow for decision-making based on the concept of ecosystem services, yet to do so requires quantification and valuation of the services and their importance to society. The SQUID does indirectly address the value of soil functions but not of ecosystem services, since 'experts' were asked about the importance of a soil function for ecosystem services, but not about the importance of the ecosystem service for society. The framework developed by Fossey *et al.* [90] could potentially be widened into including land use other than agricultural. However, it is unclear how the gains and losses of services are weighted as supply-demand is not taken into consideration; more of a specific service is not always better on a larger scale. Finally, despite the awareness of the interdependencies between ecosystem services as also exist between functions, none of the abovementioned approaches have explicitly accounted for this, as they are based on the current state of land use and soil.

Jost *et al.* performed a case-study to estimate the change of soil functioning over time [32]. This change was driven by changes in soil organic carbon in the top 30 cm, in turn driven by land use change and climate change. The authors compared the extent to which crop and grasslands in the Mostviertel region (Austria) performed the five often prioritised agricultural soil functions (i) nutrient storage, ii) water regulation, iii) productivity, iv) habitat, v) carbon sequestration) in 2017 relative to the year 2100, for different land use and climate change scenarios. The methods to estimate the level of soil functioning (five levels; very good to very poor) were based on existing assessment schemes using soil and land use data, adapted to the Mostviertel region, at a 1-km grid resolution. Pedotransfer functions were used to replace missing data. The combined effects of the change in land use and climate resulted in both increased and decreased performance of the different soil functions. The authors stressed that their case-study highlights the need to explore possible futures of the effects of land use and climate on soil functions. Moreover, Jost *et al.* advocated the integration of a wide range of actors in land use planning instead of restricting the decision-making to authorities alone, since the local actors such as farmers have a significant influence on the actual level of performance of soil functions [32].

Muradian and Gómez-Baggethun argued that the concept of ecosystem services may continue to fail to improve the attitude of people to, and their dealing with, the natural environment sufficiently [98]. They argue that an anthropocentric and utilitarian representation of human-nature relationships are major causes of the global environmental crises, and advocate for a rigorous move away from these anthropocentric and utilitarian centred concepts, such as ecosystem services. As an alternative, Muradian and Gómez-Baggethun pledged focussing on care-concepts based on responsibility [98]. This trails back to Potschin and Haines-Young and Robinson *et al.,* who both argued that focussing on the outputs of processes (services instead of functions) allows the underlying systems to be overlooked [99,100].

As an alternatively to the use of ecosystem services, spatial planning could be driven by an assessment of soil functions. Although the concept of soil functions is not free of anthropocentric bias (assessments are still based on what human prioritise) it does focus to a much larger extent on the underlying system that produces ecosystem services.

This, in combination with the high interdependencies between soil functions, seems a promising approach in protecting the natural environment. As a case study, Schulte *et al.* quantified and mapped the supply and demand for the five prioritized agricultural soil functions on a national scale in the Republic of Ireland and could thereby identify pathways to manage soil functions so that the supply would match the demand, such as a conversion of land use and a change of soil management [38].

The review above shows progress made in the assessment of soil functioning (whether ecosystem services or soil functions) in a spatial planning-context. However, in addition to drawbacks highlighted, an assessment suitable for road projects should allow quantification of the impact of the new road (section) on soil functions, limiting soil threats and quantification of the benefits of compensation measures to restore and enhance soil functioning.

3.2 Defining the scope for the frameworks in road context

The need to protect soils is high, taking into consideration that:

- Soil is a natural dynamic system in which many biological, chemical and physical processes vital to life take place;
- The ability of soils to conduct these processes sustainably is severely impacted mainly by anthropogenic-induced soil degrading processes that change soil functional (biological, chemical and physical) characteristics;
- Soil degradation has over the past decades largely been disregarded in decision making (e.g. in spatial planning and soil management);
- The results of soil degradation processes can be linked to pressing socio-environmental challenges such as climate change and the loss of biodiversity, and;
- Degraded soil cannot always be compensated for due to the high heterogeneity of soils and their environment, spatial-temporal variation in the demand and supply, and the high pressure on land.

Road projects can contribute to protecting soil on account that:

- Expansion of the road network usually equals sealing of soils (naturally vegetated or cultivated for forestry or agricultural), as new road corridors rarely cross settlements;
- Sealing of soils interrupts the exchange between soil and the atmosphere and thereby diminishes many soil processes, and;
- Additional areas (in excess of the land sealed) are converted to road verges, and/or used temporarily in the building phase, for storage and transport, the latter during which soil is potentially exposed to a great number of (other) soil degrading processes;
- Roads may result in soil sealing of a far larger area due to the attraction of for example for gas stations, rest stops, shopping centres and industrial estates.

A suitable framework for use in road projects will protect soil functioning due to a common approach of raising awareness and conscious decision making of the placement of the road and construction sites (among others, used for storage, buildings or as work roads) as well as the planning and execution of construction and mitigating measures. Various concepts have been developed that can be used in accounting for the impact of human actions on soil: the concepts of soil threats (Chapter 2), of soil functions (Appendix 1, and of (soil) ecosystem services (Appendix 2).

The three concepts are closely related, namely:

- (*soil*) *ecosystem services* are quantifiable outputs of (soil) functions;
- (*soil) functions* are bundles of (soil) processes, and;
- *Soil threats* are processes that negatively affect the extent to which soil processes can take place.
- Note that: (soil) processes are driven by a soils functional (biological, chemical and physical) characteristics and fluxes between soil, the atmosphere and bedrock.

The concept of (soil) ecosystem services has developed out of the wish to add weight in decision making on behalf of the natural environment:

- Ecosystem services are quantifiable outputs of environmental functions;
- 'Soil ecosystem services' were introduced to stress the importance of soils in the environment;
- Ecosystem services need – like any type of service – to be used to have value, but do not necessarily need to be expressed in economic terms;
- Ecosystem services address interdependencies between human and the environment;
- Different levels of provision can (theoretically) be distinguished: flow, demand, potential supply, capacity, capability, and potential maximum supply;
- Ecosystem services are interdependent: maximising the production of a single service will limit the production of another service;
- (soil) Ecosystem services are largely driven by soil functions;

(but) The concept of ecosystem services has not yet succeeded to add weight in decision making on behalf of the environment, among others because:

- Ecosystem services are confused or mixed with soil functions;
- Differentiating between intermediate and final ecosystem services, needed to prevent double accounting, is complicated;
- Direct quantifications of ecosystem services are lacking;
- Ecosystem modelling is complicated by the lack of (empirical) models for different pedoclimatic conditions;
- The value of services is highly influenced by supply and demand, as well as based on the stakeholders awareness and preferences.

The concept of soil functions was introduced to promote a sustainable use of soils:

- Soil functions are bundles of soil biological, chemical and physical interactions;
- Soil functions determine to a large extent the capacity of soils to deliver ecosystem services;
- All soils are, in principle, able to fulfil all functions but they cannot all functions to the same extent;
- Soil functions are assessed through measurable soil function characteristics or outputs;

(but) The concept of soil functions has not yet successfully been used to asses a wide range of soil functions across different land uses, namely:

- Five soil functions are often prioritised: i) primary productivity, ii) nutrient cycling, iii) water regulation, iv) climate regulation, and v) biodiversity, but the relevance of these functions may vary between geographical regions and land use;
- Soil functions are partly interdependent because of shared soil processes and characteristics;
- Not all synergies and trade-offs are understood, and there is evidence that the intensity may varies between climatic regions;

- Soil (quality) assessment are generally based on the current use and management of land, while only a few assessments exist that consider changes over time (such as land-use or climate);
- The optimal conditions for soil functions to be performed differ between pedoclimatic conditions.

The concept of soil threats emerged from political commitment to soil protection:

- Eight soil threats were highlighted by the European Commission: i) (wind and water) erosion, ii) decline in organic matter, iii) (local and diffuse) contamination, iv) sealing, v) compaction, vi) decline in biodiversity, vii) salinization, and viii) floods and landslides;
- Most soil threats are naturally occurring processes, yet a soil threat is, by definition, aggravated by human activity;
- Soil threats can be actively prevented if driving forces are recognised;
- For various soil threats, indicators to assess the likeliness and/or severity and extent of soil degradation by soil threats have been established.

These considerations lead us to propose two frameworks to assess and limit the impact of road projects: one for the planning phase and one for the construction phase. In the planning phase, we suggest focussing on soil functions (Chapter 2, Appendix 1). For one, soil functions is a more straightforward concept than ecosystem services, the latter which has many different levels (of supply) which have rarely been quantified, let alone valued. Case-studies of the performance of soil functions have shown that soil functions can be estimated from measurable soil properties and information on land use, and that supply and demand can be taken into account. Moreover, since soil functions determine to a large extent the capacity of soils to deliver ecosystem services, we need to be fully aware of soil functions before we are able to comprehend ecosystem services. In the construction phase, it is critical to not change soil biological, chemical and physical attributes to the extent that soil functions are impaired. During construction, one should be proactive to prevent soil degradation and save resources. This is best done through managing soil threats (Chapter 2).

4 A framework to assess and limit negative effects on soil in the planning phase

Intuitively, there may be an urge to protect soils with an apparent good soil quality; e.g. soils with a high primary productivity or soils that support biodiversity well. Similarly, restoring soil functions might intuitively be allocated to soils with an apparent poor quality, e.g. soils with a low primary productivity. However, more of something is not always better. Due to the interdependencies between soil functions, soil functions may be optimized but cannot all be maximized. This emphasizes the need for balancing the supply and demand of soil functions. This is a crucial aspect in the framework outlined below. However, from the perspective of protecting soil functions, it is critical to start with the questions 'Is this road really necessary?' and 'Does it need to have these dimensions?'. The less soil is sealed, the more soil is available to perform other soil functions critical for life on Earth. The conceptual framework outlined here assumes these questions have been asked and answered positively. This framework then allows limiting the impact of roads on soils in the planning phase in four stages (Figure 8).

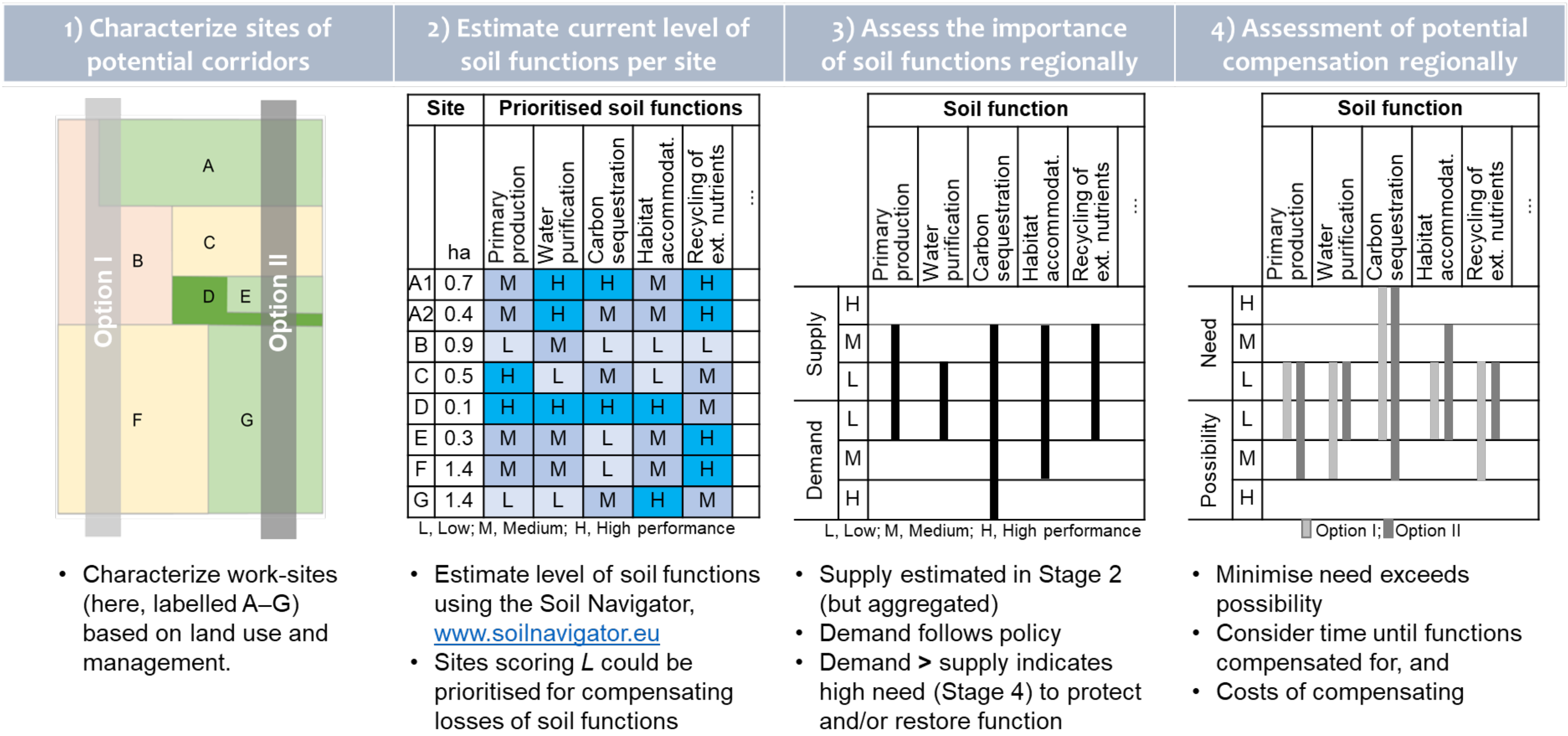
1) Characterise sites of potential corridors

From a soil protection perspective, the earlier the impact of road projects on soils is included, the better. The latest stage at which potential impacts on soil functions should be included is when multiple options of the new road-corridor are on the table. For the different optional corridors, the land-use and the intensity of soil management of the potential required sites need to be characterised, and important soil functions present should be listed (Figure 8, stage 1). Inclusion of the following functions are proposed: i) primary productivity, ii) nutrient cycling, iii) water regulation, iv) climate regulation, and v) biodiversity. These functions have been studied in detail, and their understanding and assessment is most advanced. Narrowing the focus down to fewer functions falls short in recognition of the multifunctionality that soils have. Specific areas may have a known high level of another function, such as preservation of geogenic and cultural heritage, or other functions than currently specified may appear crucial. If so, these functions can be added to or replace one of the soil functions previously mentioned. Their importance (stage 2) will be estimated (expert knowledge) as M (medium) or H (high) performing. Assessment methods other than expert knowledge may be developed.

2) Estimate current level of soil functions

When the sites have been characterised (stage 1), the current level of the performance of a soil's functions should be estimated (stage 2). This can be done using Soil Navigator, www.soilnavigator.eu (Video tutorials are currently available in English, German, French and Danish), which uses data of a soils biological, chemical and physical attributes as well as land use and pedoclimatic information to estimate a low, medium or high (L, M or H) performance. Management details need to be provided by the land user or other local actor, climatic and topographic data will mostly be known, but soil attributes (Table 4) need to be tested for in a lab (in particular the biological and chemical attributes), estimated or defined on-site. For all attributes, values come in ranges rather than exact values.

Soil Navigator allows small-scale assessment that can capture the great variability of soils' performance of soil functions. Soils inhibit great variation in functional characteristics, among others due to genesis, (local) climate and soil management. Therefore, the performance of soil functions may be highly variable on short distances even on similar land use. Zwetsloot *et al.* has shown it is possible to estimate the current level of the provision of soil functions on parcel-level using Soil Navigator [31].



Site	ha	Primary production	Water purification	Carbon sequestration	Habitat accommodat.	Recycling of ext. nutrients	…
A1	0.7	M	H	H	M	H	
A2	0.4	M	H	M	M	H	
B	0.9	L	M	L	L	L	
C	0.5	H	L	M	L	M	
D	0.1	H	H	H	H	M	
E	0.3	M	M	L	M	H	
F	1.4	M	M	L	M	H	
G	1.4	L	L	M	H	M	

L, Low; M, Medium; H, High performance

Figure 8. The proposed conceptual framework for the assessment of the impact of the planning of new roads on soils. Four stages are outlined, which allow deciding on the corridor for which the impact of the loss of soil functions is least (supply equals or exceeds the demand), or most compensable.

Currently, Soil Navigator is validated for a limited land use (crop and arable), but it may as well be applied to other land uses such as forest and peatlands. Namely, soil functions are intrinsic to a soil and processes that take place are related to a soil's attributes and the soil's environment; the land use has an effect on the soil attributes, which in turn will translate into a certain level of soil functional performance. Note that the model has been validated for five European countries (Austria, Denmark, France, Germany and Ireland). Although this covers some geographical differences, further validation is desirable. These results of stage 2 serve as the 'supply' in stage three (see below).

Table 4. Soil attributes that serve as optional input data in the Soil Navigator tool, www.soilnavigator.eu. Online, an explanation is provided for each attribute. D = default value available; E = can be estimated; M = can be defined on-site; T = requires a soil test. Values are in ranges.

Biological	Chemical	Physical
Not available	pH, T	Organic/mineral, E
Bacterial biomass, T	Cation exchange capacity, ET	Texture, E
Fungal biomass, T	C:N ratio, T	Clay content, DE
Earthworm richness, M	N:P ratio, T	Soil crusting/capping, M
Earthworm abundance, M	Plant-available P, T	Thickness of organic layer, M
Nematode richness, T	Plant-available K, T	Potential rooting depth, E
Nematode abundance, T	Plant-available Mg, T	Groundwater table depth, E
Microarthropod richness, T	Salinity, T	Soil organic carbon content, EM
Microarthropod abundance, T		Soil organic matter content, E
Enchytraeid richness, T		Soil bulk density, EM
Enchytraeid abundance, T		Drainage class, E

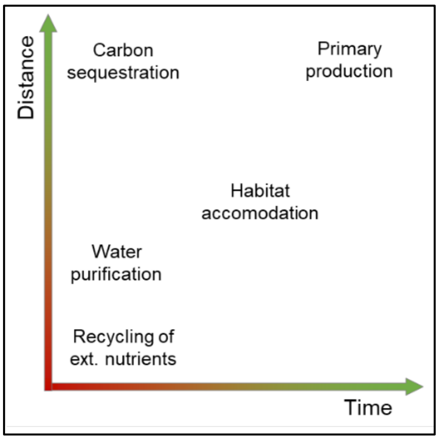
3) Assess the importance of the soil functions

Different soil functions share processes. Therefore, the performance of all functions at a particular (parcel of) soil may be optimised, but not maximised. Maximisation of a particular soil function will cost the performance of another soil function due to trade-offs, while some functions may reinforce each other due to synergies. The importance of soil functions thus relates to the balance of supply (stage 2) and demand. In order to compare the supply and demand, the demand should also be estimated in terms of low, medium or high (L, M or H).

The demand for soil functions, being bundles of soil processes rather than quantifiable outputs, may best be quantified by (local) governments. Assessment of the demand on small-scale will be stakeholder-biased. For example, a farmer or forest manager may strive for a biomass output with a quality as high as possible, which may be at the expense of water quality or soil biodiversity. Protecting soils and their functioning is, however, a major task in terms of the sustainability of life on Earth for the generations to come, thereby the demand for soil functions should be emphasized by governmental institutions. Moreover, while the supply and demand may be unbalanced on parcel-scale, they may be in balance on a slightly larger scale. Schulte *et al.* made such a comparison of supply and demand on a national scale, where the demand is based on EU policy (e.g., number of dairy cows represented the demand for primary productivity and the area needed for the disposal of slurry from pig farms and sewage sludge represented the demand for nutrient cycling) [38]. This approach should be adapted by different regions; the relevance of proxy-indicators may vary. Also, the approach need to be elaborated to include multiple proxy-indicators instead of a single, which will provide a more complete overview of the demand for specific soil functions.

4) Assessment of potential compensation

In the last stage, the need for and the effectiveness of compensation measures is evaluated for the soil that becomes sealed. The 'need' is evaluated based on the (intra-regional) assessment of the demand and supply of different soil functions (stage 3). If the demand exceeds the supply, the possibility of compensating should be assessed. Here, two aspects are of particular importance: i) at which distance can the loss of soil functioning be compensated for, and ii) when should the compensation be realised (Figure 9)? For example, the ability of soil to recycle external nutrients is limited when agricultural land is taken out of production, as no manure can be spread. It is desirable to compensate for this rapidly and on a short distance. The urgency of compensation for the loss of carbon sequestration can be considered high due to the global importance of soil carbon storage, but in many cases the distance at which it will be compensated might matter to a lesser extent. In different situations, the relative importance of time and distance of compensation can differ. Also, for different geoclimatic and political regions and between land-uses, urgency in terms of time and requirements in terms of distance will vary.



It is recommended to compensate for the loss of soil functions on parcels with similar land use from the parcel where the functions are lost. This was also suggested by [101]. It prevents an unintended cascade of performance of soil functioning due to interdependencies between functions, and differences in the soil attributes and pedo-climatic conditions. Logically, compensation is most effective on parcels that have a lesser soil performance. However, the cost of compensation measures may also be needed to consider, and, lastly, the extent to which technical measures are acceptable in replacing a soil function (such as CCS, carbon capture and storage).

Figure 9. Illustration of the relevance of the acuteness and localness of accomplishing compensation measures for the loss of soil functions. Red indicates that compensation should be accomplished rapidly/nearby, whereas green indicates that compensation measures can be realised further in time/at greater distance. In this example, the five often-prioritised soil functions of agricultural land are included. The positioning of the functions within the plot are expected to differ between geoclimatic regions.

Compensating for soil functions lost from a soil being sealed requires changes in land use, management, or site-reconstructions. Practical examples are provided by Haraldsen and Tobias [11], Geiges *et al.* [10] and by Geiges and Tobias [91]. To provide another example, water purification could be improved by increasing water infiltration. This can, for example, be done by levelling sites (i.e. removing depressions). In this approach, priority is given to compensation of soil functions for which the demand exceeds the supply. These soil functions should then score higher in the re-assessment after successful compensation. At the same time, soil degradation is only prevented if none of the soil function scores lower than at the initial assessment.

Once compensation has been finalised, it should also be assessed for to make sure that it was successful or to identify why it was not successful. For on-site compensation measures, the Soil Navigator tool could be re-run on the sites where compensation took place. For example, if from Figure 8 corridor Option 1 is chosen, the loss of soil productivity could be compensated for on parcel G (scored L, low). In the re-assessment, after compensation, productivity should score M (medium) or H (high). In this case, the result also depends on the farmer's soil management. In case compensation has not the desired result, a soil-expert and/or the actors involved should be able to evaluate the compensation measures versus farm management and identify what caused the lack of success.

4.1 Summary, and data and knowledge gaps

The four stages together allow decision of the corridor for which the impact of the loss of soil functions is least (supply equals or exceeds the demand), or most compensable. For this framework to have practical relevance, the following needs to be decided and developed:

- (stage 1) Who is in charge of prioritising soil functions? Will these be mapped?
- (stage 2) SoilNavigator is an example of a model that can be used to estimate the performance of soil functions on a small-scale. It has been validated for grassland and arable land in five countries. Hence, there is a need to validate the model-performance for other land uses and countries/regions. Alternatively, different models can be used for different countries/regions, but these should be validated too.
- (stage 2) [38] stressed that their study on a national scale served as a case study that was used to illustrate the framework they developed. This approach has not been validated, and for it to serve protection of soil functioning well, multiple indicators per soil function should be incorporated. The indicators and soil functions are likely to vary between countries/regions. Maps should be available preferably at a scale that reveals local variations, but the more precise, the more time and resources the investigation will take.
- (stage 3) The demand for each soil function should be established based on multiple proxy-indicators (because of trade-offs).

5 A framework to limit negative effects on soil in the construction phase

Once the road corridor has been appointed, a second operative framework (Figure 10) is needed to limit the risk of soil degradation during construction. The impact on soil of the temporary construction sites, used for, among others, storage, buildings or as work roads, is minimal if the performance of soil functions afterwards equals or exceeds the performance of the functions beforehand. The proposed framework here exists of four stages, of which stage 1–3 should be performed before the temporary construction site is taken in use.

1) Define target values of soil functional characteristics

The optimal soil functional characteristics vary between soil functions, land-use and between pedoclimatic zones. Local management plays a key role in the actual functional characteristics, for example due to difference in farming practices. When land use is temporarily changed (e.g. for months or years), target values of functional characteristics should be set keeping in mind the final land use. For example, agricultural soil may be temporarily used during road construction, but might not necessarily be agricultural soil afterwards; instead, it may be converted to a road verge or natural grassland. Target values should preferably be set based on measurements on sites with the final-land use, in comparable pedoclimatic conditions (same soil type and climate). Established indicators and threshold values are presented in Table 5.

2) Perform site-based risk assessments

When natural and cultivated land is used as a (temporal) construction site, soils are exposed to potential threats. Temporal construction sites may be used for stockpiling, storage, office and as parking lot. This may take place directly on the existing soil surface (or after cutting/mowing), but particularly for long-term use (years), soil may often be sealed for example with a layer of gravel (with or without geotextile), with ground protection mats or even with asphalt.

While specific road building activities and the materials used may vary widely between projects and countries, the activities can be placed into three different groups: i) static loading (stocking), ii) material removal, and iii) material addition. Traffic is included both in material removal and addition. Activities within these three groups can trigger soil threats (Table 6). It is important to reduce the risk of these threats, as soil threats change soil functional characteristics to the extent that the ability of the soil to perform certain functions is reduced. Notice that the risk of soil compaction and loss of biodiversity occur in static loading, material removal and material addition. Soil compaction can result from both static and dynamic loading, the latter which is the case when traffic removes or adds material. Biodiversity is extremely sensitive to changes in soil or the environment and has been noticed to not only be a soil threat in itself, but also to suffer from all other soil threats.

Static loading (or stocking) leads to sealing of soil. It can also cause soil contamination due to pollution, compaction due to exerted stresses exceeding soil strength, a decline in biodiversity due to disturbed gas- and water transport, and floods and landslides if loaded on a slope. Material removal can cause wind and water erosion due to removal of vegetation and reduction of aggregate stability, a decline in organic matter and biodiversity if topsoil layer is removed and soil deeper down aerated and to soil compaction when traffic exerts mechanical stresses higher than the soil strength. Adding of material can result in contamination and a decline in biodiversity if (some of) the added material remains but is alien to the site, to compaction when traffic exerts mechanical stresses higher than the soil strength.

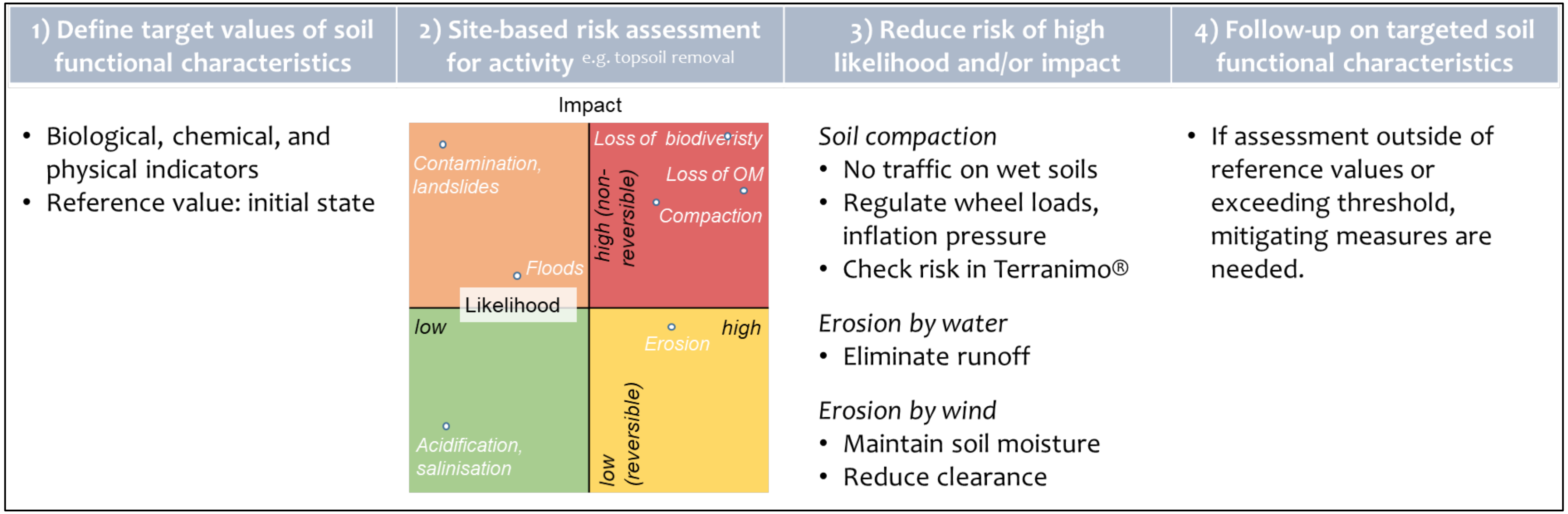


Figure 10. The proposed conceptual framework for limiting the impact of temporary use of soils in the road building phase.

Table 5. Widely established soil biological, chemical and physical indicators and threshold/target values.

Indicator	Land use	Lower threshold	Upper threshold	Optimal	Comment	Source
Biological indicators						
To be defined		To be determined	To be determined	To be determined	requires sub-indicators by species (functional) group)	[1]
Chemical indicators						
Soil organic carbon (SOC)	mineral cropland	0.5–1.9%	1.2–3.2%	0.7–2.7	Thresholds vary with soil texture class, type of fertiliser and the water balance during summer.	[1]
Nitrogen (N)	Agriculture		NH_3 in air: 1–3 [mg NH^3 m^{-3}] NO_3 in ground water: 50 [mg NO_3 l^{-1}] N in surface water: 1.0–2.5 [mg N l^{-1}]		Mineral N: sum of available NH_4 and NO_3	[1]
Phosphorus (P)	Agriculture			P-concentration 25-35	Extractable P concentration < optimum (value range refers to Mehlich 3-ICP; also available: P-Bray P1 and Olsen P)	[1]
C/N ratio	Forest	25–30		> 30	C/N ratio in the organic layer	[1]
pH	Agriculture	5.0–5.9		6.0–7.0	Varies between crops and might differ for other crops	[1]
Base cation : inorganic aluminium levels	Forest	0.5	2.0	1	Base Cation: Ca+Mg+K	[1]
Heavy metals Cd, Cu, Pb and Zn	Cropland		Country-specific		Country-specific values vary broadly and are not necessarily comparable. Stratification by land use and soil texture	[1]

Indicator	Land use	Lower threshold	Upper threshold	Optimal	Comment	Source
Physical indicators						
Bulk density	Not specified	1.2 Mg m^{-3}	1.9 Mg m^{-3}	1.2–1.6 Mg m^{-3}	From clay to silt and sand	[1]
Degree of compactness	Arable			87%		[2] [3]
Air-filled pore space	Not specified	5% or 10%			at soil matric potential of -6 kPa (near-saturated)	[1] or [4]
Total pore space	Not specified	35%			irrespective of texture effects)	[1]
Saturated /unsaturated hydraulic conductivity	Not specified	< 10 cm d^{-1}				[1]
Air permeability	Not specified	$12*10^{-4}$ cm s^{-1} or 1 μm^2			In particular loamy, silty and clayey soils at high water content and/or weak aggregation	[1] or [5]
Oxygen diffusivity	Not specified	$1.5*10^{-8}$ m^2 s^{-2}			In particular loamy, silty and clayey soils at high water content and/or weak aggregation	[1]
Oxygen diffusion	Not specified	< 0.005 or 0.02			For loamy and sandy soils, respectively	[1]
Soil organic carbon to clay ratio	Not specified	1:10 (0.10)		1:8 (0.125)	1:8–1:13 as minimum desired level for agricultural soils	[1] [6]
VESS or SubVESS (visual evaluation of soil structure, top or subsoil)	Not specified		3	1–3		[1] [7]
Soil organic matter	Extensive management	0.9–3.3%			Depending on soil texture class and water balance during summer.	[1]
Soil organic matter	Intensive management	1.71–4.83%			Depending on soil texture class and water balance during summer.	[1]

Indicator	Land use	Lower threshold	Upper threshold	Optimal	Comment	Source
Soil organic carbon	Fertilizes cropland	Sand: 1.0%) Silt: 1.4% Loam and clay: 1.0%				[1]
Penetration resistance	Not specified	2 MPa			Dependent on soil moisture conditions	[8]

[1] [57]; [2] [102] [3]; [103]; [4] [104]; [5] [105]; [6] [106]; [7] [107]; [8] [108].

3) Reduce the risk

For those risk assessments that result in high likelihood and/or high impact, mitigating measures have to be taken. Key-considerations of risk-mitigating measures are included in Table 6. Best-practices for avoiding, mitigating and compensating impacts on soils are reviewed by Geiges *et al.* [10].

4) Follow-up on targeted soil functional characteristics

When the construction work has completed and the construction site is returned to its original land-use, a follow-up is needed to check if the defined target values of Stage 1 are at least maintained. In some cases, even risk-reducing measures may not prevent degradation. When the follow-up measurements indicate a deterioration of the soil, further mitigating measures are needed to restore full soil functioning. The stages two and three can be used to check stakeholders' responsibilities.

5.1 Summary, and data and knowledge gaps

The four stages together will reduce the risk of (permanent) degradation of important soil functions. However, it is important to keep in mind the following:

- (stage 1, stage 4) soil functional indicators are continuously refined. Biological indicators are currently underrepresented. For many indicators, no thresholds or target values are known. The threshold and target values will often vary between soil types, land use, climatic zone and over time (for example, between seasons);
- (stage 2) practical guidelines on how to perform risk assessments are needed, targeting national/regional important soil functions.
- (stage 3) practical guidelines on how to reduce prudent soil threats should be widely available.

Table 6. Soil threats in road constructions, main risk-factors that increase the likelihood of soil degradation processes, the activity during which the risk might be high and risk-mitigating measures (during construction). Note that no X for activity does not indicate that the risk is non-existing. [1] Includes traffic. [2] At the time of writing this report, Terranimo ® is based on agricultural machines.

Soil threat	**Key-factors**	**Activity**			**Risk-mitigating measures**
		Static loading (stocking)	Material removal [1]	Material addition [1]	
Wind and water erosion	Heavy wind/rainfall, slope and bare soils		X	X	Reduce bare soil in period with high rainfall and strong wind, particular on/near slopes.
Decline in organic matter	Soil disturbance, decreasing organic matter input		X		Decrease soil disturbance, maintain organic matter input.
(local and diffuse) contamination	Pollution (biological, chemical, physical)	X		X	Prevent leaching, volatilisation, drift, mixing of pollutants to soils, alien species
Sealing	Size and base needed	X		X	Limit area and time needed (e.g. for storage or Construction trailers)
Compaction	Low soil strength, high loads, repeated loading	X	X	X	Prevent exposed mechanical stresses exceeding soil strength. Use of decision support tool Terranimo ® www.terranimo.world or www.terranimo.dk [2]
Decline in biodiversity	Other soil threats	X	X	X	See other risk-mitigating measures
Salinization	Use of salt and saltwater, poor drainage conditions			X	Prevent poor drainage conditions and limit use of salts
Floods and landslides	Low soil strength, poor drainage, high loads	X	X		Prevent poor drainage conditions

6 References

1. Veldkamp, A. & Fresco, L. O. CLUE: A conceptual model to study the conversion of land use and its effects. *Ecol. Modell.* **85**, 253–270 (1996).
2. FAO & ITPS. *Status of the World's Soil Resources (SWSR) - Main Report. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils. Intergovernmental Technical Panel on Soils* (2015).
3. Koch, A. *et al.* Soil Security: Solving the Global Soil Crisis. *Glob. Policy* **4**, 434–441 (2013).
4. Keesstra, S. *et al.* Soil-related sustainable development goals: Four concepts to make land degradation neutrality and restoration work. *Land* **7**, (2018).
5. Koch, H. J., Heuer, H., Tomanová, O. & Märländer, B. Cumulative effect of annually repeated passes of heavy agricultural machinery on soil structural properties and sugar beet yield under two tillage systems. *Soil Tillage Res.* **101**, 69–77 (2008).
6. Blum, W. E. H. Soil protection concept of the council of Europe and integrated soil research. in *Soil and environment (Vol. 1, pp. 37-47)* (eds. Eijsackers, H. J. P. & Hamers, T.) (Kluwer Academic Publisher, 1993).
7. Daily, G., Matson, P. & Vitousek, P. Ecosystem services supplied by soils. in *Nature's services: Societal dependence on natural ecosystems* (ed. Daily, G.) 113–142 (Island Press, Washington, DC., 1997).
8. Vogel, H.-J. *et al.* A systemic approach for modeling soil functions. *SOIL* **4**, 83–92 (2018).
9. European Commission (EC). EU Soil Strategy for 2030. Reaping the benefits of healthy soils for people, food, nature and climate. COM(2021) 699 final. *Communication from the Commission to the European Parliament, the council, the European economic and social committee and the committee of the regions* 40–51 (2021).
10. Geiges, T., Haraldsen, T. K. & Tobias, S. *CEDR Call 2019: Soils. Comprehensive Literature and Best-practice Review for Avoiding, Mitigating and Compensating for Impacts on Soil*. (2022).
11. Haraldsen, T. K. & Tobias, S. *CEDR Call 2019: Soils. Guidelines for soil handling in infrastructure projects from planning processes to construction work in field. Version 1.0, 14.11.2022*.
12. Greiner, L., Keller, A., Grêt-Regamey, A. & Papritz, A. Soil function assessment: review of methods for quantifying the contributions of soils to ecosystem services. *Land use policy* **69**, 224–237 (2017).
13. Kibblewhite, M. G., Ritz, K. & Swift, M. J. Soil health in agricultural systems. *Philos. Trans. R. Soc. B Biol. Sci.* **363**, 685–701 (2008).
14. Schwilch, G. *et al.* Operationalizing ecosystem services for the mitigation of soil threats: A proposed framework. *Ecol. Indic.* **67**, 586–597 (2016).
15. Bünemann, E. K. *et al.* Soil quality – A critical review. *Soil Biol. Biochem.* **120**, 105–125 (2018).
16. Dominati, E., Patterson, M. & Mackay, A. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecol. Econ.* **69**, 1858–1868 (2010).
17. Jax, K. Function and 'functioning' in ecology: What does it mean? *Oikos* **111**, 641–648 (2005).
18. Wallace, K. J. Classification of ecosystem services: Problems and solutions. *Biol. Conserv.* **139**, 235–246 (2007).
19. Boyd, J. & Banzhaf, S. What are ecosystem services? The need for standardized environmental accounting units. *Ecol. Econ.* **63**, 616–626 (2007).
20. Fisher, B. & Kerry Turner, R. Ecosystem services: Classification for valuation. *Biol. Conserv.* **141**, 1167–1169 (2008).

21. Brussaard, L. Ecosystem Services Provided by the Soil Biota. in *Soil Ecology and Ecosystem Services* 45–58 (Oxford University Press, 2012). doi:10.1093/acprof:oso/9780199575923.003.0005.
22. ten Damme, L. & Keller, T. A conceptual framework for protecting soil functioning in road projects. *Transp. Res. Part D*.
23. Commission of the European Communities. Communication of 16 April 2002 from the Commission to the Council, the European Parliament, the Economic and Social Committee and the Committee of the Regions: Towards a Thematic Strategy for Soil Protection. COM 2002, 179final. (2002).
24. Stolte, J. *et al. Soil threats in Europe: status, methods, drivers and effects on ecosystem services*. *JRC Scientific and Technical Reports* (2015). doi:10.2788/828742.
25. Tóth, G. Soil quality in the EU. in *Threaths to Soil Quality in Europe* (eds. Tóth, G., Montanarella, L. & Rusco, E.) 11–20 (European Commision - Institute for Environment and Sustainability, 2008). doi:10.2788/8647.
26. Karlen, D. L. *et al.* Soil Quality: A Concept, Definition, and Framework for Evaluation (A Guest Editorial). *Soil Sci. Soc. Am. J.* **61**, 4–10 (1997).
27. Faber, J. *et al.* Stocktaking for Agricultural Soil Quality and Ecosystem Services Indicators and their Reference Values. EJP SOIL Internal Project SIREN Deliverable 2. Report, 153 pp. (2022).
28. Drobnik, T., Greiner, L., Keller, A. & Grêt-Regamey, A. Soil quality indicators – From soil functions to ecosystem services. *Ecol. Indic.* **94**, 151–169 (2018).
29. Haygarth, P. M. & Ritz, K. The future of soils and land use in the UK: Soil systems for the provision of land-based ecosystem services. *Land use policy* **26**, 187–197 (2009).
30. Blum, W. E. H. Functions of soil for society and the environment. *Rev. Environ. Sci. Biotechnol.* **4**, 75–79 (2005).
31. Zwetsloot, M. J. *et al.* Soil multifunctionality: Synergies and trade-offs across European climatic zones and land uses. *Eur. J. Soil Sci.* **72**, 1640–1654 (2021).
32. Jost, E. *et al.* Dynamic soil functions assessment employing land use and climate scenarios at regional scale. *J. Environ. Manage.* **287**, (2021).
33. Bouma, J. *et al.* Soil information in support of policy making and awareness raising. *Curr. Opin. Environ. Sustain.* **4**, 552–558 (2012).
34. CEC. *Proposal for a Directive of the European Parliament and of the Council, establishing a framework for the protection of soil and amending Directive 2004/35/EC. Commission of the European Communities.* (2006).
35. FAO. Infographic. https://www.fao.org/assets/infographics/pdfimg/FAO-Infographic-IYS2015-soilfunctions-en.jpg (2015).
36. Coyle, C., Creamer, R. E., Schulte, R. P. O., O'Sullivan, L. & Jordan, P. A Functional Land Management conceptual framework under soil drainage and land use scenarios. *Environ. Sci. Policy* **56**, 39–48 (2016).
37. Bampa, F. *et al.* Harvesting European knowledge on soil functions and land management using multi-criteria decision analysis. *Soil Use Manag.* **35**, 6–20 (2019).
38. Schulte, R. P. O. *et al.* Making the most of our land: Managing soil functions from local to continental scale. *Front. Environ. Sci.* **3**, (2015).
39. Vrebos, D. *et al.* Spatial evaluation and trade-off analysis of soil functions through Bayesian networks. *Eur. J. Soil Sci.* **72**, 1575–1589 (2021).
40. Breure, A. M. *et al.* Ecosystem services: A useful concept for soil policy making! *Curr. Opin. Environ. Sustain.* **4**, 578–585 (2012).
41. Safaei, M., Bashari, H., Mosaddeghi, M. R. & Jafari, R. Assessing the impacts of land use and land cover changes on soil functions using landscape function analysis and soil quality indicators in semi-arid natural ecosystems. *Catena* **177**, 260–271 (2019).
42. Costanza, R. *et al.* The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260 (1997).
43. Baveye, P. C., Baveye, J. & Gowdy, J. Soil 'ecosystem' services and natural capital:

Critical appraisal of research on uncertain ground. *Front. Environ. Sci.* **4**, 1–49 (2016).
44. de Groot, R. S. Environmental functions as a unifying concept for ecology and economics. *Environmentalist* **7**, 105–109 (1987).
45. de Groot, R. S., Alkemade, R., Braat, L., Hein, L. & Willemen, L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol. Complex.* **7**, 260–272 (2010).
46. Robinson, D. A. *et al.* Natural Capital, Ecosystem Services, and Soil Change: Why Soil Science Must Embrace an Ecosystems Approach. *Vadose Zo. J.* **11**, (2012).
47. MEA. *Ecosystems and Human Wellbeing: Synthesis*. https://www.millenniumassessment.org/documents/document.356.aspx.pdf (2005).
48. Schleyer, C., Lux, A., Mehring, M. & Görg, C. Ecosystem services as a boundary concept: Arguments from Social Ecology. *Sustainability* **9**, 1–14 (2017).
49. Mehring, M., Zajonz, U. & Hummel, D. Social-ecological dynamics of ecosystem services: Livelihoods and the functional relation between ecosystem service supply and demand-evidence from socotra archipelago, yemen and the sahel region, West Africa. *Sustainability* **9**, 1–15 (2017).
50. Villamagna, A. M., Angermeier, P. L. & Bennett, E. M. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. *Ecol. Complex.* **15**, 114–121 (2013).
51. Carpenter, S. R. *et al.* Millennium Ecosystem Assessment: Research Needs. *Science (80-.).* **314**, 257–259 (2006).
52. Robinson, D. A. *et al.* Advances in Soil Ecosystem Services: Concepts, Models, and Applications for Earth System Life Support. *Vadose Zo. J.* 13 (2013) doi:10.2136/vzj2013.01.0027.
53. Bouma, J. The new role of soil science in a network society. *Soil Sci.* **166**, 874–879 (2001).
54. Choquet, P. *et al.* Comparison of empirical and process-based modelling to quantify soil-supported ecosystem services on the Saclay plateau (France). *Ecosyst. Serv.* **50**, (2021).
55. Adhikari, K. & Hartemink, A. E. Linking soils to ecosystem services - A global review. *Geoderma* **262**, 101–111 (2016).
56. Huber, S. *et al. Environmental Assessment of Soil for Monitoring: Volume I Indicators & Criteria. EUR 23490 EN/1, Office for the Official Publications of the European Communities, Luxembourg, 339pp.* (2008).
57. European Environment Agency. *Soil Monitoring in Europe*. (Luxembourg: Publications Office of the European Union, 2023, 2023). doi:10.2800/956606.
58. Pistocchi, A., Calzolari, C., Malucelli, F. & Ungaro, F. Soil sealing and flood risks in the plains of Emilia-Romagna, Italy. *J. Hydrol. Reg. Stud.* **4**, 398–409 (2015).
59. Schmidt, M. W. I. *et al.* Persistence of soil organic matter as an ecosystem property. *Nature* **478**, 49–56 (2011).
60. Jobbágy, E. G. & Jackson, R. B. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Appl.* **10**, 423–436 (2000).
61. Tanneberger, F. *et al.* The peatland map of Europe. *Mires Peat* **19**, 1–17 (2017).
62. Vanino, S. *et al.* Barriers and opportunities of soil knowledge to address soil challenges: Stakeholders' perspectives across Europe. *J. Environ. Manage.* **325**, (2023).
63. Sakshi, Singh, S. K. & Haritash, A. K. Polycyclic aromatic hydrocarbons: soil pollution and remediation. *Int. J. Environ. Sci. Technol.* **16**, 6489–6512 (2019).
64. Kole, P. J., Löhr, A. J., Van Belleghem, F. G. A. J. & Ragas, A. M. J. Wear and tear of tyres: A stealthy source of microplastics in the environment. *Int. J. Environ. Res. Public Health* **14**, (2017).
65. Ziajahromi, S., Drapper, D., Hornbuckle, A., Rintoul, L. & Leusch, F. D. L. Microplastic pollution in a stormwater floating treatment wetland: Detection of tyre particles in sediment. *Sci. Total Environ.* **713**, 136356 (2020).

66. Wik, A., Nilsson, E., Källqvist, T., Tobiesen, A. & Dave, G. Toxicity assessment of sequential leachates of tire powder using a battery of toxicity tests and toxicity identification evaluations. *Chemosphere* **77**, 922–927 (2009).
67. Wik, A. & Dave, G. Environmental labeling of car tires-toxicity to Daphnia magna can be used as a screening method. *Chemosphere* **58**, 645–651 (2005).
68. Panko, J. M., Kreider, M. L., McAtee, B. L. & Marwood, C. Chronic toxicity of tire and road wear particles to water- and sediment-dwelling organisms. *Ecotoxicology* **22**, 13–21 (2013).
69. Kreider, M. L., Panko, J. M., McAtee, B. L., Sweet, L. I. & Finley, B. L. Physical and chemical characterization of tire-related particles: Comparison of particles generated using different methodologies. *Sci. Total Environ.* **408**, 652–659 (2010).
70. Napier, F., D'Arcy, B. & Jefferies, C. A review of vehicle related metals and polycyclic aromatic hydrocarbons in the UK environment. *Desalination* **226**, 143–150 (2008).
71. Abdel-Shafy, H. I. & Mansour, M. S. M. A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. *Egypt. J. Pet.* **25**, 107–123 (2016).
72. Mirsal, I. A. *Soil Pollution. Origin, Monitoring and Remediation. 2nd Edition*. (2004).
73. Jayesingha, M., Dietrich, M., Hanslin, H. M. & Haraldsen, T. K. *CEDR Call 2019: Soils. Optimising reuse of soil in road projects - focus on alien invasive species and soil pollution. First draft.* (2022).
74. Ward, M., McDonnell, K., Metzger, K. & Forristal, P. D. The effect of machine traffic zones associated with field headlands on soil structure in a survey of 41 tilled fields in a temperate maritime climate. *Soil Tillage Res.* **210**, 104938 (2021).
75. Matthees, H. L., Hopkins, D. G. & Casey, F. X. M. Soil property distribution following oil well access road removal in North Dakota, USA. *Can. J. Soil Sci.* **98**, 369–380 (2018).
76. EEA - European Environment Agency. CORINE Land cover and change accounts 2000-2018. https://www.eea.europa.eu/data-and-maps/dashboards/land-cover-and-change-statistics (2019).
77. Keller, T., Sandin, M., Colombi, T., Horn, R. & Or, D. Historical increase in agricultural machinery weights enhanced soil stress levels and adversely affected soil functioning. *Soil Tillage Res.* **194**, 12 (2019).
78. Schjønning, P. *et al.* Soil compaction. Chapter 6. in *Soil threats in Europe: status, methods, drivers and effects on ecosystem services* (ed. Stolte, J., Tesfai, M., Øygarden, L., Kværnø, S., Keizer, J., Verheijen, F., Panagos, P., Ballabio, C., Hessel, R.) (Soil Threats in Europe; EUR 27607 EN, 2016). doi:10.2788/828742.
79. Saffih-Hdadi, K. *et al.* A method for predicting soil susceptibility to the compaction of surface layers as a function of water content and bulk density. *Soil Tillage Res.* **105**, 96–103 (2009).
80. Schjønning, P., Lamandé, M., Keller, T., Pedersen, J. & Stettler, M. Rules of thumb for minimizing subsoil compaction. *Soil Use Manag.* **28**, 378–393 (2012).
81. van den Akker, J. J. H. & Soane, B. Compaction. in *Encyclopedia of Soils in the Environment* (ed. Hillel, D.) vol. 4 285–293 (2005).
82. Stettler, M. *et al.* Terranimo ® - a web-based tool for evaluating soil compaction. *Landtechnik* **69**, 132–137 (2014).
83. Soane, B. D., Blackwell, P. S., Dickson, J. W. & Painter, D. J. Compaction by agricultural vehicles: A review II. Compaction under tyres and other running gear. *Soil Tillage Res.* **1**, 373–400 (1981).
84. Norrström, A. C. & Jacks, G. Concentration and fractionation of heavy metals in roadside soils receiving de-icing salts. *Sci. Total Environ.* **218**, 161–174 (1998).
85. Norrström, A. C. & Bergstedt, E. The impact of road de-icing salts (NaCl) on culloid dispersion and base cation pools in roadside soils. *Water. Air. Soil Pollut.* **127**, 281–299 (2000).
86. Asensio, E. *et al.* Accumulation of de-icing salt and leaching in Spanish soils

surrounding roadways. *Int. J. Environ. Res. Public Health* **14**, (2017).
87. Bagstad, K. J., Semmens, D. J., Waage, S. & Winthrop, R. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosyst. Serv.* **5**, 27–39 (2013).
88. Oliveira, E., Leuthard, J. & Tobias, S. Spatial planning instruments for cropland protection in Western European countries. *Land use policy* **87**, 104031 (2019).
89. Wolff, G. Das Bodenschutzkonzept Stuttgart (BOKS). (2006).
90. Fossey, M. *et al.* A Framework to Consider Soil Ecosystem Services in Territorial Planning. *Front. Environ. Sci.* **8**, 1–12 (2020).
91. Geiges, T. & Tobias, S. CEDR Call 2019: Soils. Expert survey and workshop on best practices for soil protection. 1–38 (2022).
92. Renard, K. G., Yoder, D. C., Lightle, D. T. & Dabney, S. M. *Universal soil loss equation and revised universal soil loss equation. In Morgan, R.P.C. and Nearing, M.A., Eds., Handbook of erosion modelling.* (Blackwell Publ., Oxford, UK, 2011).
93. Mueller, L., Schindler, U., Behrendt, A., Eulenstein, F. & Dannowski, M. The Muencheberg Soil Quality Rating (SQR). (2007).
94. Aveline, A., Rousseau, M. L., Guichard, L., Laurent, M. & Bockstaller, C. Evaluating an environmental indicator: Case study of MERLIN, a method for assessing the risk of nitrate leaching. *Agric. Syst.* **100**, 22–30 (2009).
95. Aimon-Marie, F., Angevin, F. & Guichard, L. MERLIN : une méthode agronomique pour apprécier les risques de pollution diffuse par les nitrates d'orgine agricole. *Agrotranfert Lusignan* 27 (2001).
96. Hassink, J. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *1Plant Soil* **191**, 77–87 (997).
97. Gabrielle, B. *et al.* Process-based modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale. *Global Biogeochem. Cycles* **20**, n/a-n/a (2006).
98. Muradian, R. & Gómez-Baggethun, E. Beyond ecosystem services and nature's contributions: Is it time to leave utilitarian environmentalism behind? *Ecol. Econ.* **185**, (2021).
99. Robinson, D. A. *et al.* Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biol. Biochem.* **57**, 1023–1033 (2013).
100. Potschin, M. B. & Haines-Young, R. H. Ecosystem services: Exploring a geographical perspective. *Prog. Phys. Geogr.* **35**, 575–594 (2011).
101. FAO. *Overview of land degradation neutrality (LDN) in Europe and Central Asia*. *Overview of land degradation neutrality (LDN) in Europe and Central Asia* (2022). doi:10.4060/cb7986en.
102. Håkansson, I. A Method for Characterizing the State of Compactness of the Plough Layer. *Soil Tillage Res.* **16**, 105–120 (1990).
103. Naderi-boldaji, M. & Keller, T. Degree of soil compactness is highly correlated with the soil physical quality index S. *Soil Tillage Res.* **159**, 41–46 (2016).
104. Håkansson, I. & Lipiec, J. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. *Soil Tillage Res.* **53**, 71–85 (2000).
105. Ball, B. C., O'Sullivan, M. F. & Hunter, R. Gas diffusion, fluid flow and derived pore continuity indices in relation to vehicle traffic and tillage. *J. Soil Sci.* **39**, 327–339 (1988).
106. Jensen, J. L. *et al.* Relating soil C and organic matter fractions to soil structural stability. *Geoderma* **337**, 834–843 (2019).
107. Ball, B. C. *et al.* Visual soil evaluation: A summary of some applications and potential developments for agriculture. *Soil Tillage Res.* **173**, 114–124 (2017).
108. Bengough, A. G., McKenzie, B. M., Hallett, P. D. & Valentine, T. A. Root elongation, water stress, and mechanical impedance: A review of limiting stresses and beneficial root tip traits. *J. Exp. Bot.* **62**, 59–68 (2011).

Appendix 1. Soil functions

The concept of soil functions was introduced by Blum [1,2] to promote sustainable use of soils. Assessments of the current and potential soil functions are considered important tools for spatial planning and policy making [3].

Soil functions are, in principle, intrinsic to a soil, as they are bundles of soil biological, chemical and physical processes that take place regardless of human interest. Soil functions determine largely the capacity of soils to deliver ecosystem services [4], hence the concept of soil functions focusses often on those functions that bring the stakeholder benefits. Prioritisation of soil functions strongly depends on the individuals awareness and preferences [5,6].

It is generally considered that all soils are able to fulfil all functions, although it is known they do not do so to the same extent [5,7–9]. Soil functions are often partially interdependent because they share some soil processes and characteristics [10]. This results in synergies and trade-offs between functions, which are likely to vary over time and space due to differences in pedoclimatic characteristics and management. The understanding of the different interactions across different scales are crucial for a functioning concept [8]. The concept of soil functions thus emphasizes the multifunctionality of soils, which may be optimised, but cannot be maximised [8,10]. Differences in the extent to which soil fulfils a certain function can be explained by differences in biological, chemical and physical functional characteristics of soils, and by soil extrinsic factors such as climate, topography and management, which vary in time and space [10–12]. For example, Coyle *et al.* showed that improving drainage conditions can be expected to increase primary productivity, but at the expense of carbon sequestration [7].

A1.1 The origin and early development

Blum argued that sustainable use of soil is only possible by a temporal and/or spatial harmonisation of soil functions [1,2]. For this reason, Blum introduced the concept of soil functions and defined interactions between the uses of soil functions and their uses in space and time. Three ecological and three non-ecological soil functions were highlighted: i) the production of biomass, ii) the protection of humans and the environment, iii) containing a gene reservoir, iv) providing a physical basis for human activities, v) providing a source of raw materials, and vi) preserving geogenic and cultural heritage [2]. Soon, a seventh function was added by the European Commission in the proposed Soil Framework Directive [13]: the ability to store carbon. These seven functions were prioritised as they were considered being particularly vulnerable to soil threats [14]. This highlights that the concept of soil functions is not static but driven by pressing socio-environmental challenges, climate change and biodiversity.

A large number of approaches for evaluating and monitoring soil functions have been reviewed [12] and further developed [7–10]. Functions themselves are not well-defined soil properties, but they emerge from a multitude of interactions between biological, chemical and physical processes. Assessments of soil functions are based on measurable soil properties. Consequently, monitoring of soil functions is closely linked to soil quality assessments [8,12,15].

Despite the known multifunctionality, assessments generally include only a limited number of soil functions. This selection can vary between land uses or locations. For example, agricultural soils are mainly considered for their function of primary productivity, but assessments generally also include nutrient cycling, water regulation, climate regulation, and biodiversity [8,16]. These soil functions are an umbrella of different soil sub-functions. For example, the main soil function 'water regulation' includes, among others, the sub-functions 'water storage' and 'water seepage'. Or, the main function 'climate regulation' can include the sub-functions 'carbon storage', 'thermal regulation' or 'greenhouse-gas regulation'. In different land-uses, different (main or sub) functions can be prioritised. For example, Safaei *et al.* assessed the soil functions in terms of soil stability, infiltration, and nutrient cycling for in semi-arid forests and rangelands [17]. In peat lands, greenhouse-gas regulation may be prioritised.

Synergies and trade-offs

Soil functions share certain soil processes, therefore changes made to improve a single function will generally affect other soil functions [8,10]. These interdependencies can turn out positive due to synergies, or negative due to trade-offs. Potential synergies and trade-offs between soil functions need to be anticipated for the concept of soil functions to serve as a tool in spatial planning and policy making [3].

It is expected that changes in land-use or management favouring a single soil function will always affect at least one other soil function negatively. This is an important cause of conflicting management recommendations and policy initiatives, as for example management that maximises primary production inadvertently affects the soil functions water purification and habitat. The direction (positive or negative) of the effect of a change in one function on another function is generally expected to be the same for different environmental zones, although Vrebos *et al.* anticipated a positive correlation between primary productivity and nutrient cycling in most of Europe, but a negative correlation between these functions in the Mediterranean North [10]. The magnitude of trade-offs will differ between environmental zones [8,10]. Difference in the direction and magnitudes of the interdependencies minimises the possibility of generalisation of the impact of changes of land use or management on multiple soil functions. The differences are expectedly explained by differences in soil biological, chemical and physical components, soil management and climatic conditions [8,18].

Soil quality assessments

Soil quality assessments were initially based on visual evaluations, and used to check the suitability of a field for crop growth. Nowadays, they are often based on a combination of visual, analytical and digital approaches and may be used as part of a monitoring program, for educating purposes, or as a basis for management recommendations [12]. Soil quality assessments are based on measurable properties or outputs, used to assess the current situation and/or estimate the situation under different management [12,15,19]. One example that does both is Soil Navigator (www.soilnavigator.eu). Soil Navigator estimates the current level (low, medium or high) of five soil functions simultaneously (i) primary productivity, ii) water purification and regulation, iii) biodiversity and habitat provision, iv) nutrient cycling, and v) climate regulation), and comes with management recommendations if a different level of soil functions is desired [20]. The model requires input data on the soil's biological, chemical and physical components as well as on soil extrinsic factors (management, climate and topography), and has currently been validated for crop- and grassland in five countries (Austria, Denmark, France, Germany and Ireland).

Indicators and values

Soil quality assessments are based on measurable soil properties, optionally supplemented by information on land use (Table A2.1 and A2). Assessments are often simplified by focussing on specific or aggregated indicators, among others because assessing a large variety of soil attributes increases labour and expenses. A large spectrum of indicators can also be problematic in comparison and may be more prone to errors [12,21]. Aggregating indicators into one 'soil quality index' is however not desired, because different soil functions or different land use may thrive with a different set of soil attributes. For example, a pH < 5 may be suitable for a forest soil, but not for grasslands. Moreover, our understanding of the interaction between indicators will continuous to develop. Nevertheless, Drobnik *et al.* demonstrated that even a highly aggregated index can distinguish spatial patterns [15].

Aiming for high effectiveness of soil assessments (minimising labour and costs, maximising the output), minimum datasets are required. Minimum dataset usually contain chemical and physical soil characteristics, as among others shown in the review by Bünemann *et al.*, who studied 65 published minimum datasets used for soil quality assessments [12]. Bone *et al.* encourages the use of cross-functional indicators, i.e., indicators applicable to several soil functions [21]. Generally, soil biological indicators have been underrepresented, despite being considered potentially most informative indicators due to their sensitivity to environmental changes [8,12,22]. Moreover, soil extrinsic factors (e.g. climate, topography, management) are hardly included [10,12] and soil structural characteristics are largely underrepresented too.

Quantification of indicators is only useful if its value can be put in perspective. Roughly, interpretation of the value relates to i) threshold values, ii) reference, or iii) target values. Threshold values relate to minimum or maximum values beyond which processes are impaired. For example, root penetration is largely restricted if the soil mechanical resistance is higher than 2 MPa, and an air-filled pore space of less than 10% might indicate anaerobic conditions. Reference values could either be those of a natural (native) soil, those of a soil with the desired functionality, or in case of temporal use, those of the original soil before temporal land-use-change (e.g. temporal roads or storage). Target values are generally based on expert knowledge, reflecting for example 'near optimal' or 'poor' conditions.

Threshold, reference and target values tend to vary among others with soil type and land-use [10,23]. Values may even vary between crop species or seasons [24]. Therefore, rather than exact values, the values may be given in ranges instead. The smaller the scale at which an assessment is made, the narrower the range can be [10,25]. Bünemann *et al.* advocated the establishment of non-linear scoring functions such as 'more/less is better', 'most common', or 'optimum/undesired range' [12]. They argued that the establishment of exact values will be subject to changes, among others because our understanding of soil functions will continuously develop. Bünemann *et al.* also stressed that the values may need reconsideration since many have been developed mainly by scientists, while the end-users such as farmers or advisors usually played insignificant roles [12].

Table A1.1. Indicator terms from soil quality assessments publications in indicator categories for soil biological, chemical and physical functional characteristics. Source: Bünemann *et al.*, [12], Supplementary 3.

	Indicator category	**Indicator terms included**
Biological	soil respiration	soil respiration
	microbial biomass	microbial biomass, microbial C, microbial N, microbial P, substrate-induced respiration, bacterial biomass, fungal biomass
	N mineralization	N.mineralization (aerobic or anaerobic), mineralizable N
	earthworms	earthworms (species or biomass), incl. potworms (Enchytraeidae), biopores if counted to assess earthworms
	enzyme activities	enzyme activities, phosphatase, urease, dehydrogenase
	root health	root health, soil-borne pest pressure, root system development
	nematodes	nematodes (functional groups, density, diversity)
	microbial diversity	microbial diversity, microbial community composition, Biolog, total species number, fatty acid profiles, bacterial diversity, DNA-based methods
	soil faunal diversity	soil faunal diversity, total species number, microarthropod diversity and density
	metabolic quotient	qCO_2, respiration/microbial C, metabolic quotient
	microbial activity	microbial activity, bacterial activity, thymidine incorporation, odour as sign of microbial activity
	other microbial N cycling processes	potential denitrification, nitrification, denitrification, potential ammonium oxidation
	Cmic/TOC	Microbial C /total organic C
	N fixation/fixing bacteria	N fixation/fixing bacteria

Chemical	total organic matter/carbon	total organic carbon, total organic matter, soil color and odor (if related to organic matter)
	pH	pH
	available P	available P, often as part of nutrient availability
	available K	available K, often as part of nutrient availability or extractable Ca, Mg, K
	total N	total N, Ntot
	cation exchange capacity	cation exchange capacity, exchangeable cations
	electrical conductivity	electrical conductivity, electromagnetic ground conductivity
	available N	Available N, mineral nitrogen, often as part of nutrient availability
	heavy metals	heavy metals (total or available)
	other macronutrients	total and available Mg, S, Ca
	micronutrients	total and extractable Al, Fe, Mn and other micronutrients
	labile C and N	labile C and N, active C, particulate organic matter, oxidizable C, $KMnO_4$-extractable C, light fraction C and N, water-extractable C, water-extractable organic N
	sodicity, salinity	sodicity, exchangeable Na (ESP), Na adsorption ratio, salinity
	base saturation	base saturation, exchangeable acidity
	carbonate content	carbonate content
	total P	total P
	total K	total K (part of a whole list of total elements)
	organic pollutants	organic pollutants, PAH, Polychlorinated Biphenyls (PCB), xenobiotics loadings, insecticides
	C/N	C/N
	^{137}Cesium distribution	^{137}Cesium distribution (as an indicator of erosion)
Physical	water storage	water-holding capacity, water content, sorptivity, water-filled pore space, water retention, field capacity, permanent wilting point, plant-available water content, Ksat
	bulk density	bulk density
	texture	particle-size distribution, soil texture (sand, silt, clay)
	structural stability	aggregate stability, shear strength, tilth and friability, structure, consistence, slake test
	soil depth	soil depth, topsoil depth, maximum rooting depth, layer thickness
	penetration resistance	penetration resistance, previous consolidation
	hydraulic conductivity	hydraulic conductivity
	porosity	porosity, macroporosity, air capacity
	aggregation	aggregation, aggregate size distribution, pedality
	infiltration	infiltration rate
	stone content	stone content
	soil temperature	soil temperature
	particle density	particle density
	surface characteristics	surface characteristics, surface conditions, surface residues
	clay characteristics	clay characteristics, mineralogy, water-dispersible clay, soil color (if related to clay characteristics)
	biopores	biopores

Table A1.2 Soil function assessment methods developed for and applied in the European temperate climate zone. Source: Jost *et al.* Supplementary 3 [26]

SOC = soil organic carbon, RF = regression functions, PTF = pedotransfer functions, SQT = semi-quantitative lookup tables, n.s.= not specified

Indicators	***Scale of application***	***Reference soil depth (cm)***	***Methods for indicator derivation***	***Input parameters***													
				SOC content	*pH-value*	*Bulk density*	*Drainage*	*Carbonate content*	*Hydromorphy*	*Humus form*	*Electrical conductivity*	*Texture (sand, silt, clay)*	*Stone content*	*Parent material*	*Soil horizon thickness /soil depth*	*Soil type*	*Others*
Soil Nutrient Storage Function																	
Effective cation-exchange-capacity	Regional	0-100	PTF, SQT	x	x	x						x	x		x		
Regulation of nutrient loss	Plot to landscape	0-100	PTF, SQT	x		x	x					x	x	x	x		Topography
Soil Water Regulation Function																	
Water storage capacity, saturated soil hydraulic conductivity	Regional	0-100	PTF, SQT	x		x						x	x		x		
Water storage potential, water regulation potential	Regional	0-100	PTF, indexing	x		x						x	x		x		

Table A1.2 cont.

Indicators	Scale of application	Reference soil depth (cm)	Methods for indicator derivation	Input parameters													
				SOC content	pH-value	Bulk density	Drainage	Carbonate content	Hydromorphy	Humus form	Electrical conductivity	Texture (sand, silt, clay)	Stone content	Parent material	Soil horizon thickness /soil depth	Soil type	Others
Soil Productivity Function																	
Crop yield potential	Plot to landscape	0-100	SQT	x		x		x	x			x	x		x		Topography, risk potentials
Soil usage capacity	Plot to landscape	0-100	SQT	x	x	x		x	x		x	x	x		x		Topography, oxygen, risk potentials
Agricultural suitability	Plot to landscape	0-100	SQT	x	x	x	x					x	x		x		Length of growing period
Soil suitability for sustainable intensification (Soil resilience and performance)	Local to national	n.s.	PTF, SQT	x	x	x						x			x		Topography
Soil as site for biomass and wheat production	Local to national	0-100	PTF, SQT	x	x	x						x			x		Topography
Soil Habitat Function																	
Indicator species: earthworms (abundance, number of taxa, Shannon-Index)	Continental	0 – 30	RF, indexing	x	x							x					Topography, latitude/longitude

Table A1.2 cont.

Indicators	***Scale of application***	***Reference soil depth (cm)***	***Methods for indicator derivation***	***Input parameters***													
				SOC content	*pH-value*	*Bulk density*	*Drainage*	*Carbonate content*	*Hydromorphy*	*Humus form*	*Electrical conductivity*	*Texture (sand, silt, clay)*	*Stone content*	*Parent material*	*Soil horizon thickness /soil depth*	*Soil type*	*Others*
Soil biodiversity potential	Continental	n.s.	SQT	x	x							x					Potential evapotranspiration, biomass productivity
Potential habitat for soil organisms indicator (based on Biological Soil Quality (Parisi et al.*, 2005))*	Regional	0-30	SQT, indexing	x		x											
Hemeroby index (magnitude of anthropogenic impact)	Local to national	n.s.	SQT, indexing				x										
Soil capacity to provide niches for rare plant species	Regional	0-100	PTF, SQT	x		x						x			x	x	
Soil Carbon Sequestration Function																	
Soil carbon saturation deficit	Regional to national	0-10	PTF, RF	x		x						x	x		x		

Table A1.2 cont.

Indicators	***Scale of application***	***Reference soil depth (cm)***	***Methods for indicator derivation***	***Input parameters***													
				SOC content	*pH-value*	*Bulk density*	*Drainage*	*Carbonate content*	*Hydromorphy*	*Humus form*	*Electrical conductivity*	*Texture (sand, silt, clay)*	*Stone content*	*Parent material*	*Soil horizon thickness /soil depth*	*Soil type*	*Others*
Carbon potential sequestration, carbon potential loss (SOC-specific range boundaries)	Regional	0-30	PTF	x		x	x					x	x				Topography, Flooding occurrence
Relative temporal SOC-stock change	> regional	0-30	PTF, RF	x								x					Time after land use change, management intensity

Appendix 2. Ecosystem services

The concept of ecosystem services has, since decades, revolved around the wish to quantify the value of environmental services. This value has been expected to add weight on behalf of natural systems in decision-making, and thereby protect these systems [27–31]. While an ecosystem service in itself are to be understood as the quantifiable functional outputs of complex natural processes [32], i.e., of soil functions, the concept of ecosystem services addresses the interdependencies between nature and human wellbeing [33,34]. In fact, ecosystem services cannot be discussed without considering both nature and society. On the one hand, people influence the extent to which services can be produced through utilisation of services and management of land. On the other hand, the value of services varies depending on, among others, spatial and temporal variation of supply and demand. Accounting for ecosystem services requires i) identification or classification of services, ii) quantification of the services, and iii) valuation of services.

Most services are delivered locally, but the supply is influenced by processes at a much larger scale. Decisions on land use, including management, may thereby influence the services generated. Not all changes of soil intrinsic or extrinsic factors will significantly affect the generation of an ecosystem service, but when they do, it generally affects multiple services provided by that system [30]. As soil functions, ecosystem services are highly interdependent: both synergies (an ecosystem functions or services strengthens another) and trade-offs (an ecosystem function or service suppresses another) exist [35,36]. Carpenter *et al.* observed that many studies tended to assess a single or a few ecosystem services [36]. The authors attributed this to the complexity of the concept of ecosystem services. However, doing so explicitly excludes synergies and trade-offs with other services. As a consequence of the multifunctionality, optimisation of the generation of a single ecosystem service will often cascade and affect the generation of other ecosystem services. This effect may not be restricted to the study area but possibly extend beyond the areas boundaries, meaning that local changes are felt at greater distance [36].

The origin and early development

According to Baveye *et al.* [28] and De Groot *et al.* [30], the origin of the concept of ecosystem services dates back to the 1960's, when researchers sought ways to quantify the value of 'environmental services', being confident that this would raise awareness in society about the importance of protecting nature [28] and [30]. However, they did not succeed, and the lack of success resulted in a decline of the popularity of the concept in science and politics. For years, exploring ways to account for environmental services has been back on the agenda, as it still is considered to having the greatest potential in protecting the natural environment [31,37]. This has developed into the 'concept of ecosystem services'. To date, however, researchers still struggle to complete the objective of quantifying the value of services [28,38].

Likely the first international instrument promoting the protection of natural systems was the World Conservation Strategy (WCS). The WCS was launched in 1980 by the International Union for Conservation of Nature and Natural Resources, founded in 1948, to "*stimulate a more focused approach to living resource conservation and to provide policy guidance on how this can be carried out*" [39]. Three objectives made up the core of the WCS: *i) maintaining essential ecological processes and life-support systems; ii) preserving genetic diversity, and; iii) ensuring the sustainable utilisation of species and ecosystems*. The WCS was drafted with the involvement of over 450 governmental agencies and conservation organisations in over 100 countries. However, the WCS did not have sufficient impact, as was widely agreed upon during the Conference on Conservation and Development, *Implementing the World Conservation Strategy* in 1986. The participants reckoned the WCS lacked appeal to economists and policymakers as well as to society in general [29].

The lack of appeal and consideration of environmental services in decision making were thought to be twofold: i) a different interpretations of key-terms [29] and ii) a lack of economic valuation [27]. After the WCS, De Groot introduced a conceptual framework adopting a broader [than the WCS] concept of the goods and services provided by the natural environment [29]. In this framework, De Groot presented the functional interrelation between 'natural processes and components' and 'human needs and activities'; highlighting that natural processes and components provide the goods and services that fulfil human needs, while, *vice versa*, human needs affect natural processes and components. De Groot used the term 'ecosystem values' when referring to 'the goods and services provided by the natural environment'; the term 'ecosystem services' may have been introduced by [27]. The concept of ecosystem services was greatly popularised by the Millennium Ecosystem Assessment (MEA) [40].

The MEA was designed to aid decision-makers in evaluating the consequences of ecosystem changes on human well-being [36]. The framework created within the Millennium Ecosystem Assessment [40] includes, like De Groots (1987) framework [29], linkages between the natural environment and society, albeit named 'ecosystem services' and 'constituents of well-being'.

Since the MEA, a great number of ecosystem services frameworks have been developed and reviews have been performed, refining the concept of ecosystem services. Many of these frameworks can, in their essence, be reduced to De Groots (1987) framework [29], since most contemplate the interlinkages between the natural environment and society too. Newer frameworks are not always intended to be an improvement of, and thereby a replacement of, an earlier framework. Instead, continuous development and adaptation of conceptual models, in general, is much desired [41]. It is not to expect that one framework will adequately fit all purposes of assessments or all different context in which ecosystem services may be assessed. Therefore, the ecosystem services concept has to be seen as an evolving concept, and earlier concepts need to be revalidated.

'Soil ecosystem services' is a distinction made after it was observed that frameworks and regulations did not generally promote the understanding of the critical role that soils have in the provision of ecosystem services [42–45]. Namely, frameworks addressing ecosystem services have often reduced soil processes to intermediate services, despite the knowledge that the capability of ecosystem services highly varies between soils [6,46–48].

Classification of ecosystem services

Accounting for ecosystem services requires first of all identification and classification of the services. Classification systems in particular are considered needed, among others, for comparison across time, locations and management, and for prevention of double counting of intermediate and end-services [11]. Classification of ecosystem services appears particularly challenging as 'processes or functions' and 'services' are mixed up in many classification systems and frameworks [11,49,50]. Using the two phrases interchangeably is particularly problematic since functions and generated services are not necessarily one-to-one comparable; it is not generally one specific function that produces a certain ecosystem service, but a single ecosystem service is often the result of multiple (sub)functions. For example, crop production also requires a certain water regulation. Moreover, one function may contribute to different services, the way that nutrient cycling results in food, biodiversity and save portable water. Costanza *et al.* called for a clear distinction between ecosystem processes or functions and ecosystem services [27].

Part of the complexity of classifying services results from debates about the concept's key-terms. Early definitions of ecosystem services included 'benefit', such as in the MEA [40], which defined ecosystem services as "*the benefits people can obtain from ecosystems*". The use of the term benefit has, however, been heavily questioned. Ecosystem services are functional outputs generated by ecosystem components and processes. These functional outputs may or may not be used, and therefore they may or may not benefit society [11,49,51].

There is a wide range of different ecosystem service categorizations described in the literature [40,41,50]. The MEA (2005) [40] differentiated between supporting, provisioning, regulating and cultural ecosystem services, and this classification system has been widely used (see [41]). Within these classifications, soils are considered 'supporting', as in supporting aboveground ecosystems in generating ecosystem services, but not considered for the services soils provide themselves [8,46,52]. A Common International Classification of Ecosystem Services (CICES, available at https://cices.eu/) has been developed based on environmental accounting by the European Environment Agency (EEA). It adopted the classification as used within the Millennium Ecosystem Assessment [40], but is revised based on experience of the user community [53]. CICES classifies ecosystem services into cultural, provisioning, and regulation and maintenance, but differentiates between biotic and abiotic outputs [54]. Among others Drobnik *et al.* [15] have linked to CICES in developing their soil quality index.

Quantification of ecosystem services

As for soil functions, spatial-temporal variation in the generation of ecosystem services is great. This is, among others, highlighted by Choquet *et al.* who observed a great variability between the organic carbon saturation on forested Planosols and Gleysols on the Saclay plateau (France) [45]. Except for spatial-temporal differences, ecosystem services have another aspect of quantification that is crucial to consider: differences in the level of provision.

The notion that ecosystem services may or may not benefit society highlights that different levels of provision can be distinguished. In acknowledgement of this, Villamagna *et al.* drafted a framework based on literature review to help advance a common language [35]. Their framework existed out of four distinct ecosystem components: capacity, pressures of anthropogenic and natural stressors, demand, and flow. Three of these – capacity, demand, and flow – relate to different levels of provision of the services: capacity to potentially generated services based on biophysical and social properties (i.e., current conditions and use), demand for the services desired by society, and flow to services used by society. Faber *et al.* suggested renaming capacity with 'potential supply' [55]. Potential supply can exceed the capability due to 'human derived capital' in forms of labour, technology and capital [56]. This human capital is, however, often ignored in ecosystem service research [33].

The demand may exceed the potential supply, but the potential supply can also exceed the demand. For example, a farms potential supply of tomatoes may exceed the demand, or a watersheds provision of clean water fails to deliver the amount of water in certain quality. The balance between the different levels of supply (potential, capacity, capability and potential maximum) and flow of ecosystem services is often offset, because supply and demand are often mismatched spatially or over time [35,36,51]. The difference between potential supply and flow may be lost if not used or not stocked.

While renaming 'capacity with 'potential supply', Faber *et al.* described two additional levels of supply: capacity and capability [55]. Both refer to the ability to sustainably generate ecosystem services based on context properties like soil type and land use, but irrespective of its existing condition and management. In other words, a change of management may increase the output. The difference between capacity and capability is explained by the effect on *other* ecosystem services: 'capacity' is used to refer to the level of provision that does not affect the provision of other services, while 'capability' is used to refer to the level at which other services are affected. The land use in Faber *et al.* is limited to agricultural land [55]. If considering different land uses, the level of provision may be increased by a change in land use. We call this the potential maximum supply. Some levels of provision are susceptible to changes of natural and anthropogenic pressures. For example, changes in lifestyle can change the level of demand and flow, and changes of management or biophysical conditions alter the potential supply.

The various levels of provision are mostly still hypothetical, as direct measurements of (soil) ecosystem services are largely lacking [45].

Valuation of ecosystem services

Historically, (goods and) services have a clear economic implication [29]. Nevertheless, ecosystem services do not necessarily need to be expressed in economic terms, as highlighted by Costanza *et al.* [27]: "*We can choose to make these valuations explicit or not … ; as long as we are forced to make choices, we are going through the process of valuation*". Similarly to valuing of soil functions [5,6], valuing ecosystem services strongly depends on a person's awareness and preferences.

Ecosystem services must, like all services, be used by someone to have (economic) value [51]. Consequently, valuation of current ecosystem services is suggested to be based on the 'flow' of services [27]. However, among others Robinson *et al.* argued that a focus on valuing the final services will be counterproductive in the long run, as a focus on the output would not do justice to the system that delivers them [52]. In other words, valuing the final ecosystem services would not necessarily protect a sustainable use of soils. Similarly, Potschin and Haines-Young argued that the importance and costs of maintaining the characteristics and structures of the systems that underpin the generation of ecosystem services cannot be, but currently too often are, overlooked [57].

The value of services is highly influenced by the levels of potential supply and demand. McDonald highlighted a few studied examples [51], such as that open space used for the day-to-day recreation is demanded within several kilometres of residency, whereas potable water may be drawn from a larger region, but usually within a few hundred kilometres. Beyond these 'borders', a soil may potentially or actually deliver a large amount of ecosystem services (e.g. offer open space or potable water), but without a demand for them, they will not flow and therefore will not, in terms of conservative accounting, be valuable.

Actual valuation of ecosystem services and using this in decision making is an underexplored area, mostly because economic studies to support values are missing [58]. In 1997, Costanza *et al.* estimated the total value of ecosystem services on a global scale at twice the worlds GNP [27]. As discussed by Breure *et al.*, other studies showed that this most likely was an underestimation [6].

References in Appendices

1. Blum, W. E. H. Soil protection concept of the council of Europe and integrated soil research. in *Soil and environment (Vol. 1, pp. 37-47)* (eds. Eijsackers, H. J. P. & Hamers, T.) (Kluwer Academic Publisher, 1993).
2. Blum, W. E. H. Functions of soil for society and the environment. *Rev. Environ. Sci. Biotechnol.* **4**, 75–79 (2005).
3. Creamer, R. E., Hagens, M., Baartman, J. & Helming, K. Editorial for special issue on "understanding soil functions – from ped to planet". *Eur. J. Soil Sci.* **72**, 1493–1496 (2021).
4. Greiner, L., Keller, A., Grêt-Regamey, A. & Papritz, A. Soil function assessment: review of methods for quantifying the contributions of soils to ecosystem services. *Land use policy* **69**, 224–237 (2017).
5. Bampa, F. *et al.* Harvesting European knowledge on soil functions and land management using multi-criteria decision analysis. *Soil Use Manag.* **35**, 6–20 (2019).
6. Breure, A. M. *et al.* Ecosystem services: A useful concept for soil policy making! *Curr. Opin. Environ. Sustain.* **4**, 578–585 (2012).
7. Coyle, C., Creamer, R. E., Schulte, R. P. O., O'Sullivan, L. & Jordan, P. A Functional Land Management conceptual framework under soil drainage and land use scenarios. *Environ. Sci. Policy* **56**, 39–48 (2016).
8. Zwetsloot, M. J. *et al.* Soil multifunctionality: Synergies and trade-offs across European climatic zones and land uses. *Eur. J. Soil Sci.* **72**, 1640–1654 (2021).
9. Schulte, R. P. O. *et al.* Making the most of our land: Managing soil functions from local to continental scale. *Front. Environ. Sci.* **3**, (2015).

10. Vrebos, D. *et al.* Spatial evaluation and trade-off analysis of soil functions through Bayesian networks. *Eur. J. Soil Sci.* **72**, 1575–1589 (2021).
11. Boyd, J. & Banzhaf, S. What are ecosystem services? The need for standardized environmental accounting units. *Ecol. Econ.* **63**, 616–626 (2007).
12. Bünemann, E. K. *et al.* Soil quality – A critical review. *Soil Biol. Biochem.* **120**, 105–125 (2018).
13. CEC. *Proposal for a Directive of the European Parliament and of the Council, establishing a framework for the protection of soil and amending Directive 2004/35/EC. Commission of the European Communities.* (2006).
14. FAO & ITPS. *Status of the World's Soil Resources (SWSR) - Main Report. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils. Intergovernmental Technical Panel on Soils* (2015).
15. Drobnik, T., Greiner, L., Keller, A. & Grêt-Regamey, A. Soil quality indicators – From soil functions to ecosystem services. *Ecol. Indic.* **94**, 151–169 (2018).
16. Schulte, R. P. O. *et al.* Functional land management: A framework for managing soil-based ecosystem services for the sustainable intensification of agriculture. *Environ. Sci. Policy* **38**, 45–58 (2014).
17. Safaei, M., Bashari, H., Mosaddeghi, M. R. & Jafari, R. Assessing the impacts of land use and land cover changes on soil functions using landscape function analysis and soil quality indicators in semi-arid natural ecosystems. *Catena* **177**, 260–271 (2019).
18. Dade, M. C., Mitchell, M. G. E., McAlpine, C. A. & Rhodes, J. R. Assessing ecosystem service trade-offs and synergies: The need for a more mechanistic approach. *Ambio* **48**, 1116–1128 (2019).
19. Vogel, H.-J. *et al.* Quantitative Evaluation of Soil Functions: Potential and State. *Front. Environ. Sci.* **7**, 15 (2019).
20. Debeljak, M. *et al.* A Field-Scale Decision Support System for Assessment and Management of Soil Functions. *Front. Environ. Sci.* **7**, 1–14 (2019).
21. Bone, J. *et al.* Soil quality assessment under emerging regulatory requirements. *Environ. Int.* **36**, 609–622 (2010).
22. Vogel, H.-J. *et al.* A systemic approach for modeling soil functions. *SOIL* **4**, 83–92 (2018).
23. Lilburne, L. R., Hewitt, A. E., Sparling, G. P. & Selvarajah, N. Soil Quality in New Zealand. *J. Environ. Qual.* **31**, 1768–1773 (2002).
24. Arvidsson, J. & Håkansson, I. Response of different crops to soil compaction — Short-term effects in Swedish field experiments. *Soil Tillage Res.* **138**, 56–63 (2014).
25. Bouma, J. *et al.* Soil information in support of policy making and awareness raising. *Curr. Opin. Environ. Sustain.* **4**, 552–558 (2012).
26. Jost, E. *et al.* Dynamic soil functions assessment employing land use and climate scenarios at regional scale. *J. Environ. Manage.* **287**, (2021).
27. Costanza, R. *et al.* The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260 (1997).
28. Baveye, P. C., Baveye, J. & Gowdy, J. Soil 'ecosystem' services and natural capital: Critical appraisal of research on uncertain ground. *Front. Environ. Sci.* **4**, 1–49 (2016).
29. de Groot, R. S. Environmental functions as a unifying concept for ecology and economics. *Environmentalist* **7**, 105–109 (1987).
30. de Groot, R. S., Alkemade, R., Braat, L., Hein, L. & Willemen, L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol. Complex.* **7**, 260–272 (2010).
31. Robinson, D. A. *et al.* Natural Capital, Ecosystem Services, and Soil Change: Why Soil Science Must Embrace an Ecosystems Approach. *Vadose Zo. J.* **11**, (2012).
32. Kibblewhite, M. G., Ritz, K. & Swift, M. J. Soil health in agricultural systems. *Philos. Trans. R. Soc. B Biol. Sci.* **363**, 685–701 (2008).
33. Schleyer, C., Lux, A., Mehring, M. & Görg, C. Ecosystem services as a boundary concept: Arguments from Social Ecology. *Sustainability* **9**, 1–14 (2017).
34. Mehring, M., Zajonz, U. & Hummel, D. Social-ecological dynamics of ecosystem

services: Livelihoods and the functional relation between ecosystem service supply and demand-evidence from socotra archipelago, yemen and the sahel region, West Africa. *Sustainability* **9**, 1–15 (2017).
35. Villamagna, A. M., Angermeier, P. L. & Bennett, E. M. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. *Ecol. Complex.* **15**, 114–121 (2013).
36. Carpenter, S. R. *et al.* Millennium Ecosystem Assessment: Research Needs. *Science (80-.).* **314**, 257–259 (2006).
37. Hauck, J., Görg, C., Varjopuro, R., Ratamäki, O. & Jax, K. Benefits and limitations of the ecosystem services concept in environmental policy and decision making: Some stakeholder perspectives. *Environ. Sci. Policy* **25**, 13–21 (2013).
38. Ekins, P. Environmental sustainability: From environmental valuation to the sustainability gap. *Prog. Phys. Geogr.* **35**, 629–651 (2011).
39. IUCN. *World conservation strategy*. (1980).
40. MEA. *Ecosystems and Human Wellbeing: Synthesis*. https://www.millenniumassessment.org/documents/document.356.aspx.pdf (2005).
41. Fisher, B., Turner, R. K. & Morling, P. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* **68**, 643–653 (2009).
42. Robinson, D. A. *et al.* Advances in Soil Ecosystem Services: Concepts, Models, and Applications for Earth System Life Support. *Vadose Zo. J.* 13 (2013) doi:10.2136/vzj2013.01.0027.
43. Daily, G., Matson, P. & Vitousek, P. Ecosystem services supplied by soils. in *Nature's services: Societal dependence on natural ecosystems* (ed. Daily, G.) 113–142 (Island Press, Washington, DC., 1997).
44. Bouma, J. The new role of soil science in a network society. *Soil Sci.* **166**, 874–879 (2001).
45. Choquet, P. *et al.* Comparison of empirical and process-based modelling to quantify soil-supported ecosystem services on the Saclay plateau (France). *Ecosyst. Serv.* **50**, (2021).
46. Dominati, E., Patterson, M. & Mackay, A. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecol. Econ.* **69**, 1858–1868 (2010).
47. Koch, A. *et al.* Soil Security: Solving the Global Soil Crisis. *Glob. Policy* **4**, 434–441 (2013).
48. Adhikari, K. & Hartemink, A. E. Linking soils to ecosystem services - A global review. *Geoderma* **262**, 101–111 (2016).
49. Fisher, B. & Kerry Turner, R. Ecosystem services: Classification for valuation. *Biol. Conserv.* **141**, 1167–1169 (2008).
50. Wallace, K. J. Classification of ecosystem services: Problems and solutions. *Biol. Conserv.* **139**, 235–246 (2007).
51. McDonald, R. Ecosystem service demand and supply along the urban-to-rural gradient. *J. Conserv. Plan.* **5**, (2009).
52. Robinson, D. A. *et al.* Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biol. Biochem.* **57**, 1023–1033 (2013).
53. Haines-Young, R. & Potschin, M. CICES. https://cices.eu/ (2022).
54. Haines-Young, R. & Potschin, M. B. Common international classification of ecosystem services (CICES) V5.1 and Guidance on the application of the revised structure. Available from www.cices.eu. (2018).
55. Faber, J. *et al.* Stocktaking for Agricultural Soil Quality and Ecosystem Services Indicators and their Reference Values. EJP SOIL Internal Project SIREN Deliverable 2. Report, 153 pp. (2022).
56. Jones, L. *et al.* Stocks and flows of natural and human-derived capital in ecosystem services. *Land use policy* **52**, 151–162 (2016).
57. Potschin, M. B. & Haines-Young, R. H. Ecosystem services: Exploring a geographical perspective. *Prog. Phys. Geogr.* **35**, 575–594 (2011).

58. Bagstad, K. J., Semmens, D. J., Waage, S. & Winthrop, R. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosyst. Serv.* **5**, 27–39 (2013).