Supporting Information 1

SHERPA = Soil Health Evaluation, Rating Protocol and Assessment

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General aim of SHERPA:

The aim of SHERPA is to develop a soil health determination key at European scale. Here, the general framework is described, where we seek to be adaptable for refinement at local and regional scales. Considering this, the key is suitable for digital mapping, to be programmed in the open source R Software platform. We plan to keep the structure open (e.g., in a sense of non-final, like the philosophy of R itself) for later additions, corrections, extensions in areas or knowledge of area specifics, improvements, and sharing with other collaborators.

The Intergovernmental Technical Panel on Soils (ITPS) defines soil health as "*the ability of the soil to sustain the productivity, diversity, and environmental services of terrestrial ecosystems*" (FAO, 2024). SHERPA is based on a tangible definition of soil health in a pragmatic way for future mapping. Our working definition of soil health for SHERPA is that a soil is healthy if its natural functions in relation to its land use type are not subject to degradation in any significant way. This definition enables an appropriate consideration of the diversity of soils. As we cannot always differentiate if a soil is degraded by natural hazards or human impact, the cause of disturbance might sometimes be natural hazards. For instance, a landslide in alpine grasslands might be triggered by an avalanche. Whether or not this avalanche is originally triggered by natural hazards, or human-induced due to prior vegetation damage or by climate change will not be assessed with this key.

We follow the latest version of the World Reference Base (WRB) for the definition of all parameters in Soil Resources, 4th edition, 2022 (WRB, 2022) with the following exceptions: Regarding soil structure, every aggregate formation with size > 50 mm was attributed to cloddy assuming that this size of aggregates would always indicate low soil health in Europe regardless of exact aggregate form. However, as ploughing will always result in larger aggregate sizes, the latter should be assessed at least eight weeks after ploughing to allow for potential regeneration.

Structure of SHERPA

The aim is to keep the SHERPA key as parsimonious as possible, suitable for large scale monitoring. Increasing the number of indicators will not only increase collinearity as well as the complexity of the relationships between indicators and management options, but will also result in increased costs of monitoring (Bünemann et al., 2018). We propose a simple key, with the chosen parameters being reduced to the feasible minimum programmed in the open source R Software platform. We plan to keep the structure open (e.g., in a sense of non-final, like the philosophy of R itself) for later additions, corrections, extensions in areas or knowledge of area specifics, improvements, and sharing with other collaborators. The logic of the key follows in **Part 1** a "*ruling out*" principle: the assessment of potential degradation factors will lead to negative soil health scores, to be subtracted from the general soil health index to be assessed in **Part 2 (Figure 1,** main manuscript**)**. The second Part 2 of the SHERPA follows basic soil science criteria to define soil health.

Part 1 soil health assessment

For Part1, we consider the following main soil degradation processes dependent on relevance and data availability for the three land-use types (Table S1): soil erosion (as the sum of wind, water, harvest and tillage for arable soils (Borrelli et al., 2023) and water erosion only for grasslands and forests), land sliding, heavy metal contamination and nitrogen surplus. Phosphorus surplus, pesticide input, salinization, compaction and soil organic carbon loss were considered in arable soils and grasslands, while phosphorus mining was only considered in arable soils (for all scientific background as well as justification of score assignment please see below). Each of the degradation factors will be assigned a negative score from 0 (no degradation) to -9 (worst degradation influence). For data sources and spatial resolution see Supporting Information II, Table S2).

Part 2 soil health assessment

Regarding the intrinsic health of a soil, it has been argued that it will be challenging to find natural soils that can act as a reference, especially for healthy agricultural soils and that the challenge for soil laws will be how to develop gold standards for healthy soils (van der Putten et al., 2023). We argue that soil scientists know the basic soil properties that indicate healthy soils under specific land use conditions and environmental settings, and we followed these criteria to define intrinsic soil health in Part2.

The intrinsic health of a soil is assessed with a positive scoring between 1 (low soil health) to 10 (healthy soil). SHERPA will classify forest soils according to their Köppen-Geiger climate class (Beck et al., 2018; Peel et al., 2007), geology considering mineralogical nutrient and buffer capacity, altitude, humus layer structure and signs and spatial extent of humus layer disturbance (see below). Grasslands are separated into permanent and non-permanent grasslands, where the latter will be assessed together with arable soils, orchards and vineyards. Permanent grasslands are classified by their fractional vegetation cover, assuming that a closed vegetation cover will indicate soil health in grasslands considering the soil degradation factors assessed and subtracted in Part1. Arable soils, orchards, vineyards and non-permanent grasslands are classified according to their extent of vegetation cover throughout the year (e.g., cover crops, mulching, plant residue cover, intermittent crops), soil structure and the type of fertilizer used (as the use of organic over mineral fertilizer use has been shown to significantly enhance soil health, see section 2.3 below for discussion). For wetlands, drainage and vegetation cover would be the main indicators (please note that even though we are currently developing a framework for wetland assessment and mapping, assessing wetlands were beyond the scope of this current manuscript).

An example calculation of the SHERPA would be a beech forest with humus form mull and granular soil structure, but high atmospheric nitrogen deposition. This might be assigned a soil health index of 8-10 in Part 2; however, the humus form might have been influenced by surplus nitrogen deposition, thus simulating a better humus form than what would naturally be there. Therefore, negative scores will be assigned to be subtracted from the basic soil health scores assigned in Part 2 to reflect the human disturbance of this system due to nitrogen deposition. Please note that following the concept of Part 1, a soil health index of 1 is not the lowest that can be assigned in SHERPA. If multiple degradation processes are co-existing (e.g., soil erosion rate of 12 t ha^{-1} yr^{-1} (subtraction of 8 scores) and a copper concentration of 80 mg kg^{-1} (subtraction of 6 scores), the assigned soil health index will result in a negative value.

As we cannot always differentiate if a soil is degraded by natural hazards or human impact, the cause of disturbance might sometimes be natural hazards. For instance, a landslide in alpine grasslands might be triggered by an avalanche. Whether or not this avalanche is originally triggered by natural hazards, or human induced due to prior vegetation damage or by climate change will not be assessed with this key.

Justification of the soil health definition used and separating soil health from soil quality

Many reviews do not differentiate clearly between soil quality and soil health or even consider the terms equivalent (Bünemann et al., 2018; Doran and Parkin, 1994). We suggest to follow the concept that different from soil quality, which is largely chemical in focus and mostly used to characterize the status of soil to sustain crop productivity, soil health is a more holistic concept (Lehmann et al., 2020). It is based on the recognition of the ecosystem services that soils provide. Soil quality refers to the capacity of soil to function for a specific use and generally assesses soil condition in regard to this use using indicators. Soil health is a broader more dynamic concept that embodies the soil as a living system and hence it has a heavy emphasis of the soil as a living entity with the vitality of that ecosystem assessed through emergent characteristics. With that said, we would like to point out that an in-depth discussion of soil quality, soil health and related definitions is found in (Bünemann et al., 2018) and is beyond the scope of what we aim at here. However, not clearly separating the concept of soil quality from soil health resulted in heavy criticism of the combined concept, claiming it would transform soil science into a value system, with being biased towards certain soil types and limited number of annual crops that provide cheap food (Letey et al., 2003; Sojka and Upchurch, 1999; Sojka et al., 2003 discussed in (Bünemann et al., 2018)). Other soil scientists recommended a broad program of research for soil health to identify dynamic emergent properties, with resilience being especially considered (Harris et al., 2022). (Lehmann et al., 2020) suggests that soil health could become an established scientific field to which many disciplines can contribute, for example, by listing their specific discipline’s research also under the keyword ‘soil health’. Some of such recent assessments of soil health concepts and approaches conclude that continuous research, education, and outreach efforts are warranted to promote localized development, adoption, and implementation of soil health assessment and management (Guo, 2021; Lehmann et al., 2020). (Lehmann et al., 2020) even recommend that soil scientists should embrace soil health as an overarching principle to which to contribute knowledge that contributes to sustainability goals rather than only a property to measure. However, the latter would render it impossible to use soil health as an indicator for land management assessment and policy making. We suggest to endeavor emergent soil profile pedological characteristics to assess the expected vitality in SHERPA and combine this with characteristics that are indicative of soil degradation of this state to come to an ultimate health assessment.

Reviewing soil health and soil quality approaches and indicators, (Bünemann et al., 2018) concludes that all studies have one conceptual condition that a chosen indicator must be related to a given soil threat, function or ecosystem service and be relevant and that this is not of great use, if soil quality/ soil health assessment is not targeting a specific soil threat, function or ecosystem service. We respond to this, that intrinsic soil properties like soil structure (cropland soils) or humus layer development (forests) are ultimately indicators for the functioning and health of not only soils but whole ecosystems, as they are the result of many internal and external processes, functions and drivers. They are in essence emergent characteristics, on which soil health should likely be assessed.

Part 1: Assessing of soil degradation processes (ruling – out principle)

This part was originally based on the concept previously published and documented in the EUSO Soil Health Dashboard (ESDAC, 2024) but going beyond in terms of parameters considered and developing quantitative assessments. Negative scores will be assigned, if degradation processes are identified, to be subtracted from the intrinsic soil health assessed in Part 2 of this key.

The following threats of the EUSO Soil Health dashboard are substantiated **not** considered in Part 1 of SHERPA:
- loss of soil biodiversity (part 2 of the SHERPA assumes that soil structure and/or organic layer will reflect the state of soil biodiversity),
- peatland degradation (which will be assessed differently in Part 2.4. but not considered in the current manuscript) and
- soil consumption/soil sealing as the SHERPA addresses soil health and sealed surfaces cannot be considered soils anymore. Instead it would need a soil loss statistic.

In addition to the above-mentioned threats, (and in addition to the soil threats which are considered in the EUSO Soil Health dashboard) landslide density and pesticide application will be considered by SHERPA. Pesticides will be considered in Part 1 for cropland and grassland soils. The contamination with other organic pollution, whether that be from point source or diffuse pollution, is not considered yet and needs to be addressed for future relevance (hydrocarbons such as PAHs, PCBs, PFASs, Per- and polyfluorinated substances, micro- and nano-plastic pollution).

1.1 Soil erosion and land sliding

Soil erosion will be assessed in **Table 1.1.1**. as the sum of cumulative soil erosion rates from water (Panagos et al., 2020), wind (Borrelli et al., 2017), tillage (Borrelli et al., 2023) and harvest (Borrelli et al., 2023; Panagos et al., 2019). Landslides will be assessed in a separate, second table (**Table 1.1.2**.).

The lower soil tolerance limit which separates non-affected (healthy) soil from degradation through erosion follows the Minimum Environmental Standards (MES) defined in (Bazzoffi, 2009) claiming the first harmful effects with soil erosion of 0.5 t ha^{-1} yr^{-1}. Soil erosion is causing long term soil degradation if erosion rates exceed soil formation. The below suggested rates are thus in agreement with (Verheijen et al., 2009) assessing soil erosion exceeding soil formation in many soils already at 0.3 t ha^{-1} yr^{-1} but in most soils at 1 t ha^{-1} yr^{-1}. By following the five erosion classes of (Borrelli et al., 2023) (0-1, 1-2, 2-5, 5-10, >10 t ha^{-1} yr^{-1}) it would imply that serious degradation of > 10 t ha^{-1} yr^{-1} would result in subtracting a maximum of 9 scores from the SHERPA score of Part 2. Thus, **Table 1.1.1**. was developed in between the two level values of (Bazzoffi, 2009; Verheijen et al., 2009) and (Borrelli et al., 2023).

Table 1.1.1: Cumulative erosion rates of water, wind, tillage, harvest or post-fire

Cumulative erosion rates of water, wind, tillage and harvest	**Subtract scores**
Cumulative erosion rate < 0.5 t ha^{-1} yr^{-1}	0
Cumulative erosion rate > 0.5 - 1 t ha^{-1} yr^{-1}	1
Cumulative erosion rate > 1 - 2 t ha^{-1} yr^{-1}	2
Cumulative erosion rate > 2 - 3 t ha^{-1} yr^{-1}	3
Cumulative erosion rate > 3 - 4 t ha^{-1} yr^{-1}	4
Cumulative erosion rate > 4 - 5 t ha^{-1} yr^{-1}	5
Cumulative erosion rate > 5 - 6 t ha^{-1} yr^{-1}	6
Cumulative erosion rate > 6 - 8 t ha^{-1} yr^{-1}	7
Cumulative erosion rate > 8 - 10 t ha^{-1} yr^{-1}	8
Cumulative erosion rate > 10 t ha^{-1} yr^{-1}	9

The Synoptic Pan-European Landslide susceptibility map (ELSUS 1000 v1 Map; (Gunther et al., 2014)) divides landslide risk into 5 classes ranging from very low to very high. However, the map only provides hazard classes and does not give actual densities. The Landen-Map (Hollis et al., 2022) represents landslide densities with data from 24 countries assessing an area of 210,544 km^2 of landslide prone area. Here, landslide density is stratified in three different classes: 1-3, 4-10 and 11-300 landslides/km^2. For SHERPA, the two approaches were combined to the below classes: we generally followed (Gunther et al., 2014) but subdivided the 5 classes into 10 classes to be consistent within SHERPA. We used the general range and distribution of the quantitative landslide density of the Landen-Map of (Hollis et al., 2022). Ideally, the key would refer to the area affected by landslides in m^2 per km^2, however such data is not available on European scale.

Table 1.1.2: Landslide density

Landslide density	**Subtract scores**
No landslides	0
Landslides < 1 per km^2	1

Landslides > 1 - 2 per km^2	2
Landslides > 2 - 4 per km^2	3
Landslides > 4 - 6 per km^2	4
Landslides > 6 - 10 per km^2	5
Landslides > 10 - 25 per km^2	6
Landslides > 25 - 50 per km^2	7
Landslides > 50 - 100 per km^2	8
Landslides > 100 per km^2	9

The above table would be a suitable classification, however at today's state of data availability (Hollis et al., 2022), a differentiation in only four classes are available: High (11-268/km²) = scores 6, Medium (4-10/km²) = score 4, and Low (1-3/km²) = score 1. As such, we used the following scoring table for this study:

Landslide density following mapping data of (Hollis et al., 2022)	**Subtract scores**
No landslides	0
low 1 – 3 km^{-2}	1
medium 4 - 10 km^{-2}	4
high 11-268 km^{-2}	6

As the classes by (Hollis et al., 2022) are very rough indeed, we realize that landslide density insert quite some uncertainty in the soil health assessment. Thus, adequate landslide data with finer stratified classes would be needed, which should be available from remote sensing data in the near future.

1.2. Assessment of heavy metal contamination

Due to the lack of any formal or widely accepted general thresholds for soil contamination, the assessment of heavy metal contaminations is based on the European Council directive on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (EC Directive 86/278/EEC,(EUR-Lex, 2024)). Following this directive, the concentration values of heavy metals in soils is set to subtract 9 scores according to the average value of the limit value given in the directive (e.g. in Annex IA the limit values for copper in soils are set as a range of 50-140 mg kg^{-1} of dry matter. The average value would be 95 mg kg^{-1}.). A healthy soil (meaning no scores will be subtracted from the soil health) is defined up to a heavy metal concentration of the third lowest class of the current mapped concentration based from EUSO Dashboard sources (ESDAC, 2024). For example, we took the third lowest class of (Ballabio et al., 2021) for copper, which would be 16 - 22 mg/kg and set this as the upper limit concentration for a healthy soil. The classes in between this range (e.g., limit value of the EC Directive 86|278 and the "*healthy soil*" according to today's distribution of heavy metals in Europe) were distributed in equal steps. As of December 2023, only copper (Ballabio et al., 2018), mercury (Ballabio et al., 2021) and zinc (Van Eynde et al., 2023) are included in the assessment of the EUSO Dashboard sources (ESDAC, 2024). Contamination by cadmium was also considered as published by (Ballabio et al., 2024). For lead and nickel, the soil health value of 15% of the medium limit value of The EC Directive 68|278|EEC (EUR-Lex, 2024) was taken as defining a healthy soil. The assigned scores will then be subtracted in Part 2 of the soil health index. Note that for each considered heavy metal, scores are subtracted cumulatively, e.g., a mercury contamination of 180 µg kg^{-1} and a cadmium concentration of 0.8 mg kg^{-1} would result in subtraction of 4 scores (**Table 1.2.**) in Part 2. The latter considers the strong negative effects of multi and cross contamination. Note that no limit values for chromium, cobalt, antimony or arsenic were defined in the EC Directive 68|278|EEC (EUR-Lex, 2024). In (Tóth et al., 2016) the distribution of the latter heavy metals is mapped across Europe. As such, for chromium, cobalt, antimony and arsenic, the third lowest class of heavy metal concentration from (Tóth et al., 2016) were taken as "healthy soil", while the lower boundary of the highest class was taken as limit value to subtract 9 scores. Especially for arsenic, the current assessment seems unsatisfying, as while this element is strikingly underrepresented in European and national legal frameworks, its high toxicity calls for more specific evaluation and stricter regulation.

Following the structure of the SHERPA, geogenic high content of heavy metals in soils will be classified as low soil health. For practical purposes this is useful, as these areas and soils should not be used for drinking water production, partly not usable for livestock grazing or recreational areas, where small children might play and would be exposed to potential uptake of soil material. However, eventually these areas could be marked with striped or gridded pattern, to indicate the geogenic origin of the high heavy metal content, the knowledge of which might be useful for management or planning options both for possible remediation action but also for, e.g., any kind of construction. Construction on, for example, geogenic high Arsenic content soils will result in substantial extra costs for disposing the excavated soil.

We realize that the setting of the upper and lower boundaries of each heavy metal might seem arbitrary for the moment and values need to be based on further literature research for each heavy metal separately. Here, we demonstrate the concept and structure envisaging future scientifically based limit values for each element. Also, the list of heavy metals is not complete, but we could only include elements which are considered in available guidelines, monitoring and mapping. In case of soil contamination with an element missing in table 1.2, the local (national) regulation should be considered. In such a case, 9 scores should be subtracted if the concentration of element exceeds the local guideline/intervention value (i.e., the concentration which requires soil remediation).

For now, SHERPA will target the concentration in the surface layer for potential heavy metal contamination. The upper most 20 cm will be assessed to align with the LUCAS sampling, preferably analyzed in 10 cm step layers (0-10, 10-20 cm) to avoid dilution effects. If this would be beyond feasibility, or data availability, the recommendation would be to target a bulked value of the upper most 20 cm for analysis. Concentration is targeted rather than stock of heavy metals as the concentration regulates solubility to the aquifer (if any) as well as toxicity to plants and grazing animals.

Table 1.2: Limit concentrations of heavy metals in any of the upper soil surface layers (main rooting depth: 0-10, 10-20, 20-30 cm) defining scores to be subtracted from soil health values. Note that for each element, scores are subtracted cumulatively. For comparison, the grey shaded rows give legal limit values for soil concentrations in the European Council directive on the protection of the Environment EC Directive 68|278|EEC (EUR-Lex, 2024), guideline (Richtwerte) and test values (Prüfwerte) of the Swiss soil ordinance (Fedlex, 2024)and test values (Prüfwerte) of the German Bundes-Bodenschutz- und Altlastenverordnung (BBodSchV, (Bundesministerium für Umwelt, 2024))

Cu (mg/kg)	**Hg (µg/kg)**	**Zn (mg/kg)**	**Cd (mg/kg)**	**Ni (mg/kg)**	**Pb (mg/kg)**	**Sb (mg/kg)**	**As (mg/kg)**	**Cr (mg/kg)**	**Co (mg/kg)**	**Scores to subtract**
22	23	33	0.5	9	23	0.80	10	70	20	0
32	179	57	0.6	14	42	1.30	30	90	25	1
41	332	81	0.8	20	61	1.80	50	110	30	2
50	485	105	1.0	25	80	2.30	70	130	35	3
59	638	129	1.2	31	99	2.80	95	155	40	4
68	791	153	1.4	36	118	3.20	120	180	45	5
77	944	177	1.6	42	137	3.80	145	205	50	6
86	1097	201	1.8	47	156	4.30	175	230	55	7
95	1250	225	2.0	53	175	5.00	200	250	60	8
>95	>1250	>225	>2	>53	>175	>5	>200	>250	>60	9
50 - 140	1000 - 1500	150 - 300	1 - 3	30 - 75	50 - 300	n.d.	n.d.	n.d.	n.d.	EC Directive
150	n.d.	n.d.	2	n.d.	200 -300	n.d.	n.d.	n.d.	n.d.	Test value CH
40	500	150	0,8	50	50	n.d.	n.d.	50	n.d.	Guideline CH
60/40/ 20	1000/ 500/ 1000	200/15 0/60	1,5/1/ 0.4	70/50/ 15	100/70 /40	n.d.	n.d.	100/ 60/ 30	n.d.	Precaution ary value, D, clay/silt/ sand

1.3. Assessment of nitrogen surplus

The assessment of nitrogen surplus is based on a meta-analysis of the literature and on the concept of critical loads, defined in the Convention on Long-range Transboundary Air Pollution (UNECE) as a "*quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge*" (United Nations, 2015). Global changes in the nitrogen cycle are among the nine planetary boundaries identified by (Rockström et al., 2009) to define a safe operating space for humanity. This boundary was already exceeded in 2009, and nitrogen excesses are currently in a high-risk zone (Richardson et al., 2023), exposing humanity to rapid changes of unprecedented magnitude, threatening major planetary balances.

The assessment of nitrogen surplus in all the systems studied assumes that, if the quantity of nitrogen added by deposition or by fertilizer significantly alters ecosystem functioning compared to the no-nitrogen input situation, we consider the system degraded.

Because of the differences in nitrogen dynamics between different land uses, the nitrogen surplus in the soils of the four different land uses assessed as part of SHERPA (i.e., forests, grasslands, cropland soils, and wetlands) will be treated separately (**Table 1.3**). As nitrogen surplus is one of the main dominant threats in all considered land use type, we discuss below the background and justification of the scoring in detail.

Table 1.3: Separation of land use types

Land use type	**Go to**
Forests	1.3.1
Grasslands	1.3.2
Cropland soils	1.3.3
Wetlands	n.c.*

* n.c. = not considered yet. Please note that even though we are currently developing a framework for wetland assessment and mapping, assessing wetlands were beyond the scope of this current manuscript

1.3.1. Assessing nitrogen surplus in forests

The functioning of forest soils is the result of multifactorial combinations in which site conditions, the dominant tree species, the type of humus, the epigeous and endogenous plant, animal and fungal communities, and landuse history have important influences, in addition to their physico-chemical characteristics.

As the soil is part of this ecosystem with a fragile equilibrium, the input of excess nitrogen (caused in semi-natural forests mainly by atmospheric deposition) has multiple effects on soil processes such as eutrophication and acidification (Nellemann and Thomsen, 2001); (de Vries, 2021), nitrogen leaching (Dise and Wright, 1995); (Gundersen, 1995); (Thimonier et al., 2010) nutrient cycling, and decomposition of the litter (Aber et al., 1998); (Knorr et al., 2005). Several studies report a shift in the composition of microbial communities in reaction to chronic nitrogen deposition, accompanied by a reduction in phenol oxydase activity, an enzyme produced by white-rot fungi capable of degrading lignin (Carreiro et al., 2000); (Frey et al., 2004). This degradation of soil health has visible repercussions on the health of forest stands, altering their nutrition (Braun et al., 2010); leading to nutritional imbalances (Peñuelas et al., 2012; Waldner et al., 2015)), physiological and phenological functioning and their root system (Braun et al., 2005). These fundamental changes alter their resistance to climatic hazards and pests (Bobbink and Hettelingh, 2011). Nitrogen surpluses also have a significant impact on biodiversity, with the development of nitrophilous plant communities under the canopy (Pitcairn et al., 1998) (Bobbink et al., 2010); (Talhelm et al., 2013); (Xie et al., 2024), changes in the composition and functioning of nematode (Eisenhauer et al., 2012), and microbial and fungal communities (Carreiro et al., 2000); (Sinsabaugh et al., 2002); (Frey et al., 2004); (Waldrop et al., 2004); (Treseder, 2008); (Suz et al., 2021). It also induces significant stress conditions for sensitive indicators such as lichens and algae (Lu et al., 2008); (Carter et al., 2017)

If cumulative input of N over longer time periods exceeds a certain threshold, forest ecosystems react with nitrogen leaching from soils with the accompanying consequences such as eutrophication, acidification and changes in biodiversity in adjacent deeper soil layers, aquifers as well as fresh and ocean waters (Aber et al., 1998; Dise et al., 1998; Payne et al., 2019; Rothwell et al., 2008).As most of these changes are visible on a macroscopic scale, they provide a potentially valuable approach for indirect assessment of soil health. As forests have a greater buffer capacity and less nitrogen export compared to cropland soils, we will consider the average deposition over the last 5 years as moving averages of nitrogen deposition.

For now, the assessment of nitrogen surplus is based on natural, undrained forests, without fertilizer inputs, and with an established age sufficient to consider the soil essentially free from the influence of past agricultural use.

In this section, SHERPA will target total atmospheric deposition (wet and dry), as they are the major source of nitrogen in these ecosystems.

As ground-living and epiphytic lichens and algae have been shown in the literature to be highly sensitive to atmospheric nitrogen deposition (Giordani et al., 2014), they have been used as indicators for the first category of the **Table 1.3.1**, with a subtracting score of 1.

Next, the deposition threshold of 5 kg N ha^{-1} yr^{-1} marks the first changes in the dynamics of litter decomposition (Knorr, Frey and Curtis, 2005) and in the processes of the most sensitive forest soils (Nordin et al., 2005); (Bobbink et al., 2022); (Rihm and Achermann, 2016) giving rise to the second category with a subtracting score of 2.

The input of 10 kg N ha^{-1} yr^{-1} marks changes at a larger scale as nitrate leaching and consequent acidification becomes more significant (Achermann, 2002; de Vries et al., 2014; Dise and Wright, 1995; Forsius et al., 2021; Gundersen et al., 2006; Langusch and Matzner, 2002; Thimonier et al., 2010). As critical loads are supposed to be reached when an ecosystem's capacity to absorb nitrogen is saturated, we consider that this limit is a shift in most of the soil's health which justifies a subtraction of 4. With such an input, nutritional imbalances arise due to excessive nitrogen inputs compared to other elements such as K or P (Bobbink et al., 2022). This threshold also marks the beginning of apparent eutrophication with the development of nitrophilic understory plant communities and the decrease of biodiversity (Bobbink et al., 2010)

Between 10 and 25 kg N ha^{-1} yr^{-1} , classes were distributed in equal steps, subtracting one more point to each class, from 5 to 7.
Inputs greater than 25 kg N ha^{-1} yr^{-1}, as suggested by (Dise and Wright, 1995), indicate a potential decoupling of the nitrogen cycle, resulting in significant leaching on a European scale. It is also a tipping point for tree and fungal growth in deciduous forests, as suggested by (de Witte et al., 2017) :
The upper threshold has been set at > 30 kg N ha^{-1} yr^{-1} as young stands (Gundersen, 1995)and forests which have responded positively to lower inputs (Etzold et al., 2020) appear to be affected in their growth and ability to retain nitrogen at this level of atmospheric input, impacting soil functioning.

Table 1.3.1.1. Assessment of scores to be subtracted from soil health based on total atmospheric deposition of nitrogen in forest ecosystems.

Moving averages over the last 5 years (kgN ha^{-1} yr^{-1})	**Scores subtracted**
Atmospheric deposition < 2	0
Atmospheric deposition > 2 - 3	1
Atmospheric deposition > 3 - 4	2
Atmospheric deposition > 4 - 5	3
Atmospheric deposition > 5 - 10	4
Atmospheric deposition > 10 - 15	5
Atmospheric deposition > 15 - 20	6
Atmospheric deposition > 20 - 25	7
Atmospheric deposition > 25 - 30	8
Atmospheric deposition > 30	9

A second approach to nitrogen surplus assessment is to use calculated nitrogen surplus rates. For this current first conceptual assessment of SHERPA, we followed this approach, were we used the nitrogen surplus rates from (Batool et al., 2022) to calculate nitrogen surplus scores for forests (please see section 1.3.4.). Here, surplus is calculated by adding biological N fixation to total atmospheric N deposition and substracting N removal by forests (Batool et al., 2022).

1.3.2. Assessment of nitrogen surplus for grassland ecosystems

Permanent grasslands are defined here as having continuous and stable grass cover throughout the last five consecutive years (Smit et al., 2008). For Mediterranean ecosystems, we consider the last five winter and spring seasons because of drought periods potentially affecting vegetation cover during the summer months.

For now, the assessment of permanent grassland soil health is based on total nitrogen inputs (i.e. atmospheric deposition and organic or mineral fertilization) and considers the different types of grassland found across a European latitudinal and altitudinal gradient.

Soil health degradation in grasslands is associated with alteration in soil processes and physicochemical properties, including acidification, eutrophication (Horswill et al., 2008), changes in phosphorus (Johnson et al., 1999), carbon cycling (Stiles et al., 2017; Wedin and Tilman, 1996) and nitrate leaching (Phoenix et al., 2003).

The vegetation cover of grasslands is a mosaic of taxa, families and functional groups, adapted to the local edaphic and climatic conditions. Excessive nitrogen input significantly impacts grassland's biodiversity by reducing species diversity and the total species richness, particularly of herbaceous plants (e.g. Dise and Stevens, 2005; Duprè et al., 2010; MASKELL et al., 2010; Phoenix et al., 2012; Stevens et al., 2010). Competition for light (Hautier et al., 2009) and changes in soil conditions (especially an increase in acidity and nitrogen content) promote a shift towards grasses and sedges at the expense of forbs and species adapted to nutrient-poor conditions (Bassin et al., 2012; Gaudnik et al., 2011; Payne et al., 2013; VAN DEN BERG et al., 2011). While species richness generally decreases with nitrogen input into grasslands, in the particular case of highly nutrient-constrained conditions (e.g., such as alpine meadows), species richness may increase locally due to nitrogen input, leading to the loss of adapted species being replaced by less demanding and more competitive ones (Boutin et al., 2017). In this specific case, an increase in species richness is not considered to be beneficial for the ecosystem. This leads to the homogenization and simplification of vegetation communities as well as a general increase in biomass in nitrogen-limited systems (Bobbink et al., 2022; Bonanomi et al., 2006; De Schrijver et al., 2011; Onipchenko et al., 2012). Furthermore, long-term nitrogen surplus can reduce the size and the diversity of seed banks, impacting the regeneration capacity and the genetic diversity of the system (Basto et al., 2015; Phoenix et al., 2012) The excess of nitrogen threatens ecosystem sustainability, or weakens it by increasing shoot/root ratio (Bobbink, 1991; Brouwer, 1962) and promoting lodging (Lillak, 2005). It has also been proved to be a stress factor for sensitive indicators such as bryophytes and lichens (Arróniz-Crespo et al., 2008; Carroll et al., 2000; Roth et al., 2013).

Changes in edaphic conditions and plant communities can induce shifts in mycorrhizal fungi (Blanke et al., 2012). and microbial communities (Johnson et al., 1998; Zhou et al., 2015);, increase sensitivity to secondary stressors like herbivory (Stevens et al., 2011), and modify ecosystem functioning by altering plant functional traits and phenology (Phoenix et al., 2012; Stevens et al., 2018). This impacts the entire food web, the provision of ecosystem services (Haddad et al., 2000; Stevens et al., 2018; Stevens et al., 2011) and the nutritive quality of forage production (Bassin et al., 2013). As such, we argue that a change in vegetation community or species diversity in grasslands due to nitrogen input directly affects soil health.

Changes in vegetation cover and nitrate leaching are robust indicators of nitrogen surplus impacts on grassland (Emmett, 2007). For the SHERPA key assessments of nitrogen surplus effects on grassland soils, we will base our assessment on these two factors.

Significant changes in individual species (Payne et al., 2013) and vegetation composition (Stevens et al., 2011) can possess very low nitrogen inputs, particularly on acid grasslands. At more than 3 kg N ha^{-1} yr^{-1} , the community-level change points of most sensitive ecosystems are already reached (Wilkins et al., 2016). Moreover, in a gradient study of acid grassland, an average reduction in species richness of one species for every additional 2.5 kg N ha^{-1} yr^{-1} was reported for long term deposition patterns (Stevens et al., 2004; Stevens et al., 2011). These results have been used as indicators for the first category of **Table 1.3.1.2** to subtract one score. We do realize that a recovery from N deposition is not guaranteed when nitrogen deposition is reduced. As such, **Table 1.3.2** would underestimate the threat of N deposition in case of deposition reduction.

A threshold of 5 kg N ha^{-1} yr^{-1} marks a more generalized effect on oligotrophic ecosystems like acid grasslands, and grasslands composed of low-nutrient adapted species including alpine, subalpine and Mediterranean regions (Bobbink et al., 2022). This surplus can be enough to decrease the proportion of legumes in the vegetation cover (Bassin et al., 2013), impacting the nitrogen deposition at a larger scale. The largest decline in species richness and changes in species composition occurs between 0 and 20 kg N ha^{-1} yr^{-1} (Haddad et al., 2000; Roth et al., 2018; Stevens et al., 2010). Afterwards, most of the sensitive species have been replaced by ubiquitous or nitrophilous types. Most of the critical loads for the different European grassland ecosystems are also included within this range, based on biodiversity changes (Aherne et al., 2017; Bobbink et al., 2022; Jones, 2004; Rihm and Kurz, 2001; Roth et al., 2018; Tipping et al., 2013). To represent the nonlinear relationship between vegetation cover dynamics and nitrogen deposition, the categories have been distributed in equal steps between 5 and 20 kg N ha^{-1} yr^{-1} , subtracting one point for every 5 kgN/ha/yr added but subtraction of 1 point for every 10 kg N ha^{-1} yr^{-1} between 20 and 50 kg N ha^{-1} yr^{-1} .

The upper threshold has been set at > 50 kg N ha^{-1} yr^{-1} to reflect serious nitrate leaching at this point and to follow the *'safe N operating space'* defined by the European Commission's Joint Research Centre. This limit is used by the European Soil Observatory as a threshold for soil degradation in agricultural areas, including permanent grasslands. Moreover, even phosphorus-limited and eutrophic vegetation communities seem to be impacted severely by this amount of nitrogen surplus (Hornung et al., 1995; Wilson et al., 1995). In systems using permanent grassland for grazing and mowing, limiting total nitrogen inputs is necessary to reduce nitrogen losses, by minimizing nitrogen inputs into livestock feed and increasing the efficiency of its use (Oenema, 2006).

As grassland systems do have a memory effect for high nitrogen inputs, as is often evidenced by the presence of nitrophilic plants, SHERPA will use moving averages of the last 5 years of nitrogen input (as the sum of nitrogen deposition plus fertilizer input).

Table 1.3.2. provides an overall assessment of soil health in grasslands at the European scale. However, grasslands are highly sensitive ecosystems to local conditions, particularly in the context of nutrient availability and pH (Duprè et al., 2010), resulting in a wide range of sensitivity to nitrogen surpluses. This assessment aims to be stringent enough to consider the fact that grassland system productivity may not consistently reflect soil and ecosystem health, potentially resulting in underestimation of impacts, particularly if the most sensitive species have already disappeared and cumulative loads are not

considered. In the future, a more precise assessment could adapt to the local specificities of nitrogen dynamics, as in the case of cattle resting areas, and consider separately the inputs from fertilization and atmospheric deposition.

The vegetation of acid grasslands is adapted to constraining pH and nutrient availability conditions (e.g.Stevens et al., 2004; Stevens et al., 2011). These ecosystems are more sensitive and less resilient to nitrogen excess than limestone grasslands, naturally richer in nutrients and capable of accumulating more nitrogen (Phoenix et al., 2003). To assess soil health in grasslands at a local scale, it would be necessary to distinguish between them, to avoid underestimating impacts on acid grasslands and overestimating them on limestone grasslands.

Table 1.3.2. Assessment of nitrogen surplus for grassland ecosystems based on total deposition rates

Moving averages over the last 5 years (kgN ha^{-1} yr^{-1})	**Scores subtracted**
Nitrogen inputs ≤ 2	0
Nitrogen inputs > 2 - 3	1
Nitrogen inputs > 3 - 5	2
Nitrogen inputs > 5 - 10	3
Nitrogen inputs > 10 - 15	4
Nitrogen inputs > 15 - 20	5
Nitrogen inputs > 20 - 30	6
Nitrogen inputs > 30 - 40	7
Nitrogen inputs > 40 - 50	8
Nitrogen inputs > 50	9

As for the forest assessment, we followed in this current study a second approach to nitrogen surplus assessment of grasslands in using the nitrogen surplus rates from ((Batool et al., 2022), please see section 1.3.4)). Here, surplus is calculated by adding biological N fixation as well as mineral fertilizer and animal manture to total atmospheric deposition and substract N removal by harvest and pasturing (Batool et al., 2022).

1.3.3. Assessment of nitrogen surplus for cropland ecosystems

The nitrogen cycle in agro-ecosystems is a delicate balance between nitrogen inputs from fertilization, atmospheric deposition, and symbiotic fixation, and outputs through harvest and leaching to deeper soil layers and waters. The latter underscores the necessity for optimal management to maximize Nitrogen

Use Efficiency (NUE) (Oenema O et al., 2016) and to avoid over-fertilization or mining. While this is crucial for balancing food security, it is also a necessity for long-term ecosystem preservation and the prevention of fresh and ocean water eutrophication.

For SHERPA, we consider the main crop categories present in Europe (excluding flowers), whose exploitation is regulated by the CAP and the Nitrates Directive. The assessment of agricultural soil health is currently based on total nitrogen inputs, including dry and wet atmospheric deposition as well as fertilization in mineral or organic form.

In a context of demographic growth and resource strain, most cropland soils in Europe are intensively exploited. The extensive use of fertilizers to ensure necessary nitrogen inputs has overshadowed the mineralization work of soil fauna, thereby greatly reducing soil biodiversity (Nabel et al., 2021) and impacting the provision of ecosystem services (de Souza and Freitas, 2018). Changes in physico-chemical properties, particularly acidification caused by excessive nitrogen inputs disrupt the balance between fungal and bacterial communities, exacerbating the alteration of nutrient cycles in these soils (Nabel et al., 2021). The alteration of nutrient balance in the soil can limit yields by increasing lodging effects in cereals (Dahiya et al., 2018) and susceptibility to pests and herbivory (Altieri and Nicholls, 2003). Quality, especially in fruit and vegetable crops, is affected by the reduction in organoleptic attributes, content of other mineral nutrients, and synthesis of secondary metabolites (e.g., vitamins) (Albornoz et al., 2016). Some crops, notably leafy green vegetables, may accumulate excessive nitrates, with detrimental effects on consumer health (Albornoz et al., 2016).

Excessive nitrogen input to agricultural soils also leads to significant leaching of excess nitrates, exacerbated during inter-crop periods if soils are left bare (Fraters et al., 2015; Goulding, 2000; Köhler et al., 2006; Simmelsgaard and Djurhuus, 1998; Webb et al., 2000). These nitrates are transferred to waters, causing eutrophication and resulting in large-scale impacts (Billen et al., 2013). Nitrogen fixation by legumes (e.g., intercropping, cover crops) is not considered in the SHERPA tables as this nitrogen will either be exported if cover crops are harvested or will be added to the organic nitrogen pool in the soil. In the latter case we might underestimate the nitrogen surplus of the respective cropland soils.

Nitrogen can also be lost significantly in gaseous form through emissions of ammonia (NH_3), nitrogen oxide (NO), or nitrous oxide (N_2O) (Jenkinson, 2001; Velthof et al., 2009). Nitrous oxide, being a potent greenhouse gas, contributes to climate change on a large scale, as do nitrogen oxides indirectly. NH_3 emissions lead to the formation of aerosols with detrimental effects on air quality and health (Pozzer et al., 2017)

The rationale behind the assessment of nitrogen surplus in cropland ecosystems was to allow a nitrogen input above the threshold for semi-natural ecosystems (e.g., Forest **Table 1.3.1**.) based on the amount of nitrogen that is extracted as a result of harvest. Data was used from (Einarsson et al., 2021) to calculate crop specific N export by harvest averages from 2009 - 2019. Using this time period as baseline for the N scores to be subtracted might be discussed as many countries within the EU already reached a high level of N content in harvested crops due to decades of overfertilization (Albornoz, 2016). We used average data over all EU countries even though N export by harvest varies considerably between countries. However, as it is important not to base values on data from countries either over fertilizing or on countries mining N from cropland soils due to low fertilization, we consider it justified to base the SHERPA N scores for cropland soils on the EU average value.

Table 1.3.3. Assessment of nitrogen surplus for cropland ecosystems

Assessment of nitrogen surplus for cropland ecosystems (kgN.ha^{-1}·yr^{-1})	**Scores subtracted**
Atmospheric deposition ≤ 2 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	0
Atmospheric deposition > 2 - 3 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	1
Atmospheric deposition > 3 - 4 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	2
Atmospheric deposition > 4 - 5 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	3
Atmospheric deposition > 5 - 10 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	4
Atmospheric deposition > 10 - 15 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	5
Atmospheric deposition > 15 - 20 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	6
Atmospheric deposition > 20 - 25 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	7
Atmospheric deposition > 25 - 30 kgN·ha^{-1}·yr^{-1} plus N yield exported by harvest*	8
Atmospheric deposition > 30 kgN·ha^{-1}.yr^{-1} plus N yield exported by harvest*	9

*Crop specific N yield exported by harvest on average for the EU is given in **Table 1.3.3A** below.

Table 1.3.3A: Mean N yield (kg N ha^{-1} yr^{-1}) for major crop categories in the EU (excluding Cyprus and Malta) between 2009 and 2019.

Crop category	**Crop**	**N yield (kg N ha^{-1} yr^{-1})**
Cereals	Barley	83.13
	Wheat	108.76
	Grain maize	111.44
	Other cereals	63.31
Fodder	Fodder roots	112.84
	Forage legumes	188.17
	Green maize	170.47
	Temporary grassland	170.56

	Other forage crops	90.87
Oilseeds	Oilseeds	80.27
Vegetables and others	Vegetables and other	54.46
Permanent crops	Grapes	23.23
	Olives	9.71
	Other permanent crops	15.6
Other cropland	Pulses	88.3
	Potatoes	81.66
	Sugar beet	151.5

Data calculated from (Einarsson et al., 2021) .

This assessment is conducted annually. For crop rotations, it would be beneficial to adjust the modalities according to the vegetation development stage. Land management practices (i.e., timing and form of nitrogen inputs, irrigation, tillage) should also be considered for local assessment, as this factor significantly influences its fate within the ecosystem.

As for the forest and grassland assessment, we followed in this current study a second approach to nitrogen surplus assessment of croplands in using the nitrogen surplus rates from ((Batool et al., 2022), please see section and table 1.3.4)). Here, surplus is calculated by adding biological N fixation as well as mineral fertilizer and animal manure to total atmospheric deposition and substract N removal by harvest (Batool et al., 2022).

1.3.4. Assessment of nitrogen surplus for forest, grasslands and cropland ecosystems following (Batool et al., 2022)

In the current conceptual assessment of SHERPA we assigned nitrogen surplus rates for the three land use types forest, grasslands and cropland following (Batool et al., 2022). As (Batool et al., 2022) calculated nitrogen surplus for each land use type differently, we considered this approach suitable for our preliminary assessment of point data across Europe. For a detailed future evaluation of soil health with high resolution data availability and with the aim of high-resolution mapping, we recommend the above approaches for the different land use specific assessments (see section 1.3.1. -1.3.4.).

(Batool et al., 2022) calculated nitrogen surplus rates for each land use type by subtracting total outputs (N removal by harvest) from total inputs (mineral fertilizer, animal manure, nitrogen fixation, total nitrogen deposition). We used these rates to assign the nitrogen surplus scores in the current evaluation following table 1.3.4.

Table 1.3.4. Assessment of scores to be subtracted from soil health based on nitrogen surplus rates of (Batool et al., 2022) for the land use types forest, grassland and cropland.

Moving averages over the last 5 years (kg N ha^{-1} yr^{-1})	**Scores subtracted**
Atmospheric deposition < 2	0
Atmospheric deposition > 2 - 3	1
Atmospheric deposition > 3 - 4	2
Atmospheric deposition > 4 - 5	3
Atmospheric deposition > 5 - 10	4
Atmospheric deposition > 10 - 15	5
Atmospheric deposition > 15 - 20	6
Atmospheric deposition > 20 - 25	7
Atmospheric deposition > 25 - 30	8
Atmospheric deposition > 30	9

1.4. Assessment of phosphorus deficiency or excess in grasslands and cropland soils

Phosphorus (P) is essential for the growth, functioning and reproduction of all life on earth. In natural ecosystems, the P that is lost from the soil-plant cycling system has to be replaced by the slow process of rock weathering (Bouwman et al., 2009) or added via fertilizer in human managed systems (there is no equivalent to the biological N_2 fixation which is only kinetically limited but potentially not resource limited). However, if fertilization with animal waste or human excreta is not available or not organized, P fertilizers stem from non-renewable geological P deposits, which are an increasingly limited resource (this is again in contrast to N, as N fertilizer can be produced as an endless resource via the Haber Bosch process as along as energy and natural gas is available). The one-way flow of P from mineral reserves to farms (e.g., soils), to freshwaters and finally into oceans, are already considered to be beyond the safe operating space for sustainable human development (Carpenter and Bennett, 2011). The potential threats of global P limitation due to “peak phosphorus” have been discussed intensively in the recent past (e.g. (Cordell et al., 2009) (Edixhoven et al., 2014; Gilbert, 2009)). The imminent threat of such a P limitation has been restrained somewhat as obviously some P deposits had been overlooked or misclassified in the past which will theoretically last for the next 600 years of global P supply (Van Kauwenbergh, 2010). However, the socio economic and political consequences are still dramatic with the newly discovered P reserves being restricted to a small region of the Western Sahara and Morocco.

Whilst P is often a limiting macronutrient for living organisms in aquatic and terrestrial ecosystems (Paytan and McLaughlin, 2007; Sohrt et al., 2017), especially in the context of agricultural production (Alewell et al., 2020), urbanization and intensification of agriculture have caused a P enrichment of surface aquatic ecosystems. This has led to hypoxia, eutrophication (Withers and Jarvie, 2008), and a loss of biodiversity in fresh- and ocean waters (Davis et al., 2019; Lind et al., 2019; Pant, 2020). As such, regarding soil and ecosystem health, an excess of P (referring to greater P input compared to P harvested) will lead to an accumulation of P in soils and/or a loss of this nutrient mainly due to erosion which can be considered a potential ecological threat to biodiversity and ecosystem health due to its eutrophication effect in less intensively or unmanaged ecosystems. As such, well managed P budgets of soils are needed to decrease the ecologically negative impact on fresh and ocean waters due to high P input and accompanied eutrophication and hypoxia. We conclude that a sustainable balance between P input via mineral or organic fertilizer with P harvest by crops or grassland is mandatory not only for soil and water health but also for sustainable agricultural management.

Phosphorous input per hectare of cropland varies between countries and regions, ranging from as low as 0.52 kg ha^{-1} a^{-1}r in Mozambique to 85 kg ha^{-1} yr^{-1} in New Zealand (2009 data, https://ourworldindata.org based on (West et al., 2014)) (Figure 1.4.1). The latter obviously reflect the wide span from scarcity of resources to rich, resource wasting countries and, indirectly, height of gross domestic product in the countries.

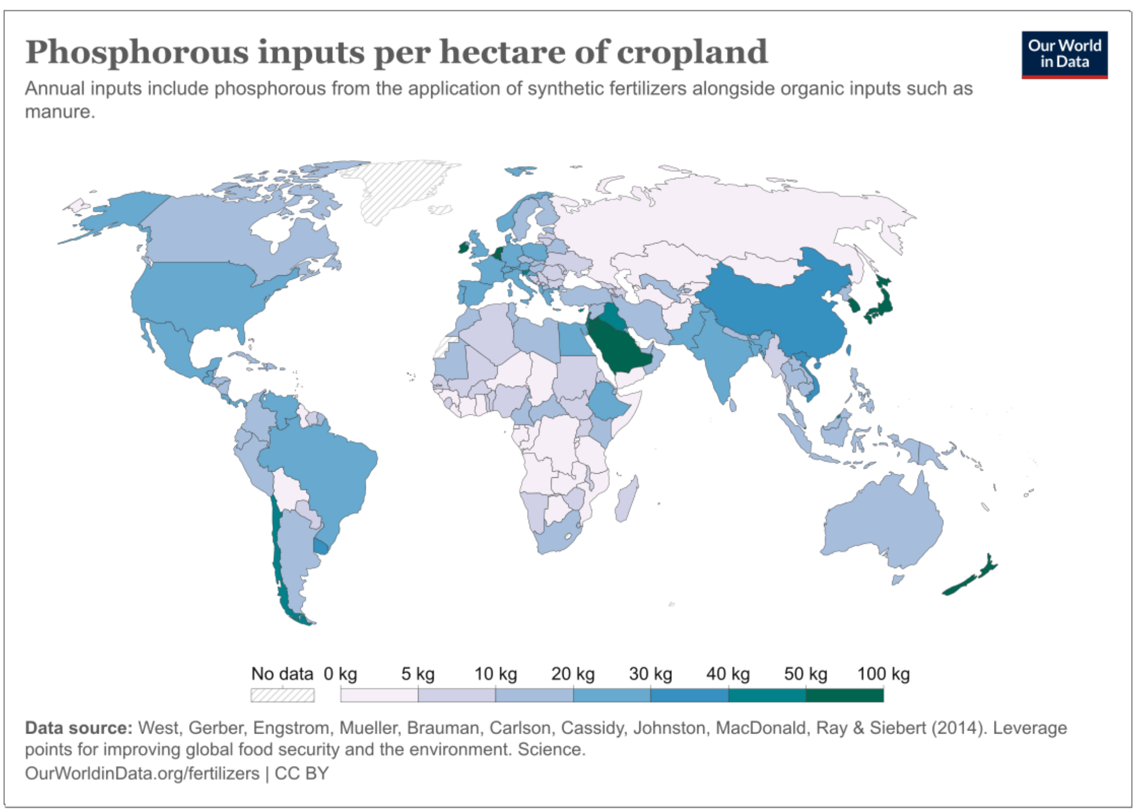


Figure 1.4.1: Phosphorus inputs per hectare of cropland (Our WorldInData, https://ourworldindata.org).

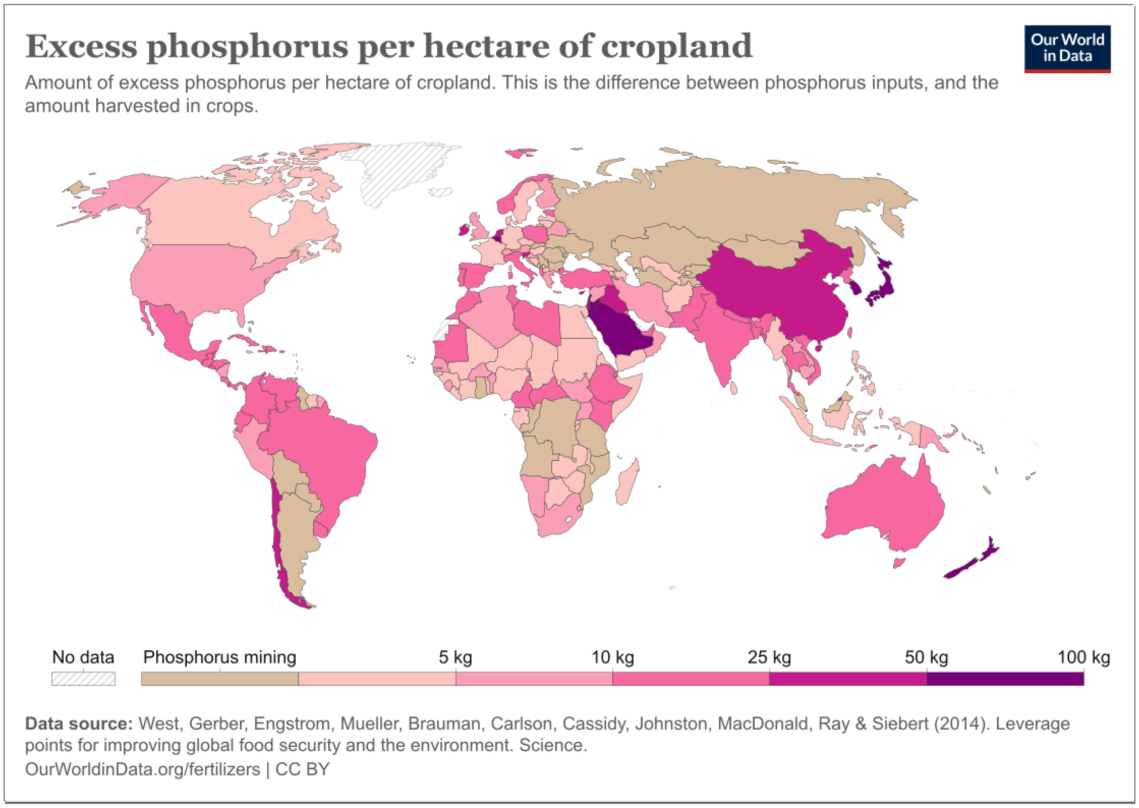


Figure 1.4.2.: Excess Phosphorus per hectare of cropland (OurWorldInData; https://ourworldindata.org)..

The amount of phosphorus fertilizer applied per hectare of cropland has increased tremendously between 1961 and 2021 globally (https://ourworldindata.org/grapher/phosphate-application-per-hectare-of-cropland?time=2021; Food and Agriculture Organization of the United Nations (2023) – with major processing by Our World in Data). Excess Phosphorus ranges from > 70 kg ha^{-1} yr^{-1} in New Zealand, Singapore and Jordan to negative values (P mining) in many eastern European and African countries (> -20 kg ha^{-1} r^{-1} in Solomon Islands, Gaum, Northern Mariana Islands) (Figure 1.4.2.). Within Europe, Malta (-5 kg ha^{-1} a^{-1}) and eastern European countries, e.g. the Republic of Moldova (-2.5 kg ha^{-1} a^{-1}) and the European part of Russia (-1.4 kg ha^{-1} a^{-1}) have a negative P balance (calculates as input as fertilizer minus output with P harvest), the so-called P mining of soils. Conversely, Belgium (66 kg $ha^{-1}r^{-1}$), Luxembourg (49 kg $ha^{-1}r^{-1}$), the Netherlands (45 kg $ha^{-1}r^{-1}$) and Ireland (41 kg $ha^{-1}r^{-1}$) are overfertilizing with high rates of excess P.

P is a limited resource, and a waste of P will lead to famines and migration in a future world due to an expected sharp decrease in agricultural productivity. Simultaneously, each kg of excess P will lead to unnecessary loss in grassland biodiversity as well as possible eutrophication and hypoxia in waters. Thus, we apply strict negative scoring for excess phosphorous (**Table 1.4.1**.).

P mining (harvesting more P with biomass export from soils compared to input with mineral and organic fertilizer) will lead to P depleted soils minimizing harvest potential. However, even though P mining will lead to P depleted soils with low agriculture production, it might be argued that this is a soil quality issue rather than a soil health issue. While older soils are generally depleted in phosphorus compared to younger soils with P release from easily weatherable material (Izquierdo et al., 2013; Walker and Syers, 1976), we are not aware that the extraction of soil P by plant uptake in natural systems will lead to major negative impacts on soil biota or unwanted soil chemical or soil physical processes. The latter considers that in natural ecosystems weathering will always release P which eventually is the limiting nutrient for vegetation growth. However, a continuous long-term P mining in agricultural soils (cropland as well as grasslands) with increasing P depletion will eventually lead to a negative impact on soil biota and might limit vegetation cover and soil organic matter cycling leading to a degradation of soil structure, enhanced erosion and general nutrient depletion of soils. As such, P mining will also be scored within SHERPA.

From modelling the phosphorus cycle in agricultural soils on a European scale using the DayCent process based model, results have shown that (considering all inputs such as mineral and organic fertilizer, chemical weathering, possible storage and sorption processes versus all outputs like harvest, residue removal, erosion and organic P leaching) led to negative P balances of -2.5 kg ha^{-1} yr^{-1} in the Czech Republic to positive P balances of 5.2 kg ha^{-1} yr^{-1} in Cyprus and Denmark (Muntwyler et al., 2024). While a full process-based modelling will inevitably be connected to a certain degree of uncertainty and different assessments varying considerably for countries and regions (Muntwyler et al., 2024; Panagos et al., 2022), the current evaluation of SHERPA only considered input from mineral and organic fertilizer versus output with P harvest similar to the approach of Our World in Data (https://ourworldindata.org/grapher/phosphate-application-per-hectare-of-cropland?time=2021). However, scores to be subtracted are scaled considering the magnitude of the full P budget of the two assessments by (Panagos et al., 2022) and (Muntwyler et al., 2024) which range from – 8 kg ha^{-1} yr^{-1} of P mining to > 10 kg ha^{-1} yr^{-1} of P excess in European soils.

Table 1.4. P Mining or P excess from mineral and organic fertilizer

P mining or P excess	Go to table
P mining [$kgP \cdot ha^{-1}yr^{-1}$]: Phosphorus input from mineral and organic fertilizer smaller then P harvested with crops or grass	1.4.1
P excess [$kgP \cdot ha^{-1}yr^{-1}$]: Phosphorus input from mineral and organic fertilizer greater then P harvested with crops or grass	1.4.2

Table 1.4.1. P mining [$kgN \cdot ha^{-1}yr^{-1}$]: Phosphorus input from mineral and organic fertilizer smaller then P harvested with crops or grass

P mining [$kgP \cdot ha^{-1}yr^{-1}$]	Scores subtracted
< 0 to - 0.5	0
< -0.5 to -1	1
< -1 to -2	2
< -2 to -3	3

< -3 to -4	4
< -4 to -5	5
< -5 to -6	6
< -6 to -7	7
< -7 to -8	8
< -8	9

Table 1.4.2. P excess [kgN$\cdot$ha$^{-1}\cdot$yr^{-1}]: Phosphorus input from mineral and organic fertilizer greater then P harvested with crops or grass

P excess [kgP$\cdot$**ha$^{-1}\cdot$yr^{-1}]**	**Scores subtracted**
≤ 1	0
> 1 to 2	1
> 2 to 3	2
> 3 to 4	3
> 4 to 5	4
> 5 to 6	5
> 6 to 7	6
> 7 to 8	7
> 8 to 9	8
> 9 to 10	9

We realize that basing the SHERPA scores merely on the balance of P inputs and outputs seems, even though logical and operational, over simplistic. The approach might overlook legacy effects—namely, soils that have already undergone intense eutrophication (P storage) or, conversely, those that have experienced massive P depletion. Thus, there might be a rationale for considering the status of the bioavailable P stock. In practical terms, if a soil has a high available P level (>45 mg kg^{-1} Olsen P), the appropriate P input should be lower than the crop export. Conversely, if the bioavailable P level is low (likely P-limiting) then inputs exceeding exports might be justified. However, this adds complexity not only on data availability of (to defined) specific P pools, but even more so how to rate certain P species and pools regarding their availability for plants or aquatic organisms. Thus, SHERPA is based on the balance of P inputs and outputs, realizing that this introduces short term uncertainties but the concept should hold if we assess longer term trends in the future.

1.5. Pesticide use in cropland and grassland

Pesticides play an ambivalent role in modern agriculture. On the one hand, they are an effective tool to control damaging crop pests, diseases, and weeds, preventing yield losses, ensuring crop quality and food security. On the other hand, the hazardous effects that pesticide use can have on human health and the environment are well documented (Bourguet and Guillemaud, 2016; Dereumeaux et al., 2020; Kim et al., 2017; Lee et al., 2019; Sánchez-Bayo and Wyckhuys, 2019; Savary et al., 2019). On cropland and non-permanent grassland, pesticides tend to be applied over large areas. In permanent grassland, usually only herbicides are applied. Nevertheless, pesticides can be found in all soils, including the extensive grassland sites, due to diffuse contamination (e.g. from neighbouring fields) (Riedo et al., 2022). This often leads to unintentional widespread dispersion across the environment, contamination of non-target areas and poisoning of species (Arias-Estévez et al., 2008; Fenner et al., 2013; Galon et al., 2021; Mottes et al., 2014; Tang et al., 2021).

Against this background, policymakers have consistently been interested in pragmatic and practical approaches to assess the environmental and human health implications of agricultural pesticide use at a national level as a basis for policy initiatives (including pesticide taxation) and communication of policy outcomes to the public (Lewis et al., 2021). To serve the various purposes, several pesticide risk indicators were developed. Most indicators are used to assess pesticide risks for (specific aspects/sub-systems of) the environment and/or human health.

Some of the early and most widely used pesticide (risk) indicators are the Environmental Impact Quotient (Kovach et al., 1992), the SYNOPS indicator which has been used in Germany and continues to be further developed (Strassemeyer et al., 2017), the Environmental Yardstick for Pesticides indicator from the Netherlands (Reus and Leendertse, 2000) and the Treatment Frequency Index which has been used in Denmark for pesticide taxation for over 30 years (Gravesen, 2003). A more exhaustive and detailed overview of the different indicators is provided by (Kudsk et al., 2018). The use of simpler measures (or a combination thereof) such as the number of interventions (i.e. the number of pesticide applications), the total amount of the applied active substances, and the active-substance ranking (i.e. the frequency with which individual active substances are used) have also been suggested (de Baan L., 2009-2012). However, these cannot be considered suitable for SHERPA because they either do not consider the extremely heterogeneous toxicity of different active ingredients in pesticides and as a result, their varying impact on soil health, or the fact that they are calculated per crop rather than for land use type. The latter inhibits soil health assessment or would make it overly complex as crop rotation would have to be considered and such data is rarely available for Europe.

More recent and elaborated indicators were developed for both assessing pesticide risks and as a basis for the taxation of pesticides. Next to the Norwegian pesticide risk indicator (Stenrød et al., 2008), the pesticide load indicator (PLI) is the most recent of these indicators. It was developed in Denmark and serves not only to monitor pesticide load (PL) but also to set quantitative targets for the reduction of adverse impact of pesticides on human health and the environment.

The PLI consists of three sub-indicators for human health, ecotoxicology and environmental fate. For each of the sub-indicators, the pesticide load is calculated per unit commercial product (e.g. kg, liter) (Kudsk et al., 2018). The rationale behind the assessment is based on the following concept of pesticide load. PL does not aim to account for the actual exposure or harm but it reflects the *"[…] relative environmental pressure that occurs due to the differing hazardous nature of the pesticides used and the variability in quantities applied*" (Lewis et al., 2021). This approach results in a higher PL when active substances in pesticide products are used in large amounts, are persistent and mobile, bioaccumulate, and are ecotoxic to many species.

From the above approaches, we regard three alternatives as most relevant for SHERPA. First, SHERPA could use the sub-indicator for environmental fate (PLI_{Fate}) which is calculated on the basis of the half-life in soil (DT_{50}), the bioaccumulation factor (BCF) and the SCI-GROW index reflecting the mobility and risk of leaching to the groundwater of the active ingredient and its major metabolites (US-EPA, 2016). Second, the full PLI (PLI_{Total}) made up of the three mentioned sub-indicators could be used. Third, another slight variation presented as a third option is the PLI_{UK} which is the result of (Lewis et al., 2021) application of the Danish PLI to UK data. The relatively similar threshold values demonstrate that the PLI is transferable to other European countries. However, the threshold values provided in Table 1.5.1. may need to be calculated for each country based on national pesticide use data. This is because pesticide products and their formulation may vary between countries and change over time (Möhring et al., 2021). In addition, the threshold values of the indicators presented represent the distribution of the calculated values on a national scale (Kudsk et al., 2018).

To calculate the PLI, a tangible and transparent public tool for indicator computation is available in the form of an R package (Möhring et al., 2021). It requires data on farmers' pesticide application (i.e. the annual pesticide statistics; using sales data instead may not reflect the actual use by farmers). A key challenge here appears to be that available pesticide use data differ in quality. Moreover, the low accessibility of national databases on registered pesticides calls for improvements. Information on pesticide properties needed for the computation such as BCF or DT_{50} can be integrated from the Pesticide Properties Database (licensed) (http://sitem.herts.ac.uk/aeru/ppdb/en/index.htm; (Lewis et al., 2016)). Upon computation of the PL for each pesticide product, the PL may be aggregated at a field and crop-, farm, regional or national level. The PLI can also be used for mapping. This has already been done for Denmark where pesticide use data is available for each farm (Kudsk et al., 2018). The maps produced thus provide detailed information on pesticide use in different regions.

Table 1.5.1 Three different pesticide load indicators for suggestion to define scores to be subtracted from the soil health key in SHERPA.

PLI_{Fate} (Kudsk et al., 2018, Fig.3C)	**PLI_{Total}** (Kudsk et al., 2018, Fig. 2B)	**PLI_{UK}** (Lewis et al., 2021, Fig. 3) (*10exp6)	**Scores to subtract**
0	0.0	0	0
0.2	0.6	0.2	1
0.3	1.2	0.5	2
0.4	1.4	1	3
0.5	1.6	1.5	4
0.7	2.0	2	5
0.9	2.4	2.5	6
1	3.0	3	7
1.2	3.6	3,5	8
1.3	4	4	9

The above approaches look promising and will result in country and region-specific pesticide risk assessment within the overall assessment of soil health within SHERPA. However, the country or region-specific risk indicators are not available yet. As such, for the current calculation of SHERPA in this study we rely on the data from (Tang et al., 2021) where the environmental pollution risk caused by 92 active pesticide ingredients in 168 countries globally is considered. Here, to be able to go large scale, a simplified approach has been developed, where a region is considered to be at risk of pollution if pesticide residues in the environment exceed the no-effect concentrations (Tang et al., 2021). The pesticide-no-effect-concentrations (PNEC) of the 92 selected active incredients (AIs) in each of the four environmental compartments (atmosphere, surface water, groundwater and soil) were defined in using an assessment factor approach with acute toxicity data (see (Tang et al., 2021) for details). To calculate the SHERPA scores of pesticide pollution to be substracted from the overall soil health, we subdivided the six risk classes of (Tang et al., 2021) (Fig. 1.5.1.) using the active incredient concentrations (Fig. 1.5.2.) to classify the ten scoring classes of SHERPA (Table 1.5.2.).

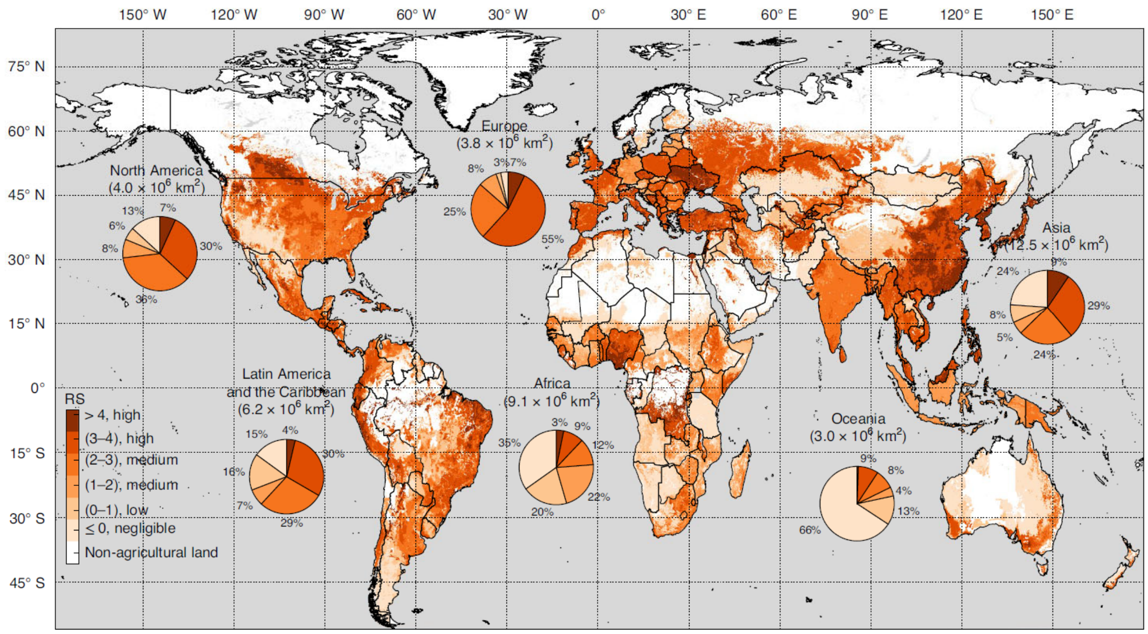


Fig. 1.5.1 Global map of pesticide RS. The map has a spatial resolution of 5 arcmin, which is approximately 10 km × 10 km at the Equator. The pie charts represent the fraction of agricultural land classed under different RS in each region, and the values in parentheses above the pie charts denote the total agricultural land in that region(Tang et al., 2021).

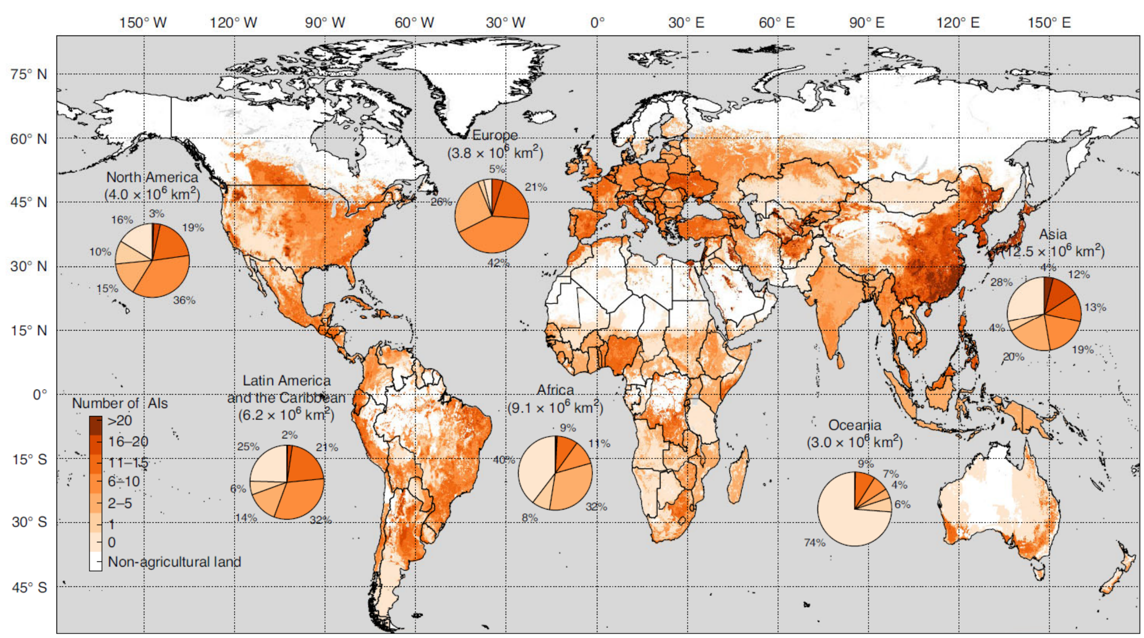


Fig. 1.5.2 Global map of the number of AIs posing risks to the environment. The map has a spatial resolution of 5 arcmin, which is approximately 10 km × 10 km at the Equator. The pie charts represent the fraction of agricultural land contaminated by different numbers of AIs in each region, and the values in parentheses above the pie charts denote the total agricultural land in that region (Tang et al., 2021).

Table 1.5.2. Pesticide Risk and Active Ingredients of (Tang et al., 2021) combined to a pesticide health score to subtract from soil health.

Pesticide Risk class (RS)	**Active ingredients** (AI)	**Scores to subtract**
≤ 0 (negligible)	0.0	0
≤ 0 (negligible)	> 0	1
0-1 (low)	> 0-10	2
1-2 (medium low)	>1 - 10	3
1-2 (medium low)	> 10 – 20	4
2-3 (medium)	> 1 – 10	5
2-3 (medium)	> 10 – 20	6
3-4 (high)	>1 - 10	7
3-4 (high)	> 10 – 20	8
>4 (very high)	> 10	9

1.6. Assessment of salinization

As salinization in recent literature is based on electrical conductivity (EC) (dS m−1) of the soil water extract (Hassani et al., 2020; Hassani et al., 2024a), a separation of calcareous (e.g., containing calcium or magnesium carbonate) from saline or alkaline soils (containing sodium chloride, sodium carbonate of sodium sulphates) is necessary (**Table 1.7.1**). Also, naturally saline or alkaline soils will not be counted as "*unhealthy*" as they form naturally saline/alkaline ecosystems with low vegetation cover and/ or halophytic vegetation with a very precious and special biodiversity. These ecosystems are either formed from the influence of sea water in coastal conditions, high geogenic salt contents or by naturally dry climatic conditions with ascending groundwater input inducing evaporation to precipitation ratios above 1.

Traditionally, the threshold value of EC of saturated soil extract ECe (indicating the ability of a water-saturated soil paste extract to conduct electrical current representative of salinity severity) has been one of the primary indicators for identifying saline soils (Hassani et al., 2020). The threshold values used for ECe, however, can differ depending on the soil classification system. In this study, we adopted ECe of 4 dS m^{-1} as the critical thresholds, reflecting the lowest agronomic limits acceptable for crops (19) which is adopted by recent papers to determine the extent of saline soil on a global scale (Hassani et al., 2020; Hassani et al., 2021; Hassani et al., 2024a; Hassani et al., 2024b). As such, SHERPA classifies this as severely degraded due to salinisation (scores of - 9 to be subtracted).

To define the salinity score to be subtracted from the soil health key in SHERPA, we analyzed the sensitivity of a wide range of plants to soil salinity. Plants tolerate salinity up to a certain threshold above which they will be negatively impacted (e.g., in the case of agricultural crops, the yield begins to decrease or in natural grasslands or forests population dynamics of flora and fauna will be altered). Using literature data, we determined salt tolerance and salinity threshold of more than 140 different plants including herbaceous and woody crops from (Maas and Grattan, 1999) as well as agricultural crops from (Maas and Hoffman, 1977). The threshold values below 2 dS/m and above 4 dS/m were excluded from the database because our scoring system is being proposed for soil with the salinity between 2 to 4 dS/m. Then, we performed a hierarchical clustering analysis (Hastie et al., 2009) on salinity threshold values to group similar salt tolerance levels into distinct clusters. A distance matrix was calculated for these values, and hierarchical clustering with the complete linkage method was applied. The dendrogram was then cut to form eight clusters, each representing a group of closely related salinity values with higher values indicating higher salinity levels. For each cluster, we defined the left and right boundaries, and used a nearest cluster center approach to assign new data points based on proximity to these centers which enabled us to determine the range of salinity for each score presented in **Table 1.7.2**.

Table 1.7.1. Separate natural and anthropogenic salinization from calcareous soils

Separate natural and anthropogenic salinization from calcareous soils	**Go to**
Soils on calcareous bedrock with HCl test indicating calcium carbonate or dolomite within the fine earth	part 2
Soils within the influence of marine waters	part 2
Soils in semi-arid regions with no anthropogenic irrigation	part 2
Anthropogenic irrigated soils with EC level above 2 dS m^{-1}	1.8.2.

Table 1.7.2. Assessing salinization of irrigated soils by electrical conductivity of soil water extract

Assessing salinization of irrigated soils by electrical conductivity of soil water extract (EC in dS m^{-1})	**subtract scores**
< 2 dS m^{-1}	0
2.0 ≤ EC < 2.2	1
2.2 ≤ EC < 2.4	2
2.4 ≤ EC < 2.6	3
2.6 ≤ EC < 2.9	4
2.9 ≤ EC < 3.1	5
3.1 ≤ EC < 3.4	6
3.4 ≤ EC < 3.7	7
3.7 ≤ EC < 4.0	8
≥ 4 dS m^{-1}	9

1.8. Assessment of compaction

Spatially, soil compaction, after erosion, is the most significant physical soil degradation process with detrimental impact on nearly all physical, chemical and biological soil processes (Smith et al., 2016). Soil productivity and ecosystem health is severely impeded by soil compaction through moisture deficits, restriction in root depth as well as increases in runoff and erosion (Batey, 2009; Shaheb et al., 2021; Zhang et al., 2024). The reduction in soil porosity and permeability reduces the soil's capacity to store air and water as well as impedes water flow, permeability and air diffusivity. The principal causes are compressive forces derived from wheels of agricultural machinery and from the trampling of animals acting on compressible soil (Batey, 2009). The effect of compressible forces to soils depends on many external factors such as weight and speed of machines/ animals, wheel slippery, number of skidding cycles, caution and expertise of machine operators and soil intrinsic factors such as initial bulk density, soil moisture, soil texture, soil structure, clay mineralogy, soil organic matter content and slope (more confined distribution of loads with increasing slope) (Batey, 2009; Cambi et al., 2015; Hamza and Anderson, 2005; Labelle et al., 2022; Zhang et al., 2024).

A number of reviews have been published in recent years on the causes, influencing factors, and effects as well as on the mitigation and prevention of soil compaction (Batey, 2009; Cambi et al., 2015; Labelle et al., 2022; Shaheb et al., 2021; Zhang et al., 2024). Furthermore, some of these reviews also assessed the effect of compaction on crop productivity (Hamza and Anderson, 2005; Lacey and Ryan, 2000; Shaheb et al., 2021; Zhang et al., 2024), tree species biodiversity (Latterini et al., 2023) or forest regeneration (Cambi et al., 2015). However, to our knowledge, the degree of compaction in quantifiable numbers is not assessed which would be necessary to link compaction to soil health effects. (Jones et al., 2003) provides tools to assess the vulnerability of soils to soil compaction both on small and large scale considering inherent susceptibility of soils (e.g. soil texture, packing density calculated form bulk density and clay content, organic matter content, soil structure) as well as climatic dryness/subsoil wetness or actual soil moisture status. However, developed maps represent not the actual soil compaction but soil vulnerability to compaction. As such, a soil having a high risk/ high vulnerability might have a low bulk density, low packing density and loose structure, thus being a very healthy soil.

Compacted soil may also be created by natural processes, mainly by periglacial conditions creating dense indurated layers (Fitzpatrick, 1956) or naturally occurring pans (Needham et al., 2004). Thus, in some settings, hard-setting soils may form dense and impenetrable layers, unrelated to the application of anthropogenic compressive forces (Mullins et al., 1990). However, in European ecosystems, natural soil formation processes (mostly in forests and grasslands) as well as cropland management (ploughing, if these systems were chosen for agricultural use) might have mitigated periglacial compaction of surface soils during (agro)ecosystem succession. Thus, we assume that this originally natural compaction by periglacial processes is not a common phenomenon in European soils and will thus not be considered by SHERPA.

Field criteria to assess compaction include water logging on the surface or in subsurface layers, increase in soil strength, a reduction in visible porosity, changes to soil structure, soil colour and particularly change in distribution of roots and irregular patterns of soil (Batey and McKenzie, 2006; Spoor, 2006). Bulk density and packing density have been proposed as a proxy for soil compaction (Panagos et al., 2023) or soil's vulnerability to compaction (Jones et al., 2003). While data is available at European scale (Panagos et al., 2024) and compacted soils are related to high bulk and packing density, both parameters might not necessarily indicate anthropogenic compaction and soils with medium to even low bulk and packing density might not be free from compaction and/or damage from ruts/machine tracks and life stock trails.

For large scale assessment of compaction as aimed at with SHERPA, water logging and signs of machinery tracks/ruts or livestock trails seem the best indicator to choose. Machinery tracks (also called ruts) are the result of vertical and horizontal soil displacement to either the middle or the sides of the skid trail associated with shearing stresses and soil compression in moist or wet soils (Horn et al., 2007). Beyond a critical water content, in fact, tyre or track forces cause soil displacement and rut formation rather than simple compaction. On flat terrain, ruts are collectors of rain or depressions where the water table surfaces, while on slopes they are preferential routes for runoff, which might increase with depth over time due to erosion. We argue that when porosity is reduced in such a way that water flow, oxygen saturation and water percolation is affected, negative soil health effects will definitely occur. In this case, assessing the degree of compaction (or the exact change in bulk and packing density) might be less important over assessing the duration the compaction and its effects on water and gas fluxes in the soil lasts (which is, of course related to the degree of compaction). As such, we propose to map the percent of the observed area affected by compaction as well as the duration of these effects (e.g. in yearly time steps). As observations of compaction result in a wide variation in the degree of compaction encountered in fields, both laterally and vertically (Batey, 2009), we suggest a systematic grid approach, where whenever signs of water logging after rain events and/ or machinery tracks/ life stock trails are visible, the assessed area will be counted as compacted. As such, the following tables (**Table 1.8.1, Table 1.8.2. and Table 1.8.3**) were developed.

1.8.1. Compaction of Forest Soils

Forest soil compaction and the alteration of ground morphology are crucial direct effects of forest harvesting carried out using heavy equipment (Cambi et al., 2015). As forest is often stocked on sloped soils, compaction, especially when confined in ruts, might have dramatic ramifications in terms of runoff and erosion not only of the most fertile soil compartment (i.e., the topsoil) but also resulting in deep soil incisions (Cambi et al., 2015). In compacted soils, forest regeneration can be impeded or even prevented for long time periods (Cambi et al., 2015). As in agricultural land, high-pressure tyres, high axle loads in combination with frequent machine passage will cause subsoil compaction (Cambi et al., 2015). However, in contrast to agricultural soils where surface soils might be ploughed, machine tracks will be seen permanently and might thus be a continued sign of surface and subsoil compaction.

Forest soils with a naturally lower bulk density compared to cropland and grassland use, are particularly prone to compaction. (Lacey and Ryan, 2000) suggest consideration of litter displacement and/or soil displacement as well as heaps formed from harvesting residues in addition to considering ruts/ machine tracks. As such, we separated the tables to assess soil compaction for forest versus grassland or cropland use.

Table 1.8.1. Forest soils – area (%) with signs of disturbance of mineral soil or humus layer due to machinery tracks/ruts and/or signs of water logging

Forest soils - area in spatial percent with signs of disturbance of mineral soil or humus layer due to machinery tracks/ruts and/or signs of water logging	**subtract scores**
No signs	0
≤ 5%	1
>5 - 10%	2
> 10 - 15%	3
>15 - 20%	4
>20 - 30%	5
>30 - 40%	6
>40 - 50%	7
>50 - 60%	8
> 60%	9

1.8.2. Compaction of cropland and grassland soils

The higher frequency of machine passage, lower soil organic matter, ploughing and destruction of soil structure often leads to subsoil compaction below the plough layer in cropland soils (Batey, 2009; Ferreira et al., 2022; Shaheb et al., 2021). Subsoil compaction is of serious concern in agricultural land as it hampers water percolation through the soil profile, restricts rooting depth as well as gas exchange within the soil and, in addition, may be very persistent or even permanent with very limited mitigation possibilities (Batey, 2009; Ferreira et al., 2022; Hamza and Anderson, 2005). As such, subsoil compaction below 40 cm depth has been suggested to cause persistent and permanent reductions in crop yield (Håkansson and Reeder, 1994; Hamza and Anderson, 2005). Especially if the subsoil provides a significant proportion of the water or nutrients required by crops to meet transpiration or nutritional demands, subsoil compaction might seriously affect crop production (Batey, 2009).Ideally, subsoil vulnerability to compaction should be assessed by direct measurement of soil bearing capacity but currently no direct practical tests are available (Jones et al., 2003).Subsoil compaction is not necessarily a phenomenon seen at the surface of the soils such as water logging or machine tracks and is, with today's tools, not detectable from remote. As such, European monitoring as a ground survey will be necessary for subsoil compaction to develop detailed maps. As compaction of the surface soil and subsoil compaction might or might not occur in parallel, add up in their detrimental effects, and need different measures for assessment, both phenomena will be assessed separately.

Table 1.8.2. Cropland soils and grasslands - area units with signs of water logging at the surface one day after rain events and/or ruts/machine tracks

. Cropland soils and grasslands - areal units with signs of water logging at the surface one day after rain events and/or ruts/machine tracks	**subtract scores**
no signs	0
≤ 5%	1
>5 - 10%	2
> 10 - 15%	3
>15 - 20%	4
>20 - 30%	5
>30 - 40%	6
>40 - 50%	7
>50 - 60%	8
more than 60%	9

Table 1.8.3. Cropland soils and grasslands showing signs of subsoil (> 30 cm depth) compaction assessed in a cross-sectional area of a representative soil profile

Cropland soils and grasslands showing signs of subsoil (> 30 cm depth) compaction assessed in a cross-sectional area of a representative soil profile	**subtract scores**
No signs	0
Bulk density below plough layer increasing but no indication of inducing any stagnic properties nor restriction of rooting depth	1
Bulk density below plough layer increasing with first signs of stagnic properties detectable but no indication of a restriction in rooting depth	2
Bulk density below plough layer increasing with clearly visible stagnic properties but no indication of a restriction in rooting depth	3
Bulk density below plough layer increasing with clearly visible stagnic properties and /or first indications of restrictions of rooting depth	4
Bulk density below plough layer increasing with indications of restrictions of rooting depth in 10 - 20% of the cross-sectional area	5
Bulk density below plough layer increasing with indications of restrictions of rooting depth in 21 - 30% of the cross-sectional area	6
Bulk density below plough layer increasing with indications of restrictions of rooting depth in 31 - 40% of the cross-sectional area	7
Bulk density below plough layer increasing with indications of restrictions of rooting depth in 41 - 50% of the cross-sectional area	8
Bulk density below plough layer increasing with indications of restrictions of rooting depth in > 50% of the cross-sectional area	9

The above scoring would be the best to follow according to soil science knowledge. However, data is not available on European scale. As such, for the current assessment in this study we followed an approach relying on how bulk densities dependent on specific soil texture classes will affect plant growth (Nyéki et al., 2017) quoted from (USDA, 1987). The bulk density data was obtained from (Panagos et al., 2024) and the soil texture classes from (Ballabio et al., 2016).

Table 1.8.4. Soil health scoring based on the general relationship of soil bulk density to root growth classified for different soil texture groups according to (USDA, 1987) quoted in (Nyéki et al., 2017).

Soil texture	Bulk densities suitable for plant growth (g/cm^3)	Bulk densities that affect root growth (g/cm^3)	Bulk densities that restrict root growth (g/cm^3)
Sands, loamy sands	<1.60	1.69	>1.80
Sandy loams, loams	<1.40	1.63	>1.80
Sandy clay loams, clay loams	<1.40	1.60	>1.75
Silts, silt loams	<1.40	1.60	>1.75
Silt loams, silty clay loams	<1.40	1.55	>1.65
Sandy clays, silty clays, clay loams	<1.10	1.49	>1.58
Clay (>45% clay)	<1.10	1.39	>1.47
Scores subtracted	0	3	9

1.9. Assessment of soil organic matter loss

Another parameter which is ubiquitous as an indicator in all soil health assessment studies is soil organic carbon (SOC). Soil organic matter is the primary food source for soil microorganisms, and soil organic matter has such a profound effect on numerous soil physical, chemical, and biological properties that organic matter management is crucial in maintaining soil health (Larkin, 2015). In addition to providing nutrients and energy for the soil biota, organic matter stabilizes soil structure and water relations (used here to refer to all aspects of water infiltration, storage, availability, etc.) and increases soil fertility (Larkin, 2015). Changes in SOC response to management, soil threats or land use are difficult not only to detect, because the soil pool is so large but even more so to predict or understand, as, e.g., increases in SOC can indicated a change towards healthier, nutrient enriched soils with an active soil community but might also indicate a decline in soil biodiversity and inhibition of degradation processes. The same holds true for soil organic matter fractions such us labile or active carbon, particulate organic matter, permanganate-oxidizable carbon, or dissolved organic carbon, all of which are more difficult to quantify but do not indicate clear relationships to certain soil processes or functions (Bünemann et al., 2018). Thus, as we do agree that soil organic carbon will always be an important indicator in any soil survey, soil monitoring or stakeholder assessment when expert judgement is involved in decision based iterative judgements, SHERPA does not include absolute values of SOC but rather decreasing trends over time. We do think that a steady, ongoing loss of SOC in grasslands or cropland soils is clearly pointing towards a loss in soil health. Thus, SHERPA does not consider absolute SOC or any of its fractions as an indicator, but the steady, ongoing loss or SOC in cropland soils. We used the data of (De Rosa et al., 2024) and based on their European maps and classification developed the following scoring (Table 1.9). Note that the soil loss data was available in gC kg^{-1}9yr^{-1}, and we used the bulk density from (Panagos et al., 2024) to convert it into tC ha^{-1}9yr^{-1}.

Table 1.9 Soil organic carbon (SOC) loss for the 2009–2018 period [t C ha^{-1}·yr^{-1}]

C loss [t C ha^{-1}·9yr^{-1}]	**Scores subtracted**
0 to positive (e.g., SOC accumulation)	0
< 0 to -0.5	1
< -0.5 to -1.5	2
< - 1.5 to -3	3
< -3 to -4	4
< -4 to -5	5
< - 5 to -6	6
< -6 to -7	7
< -7- to -8	8
< -8	9

Part 2: Definition of Soil Health based on emergent soil profile characteristics

In **Part 1**, depending on human induced soil degradation and/or negative anthropogenic impacts on soils, scores are attributed to be subtracted from the scores obtained here in assessing the intrinsic health of soils. This might result in overall negative values, depending on soil threats accumulating. Soil health index of **Part 2** is scaled from 1-10, with 1 being the lowest soil health, 10 being the healthiest soil.

Table 2.0. Land Use – First Level Identifier

0. Land use – First level identifier	**Go to**
Non-drained Forests and bushland	2.1
Non-drained Grassland including dwarf shrubs	2.2
Non-drained Cropland, orchards, vineyards, non-permanent grassland	2.3
Wetlands/organic soils[1] with organic carbon > 20% and any land drained independent of land use	n.c.*

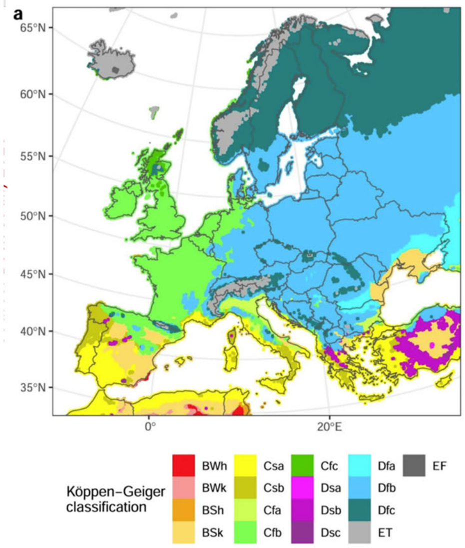
[1]according to (WRB, 2022): Organic soils = If OC>20 % and mineral soils = if OC<20 %.

* n.c. = not considered yet. Please note that even though we are currently developing a framework for wetland assessment and mapping, assessing wetlands were beyond the scope of this current manuscript

2.1. Intrinsic soil health assessment of forests

Soil health classification will follow a combination of climate classes (considering longitude and latitude, altitude, precipitation and aspect), nutrient and acidity status of the soil forming bedrock strata, humus layer development and possible disturbance. To develop the below tables we used the International Co-operative Program on Assessment and Monitoring of Air Pollution Effects on Forest (ICP forest; http://icp-forests.net/) data set (Ferretti and Fischer, 2013; Haußmann and Fischer, 2004; Puletti et al., 2019) to develop respective combinations of the above mentioned parameters to assign soil health scores. We argue that in a healthy forest ecosystem we can expect to find (i) either either humus layer mull (e.g., no organic humus layer on the surface) on nutrient rich soils of temperate or Mediterranean climate or (ii) if conditions progress to cooler and humid climates and/or nutrient poorer and/or more acidic conditions a fully developed humus layer covering the mineral soil. Considering all ruling out criteria of Part 1, we could not really think of situations of degraded forest soils if the humus layer is either mull on nutrient rich low altitude situations or fully developed with 100% aerial coverage of the underlying mineral soil in cool and humid and/or acid and/or high-altitude systems.

As the ICP forest data did not indicate principal differences between the classes Csa, Cfb, Dfc and Dfc of the Köppen-Geiger climate classification for Europe (Beck et al., 2020; Peel et al., 2007) regarding organic humus layer thickness or development, these classes were merged to one respective table. We separated ET (polar, tundra and alpine, **Figure 2.1**) as well as all ecosystems with altitudes above 1400 m and/or forests soils with sand content of ≥.85% from all other forest ecosystems. In these forest ecosystems, we expect humus forms other than mull, e.g. even in healthy forest ecosystems delayed degradation of litter and thus a closed organic humus layer as surface cover can be expected.



Source: Peel et al., 2007

Table 1. Description of Köppen climate symbols and defining criteria.

1st	2nd	3rd	Description	Criteria*
A			Tropical	$T_{cold} \geq 18$
	f		- Rainforest	$P_{dry} \geq 60$
	m		- Monsoon	Not (Af) & $P_{dry} \geq 100–MAP/25$
	w		- Savannah	Not (Af) & $P_{dry} < 100–MAP/25$
B			Arid	$MAP < 10 \times P_{threshold}$
	W		- Desert	$MAP < 5 \times P_{threshold}$
	S		- Steppe	$MAP \geq 5 \times P_{threshold}$
		h	- Hot	$MAT \geq 18$
		k	- Cold	$MAT < 18$
C			Temperate	$T_{hot} > 10$ & $0 < T_{cold} < 18$
	s		- Dry Summer	$P_{sdry} < 40$ & $P_{sdry} < P_{wwet}/3$
	w		- Dry Winter	$P_{wdry} < P_{swet}/10$
	f		- Without dry season	Not (Cs) or (Cw)
		a	- Hot Summer	$T_{hot} \geq 22$
		b	- Warm Summer	Not (a) & $T_{mon10} \geq 4$
		c	- Cold Summer	Not (a or b) & $1 \leq T_{mon10} < 4$
D			Cold	$T_{hot} > 10$ & $T_{cold} \leq 0$
	s		- Dry Summer	$P_{sdry} < 40$ & $P_{sdry} < P_{wwet}/3$
	w		- Dry Winter	$P_{wdry} < P_{swet}/10$
	f		- Without dry season	Not (Ds) or (Dw)
		a	- Hot Summer	$T_{hot} \geq 22$
		b	- Warm Summer	Not (a) & $T_{mon10} \geq 4$
		c	- Cold Summer	Not (a, b or d)
		d	- Very Cold Winter	Not (a or b) & $T_{cold} < -38$
E			Polar	$T_{hot} < 10$
	T		- Tundra	$T_{hot} > 0$
	F		- Frost	$T_{hot} \leq 0$

*MAP = mean annual precipitation, MAT = mean annual temperature, T_{hot} = temperature of the hottest month, T_{cold} = temperature of the coldest month, T_{mon10} = number of months where the temperature is above 10, P_{dry} = precipitation of the driest month, P_{sdry} = precipitation of the driest month in summer, P_{wdry} = precipitation of the driest month in winter, P_{swet} = precipitation of the wettest month in summer, P_{wwet} = precipitation of the wettest month in winter, $P_{threshold}$ = varies according to the following rules (if 70% of MAP occurs in winter then $P_{threshold}$ = 2 x MAT, if 70% of MAP occurs in summer then $P_{threshold}$ = 2 x MAT + 28, otherwise $P_{threshold}$ = 2 x MAT + 14). Summer (winter) is defined as the warmer (cooler) six month period of ONDJFM and AMJJAS.

Start with Csa temperate, dry summer, hot summer
Cfb temperate, without dry season, warm summer
Dfb cold, without dry season, warm summer
Dfc cold, without dry season, cold summer
ET polar, Tundra

Figure 2.1: Description of Köppen-Geiger climate classification for Europe

Underlying bedrock strata will be assigned according to region specific maps (using the ones with the best spatial resolution). Geological formations were translated into classes relating to their mineralogical nutrient and acidity content (see Supporting Information 2, Table S2.4.). Here, it is critical to have information on the top stratums, covering the source rock formations, as soils develop on these top stratums (e.g., Quaternary layers like aeolian, fluvial, glacial deposits). In many areas of Europe, this information might not be available yet, which might thus be an important demand to politics and regional and national authorities, to make this data available.

The humus layer has long been acknowledged as a fundamental component of numerous biological and physico-chemical processes that are vital for soil development and the functioning of terrestrial ecosystems within forest environments (Ponge et al., 2007 Consequently, it serves as a critical indicator of forest soil health. However, it is only recently that the concept emerged of humus forms as a digest of major processes which shape and stabilize ecosystems, pointing to the need for a better and worldwide assessment of diagnostic characters of humus forms (Ponge, 2003; Zanella et al., 2011). Even though the general concept and classification of three different humus layer types is generally agreed upon worldwide (Mull, Moder, Mor/Raw humus) many region and country specific differences and specialties exist in the exact classification (Zanella et al., 2011). Thus, we use occurrence and thickness of humus layer rather than humus forms as well as the general occurrence of a closed humus layer cover above the mineral soil as identifiers for soil health. In this, we relate to the data set of the ICP-Forest (Ferretti and Fischer, 2013; Haußmann and Fischer, 2004; Puletti et al., 2019) on calcareous or nutrient rich bedrock strata to delineate tables of soil health scores dependent on humus horizon thickness (Oi, Oe, Oa) stratified by climate, geology and altitude. On bedrock strata which are nutrient poor or even acid in weathering as well as on high altitude forest ecosystems, we rely on closed humus layer coverage of the soil surface to indicate a healthy forest ecosystem (please keep in mind that soil degradation threats like nitrogen surplus, compaction, pollution or erosion are considered in Part 1 as the ruling out criteria).

Plantation forests will be assessed with the same key as natural or semi-natural forests, as we can expect the same processes and ecosystem functions in all these forested systems. If plantation forests are in a transition phase from cropland or grassland to forest systems or are intensively managed, this indicates a lower ecosystem stability which will be considered by SHERPA as we do not expect a fully developed humus layer coverage and/or signs of compaction (tracks of heavy machinery). If pesticides are used or degradation due to compaction is noted this will be considered by the ruling out criteria of part 1 of SHERPA.

In summary, to assess forest soil health, the following tables of the key will ask for (i) climate classes according to Köppen - Geiger, (ii) altitude (separating the high-altitude mountain soils), (iii) soil forming bedrock layers considering mineralogy and acidity content and finally (iv) organic layer thickness and possible disturbance.

Table 2.1. Forest – climate classification according to Köppen-Geiger, altitude and soil texture?

Climate classification according to Köppen-Geiger, altitude and soil texture?	Go to
Csa (temperate, dry summer, hot summer), Cfb (temperate, without dry season, warm summer), Dfb (cold, without dry season, warm summer) or Dfc (cold, without dry season, cold summer) and sand content < 85%*	2.1.1.
ET polar, Tundra (and alpine) and/or altitude above 1400 m and/or sand content > 85%*	2.1.2.

**following WRB definition of soil texture sand (WRB, 2022)*

Table 2.1.1 Forest, climate temperate, dry summer, hot summer (Csa), temperate, without dry season, warm summer (Cfb), cold without dry season, warm or cold summer (Dfb or Dfc) – Geology[1] and altitude?

Forest, climate temperate, dry summer, hot summer (Csa), temperate, without dry season, warm summer (Cfb), cold without dry season, warm or cold summer (Dfb or Dfc) – Geology[1] and altitude	Go to
Calcareous bedrock or siliceous bedrock containing carbonates or siliceous base rich (e.g., Basalt, Opalinus clay) below 1400 m altitude	2.1.1.1
Siliceous medium base content (e.g. Granite) below 1400 m altitude	2.1.1.2

[1]for the defined geological classes see region specific tables in Appendix 2.1

Table 2.1.1.1. Forest, climate Csa, Cfb, Dfb or Dfc, Calcareous bedrock or siliceous bedrock containing carbonates or siliceous base rich bedrock below 1400 m altitude - organic layer thickness

Forest, climate Csa, Cfb, Dfb or Dfc, Calcareous bedrock or siliceous bedrock containing carbonates or siliceous base rich bedrock below 1400 m altitude – organic layer composition?	**Soil health**
Signs of disturbance of the mineral soil or of the closed coverage of the mineral soil by humus layer	go to 2.1.1.1A
Only litter layer[1] present, no developed O horizons2	10
Only litter layer[1] present, patches of Oi horizon developing	9
Oi ≤ 1 cm	8
Oi ≤ 1 cm and Oe ≤ 0.5 cm	7
Oi 1 -2 cm and Oe 0.5 - 1 cm	6
Oi 2 - 3 cm and Oe 1 - 2 cm	5
Oi 2 - 3 cm and Oe 1 - 2 cm and patches of Oa developing	4
Oi 2 - 3 cm and Oe 1 - 2 cm and Oa < 1 cm	3
Oi 2 - 5 cm and Oe 1 - 3 cm and Oa 1 -2 cm	2
Oi 2 - 5 cm and Oe 1 - 3 cm and Oa > 2 cm	1

[1]A litter layer is a loose layer that contains > 90% (by volume, related to the fine earth plus all dead plant remnants) recognizable dead plant tissues (e.g. undecomposed leaves)
[2]Oi - Organic material in an initial state of decomposition; after gently rubbing, > two thirds of the volume (related to the fine earth plus all dead plant remnants) consist of recognizable dead plant tissues
Oe – Organic material in an intermediate state of decomposition; after gently rubbing, ≤ two thirds and > one sixth of the volume (related to the fine earth plus all dead plant remnants) consist of recognizable dead plant tissues
Oa – Organic material in an advanced state of decomposition; after gently rubbing, ≤ one sixth of the volume (related to the fine earth plus all dead plant remnants) consists of recognizable dead plant tissues;

Table 2.1.1.1A. Forest, climate Csa, Cfb, Dfb or Dfc, Calcareous bedrock or siliceous bedrock containing carbonates or siliceous base rich bedrock below 1400 m altitude - with signs of disturbance of mineral

2.1.1.1A Forest, climate Csa, Cfb, Dfb or Dfc, Calcareous bedrock or siliceous bedrock containing carbonates or siliceous base rich bedrock below 1400 m altitude - with signs of disturbance of mineral soil or organic humus layer?	**Soil health**
Up to 10% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	8
> 10 - 15% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	7
> 15 - 20% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	6
> 20 – 30% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	5
> 30 - 40- % of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	4
> 40 - 50% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	3
> 50 - 60% of the area with signs of disturbance of the mineral soil or of a closed coverage of the mineral soil by organic humus layer*	2
> 60% of the area with signs of the mineral soil or of disturbance of a closed coverage of the mineral soil by organic humus layer*	1

** note that disturbance of humus layer and compaction of forest soil by machine tracks is not considered here, but in table 1.9 of part 1 of SHERPA. Here, the disturbance of the humus layer by overstocking with game, touristic use, construction (e.g., windfarming), snow processes or forms of erosion that are not considered in Table 1.1.1 (soil erosion) or Table 1.1.2. (landslide density).*

Table 2.1.1.2 Forest, climate Csa, Cfb, Dfb or Dfc, siliceous medium base bedrock below 1400 m altitude

Forest, climate Csa, Cfb, Dfb or Dfc, siliceous medium base bedrock below 1400 m altitude - organic layer composition?	**Soil health**
Oi ≤ 1 cm and no signs of disturbance of a closed coverage of the mineral soil by humus layer	10
Oi ≤ 1 cm and Oe ≤ 0.5 cm and no signs of disturbance of a closed coverage of the mineral soil by humus layer*	9
Oi 1 -2 cm and Oe 0.5 - 1 cm or less than 10% of the area with signs of disturbance a of closed coverage of the mineral soil by humus layer*	8
Oi 2 - 3 cm and Oe 1 - 2 cm or less than 15% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	7
Oi 2 - 3 cm and Oe 1 - 2 cm and patches of Oa developing or less than 20% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	6
Oi 2 - 3 cm and Oe 1 - 2 cm and Oa < 1 cm or less than 30% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	5
Oi 2 - 5 cm and Oe 1 - 3 cm and Oa 1 -2 cm or less than 40% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	4
Oi 2 - 5 cm and Oe 1 - 3 cm and Oa 2 - 3 cm or or less than 50% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	3
Oi 2 - 5 cm and Oe 2 - 5 cm and Oa 3 - 4 cm or or less than 60% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	2
Oi 2 - 5 cm and Oe 2 - 5 cm and Oa > 4 cm or more than 60% of the area with signs of disturbance of a closed coverage of the mineral soil by humus layer*	1

** note that disturbance of humus layer and compaction of forest soil by machine tracks is not considered here, but in table 1.9 of part 1 of SHERPA. Here, the disturbance of the humus layer by overstocking with game, touristic use, snow processes or forms of erosion that are not considered in Table 1.1.1 (soil erosion) or Table 1.1.2. (landslide density).*

Table 2.1.2. Forest, climate ET polar, Tundra (and alpine) and/or altitude above 1400 m and/or sand content > 85% - organic layer coverage of soil surface?

Forest, climate ET polar, Tundra (and alpine) and/or altitude above 1400 m and/or sand content > 85% - organic layer coverage of soil surface?	Soil health
100% soil surface covered with closed organic humus layer, no signs of disturbance	10
> 95% soil surface covered with closed organic humus layer, less than 5% of the area with signs of disturbance of the organic humus layer	9
90% soil surface covered with closed organic humus layer, less than 10% of the area with signs of disturbance of the organic humus layer	8
80% soil surface covered with closed organic humus layer, less than 20% of the area with signs of disturbance of the organic humus layer	7
70% soil surface covered with closed organic humus layer; less than 3% of the area with signs of disturbance of the organic humus layer	6
60% soil surface covered with closed organic humus layer, up to 40% of the area with signs of disturbance of the organic humus layer	5
50% soil surface covered with closed organic humus layer, up to 50% of the area with signs of disturbance of the organic humus layer	4
40% soil surface covered with closed organic humus layer, up to 60% of the area with signs of disturbance of the organic humus layer	3
30% soil surface covered with closed organic humus layer, up to 70% of the area with signs of disturbance of the organic humus layer	2
<30% soil surface covered with closed organic humus layer, more than 70% of the area with signs of disturbance of the organic humus layer	1

Due to the absence of data on a European scale regarding the disturbance of the organic humus layer in forests, we relied on the thickness of the organic humus layer only in this study. Consequently, SHERPA is likely to overestimate soil health in forests given the current state of data availability; however, this information will be critically needed in the future.

2.2. Intrinsic soil health assessment of grassland soils

We separate permanent from non-permanent grasslands defining permanent grasslands as all grasslands, which had continuous, stable grass cover throughout the last five years (Smit et al., 2008) or, in Mediterranean areas, throughout the last five winter and spring seasons. In contrast, non-permanent grasslands, when subjected to intensive use, are routinely ploughed under to remove unproductive grass species and herbs, typically every 3 to 4 years, depending on the region and local land management practices. Alternatively, these areas may be utilized in different ways and subsequently re-seeded as grasslands after a designated period. After tillage and, often, herbicide treatment (leaving sometimes the soil bare for a few months) highly productive grass species with low diversity (3-4 species) are re-sown to create a high productive pasture/ meadow for intensive production. These grasslands are characterized by a low biodiversity which is not connected to the site-specific characteristics. However, we cannot classify these systems *per se* rigorously as “unhealthy”, because the management aims and agricultural practices are parallel to cropland systems. We will thus classify non-permanent grasslands together with cropland soils (**Tables 2.4**.).

The rationale for permanent grasslands follows the concept that usually vegetation cover is a safe indicator for soil health of grassland soils (considering that any form of contamination including pesticide treatment is assessed in Part 1). Wherever we have degradation by livestock (trails as well as sheet erosion due to overgrazing), construction, land sliding, erosion, snow ablation or avalanche activity this will show up in a reduced vegetation cover. However, there are some exceptions (see Supplementary Information 2, Figure S2.3) like livestock resting places as well as places around farms, settlements with heavy manure or waste water input which might have high vegetation cover but not necessarily a diverse community typical for permanent grasslands. Instead, we will find monocultures or low-diverse cultures from e.g., Rumex spec., Epilobium spec. (see Figure S2.3, example of Rumex mats with some Epilobium monoculture at the edge, example from Val Piora Alpe, Ticino, Switzerland). This is mostly very local. Here, spectral indices could be developed reflecting the heterogeneity of grasslands. E.g., if we have dense Rumex communities (or, another example, Calamagrostis mats) with 1 to 2 species only, we should have a very homogenous spectral reflectance. However, as this is very local, often close to settlements or alpine huts, these areas will not be considered at the moment but will be left for future projects to be covered. In any case, they will be scored negatively in part 1 of SHERPA, as they are subject to high nitrogen and phosphorus surplus and prone to compaction.

Regarding species diversity, we expect that with permanent grassland development for more than 5 years, species composition will adapt to a) ecological zone parameters and b) land use. However, grasslands are managed ecosystems, like cropland soils. If we would subtract scores, just because the grassland is not natural any more, this would also mean, that agricultural use would by definition never be assessed as good soil health. Eventually, separate tables for natural grasslands could be developed, however, we do not really have many natural grasslands in Europe except the higher alps. The latter zones would be classified with high soil health score, already, except they have high rates of erosion.

In summary, the key for the grasslands soils follows the order of (i) asking for permanence of grassland over winter and spring in the Mediterranean and full year-round in all other climate zones for more than 5 years, and (ii) mapping fractional vegetation cover. Human induced degradation as compaction, nitrogen or phosphorus surplus, erosion and land sliding or pesticide input will be assessed in Part 1 of this key.

2.2. Grassland – permanence of grassland?	**Go to**
Grassland vegetation permanent all year round > 5 years **or** Mediterranean climate (Köppen-Geiger Class B (arid), Csa or Csb (temperate, dry summer, warm or hot summer)) with vegetation cover permanent during winter and spring > 5 years	2.2.1
Grassland non-permanent last five years (follow tables for cropland)	2.3.

2.2.1. Permanent Grassland- Vegetation cover (VC) by FVC* (= living vegetation)	**Soil health**
VC 100%	10
VC < 100 - 98%	9
VC < 98 - 95%	8
VC < 95- 90%	7
VC < 90 -80%	6
VC < 80 - 75%	5
VC < 75 - 70%	4
VC < 70 - 65%	3
VC < 65 – 60 %	2
VC < 60%	1

**Fractional vegetation cover (= living vegetation) will be assessed over the last 5 years for Dfb and Dfc climate classes as an average of 4 times per year seasonally distributed (e.g., four satellite images evaluated) or for climate classes Csa and Cfb (mediterranean grasslands) over 5 years or for 4 times per winter/spring season*

2.3. Intrinsic soil health assessment of cropland soils

As soil threats and soil degradation is considered in part 1 of SHERPA, we consider three main attributes as decisive for the intrinsic soil health of cropland soils: surface cover as fractional vegetation cover throughout the year (FVC), mineral versus organic fertilizer addition and soil structure. We realize that the first two attributes are driving factors of soil health, while the third, soil structure is an intrinsic soil property directly indicating the status of soil health.

Vegetation cover throughout the year is an important factor of soil health as it reduces erosion, conserves moisture, reduces temperature, intercepts rainfall and suppresses weed growth (Larkin, 2015). Furthermore, it provides habitat for soil organisms as living plants provide the most readily available food source for soil microbes in the rhizosphere, an area of concentrated microbial activity, which is the most active part of the soil ecosystem with readily available food and peak nutrient and water cycling (Larkin, 2015). Thus, growing plants throughout the year (long-season crops or multiple short season crops, rotations, cover crops) helps the soil-food web and helps cycle the nutrients that plants need to grow (Larkin, 2015).

It is generally discussed that soil management with organic fertilizer will intensify soil health, due to the positive relationship between organic fertilizer input, increase in soil organic matter and carbon sequestration, cation exchange capacity, pH and acid buffering capacity, decrease in bulk density, improvement of soil structure, water retention and infiltration, increase in permeability, fungal and bacterial diversity as well as microbial activity, plant nutrient supply, fruit quality and even suppression of plant pathogens and diseases (Hatano et al., 2024; Khasawneh and Othman, 2020; Larkin, 2015;

Lehmann et al., 2020; Rayne and Aula, 2020). While green manure has been discussed to be superior in its effects on soil health compared to animal products (Khasawneh and Othman, 2020), animal manure applications have also been concluded to be beneficial for all of the above discussed improvements (Larkin, 2015; Rayne and Aula, 2020). Conserving and/or maintaining existing soil organic matter levels needs regular additions of organic matter to replenish soil resources and improve soil health. Organic matter can be added through crop residues, rotations, and cover crops, as well as via off-field sources of organic amendments such as compost, manures, and mulches (Larkin, 2015). There has been some evidence, that the exclusive use of swine manure, even though having many beneficial effects, might have some negative effects like decreasing bulk density (Yost et al., 2022). However, we consider the overwhelming evidence of studies on positive effects of organic fertilizer crucial. An overapplication of organic fertilizer will, of course, have detrimental effects not only on the environment (e.g., nitrogen, phosphorus leaching) but are also an unappealing management form from a labor and cost perspective. However, the latter is not needed to be considered for soil health. And, as we consider nitrogen and phosphorus surplus in part 1 of SHERPA, here the percent of organic fertilizer of the total fertilizer input will be considered as a generally beneficial factor.

Management practices such as tillage versus conservational tillage (no till or stripping) is not considered in SHERPA, due to the contradictory effects on soil health. While tillage might increase soil erosion (which is considered in part 1), conservational tillage might be beneficial in increasing soil's penetration resistance, organic carbon content and biota biomass, but has also been shown to lead to higher compaction and sealing (also increasing erosion) and a greater number of root feeding nematodes (Khasawneh and Othman, 2020).

It might also seem surprising that soil texture is not considered in the below tables, especially as compaction and soil structure is, of course, strongly dependent on soil texture. However, as we strive for healthy soils, we need land use and management that is adapted to soil specific characteristics. As such, a soil which is texture dependent very prone to compaction and/or angular structure (e.g., clay rich) needs adjusted management practices which prevents compaction and promotes biological turnover supporting zoogenic soil structure (e.g., granular), otherwise it cannot be considered healthy. Please note that soil erosion, soil compaction (with consideration of soil texture), overfertilization and erosion as well as contamination with or application of pesticides is considered in part 1 as ruling out principles.

We realize that FVC, soil structure and the type of fertilizer all influence each other and are not independent of each other. As such, we do not take the sum of all three, but the mean of soil health scores developed from each of the attributes to come to a final soil health score for cropland soils. As such, (co-)dependencies in driving factors will be averaged out.

Thus, intrinsic soil health of cropland soils (SHS) will ideally be calculate as

$$SHS = Mean(SHS_{FVC} + SHS_{SoilStructure} + SHS_{Fertilizer}) \qquad \text{(Equation 2.3.1.)}$$

With SHS_{FVC} being the soil health score due to fractional vegetation throughout the year (defined in Table 2.3.1), $SHS_{SoilStructure}$ the soil health score due to soil structure (defined in Table 2.3.2.) and $SHS_{Fertilizer}$ the soil health score due to the fraction of organic fertilizer of total fertilizer input (Table 2.3.3.). If soil structure is missing we suggest to follow Equation 2.3.2. with the restriction that soil structure is an important indicator of intrinsic soil health and that the uncertainty in soil health assessment is likely increasing considerably:

$$SHS = Mean(SHS_{FVC} + SHS_{Fertilizer}) \qquad \text{(Equation 2.3.1.)}$$

To assess SHS_{FVC} we considered that up to two-month (2/12 = 17 % of the year) crops might be in a dry stage and thus will not be mapped as living vegetation (note that the FVC is only identifying living vegetation). We consider a full soil cover considering up to month of dry vegetation as maximum soil health score of 10 (meaning that the temporal FVC should be between 80-100%). We define the limit value to the lowest soil health score if the soil is left bare 5 or more months of the year (5/12 of the year = 42% plus a possible two month of dry vegetation would be 7/12 = roughly 60% of the time non or non-living vegetation, thus 40% of the time living vegetation.) The ranking of the soil health score classes in between are equally scaled.

Table 2.3.1 Soil health score of fractional vegetation cover of cropland soils, orchards, vineyards

2.3.1. Cropland Soil Vegetation cover (FVC* = living vegetation) as a spatial and temporal mean within 300 * 300 m pixel in 2018	**SHS_{FVC}**
FVC > 80 - 100%	10
FVC < 75 - 80%	9
FVC < 70 - 75%	8
FVC < 65 - 70%	7
FVC < 60 - 65%	6
FVC < 55 - 60%	5
FVC < 50- 55%	4
FVC < 45- 50%	3
FVC < 40 -45%	2
FVC < 40%	1

**Fractional vegetation cover (= living vegetation) was assessed in 2018 as an average of 36 images in 10 days interval per year seasonally distributed (e.g., proba V satellite images)*

Regarding soil structure, we consider zoogenic structures like granular an indicator of a healthy, biological active and divers soil. With subangular soil structure biological activity is clearly decreasing and drops to near zero in angular, blocky, cloddy or platy structured soils.

Table 2.3.2 Soil health score of structure of cropland soils, orchards, vineyards

2.3.2. Soil health score of soil structure in A horizons following (WRB, 2022) in cropland soils, orchards, vineyards	**$SHS_{SoilStructure}$**
Prevailing granular structure	10
Prevailing subangular structure	8
Prevailing angular or cloddy structure with aggregates <100 mm	5
Prevailing angular or cloddy structure with aggregates >100 mm, or columnar, or platy structure	3
Prevailing single grain or massive structure	1

Table 2.3.3 Soil health score type of fertilizer used in cropland soils, orchards, vineyards

2.3.3. Percent of organic fertilizer from total fertilizer input in cropland soils, orchards, vineyards (Average 2013-2019, (Tian et al., 2022))	**$SHS_{Fertlizer}$**
Organic fertilizer only*	10

95 - 100%	9
< 80 - 95%	8
< 65 - 80%	7
< 50 - 65%	6
< 35 – 50%	5
< 20 - 35%	4
< 10– 20%	3
> 0 – 10%	2
Mineral fertilizer only	1

We consider assessing the intrinsic soil health score in cropland soil following equation 2.3.1 including soil structure as the best and scientifically sound way to assess soil health in cropland soils. However, this information is not available yet on European scale. Thus, for the current study we followed a combination of vegetation cover throughout the year with the amount of organic versus mineral fertilizer input (Equation 2.3.2). We realize that this is most likely increasing the uncertainty of the soil health score in cropland soils, as we do not see the actual state of soil health as it is represented in the soil structure. But for the time being, we consider the following the best way to move forward.

References

Aber, J., McDowell, W., Nadelhoffer, K., Magill, A., Berntson, G., Kamakea, M., McNulty, S., Currie, W., Rustad, L. and Fernandez, I., 1998. Nitrogen Saturation in Temperate Forest Ecosystems: Hypotheses revisited. BioScience, 48(11): 921-934.

Achermann, B.B., R., 2002. Empirical Critical Loads for Nitrogen. Expert Workshop, Berne, 11-13 November 2002. Environ. Doc. No. 164.

Aherne, J., Henry, J. and Wolniewicz, M., 2017. Development of Critical Loads for Ireland: Simulating Impacts on Systems (SIOS). Environmental Protection Agency, Johnstown Castle, Ireland.

Albornoz, F., 2016. Crop responses to nitrogen overfertilization: A review. Scientia Horticulturae, 205: 79-83.

Albornoz, F., Fanelli, S. and Hallak, J., 2016. Survival in export markets. Journal of International Economics, 102(C): 262-281.

Alewell, C., Ringeval, B., Ballabio, C., Robinson, D.A., Panagos, P. and Borrelli, P., 2020. Global phosphorus shortage will be aggravated by soil erosion. Nature Communications, 11(1): 4546.

Altieri, M.A. and Nicholls, C.I., 2003. Soil fertility management and insect pests: harmonizing soil and plant health in agroecosystems. Soil and Tillage Research, 72(2): 203-211.

Arias-Estévez, M., López-Periago, E., Martínez-Carballo, E., Simal-Gándara, J., Mejuto, J.-C. and García-Río, L., 2008. The mobility and degradation of pesticides in soils and the pollution of groundwater resources. Agriculture, Ecosystems & Environment, 123(4): 247-260.

Arróniz-Crespo, M., Leake, J.R., Horton, P. and Phoenix, G.K., 2008. Bryophyte physiological responses to, and recovery from, long-term nitrogen deposition and phosphorus fertilisation in acidic grassland. New Phytologist, 180(4): 864-874.

Ballabio, C., Jiskra, M., Osterwalder, S., Borrelli, P., Montanarella, L. and Panagos, P., 2021. A spatial assessment of mercury content in the European Union topsoil. Science of The Total Environment, 769: 144755.

Ballabio, C., Jones, A. and Panagos, P., 2024. Cadmium in topsoils of the European Union – An analysis based on LUCAS topsoil database. Science of The Total Environment, 912: 168710.

Ballabio, C., Panagos, P., Lugato, E., Huang, J.-H., Orgiazzi, A., Jones, A., Fernández-Ugalde, O., Borrelli, P. and Montanarella, L., 2018. Copper distribution in European topsoils: An assessment based on LUCAS soil survey. Science of The Total Environment, 636: 282-298.
Ballabio, C., Panagos, P. and Monatanarella, L., 2016. Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma, 261: 110-123.
Bassin, J.P., Kleerebezem, R., Dezotti, M. and van Loosdrecht, M.C.M., 2012. Simultaneous nitrogen and phosphate removal in aerobic granular sludge reactors operated at different temperatures. Water Research, 46(12): 3805-3816.
Bassin, S., Volk, M. and Fuhrer, J., 2013. Species Composition of Subalpine Grassland is Sensitive to Nitrogen Deposition, but Not to Ozone, After Seven Years of Treatment. Ecosystems, 16: 1105-117.
Basto, S., Thompson, K., Phoenix, G., Sloan, V., Leake, J. and Rees, M., 2015. Long-term nitrogen deposition depletes grassland seed banks. Nature Communications, 6(1): 6185.
Batey, T., 2009. Soil compaction and soil management – a review. Soil Use and Management, 25(4): 335-345.
Batey, T. and McKenzie, D., 2006. Soil compaction: identification directly in the field. Soil Use and Management, 22(2): 123-131.
Batool, M., Sarrazin, F.J., Attinger, S., Basu, N.B., Van Meter, K. and Kumar, R., 2022. Long-term annual soil nitrogen surplus across Europe (1850–2019). Scientific Data, 9(1): 612.
Bazzoffi, P., 2009. Soil erosion tolerance and water runoff control: minimum environmental standards. Regional Environmental Change, 9(3): 169-179.
Beck, H.E., Zimmermann, N.E., McVicar, T.R., Vergopolan, N., Berg, A. and Wood, E.F., 2018. Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific Data, 5(1): 180214.
Beck, H.E., Zimmermann, N.E., McVicar, T.R., Vergopolan, N., Berg, A. and Wood, E.F., 2020. Publisher Correction: Present and future Köppen-Geiger climate classification maps at 1-km resolution. Sci Data, 7(1): 274.
Billen, G., Garnier, J. and Lassaletta, L., 2013. The nitrogen cascade from agricultural soils to the sea: Modelling nitrogen transfers at regional watershed and global scales. Philosophical transactions of the Royal Society of London. Series B, Biological sciences, 368: 20130123.
Blanke, V., Bassin, S., Volk, M. and Fuhrer, J., 2012. Nitrogen deposition effects on subalpine grassland: The role of nutrient limitations and changes in mycorrhizal abundance. Acta Oecologica, 45: 57-65.
Bobbink, R., 1991. Effects of nutrient enrichment in Dutch chalk grassland. . Journal of Applied Ecology, 28: 28-41.
Bobbink, R. and Hettelingh, J.-P., 2011. Review and revision of empirical critical loads and dose-response relationships. National Institute for Public Health and the Environment (RIVM). RIVM Report.
Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.-W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L. and De Vries, W., 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications, 20(1): 30-59.
Bobbink, R., Loran, C. and Tomassen, H., 2022. Review and revision of empirical critical loads of nitrogen for Europe.
Bonanomi, G., Caporaso, S. and Allegrezza, M., 2006. Short-term effects of nitrogen enrichment, litter removal and cutting on a Mediterranean grassland. Acta Oecologica, 30(3): 419-425.
Borrelli, P., Lugato, E., Montanarella, L. and Panagos, P., 2017. A New Assessment of Soil Loss Due to Wind Erosion in European Agricultural Soils Using a Quantitative Spatially Distributed Modelling Approach. Land Degradation & Development, 28(1): 335-344.

Borrelli, P., Panagos, P., Alewell, C., Ballabio, C., de Oliveira Fagundes, H., Haregeweyn, N., Lugato, E., Maerker, M., Poesen, J., Vanmaercke, M. and Robinson, D.A., 2023. Policy implications of multiple concurrent soil erosion processes in European farmland. Nature Sustainability, 6(1): 103-112.

Bourguet, D. and Guillemaud, T., 2016. The Hidden and External Costs of Pesticide Use. In: E. Lichtfouse (Editor), Sustainable Agriculture Reviews: Volume 19. Springer International Publishing, Cham, pp. 35-120.

Boutin, M., Corcket, E., Alard, D., Villar, L., Jiménez, J.-J., Blaix, C., Lemaire, C., Corriol, G., Lamaze, T. and Pornon, A., 2017. Nitrogen deposition and climate change have increased vascular plant species richness and altered the composition of grazed subalpine grasslands. Journal of Ecology, 105(5): 1199-1209.

Bouwman, A.F., Beusen, A.H.W. and Billen, G., 2009. Human alteration of the global nitrogen and phosphorus soil balances for the period 1970–2050. Global Biogeochemical Cycles, 23(4).

Braun, S., Cantaluppi, L. and Flückiger, W., 2005. Fine roots in stands of Fagus sylvatica and Picea abies along a gradient of soil acidification. Environ Pollut, 137(3): 574-9.

Braun, S., Thomas, V.F.D., Quiring, R. and Flückiger, W., 2010. Does nitrogen deposition increase forest production? The role of phosphorus. Environmental Pollution, 158(6): 2043-2052.

Brouwer, R.N.I.o.t.D.o.D.M.i.P., 1962. Nutritive Influences on the Distribution of Dry Matter in Plant. Netherlands Journal Agricultural Science, 10, 399-408. https://doi.org/10.18174/njas.v10i5.17581

Bundesministerium für Umwelt, N., nukleare Sicherheit und Verbraucherschutz, 2024. Bundes-Bodenschutz- und Altlastenverordnung.

Bünemann, E.K., Bongiorno, G., Bai, Z., Creamer, R.E., De Deyn, G., de Goede, R., Fleskens, L., Geissen, V., Kuyper, T.W., Mäder, P., Pulleman, M., Sukkel, W., van Groenigen, J.W. and Brussaard, L., 2018. Soil quality – A critical review. Soil Biology and Biochemistry, 120: 105-125.

Cambi, M., Certini, G., Neri, F. and Marchi, E., 2015. The impact of heavy traffic on forest soils: A review. Forest Ecology and Management, 338: 124-138.

Carpenter, S.R. and Bennett, E.M., 2011. Reconsideration of the planetary boundary for phosphorus. Environmental Research Letters, 6(1): 014009.

Carreiro, M., Sinsabaugh, R., Repert, D. and Parkhurst, D., 2000. Microbial enzyme shifts explain litter decay responses to simulated nitrogen deposition. Ecology, 81(9): 2359-2365.

Carroll, J.A., Johnson, D., Morecroft, M., Taylor, A., Caporn, S.J.M. and Lee, J.A., 2000. The effect of long-term nitrogen additions on the bryophyte cover of upland acidic grasslands. Journal of Bryology, 22(2): 83-89.

Carter, T.S., Clark, C.M., Fenn, M.E., Jovan, S., Perakis, S.S., Riddell, J., Schaberg, P.G., Greaver, T.L. and Hastings, M.G., 2017. Mechanisms of nitrogen deposition effects on temperate forest lichens and trees. Ecosphere, 8(3).

Cordell, D., Drangert, J.-O. and White, S., 2009. The story of phosphorus: Global food security and food for thought. Global Environmental Change, 19(2): 292-305.

Dahiya, S., Kumar, S., Chaudhary, C. and Chaudhary, C., 2018. Lodging: Significance and preventive measures for increasing crop production. Int. J. Chem. Stud, 6(2): 700-705.

Davis, S.J., hUallachain, D.O., Mellander, P.E., Matthaei, C.D., Piggott, J.J. and Kelly-Quinn, M., 2019. Chronic nutrient inputs affect stream macroinvertebrate communities more than acute inputs: An experiment manipulating phosphorus, nitrogen and sediment. Science of the Total Environment, 683: 9-20.

de Baan L., S.S., Daniel O. , 2009-2012. Einsatz von Pflanzenschutzmitteln in der Schweiz von 2009 bis 2012. Agrarforschung Schweiz.

De Schrijver, A., De Frenne, P., Ampoorter, E., Van Nevel, L., Demey, A., Wuyts, K. and Verheyen, K., 2011. Cumulative nitrogen input drives species loss in terrestrial ecosystems. Wiley Online Library, pp. 803-816.

de Souza, T.A.F. and Freitas, H., 2018. Long-Term Effects of Fertilization on Soil Organism Diversity. In: S. Gaba, B. Smith and E. Lichtfouse (Editors), Sustainable Agriculture Reviews 28: Ecology for Agriculture. Springer International Publishing, Cham, pp. 211-247.

de Vries, W., 2021. Impacts of nitrogen emissions on ecosystems and human health: A mini review. Current Opinion in Environmental Science & Health, 21: 100249.

de Vries, W., Dobbertin, M.H., Solberg, S., van Dobben, H.F. and Schaub, M., 2014. Impacts of acid deposition, ozone exposure and weather conditions on forest ecosystems in Europe: an overview. Plant and Soil, 380(1): 1-45.

de Witte, L.C., Rosenstock, N.P., van der Linde, S. and Braun, S., 2017. Nitrogen deposition changes ectomycorrhizal communities in Swiss beech forests. Science of The Total Environment, 605-606: 1083-1096.

De Rosa, D., Ballabio, C., Lugato, E., Fasiolo, M., Jones, A. and Panagos, P., 2024. Soil organic carbon stocks in European croplands and grasslands: How much have we lost in the past decade? Global Change Biology, 30(1): e16992.

Dereumeaux, C., Fillol, C., Quenel, P. and Denys, S., 2020. Pesticide exposures for residents living close to agricultural lands: A review. Environment International, 134: 105210.

Dise, N.B., Matzner, E. and Gundersen, P., 1998. Synthesis of Nitrogen Pools and Fluxes from European Forest Ecosystems. Water, Air, and Soil Pollution, 105(1): 143-154.

Dise, N.B. and Stevens, J., 2005. Nitrogen deposition and reduction of terrestrial biodiversity: evidence from temperate grasslands. Science in China Series C: Life Sciences, 48: 720-728.

Dise, N.B. and Wright, R.F., 1995. Nitrogen leaching from European forests in relation to nitrogen deposition. Forest Ecology and Management, 71(1): 153-161.

Doran, J.W. and Parkin, T.B., 1994. Defining and assessing soil quality. Defining soil quality for a sustainable environment. Proc. symposium, Minneapolis, MN, 1992: 3-21.

Duprè, C., Stevens, C.J., Ranke, T., Bleeker, A., Peppler-Lisbach, C., Gowing, D.J., Dise, N.B., Dorland, E., Bobbink, R. and Diekmann, M., 2010. Changes in species richness and composition in European acidic grasslands over the past 70 years: the contribution of cumulative atmospheric nitrogen deposition. Global Change Biology, 16(1): 344-357.

Edixhoven, J.D., Gupta, J. and Savenije, H.H.G., 2014. Recent revisions of phosphate rock reserves and resources: a critique. Earth Syst. Dynam., 5(2): 491-507.

Einarsson, R., Sanz-Cobena, A., Aguilera, E., Billen, G., Garnier, J., van Grinsven, H.J.M. and Lassaletta, L., 2021. Crop production and nitrogen use in European cropland and grassland 1961–2019. Scientific Data, 8(1): 288.

Eisenhauer, N., Cesarz, S., Koller, R., Worm, K. and Reich, P.B., 2012. Global change belowground: impacts of elevated , nitrogen, and summer drought on soil food webs and biodiversity. Global Change Biology, 18(2): 435-447.

Emmett, B.A., 2007. Nitrogen saturation of terrestrial ecosystems: some recent findings and their implications for our conceptual framework. Acid rain-deposition to recovery: 99-109.

ESDAC, 2024. EUSO dashboard sources of the European Soil Data Centre, European Commission. https://esdac.jrc.ec.europa.eu/esdacviewer/euso-dashboard/.

Etzold, S., Ferretti, M., Reinds, G.J., Solberg, S., Gessler, A., Waldner, P., Schaub, M., Simpson, D., Benham, S., Hansen, K., Ingerslev, M., Jonard, M., Karlsson, P.E., Lindroos, A.-J., Marchetto, A., Manninger, M., Meesenburg, H., Merilä, P., Nöjd, P., Rautio, P., Sanders, T.G.M., Seidling, W., Skudnik, M., Thimonier, A., Verstraeten, A., Vesterdal, L., Vejpustkova, M. and de Vries, W.,

2020. Nitrogen deposition is the most important environmental driver of growth of pure, even-aged and managed European forests. Forest Ecology and Management, 458: 117762.

EUR-Lex, 2024. Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. Consolidated 2022. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A01986L0278-20220101.

FAO, 2024. Food and Agriculture Organization of the United Nations; https://www.fao.org/about/about-fao/en/.

Fedlex, 2024. Ordinance on the Remediation of Polluted Sites of 26 August 1998 (Status as of 1 July 2024), 814.680, The Swiss Ferderal Council. https://www.fedlex.admin.ch/eli/cc/1998/2261_2261_2261/en.

Fenner, K., Canonica, S., Wackett, L.P. and Elsner, M., 2013. Evaluating Pesticide Degradation in the Environment: Blind Spots and Emerging Opportunities. Science, 341(6147): 752-758.

Ferreira, C.S.S., Seifollahi-Aghmiuni, S., Destouni, G., Ghajarnia, N. and Kalantari, Z., 2022. Soil degradation in the European Mediterranean region: Processes, status and consequences. Science of the Total Environment, 805.

Ferretti, M. and Fischer, R., 2013. Forest monitoring: methods for terrestrial investigations in Europe with an overview of North America and Asia.

Fitzpatrick, E.A., 1956. AN INDURATED SOIL HORIZON FORMED BY PERMAFROST. Journal of Soil Science, 7(2): 248-254.

Forsius, M., Posch, M., Holmberg, M., Vuorenmaa, J., Kleemola, S., Augustaitis, A., Beudert, B., Bochenek, W., Clarke, N., de Wit, H.A., Dirnböck, T., Frey, J., Grandin, U., Hakola, H., Kobler, J., Krám, P., Lindroos, A.-J., Löfgren, S., Pecka, T., Rönnback, P., Skotak, K., Szpikowski, J., Ukonmaanaho, L., Valinia, S. and Váňa, M., 2021. Assessing critical load exceedances and ecosystem impacts of anthropogenic nitrogen and sulphur deposition at unmanaged forested catchments in Europe. Science of The Total Environment, 753: 141791.

Fraters, D., van Leeuwen, T., Boumans, L. and Reijs, J., 2015. Use of long-term monitoring data to derive a relationship between nitrogen surplus and nitrate leaching for grassland and arable land on well-drained sandy soils in the Netherlands. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 65(sup2): 144-154.

Frey, S.D., Knorr, M., Parrent, J.L. and Simpson, R.T., 2004. Chronic nitrogen enrichment affects the structure and function of the soil microbial community in temperate hardwood and pine forests. Forest Ecology and Management, 196(1): 159-171.

Galon, L., Bragagnolo, L., Korf, E.P., dos Santos, J.B., Barroso, G.M. and Ribeiro, V.H.V., 2021. Mobility and environmental monitoring of pesticides in the atmosphere — a review. Environmental Science and Pollution Research, 28(25): 32236-32255.

Gaudnik, C., Corcket, E., Clement, B., Delmas, C., Gombert-Courvoisier, S., Muller, S., Stevens, C. and Alard, D., 2011. Detecting the footprint of changing atmospheric nitrogen deposition loads on acid grasslands in the context of climate change. Global Change Biology, 17: 3351-3365.

Gilbert, N., 2009. Environment: The disappearing nutrient. Nature, 461(7265): 716-718.

Giordani, P., Calatayud, V., Stofer, S., Seidling, W., Granke, O. and Fischer, R., 2014. Detecting the nitrogen critical loads on European forests by means of epiphytic lichens. A signal-to-noise evaluation. Forest Ecology and Management, 311: 29-40.

Goulding, K., 2000. Nitrate leaching from arable and horticultural land. Soil Use and Management, 16(s1): 145-151.

Gravesen, L., 2003. The Treatment Frequency Index: an indicator for pesticide use and dependency as well as overall load on the environment. Reducing pesticide dependency in Europe to protect health, environment and biodiversity.

Gundersen, A.G., 1995. The Environmental Promise of Democratic Deliberation

Gundersen, P., Schmidt, I.K. and Raulund-Rasmussen, K., 2006. Leaching of nitrate from temperate forests – effects of air pollution and forest management. Environmental Reviews, 14(1): 1-57.

Gunther, A., Hervas, J., Van Den Eeckhaut, M., Malet, J. and Reichenbach, P., 2014. Synoptic Pan-European Landslide Susceptibility Assessment: The ELSUS 1000 v1 Map. In: K. Sassa, P. Canuti and Y. Yin (Editors), Landslide Science for a Safer Geoenvironment. Springer, pp. p. 117-122.

Guo, M., 2021. Soil Health Assessment and Management: Recent Development in Science and Practices. Soil Systems, 5(4): 61.

Haddad, N.M., Haarstad, J. and Tilman, D., 2000. The effects of long-term nitrogen loading on grassland insect communities. Oecologia, 124: 73-84.

Håkansson, I. and Reeder, R.C., 1994. Subsoil compaction by vehicles with high axle load—extent, persistence and crop response. Soil and Tillage Research, 29(2): 277-304.

Hamza, M.A. and Anderson, W.K., 2005. Soil compaction in cropping systems: A review of the nature, causes and possible solutions. Soil and Tillage Research, 82(2): 121-145.

Harris, J.A., Evans, D.L. and Mooney, S.J., 2022. A new theory for soil health. European Journal of Soil Science, 73(4): e13292.

Hassani, A., Azapagic, A. and Shokri, N., 2020. Predicting long-term dynamics of soil salinity and sodicity on a global scale. Proceedings of the National Academy of Sciences, 117: 202013771.

Hassani, A., Azapagic, A. and Shokri, N., 2021. Global predictions of primary soil salinization under changing climate in the 21st century. Nature Communications, 12(1): 6663.

Hassani, A., Smith, P. and Shokri, N., 2024a. Negative correlation between soil salinity and soil organic carbon variability. Proceedings of the National Academy of Sciences of the United States of America, 121: e2317332121.

Hassani, F., Zhang, Y. and Kumar, S.V., 2024b. Improved representation of vegetation soil moisture coupling enhances soil moisture data assimilation in water-limited regimes: A Case Study over Texas. Water Resources Research, 60(6): e2023WR035558.

Hatano, R., Mukumbuta, I. and Shimizu, M., 2024. Soil Health Intensification through Strengthening Soil Structure Improves Soil Carbon Sequestration. Agriculture-Basel, 14(8).

Haußmann, T. and Fischer, R., 2004. The forest monitoring programme of ICP forests-a contribution to biodiversity monitoring.

Hautier, Y., Niklaus, P.A. and Hector, A., 2009. Competition for Light Causes Plant Biodiversity Loss After Eutrophication. Science, 324(5927): 636-638.

Hollis, J., Bricker, S., Čápová, D., Hinsby, K., Krenmayr, H.-G., Negrel, P., de Oliveira, D., Poyiadji, E., Gessel, S., Heteren, S. and Venvik, G., 2022. Pan-European geological data, information, and knowledge for a resilient, sustainable, and collaborative future. 53: 5-19.

Horn, R., Vossbrink, J., Peth, S. and Becker, S., 2007. Impact of modern forest vehicles on soil physical properties. Forest ecology and management, 248(1-2): 56-63.

Hornung, M., Sutton, M.A. and Wilson, R., 1995. Mapping and modelling of critical loads for nitrogen-a workshop report. NERC Institute of Terrestrial Ecology.

Horswill, P., O'Sullivan, O., Phoenix, G.K., Lee, J.A. and Leake, J.R., 2008. Base cation depletion, eutrophication and acidification of species-rich grasslands in response to long-term simulated nitrogen deposition. Environ Pollut, 155(2): 336-49.

Izquierdo, J.E., Houlton, B.Z. and van Huysen, T.L., 2013. Evidence for progressive phosphorus limitation over long-term ecosystem development: Examination of a biogeochemical paradigm. Plant and Soil, 367(1): 135-147.

Jenkinson, D.S., 2001. The impact of humans on the nitrogen cycle, with focus on temperate arable agriculture. Plant and Soil, 228(1): 3-15.

Johnson, D., Leake, J. and Lee, J., 1999. The effects of quantity and duration of simulated pollutant nitrogen deposition on root-surface phosphatase activities in calcareous and acid grasslands: a bioassay approach. New Phytologist, 141(3): 433-442.

Johnson, L.M., Harrison, J.H. and Riley, R.E., 1998. Estimation of the Flow of Microbial Nitrogen to the Duodenum Using Urinary Uric Acid or Allantoin. Journal of Dairy Science, 81(9): 2408-2420.

Jones, H.G., 2004. Irrigation scheduling: advantages and pitfalls of plant-based methods. Journal of experimental botany, 55(407): 2427-2436.

Jones, R.J.A., Spoor, G. and Thomasson, A.J., 2003. Vulnerability of subsoils in Europe to compaction: a preliminary analysis. Soil and Tillage Research, 73(1): 131-143.

Khasawneh, A.R. and Othman, Y.A., 2020. ORGANIC FARMING AND CONSERVATION TILLAGE INFLUENCED SOIL HEALTH COMPONENT. Fresenius Environmental Bulletin, 29(2): 895-902.

Kim, K.-H., Kabir, E. and Jahan, S.A., 2017. Exposure to pesticides and the associated human health effects. Science of The Total Environment, 575: 525-535.

Knorr, M., Frey, S.D. and Curtis, P.S., 2005. NITROGEN ADDITIONS AND LITTER DECOMPOSITION: A META-ANALYSIS. Ecology, 86(12): 3252-3257.

Köhler, K., Duijnisveld, W. and Böttcher, J., 2006. Nitrogen fertilization and nitrate leaching into groundwater on arable sandy soils. Journal of Plant Nutrition and Soil Science, 169: 185-195.

Kovach, J., Petzoldt, C., Degni, J. and Tette, J.P., 1992. A Method to Measure the Environmental Impact of Pesticides.

Kudsk, P., Jørgensen, L.N. and Ørum, J.E., 2018. Pesticide Load—A new Danish pesticide risk indicator with multiple applications. Land Use Policy, 70: 384-393.

Labelle, E.R., Hansson, L., Högbom, L., Jourgholami, M. and Laschi, A., 2022. Strategies to Mitigate the Effects of Soil Physical Disturbances Caused by Forest Machinery: a Comprehensive Review. Current Forestry Reports, 8(1): 20-37.

Lacey, S.T. and Ryan, P.J., 2000. Cumulative management impacts on soil physical properties and early growth of Pinus radiata. Forest Ecology and Management, 138(1): 321-333.

Langusch, J.J. and Matzner, E., 2002. Long-term modelling of nitrogen turnover and critical loads in a forested catchment using the INCA model. Hydrol. Earth Syst. Sci., 6(3): 395-402.

Larkin, R.P., 2015. Soil Health Paradigms and Implications for Disease Management. In: N.K. VanAlfen (Editor), Annual Review of Phytopathology, Vol 53. Annual Review of Phytopathology, pp. 199-221.

Latterini, F., Mederski, P.S., Jaeger, D., Venanzi, R., Tavankar, F. and Picchio, R., 2023. The Influence of Various Silvicultural Treatments and Forest Operations on Tree Species Biodiversity. Current Forestry Reports, 9(2): 59-71.

Lee, R., den Uyl, R. and Runhaar, H., 2019. Assessment of policy instruments for pesticide use reduction in Europe; Learning from a systematic literature review. Crop Protection, 126: 104929.

Lehmann, J., Bossio, D.A., Kögel-Knabner, I. and Rillig, M.C., 2020. The concept and future prospects of soil health. Nature Reviews Earth & Environment, 1(10): 544-553.

Lewis, K., Rainford, J., Tzilivakis, J. and Garthwaite, D., 2021. Application of the Danish pesticide load indicator to arable agriculture in the United Kingdom. Journal of Environmental Quality, 50(5): 1110-1122.

Lewis, K.A., Tzilivakis, J., Warner, D.J. and Green, A., 2016. An international database for pesticide risk assessments and management. Human and Ecological Risk Assessment: An International Journal, 22(4): 1050-1064.

Lillak, R., 2005. Integrating efficient grassland farming and biodiversity: proceedings of the 13th International Occasional Symposium of the European Grassland Federation. Tartu, Estonia, 29 - 31 August 2005. Tartu: European Grassland Federation (Grassland science in Europe, 10).

Lind, L., Hasselquist, E.M. and Laudon, H., 2019. Towards ecologically functional riparian zones: A meta-analysis to develop guidelines for protecting ecosystem functions and biodiversity in agricultural landscapes. Journal of Environmental Management, 249.

Lu, X., Mo, J. and Shaofeng, D., 2008. Effects of nitrogen deposition on forest biodiversity: A review. Acta Ecologica Sinica, 28: 5532-5548.

Maas, E.V. and Grattan, S., 1999. Crop yields as affected by salinity, pp. 55-108.

Maas, E.V. and Hoffman, G.J., 1977. Crop salt tolerance—Current assessment. ASCE J. Irrig. Drain. Div., 103, 115–134.

MASKELL, L.C., SMART, S.M., BULLOCK, J.M., THOMPSON, K. and STEVENS, C.J., 2010. Nitrogen deposition causes widespread loss of species richness in British habitats. Global Change Biology, 16(2): 671-679.

Möhring, N., Kudsk, P., Jørgensen, L.N., Ørum, J.E. and Finger, R., 2021. An R package to calculate potential environmental and human health risks from pesticide applications using the 'Pesticide Load' indicator applied in Denmark. Computers and Electronics in Agriculture, 191: 106498.

Mottes, C., Lesueur-Jannoyer, M., Le Bail, M. and Malézieux, E., 2014. Pesticide transfer models in crop and watershed systems: a review. Agronomy for Sustainable Development, 34(1): 229-250.

Mullins, C.E., MacLeod, D.A., Northcote, K.H., Tisdall, J.M. and Young, I.M., 1990. Hardsetting Soils: Behavior, Occurrence, and Management. In: R. Lal and B.A. Stewart (Editors), Advances in Soil Science: Soil Degradation Volume 11. Springer New York, New York, NY, pp. 37-108.

Muntwyler, A., Panagos, P., Pfister, S. and Lugato, E., 2024. Assessing the phosphorus cycle in European agricultural soils: Looking beyond current national phosphorus budgets. Science of The Total Environment, 906: 167143.

Nabel, M., Selig, C., Gundlach, J., von Der Decken, H. and Klein, M., 2021. Biodiversity in agricultural used soils: Threats and options for its conservation in Germany and Europe. Soil Organisms, 93(1): 1-11.

Needham, P., Scholz, G. and Moore, G., 2004. Physical restrictions to root growth. In: Soil guide: a handbook for understanding and managing agricultural soils (ed. G. Moore), Dept of Agriculture, Western Australia. Bulletin No. 4343: pp. 109–124. .

Nellemann, C. and Thomsen, M.G., 2001. Long-Term Changes in Forest Growth: Potential Effects of Nitrogen Deposition and Acidification. Water, Air, and Soil Pollution, 128(3): 197-205.

Nordin, A., Strengbom, J., Witzell, J., Näsholm, T. and Ericson, L., 2005. Nitrogen Deposition and the Biodiversity of Boreal Forests: Implications for the Nitrogen Critical Load. AMBIO: A Journal of the Human Environment, 34(1): 20-24, 5.

Nyéki, A., Milics, G., Kovács, A.J. and Neményi, M., 2017. Effects of Soil Compaction on Cereal Yield. Cereal Research Communications, 45(1): 1-22.

Oenema, O., 2006. Nitrogen budgets and losses in livestock systems. International Congress Series, 1293: 262-271.

Oenema O, B.F., Lammel J, Bascou P, Billen G, Dobermann A, Erisman JW, Garnett T,, Genovese G, H.T., Hillier J, Hoxha A, Lassaletta L, Jensen LS, Olazabal C, Oleszek W, Pallière C, and Powlson D, Q.M., Schulman M, Sutton MA, Van Grinsven HJM, Vis JK, Winiwarter W., 2016. Nitrogen Use Efficiency (NUE) - EU Nitrogen Expert Panel, Nitrogen Use Efficiency (NUE) – Guidance document for assessing NUE at farm level. Wageningen University, Alterra, PO Box 47, NL-6700 Wageningen, Netherlands.

Onipchenko, V.G., Makarov, M.I., Akhmetzhanova, A.A., Soudzilovskaia, N.A., Aibazova, F.U., Elkanova, M.K., Stogova, A.V. and Cornelissen, J.H.C., 2012. Alpine plant functional group responses to fertiliser addition depend on abiotic regime and community composition. Plant and Soil, 357(1): 103-115.

Panagos, P., Ballabio, C., Poesen, J., Lugato, E., Scarpa, S., Montanarella, L. and Borrelli, P., 2020. A Soil Erosion Indicator for Supporting Agricultural, Environmental and Climate Policies in the European Union. Remote Sensing, 12(9): 1365.

Panagos, P., Borrelli, P. and Poesen, J., 2019. Soil loss due to crop harvesting in the European Union: A first estimation of an underrated geomorphic process. Science of The Total Environment, 664: 487-498.

Panagos, P., De Rosa, D., Liakos, L., Labouyrie, M., Borrelli, P. and Ballabio, C., 2024. Soil bulk density assessment in Europe. Agriculture, Ecosystems & Environment, 364: 108907.

Panagos, P., Köningner, J., Ballabio, C., Liakos, L., Muntwyler, A., Borrelli, P. and Lugato, E., 2022. Improving the phosphorus budget of European agricultural soils. Science of The Total Environment, 853: 158706.

Pant, H.K., 2020. Estimation of Internal Loading of Phosphorus in Freshwater Wetlands. Current Pollution Reports, 6(1): 28-35.

Payne, R.J., Campbell, C., Britton, A.J., Mitchell, R.J., Pakeman, R.J., Jones, L., Ross, L.C., Stevens, C.J., Field, C., Caporn, S.J.M., Carroll, J., Edmondson, J.L., Carnell, E.J., Tomlinson, S., Dore, A.J., Dise, N. and Dragosits, U., 2019. What is the most ecologically-meaningful metric of nitrogen deposition? Environmental Pollution, 247: 319-331.

Payne, R.J., Dise, N.B., Stevens, C.J., Gowing, D.J., partners, B., Duprè, C., Dorland, E., Gaudnik, C., Bleeker, A. and Diekmann, M., 2013. Impact of nitrogen deposition at the species level. Proceedings of the National Academy of Sciences, 110(3): 984-987.

Paytan, A. and McLaughlin, K., 2007. The oceanic phosphorus cycle. Chemical Reviews, 107(2): 563-576.

Peel, M.C., Finlayson, B.L. and McMahon, T.A., 2007. Updated world map of the Köppen-Geiger climate classification. Hydrol. Earth Syst. Sci., 11(5): 1633-1644.

Peñuelas, J., Sardans, J., Rivas-ubach, A. and Janssens, I.A., 2012. The human-induced imbalance between C, N and P in Earth's life system. Global Change Biology, 18(1): 3-6.

Phoenix, G.K., Booth, R.E., Leake, J.R., Read, D.J., Grime, J.P. and Lee, J.A., 2003. Effects of enhanced nitrogen deposition and phosphorus limitation on nitrogen budgets of semi-natural grasslands. Global Change Biology, 9(9): 1309-1321.

Phoenix, G.K., Emmett, B.A., Britton, A.J., Caporn, S.J.M., Dise, N.B., Helliwell, R., Jones, L., Leake, J.R., Leith, I.D., Sheppard, L.J., Sowerby, A., Pilkington, M.G., Rowe, E.C., Ashmore, M.R. and Power, S.A., 2012. Impacts of atmospheric nitrogen deposition: responses of multiple plant and soil parameters across contrasting ecosystems in long-term field experiments. Global Change Biology, 18(4): 1197-1215.

Pitcairn, C.E.R., Leith, I.D., Sheppard, L.J., Sutton, M.A., Fowler, D., Munro, R.C., Tang, S. and Wilson, D., 1998. The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. Environmental Pollution, 102(1, Supplement 1): 41-48.

Ponge, J.-F., 2003. Humus forms in terrestrial ecosystems: a framework to biodiversity. Soil Biology and Biochemistry, 35(7): 935-945.

Pozzer, A., Tsimpidi, A.P., Karydis, V.A., de Meij, A. and Lelieveld, J., 2017. Impact of agricultural emission reductions on fine-particulate matter and public health. Atmos. Chem. Phys., 17(20): 12813-12826.

Puletti, N., Canullo, R., Mattioli, W., Gawryś, R., Corona, P. and Czerepko, J., 2019. A dataset of forest volume deadwood estimates for Europe. Annals of Forest Science, 76(3): 68.

Rayne, N. and Aula, L., 2020. Livestock Manure and the Impacts on Soil Health: A Review. Soil Systems, 4(4).

Reus, J.A.W.A. and Leendertse, P.C., 2000. The environmental yardstick for pesticides: a practical indicator used in the Netherlands. Crop Protection, 19(8): 637-641.

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S.E., Donges, J.F., Drüke, M., Fetzer, I., Bala, G., von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., Huiskamp, W., Kummu, M., Mohan, C., Nogués-Bravo, D., Petri, S., Porkka, M., Rahmstorf, S., Schaphoff, S., Thonicke, K., Tobian, A., Virkki, V., Wang-Erlandsson, L., Weber, L. and Rockström, J., 2023. Earth beyond six of nine planetary boundaries. Science Advances, 9(37): eadh2458.

Riedo, J., Herzog, C., Banerjee, S., Fenner, K., Walder, F., van der Heijden, M.G.A. and Bucheli, T.D., 2022. Concerted Evaluation of Pesticides in Soils of Extensive Grassland Sites and Organic and Conventional Vegetable Fields Facilitates the Identification of Major Input Processes. Environmental Science & Technology, 56(19): 13686-13695.

Rihm, B. and Achermann, B., 2016. **Critical Loads of Nitrogen and their Exceedances - Swiss contribution to the effects-oriented work under the Convention on Long-range Transboundary Air Pollution (UNECE).** . 2016 Federal Office for the Environment (FOEN), Bern, Switzerland.

Rihm, B. and Kurz, D., 2001. Deposition and critical loads of nitrogen in Switzerland, Acid rain 2000: Proceedings from the 6 th International Conference on Acidic Deposition: Looking back to the past and thinking of the future Tsukuba, Japan, 10–16 December 2000 Volume III/III Conference Statement Plenary and Keynote Papers. Springer, pp. 1223-1228.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P. and Foley, J.A., 2009. A safe operating space for humanity. Nature, 461(7263): 472-475.

Roth, R.T., Ruffatti, M.D., O'Rourke, P.D. and Armstrong, S.D., 2018. A cost analysis approach to valuing cover crop environmental and nitrogen cycling benefits: A central Illinois on farm case study. Agricultural Systems, 159: 69-77.

Roth, T., Kohli, L., Rihm, B. and Achermann, B., 2013. Nitrogen deposition is negatively related to species richness and species composition of vascular plants and bryophytes in Swiss mountain grassland. Agriculture, Ecosystems & Environment, 178: 121-126.

Rothwell, J.J., Futter, M.N. and Dise, N.B., 2008. A classification and regression tree model of controls on dissolved inorganic nitrogen leaching from European forests. Environ Pollut, 156(2): 544-52.

Sánchez-Bayo, F. and Wyckhuys, K.A.G., 2019. Worldwide decline of the entomofauna: A review of its drivers. Biological Conservation, 232: 8-27.

Savary, S., Willocquet, L., Pethybridge, S.J., Esker, P., McRoberts, N. and Nelson, A., 2019. The global burden of pathogens and pests on major food crops. Nature Ecology & Evolution, 3(3): 430-439.

Shaheb, M.R., Venkatesh, R. and Shearer, S.A., 2021. A Review on the Effect of Soil Compaction and its Management for Sustainable Crop Production. Journal of Biosystems Engineering, 46(4): 417-439.

Simmelsgaard, S.E. and Djurhuus, J., 1998. An empirical model for estimating nitrate leaching as affected by crop type and the long-term N fertilizer rate. Soil Use and Management, 14(1): 37-43.

Sinsabaugh, R.L., Carreiro, M.M. and Repert, D.A., 2002. Allocation of extracellular enzymatic activity in relation to litter composition, N deposition, and mass loss. Biogeochemistry, 60(1): 1-24.

Smit, H.J., Metzger, M.J. and Ewert, F., 2008. Spatial distribution of grassland productivity and land use in Europe. Agricultural Systems, 98(3): 208-219.

Sohrt, J., Lang, F. and Weiler, M., 2017. Quantifying components of the phosphorus cycle in temperate forests. Wiley Interdisciplinary Reviews-Water, 4(6).

Spoor, G., 2006. Alleviation of soil compaction: requirements, equipment and techniques. Soil Use and Management, 22(2): 113-122.

Stenrød, M., Heggen, H.E., Bolli, R.I. and Eklo, O.M., 2008. Testing and comparison of three pesticide risk indicator models under Norwegian conditions—A case study in the Skuterud and Heiabekken catchments. Agriculture, Ecosystems & Environment, 123(1): 15-29.

Stevens, C., David, T. and Storkey, J., 2018. Atmospheric nitrogen deposition in terrestrial ecosystems: Its impact on plant communities and consequences across trophic levels. Functional Ecology, 32.

Stevens, C.J., Dise, N.B., Mountford, J.O. and Gowing, D.J., 2004. Impact of nitrogen deposition on the species richness of grasslands. Science, 303(5665): 1876-1879.

Stevens, C.J., Duprè, C., Dorland, E., Gaudnik, C., Gowing, D.J., Bleeker, A., Diekmann, M., Alard, D., Bobbink, R. and Fowler, D., 2010. Nitrogen deposition threatens species richness of grasslands across Europe. Environmental pollution, 158(9): 2940-2945.

Stevens, C.J., Duprè, C., Dorland, E., Gaudnik, C., Gowing, D.J.G., Bleeker, A., Diekmann, M., Alard, D., Bobbink, R., Fowler, D., Corcket, E., Mountford, J.O., Vandvik, V., Aarrestad, P.A., Muller, S. and Dise, N.B., 2011. The impact of nitrogen deposition on acid grasslands in the Atlantic region of Europe. Environmental Pollution, 159(10): 2243-2250.

Stiles, W.A.V., Rowe, E.C. and Dennis, P., 2017. Long-term nitrogen and phosphorus enrichment alters vegetation species composition and reduces carbon storage in upland soil. Science of The Total Environment, 593-594: 688-694.

Strassemeyer, J., Daehmlow, D., Dominic, A.R., Lorenz, S. and Golla, B., 2017. SYNOPS-WEB, an online tool for environmental risk assessment to evaluate pesticide strategies on field level. Crop Protection, 97: 28-44.

Suz, L.M., Bidartondo, M.I., van der Linde, S. and Kuyper, T.W., 2021. Ectomycorrhizas and tipping points in forest ecosystems. New Phytologist, 231(5): 1700-1707.

Talhelm, A.F., Burton, A.J., Pregitzer, K.S. and Campione, M.A., 2013. Chronic nitrogen deposition reduces the abundance of dominant forest understory and groundcover species. Forest Ecology and Management, 293: 39-48.

Tang, F.H.M., Lenzen, M., McBratney, A. and Maggi, F., 2021. Risk of pesticide pollution at the global scale. Nature Geoscience, 14(4): 206-210.

Thimonier, A., Graf Pannatier, E., Schmitt, M., Waldner, P., Walthert, L., Schleppi, P., Dobbertin, M. and Kräuchi, N., 2010. Does exceeding the critical loads for nitrogen alter nitrate leaching, the nutrient status of trees and their crown condition at Swiss Long-term Forest Ecosystem Research (LWF) sites? European Journal of Forest Research, 129(3): 443-461.

Tian, H., Bian, Z., Shi, H., Qin, X., Pan, N., Lu, C., Pan, S., Tubiello, F.N., Chang, J., Conchedda, G., Liu, J., Mueller, N., Nishina, K., Xu, R., Yang, J., You, L. and Zhang, B., 2022. History of anthropogenic Nitrogen inputs (HaNi) to the terrestrial biosphere: a 5 arcmin resolution annual dataset from 1860 to 2019. Earth Syst. Sci. Data, 14(10): 4551-4568.

Tipping, E., Henrys, P., Maskell, L. and Smart, S., 2013. Nitrogen deposition effects on plant species diversity; threshold loads from field data. Environmental Pollution, 179: 218-223.

Tóth, G., Hermann, T., Szatmári, G. and Pásztor, L., 2016. Maps of heavy metals in the soils of the European Union and proposed priority areas for detailed assessment. Science of The Total Environment, 565: 1054-1062.

Treseder, K.K., 2008. Nitrogen additions and microbial biomass: a meta-analysis of ecosystem studies. Ecol Lett, 11(10): 1111-20.

United Nations, 2015. Updated Handbook for the 1979 Convention on Long-range Transboundary Air Pollution and its Protocols. UN. Available at: . https://doi.org/10.18356/4c3fd450-en.

USDA, 1987. Soil Mechanics Level I. Module 3 – USDA Textural Soil Classification. Study Guide. USDA, Soil Conservation Service. Stillwater, OK, USA.

VAN DEN BERG, L.J.L., VERGEER, P., RICH, T.C.G., SMART, S.M., GUEST, D. and ASHMORE, M.R., 2011. Direct and indirect effects of nitrogen deposition on species composition change in calcareous grasslands. Global Change Biology, 17(5): 1871-1883.
van der Putten, W.H., Bardgett, R.D., Farfan, M., Montanarella, L., Six, J. and Wall, D.H., 2023. Soil biodiversity needs policy without borders. Science, 379(6627): 32-34.
Van Eynde, E., Fendrich, A.N., Ballabio, C. and Panagos, P., 2023. Spatial assessment of topsoil zinc concentrations in Europe. Science of The Total Environment, 892: 164512.
Van Kauwenbergh, S.J., 2010. World Phosphate Rock Reserves and Resources. Technical Bulletin IFDC-T-75.
Velthof, G.L., Oudendag, D., Witzke, H.P., Asman, W.A.H., Klimont, Z. and Oenema, O., 2009. Integrated Assessment of Nitrogen Losses from Agriculture in EU-27 using MITERRA-EUROPE. Journal of Environmental Quality, 38(2): 402-417.
Verheijen, F.G.A., Jones, R.J.A., Rickson, R.J. and Smith, C.J., 2009. Tolerable versus actual soil erosion rates in Europe. Earth-Science Reviews, 94(1): 23-38.
Waldner, P., Thimonier, A., Graf Pannatier, E., Etzold, S., Schmitt, M., Marchetto, A., Rautio, P., Derome, K., Nieminen, T.M., Nevalainen, S., Lindroos, A.-J., Merilä, P., Kindermann, G., Neumann, M., Cools, N., de Vos, B., Roskams, P., Verstraeten, A., Hansen, K., Pihl Karlsson, G., Dietrich, H.-P., Raspe, S., Fischer, R., Lorenz, M., Iost, S., Granke, O., Sanders, T.G.M., Michel, A., Nagel, H.-D., Scheuschner, T., Simončič, P., von Wilpert, K., Meesenburg, H., Fleck, S., Benham, S., Vanguelova, E., Clarke, N., Ingerslev, M., Vesterdal, L., Gundersen, P., Stupak, I., Jonard, M., Potočić, N. and Minaya, M., 2015. Exceedance of critical loads and of critical limits impacts tree nutrition across Europe. Annals of Forest Science, 72(7): 929-939.
Waldrop, M.P., Zak, D.R., Sinsabaugh, R.L., Gallo, M. and Lauber, C., 2004. NITROGEN DEPOSITION MODIFIES SOIL CARBON STORAGE THROUGH CHANGES IN MICROBIAL ENZYMATIC ACTIVITY. Ecological Applications, 14(4): 1172-1177.
Walker, T.W. and Syers, J.K., 1976. The fate of phosphorus during pedogenesis. Geoderma, 15(1): 1-19.
Webb, J., Harrison, R. and Ellis, S., 2000. Nitrogen fluxes in three arable soils in the UK. European Journal of Agronomy, 13(2): 207-223.
Wedin, D.A. and Tilman, D., 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. Science, 274(5293): 1720-1723.
West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K. and Siebert, S., 2014. Leverage points for improving global food security and the environment. Science, 345(6194): 325-328.
Wilkins, K., Aherne, J. and Bleasdale, A., 2016. Vegetation community change points suggest that critical loads of nutrient nitrogen may be too high. Atmospheric Environment, 146: 324-331.
Wilson, E., Wells, T. and Sparks, T., 1995. Are calcareous grasslands in the UK under threat from nitrogen deposition?--an experimental determination of a critical load. Journal of ecology: 823-832.
Withers, P.J.A. and Jarvie, H.P., 2008. Delivery and cycling of phosphorus in rivers: A review. Science of the Total Environment, 400(1-3): 379-395.
WRB, 2022. World Reference Base for Soil Resources. International soil classification system for naming soils and creating legends for soil maps. 4th edition. International Union of Soil Sciences (IUSS), Vienna, Austria.
Xie, D., Duan, L., Du, E. and de Vries, W., 2024. Chapter 14 - Indicators and thresholds for nitrogen saturation in forest ecosystems. In: E. Du and W.d. Vries (Editors), Atmospheric Nitrogen Deposition to Global Forests. Academic Press, pp. 249-261.
Yost, J.L., Schmidt, A.M., Koelsch, R. and Schott, L.R., 2022. Effect of swine manure on soil health properties: A systematic review. Soil Science Society of America Journal, 86(2): 450-486.

Zanella, A., Jabiol, B., Ponge, J.F., Sartori, G., De Waal, R., Van Delft, B., Graefe, U., Cools, N., Katzensteiner, K., Hager, H. and Englisch, M., 2011. A European morpho-functional classification of humus forms. Geoderma, 164(3): 138-145.

Zhang, B., Jia, Y., Fan, H., Guo, C., Fu, J., Li, S., Li, M., Liu, B. and Ma, R., 2024. Soil compaction due to agricultural machinery impact: A systematic review. Land Degradation & Development, 35(10): 3256-3273.

Zhou, X., Fornara, D., Wasson, E.A., Wang, D., Ren, G., Christie, P. and Jia, Z., 2015. Effects of 44 years of chronic nitrogen fertilization on the soil nitrifying community of permanent grassland. Soil Biology and Biochemistry, 91: 76-83.