Optimisation of nutrient budget in agriculture

D3.2 Report with selected critical KPI with their thresholds

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Preface

The NutriBudget project aims to develop and implement a prototype of an integrated nutrient management platform (NutriPlatform) in various regions across Europe, as a decision support tool (DST) for farmers, advisors, European policy makers and regional authorities. The development of the NutriPlatform will be based on knowledge from existing and new field-tested agronomic mitigation measures linked to advanced NutriModels, which integrate various nutrient models, common data standards and relevant monitoring indicators. Thereby, NutriBudget will contribute to systemically optimizing nutrient flows and budgets across different agricultural production systems and regions in the EU to reduce nutrient losses to the environment and related impacts. The NutriModels will be able to operate at different scales: for farmers at the farm level and for regional authorities and policy makers at the regional to EU level, taking into account a holistic, sustainable and data-driven perspective on agriculture, linking the flow of nutrients between soil, water, air, plants, animals, feed and food with specific validated technological or nature-based mitigation measures within a financially viable transition route towards the desired nutrient status, as described in the Zero Pollution Action Plan and the Farm to Fork Strategy.

To assess the actual farm performance in view of agronomic and environmental targets, an integrative key performance indicator (KPI) framework has been designed to monitor the transition from the current to the desired status to have optimised farming systems (conventional, agro-ecological and organic in animal and crop production) in equilibrium with maximum agricultural performance and minimal environmental pressure. As such, this framework will guide the actual decision support as well the identification of appropriate roadmaps to reach the desired status for soil surpluses of carbon and nutrients in view of targets for soil quality, water quality, climate, biodiversity and crop production.

This report describes the derivation of thresholds (task 3.2 of the NutriBudget project), being defined as critical or target values for carbon, nitrogen, phosphorus, potassium, calcium, magnesium, sulphur, copper, zinc and cadmium surpluses (inputs minus crop uptake) in view of agronomic and environmental targets for soil health, crop production, water quality, biodiversity and climate. These values are used to derive the KPI levels (the distance between current and desired surpluses) being used in the NutriKPI framework (being developed in task 3.1). Using these thresholds allows one to apply the NutriKPI framework in evaluating the gap between current and desired budgets as well as the contribution (and assessment) of measures aimed to minimize this gap.

We greatly acknowledge all contributing partners that contributed to the development of the NutrKPI framework: Ghent University (Belgium), Luke (Finland), Yara International (Norway), PWC (France), Arvalis (France), Beta Technology Center (Spain), Wageningen University & Research (the Netherlands), the Rural Investment Support for Europe Foundation (RICE), the Universita Degli Studi di Milano (Italy), Proman Management (Austria), Sveriges Landbruksunversitet (Sweden), the Nutrient Management Institute (the Netherlands), Acqua & Sole (Italy), Impact (Belgium), Stockholms Universitet (Sweden) and the Forschungsinstitut fur Biologischen Landbouw Stiftung (Switzerland). Lastly we thank Ludwig Hermann and Salim Beyazid for reviewing this report.

Executive Summary

This report (Deliverable 3.2 of the NutriBudget project, Task 3.2) describes the derivation of thresholds (being critical limits or targets) for carbon, nutrient and metal (N, P, K, Ca, Mg, S, Cu, Zn and Cd) surpluses in European agriculture, thereby defining the local and regional targets to be used in the NutriKPI framework of the NutriBudget project.

This results for each region in five specific targets:

1. a critical N surplus in view of critical nitrate (NO_3^-) concentrations in groundwater
2. a critical N and P surplus in view of critical N and P concentrations in runoff to surface water
3. a critical N surplus in view of maximum ammonia (NH_3) emissions to the air from soil, storages and stables
4. a target surplus for N, P, K, S, Ca, Mg, Cu and Zn in view of soil quality and crop production
5. a critical target C surplus in view of soil health and the desired C sequestration for mitigating climate change

The derivation of these targets allows the NutriBudget project to inform practitioners on how to reduce the nutrient emissions from agriculture with spatial explicit targets for groundwater quality (originating from the Nitrates Directive), surface water quality (originating from the Water Framework Directive), ammonia emissions (originating from the Birds and Habitats Directive), and greenhouse gas emissions (originating from the ambitions to reduce the emissions from agriculture and to mitigate part of the greenhouse gases (GHG) emissions by carbon sequestration, the Paris Agreement). At the same time it allows one to apply nutrients appropriately in order to produce sufficient food (an agronomic and economic objective) and to maintain soil health.

The derived targets will allow the algorithms developed in NutriBudget to be applied with relevance to the farm scale (by their adoption in the NutriFarm model, WP2) and the European scale (by applying them in the MITERRA model), thereby guiding the farm decision support (via the Decision Support Tool developed in WP5), as well the assessment of the impact of measures (identified and evaluated in WP1) and the roadmaps to be evaluated in WP2.

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List of Abbreviations

ANC	Acid Neutralizing Capacity
BS	Base Saturation
C	Carbon
Ca	Calcium
CEC	Cation Exchange Capacity
EU	European Union
GHG	Greenhouse Gas
K	Potassium
KPI	Key Performance Indicator
Mg	Magnesium
N	Nitrogen
NUE	Nitrogen Use Efficiency
P	Phosphorus
PUE	Phosphorus Use Efficiency
PS	Precipitation Surplus
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
WFD	Water Framework Directive
WP	Work Package

1. Introduction

1.1 The NutriBudget project

The NutriBudget project strongly focuses on using carbon and nutrient surpluses as *effect indicators* for which specific thresholds can be defined in view of the multidimensional agronomic and environmental targets. Nutrient balances in agriculture (both on field, farm and regional level) are equivalent to inputs of nutrients minus outputs of nutrients contained in animal and plant products as well as in manure removed from agriculture. A surplus means that inputs exceed outputs in the balance, thereby causing an accumulation of nutrients into soil, which might lead to losses into water as well as into air in the case of nitrogen. A farm surplus for nutrients relates to the overall nutrient balance at the farm level, considering inputs and outputs as well as the impact of manure treatment technologies and housing systems. A soil surplus for nutrients focuses specifically on the excess nutrients present in the soil compared to what is required for plant growth. Both concepts refer to the same mass balance principle and are therefore both important for sustainable agriculture and environmental protection, as they help in managing nutrient use efficiency and minimizing the environmental impact of nutrient losses.

As explored and reviewed by Ros et al. (2023), carbon and nutrient balances are of direct relevance to policies relating to agriculture and the environment including climate change, air quality, water quality, and biodiversity. These balances have been used in market initiatives and policy instruments as indicators of the pressure put by farming on the environment and how that pressure changes over time. Note that these balances do not necessarily estimate the actual losses of nutrients to the environment, but significant nutrient surpluses are directly linked with losses, with the actual losses varying with site conditions. They also allow direct feedback on farm management, giving great opportunities to guide farming systems in their roadmap to more sustainable carbon and nutrient use in agriculture.

The additional advantage of these carbon and nutrient balances is that they can be applied on various spatial and temporal scales, allowing generic scalability and facilitating integrative assessments of farms and farming systems in view of the desired targets given by the NutriBudget project, including water quality (mainly nitrogen and phosphorus), air quality (mainly carbon and nitrogen), biodiversity (mainly ammonia), climate (mainly carbon) and soil quality (mainly carbon, cations and anions), and optimal crop and animal production.

Therefore, since these balances link to multiple agronomic and environmental performance indicators of farming systems and since they can be implemented for all agricultural systems at various spatial and temporal scales across Europe, we select them to guide the NutriBudget project which aims to "*systematically optimize nutrient flows and budgets across different agricultural production systems and regions in the EU to limit and reduce pollution due to the excessive use of nutrients and nutrient losses to the environment*".

1.2 Principles of the NutriKPI framework

The developed KPI framework (described in D3.1, Ros et al., 2023) aims to optimize carbon and nutrient flows on field, farm and regional level in view of ecosystem services related to crop production, water quality, carbon neutrality and biodiversity. Fields and farms can be assessed and monitored via specific KPIs for soil fertility (contributing to both agronomic and environmental targets, but in particular to crop production), water quality, climate and biodiversity. These KPIs, being derived from measured or calculated field and farm properties, provide a physical gauge to the desired farm management practices to improve the sustainability of farming.

A few principles are applied to the delineation of the intended operation of the farming system. First, we assume that the improvement of agricultural production also enhances the socio-economic position of the farmer, without adding this as a separate objective. The proposed NutriKPIs include specific targets for farm and soil surpluses of carbon and nutrients in view of the five objectives (i.e. crop production and soil health, ammonia emission and nature quality, nitrogen leaching and ground water quality, nitrogen and phosphorus leaching and runoff in view of surface water quality, and carbon sequestration in view of climate). The derivation of thresholds for the NutriKPIs will follow (inter)national commitments where appropriate. Second, the NutriKPI framework portrays the performance of individual farmers in the agricultural sector at the farm level. It concerns performance aspects that farmers can influence and can be determined per farm but can be aggregated to various spatial scales (field, farm, regional) relevant to the objectives. Last, the NutriKPI framework fits to the production environment of agricultural farms (calculated nutrient inputs and outputs have direct economic consequences) and accounts for the agro-ecosystem properties (i.e., soil type, climate, crop rotation, soil management) affecting (and controlling) its agronomic and environmental impacts. Therefore, the proposed NutriKPIs relate to the specific conditions in their immediate surroundings. In principle, the framework is applicable for all farm types independent of the farming strategy (conventional, agro-ecological, regenerative or organic).

The NutrKPI framework connects different spatial scales by translating performance at the farm level into contributions to objectives at higher scales and vice versa: objectives at different scales (countries, regions, sectors, supply chains) are translated into performance at the farm level.

1.3 Objective: derivation of thresholds for KPIs

Within the NutriBudget project a process-based model approach is followed to quantity carbon and nutrient surpluses on field, farm and regional scale. To evaluate the current situation in view of the desired status regarding these surpluses, one needs to define *target values* for carbon and nutrient surpluses in relation to soil health and crop production and *critical values* for nutrient surpluses in view of their potential losses to the environment (i.e. air and water). These critical and target values will be

- applied to identify the most appropriate measures to improve crop yield and to reduce nutrient losses in WP1,
- applied to evaluate the impact of agronomic measures for various farming systems and to underpin the desired road maps for European agriculture in WP2,
- used in the NutriKPI framework of the NutriBudget project in WP3, and
- will be applied to guide the farmers in their nutrient management, in a decision support tool to be developed in WP5.

Critical target values for nitrogen can be derived via the spatially explicit N-balance approach designed by De Vries et al. (2021) in relation to i) atmospheric N deposition onto natural areas to protect terrestrial biodiversity (critical N loads), ii) N concentration in runoff to surface water (2.5 mg N L^{-1}) to protect aquatic ecosystems and (iii) nitrate concentration in leachate to groundwater (50 mg NO_3 L^{-1}) to meet the EU drinking water standard. Referring to these critical values and calculating their exceedances can inform more targeted mitigation policies than flat-rate targets for N loss reductions currently mentioned in EU policies.

Sustainable P management, which aims to grow crops without P limitation while avoiding P losses to the environment is crucial to: (i) ensure sufficient food production (Koning et al., 2008), linked to the Sustainable Development Goal SDG2 (zero hunger) and (ii) avoid eutrophication in aquatic environments, linked to achieving SDG14 (life below water). Sustainable P management can be based on the “Build-up or mining and Maintenance” approach. Depending on whether soil P availability (as being assessed by a soil P test) is high, low, or optimal, P fertilizer inputs can be less than, more than, or equal to crop P removal, respectively. Recently de Vries et al. (2024) designed a spatially explicit P-balance approach to assess the current P surplus in view of both crop yield and the losses via leaching and erosion. The same approach is applicable to the other nutrients including K, Mg and Ca. Base cations play a dual role, first as nutrients and second as key elements in balancing soil acidity at optimal agronomic target values, while accounting for site conditions controlling the acid buffering capacity of the soil. For metals, one can derive critical values in view of ecotoxicological impacts on soil life, crop production (and crop quality) and the quality of aquatic ecosystems.

Soil organic carbon (SOC) is a key parameter for healthy and high-quality agricultural soils and it drives soil processes controlling both crop yield and nutrient losses to the environment. A critical value is still missing for SOC below which the soil fertility and viable productivity are compromised. Determining a critical value for SOC requires the assessment of the quantitative evidence, i.e. the nature of SOC and the properties it confers on soils, in order to justify limits adapted to a range of soil types, climatic conditions, or land management/cropping practices. It is equally important to identify possible trade-offs from an increase in SOC levels, such as interference with nutrient bioavailability (see for example Hossain et al., 2020).

The derivation of target and critical targets will vary over space given the site conditions controlling the agronomic and environmental impacts as well the environmental targets to achieve (e.g. distance to nearby nature areas, the vulnerability for nitrate leaching). This allows spatially explicit monitoring and assessment of KPIs taking into account the spatial variability in soils, climatic conditions and farming systems.

1.4 Report outline

After this introduction (**Chapter 1**), the methodology to derive critical values and targets for nitrogen and phosphorus budgets is described and illustrated in **chapter 2**. **Chapter 3** describes and illustrates the methodology for carbon, sulphur, base cations (calcium, magnesium, and potassium) and metals (copper, zinc and cadmium). **Chapter 4** describes how these thresholds are used to transform the effect indicators (being the carbon and nutrient surpluses) into Key Performance Indicators allowing the quantification and monitoring of farm performance across all regions in the EU.

This report (D3.2) describes the output of task 3.2 from WP3, resulting in a shortlist of indicators together with critical and target values for both agronomic and environmental performance, so that policy makers and farmers can monitor the actual performance of their farm regarding their transition to a more sustainable farming system (given actual and desired status, to be identified in T2.2, foreseen in D2.5). The thresholds are spatially explicit, thereby allowing broad applicability across Europe. The selected KPIs show the sustainability performance for soil, air and water quality in view of *target values* for a desired status or *critical values* for maximum acceptable adverse impacts. A detailed spatially explicit analysis of the variation of the current carbon and nutrient budgets in view of the desired ones is foreseen in D2.5 in conjunction with the application of the MITERRA-Europe model (Task 2.2).

2. Critical values and target values for nitrogen and phosphorus surpluses

2.1 Background

Over the last 50 years the EU Common Agricultural Policy has encouraged intensification of production, now typified by large-scale commercial farming of livestock and crops, with increased use of fertilizers, pesticides and other chemical inputs. This has contributed to significant, diverse and widespread environmental problems with potential long-term negative impacts on food security, future agricultural production, damage to terrestrial and aquatic ecosystem functions and services, and risks to human health.

While Europe's food demand is only projected to increase by a few percent between now and 2050 (Bruinsma, 2012), global demand is projected to increase by 60% (FAO, 2017) to 100% over the same period (Tilman et al., 2011), driven by an increasing global population and consumption of animal protein. In several regions in Europe, there are still substantial crop yield gaps, that are induced by nutrient limitation, besides other yield limiting factors. The yield gap is the difference between the yield potential in case of irrigated crops or the water limited yield potential in case of rainfed crops and the actual yield. As it is unrealistic to expect that farmers will increase yields up to the biophysical potential, a target yield is often set at 80% of the water-limited yield potential (for situations without irrigation), considered as the maximum exploitable yield for farmers under most circumstances given economic and environmental considerations (Lobell et al., 2009; Van Ittersum et al., 2013). Currently, there remains a large potential for increasing crop yields in Southern and Eastern Europe to close the gap to target yields.

Considering the yield gaps in Europe, one may assess what this means in terms of additional N and P requirements. These requirements can be estimated based on the current N and P use efficiency (NUE and PUE), defined as the fraction of applied N or P by fertilisers, manures, deposition and in case of N also biological fixation, that is taken up by a crop. Eventually, additional requirement of N or P can be derived from the target N or P that is removed from the field (being the target crop yield times the N or P content in the crop) divided by the NUE or PUE and subtracting the actual N or P input. Following the focus of the NutriBudget project on the surpluses, a potential increase in nutrient requirement due to an increased yield is automatically included by the fertilization approach matching nutrient inputs to crop demand.

Fertilizer, crop and soil management are essential to sustain food production but they are simultaneously important causes of environmental impacts if not done appropriately. Observed forms of fertilizer related impacts are damage to ecosystem biodiversity caused by the eutrophication of terrestrial ecosystems and water bodies in response to elevated N and P inputs. Here we are facing challenging dilemmas where the need to enhance food production requires an additional input of N and P (required N and P inputs), and whereas environmental protection of surface water bodies may require a reduction of the input of N and P (critical N and P inputs), depending on the site conditions.

To enable the assessment of the critical N and P inputs and surpluses, there is a need for N and P indicators with related critical limits relevant to environmental receptors, above which adverse effects can be expected. Relevant N and P indicators in this context are (i) critical N deposition on natural ecosystems, (ii) critical nitrate (NO_3) concentration in leachate to groundwater and (iii) critical N and P concentrations in runoff to surface water. When critical limits for these indicators are known, it allows an assessment of critical N and P inputs and associated N and P surpluses, being the inputs and surpluses at which such limits are reached, either on the short term (N) or the long term (P). The approach to derive critical values for nitrogen surpluses is based on de Vries et al. (2021, 2022) and for phosphorus surpluses on de Vries et al. (2022).

2.2 Critical values and target values for nitrogen surpluses

Figure 1 illustrates the linkage between N inputs by fertilizer, biological fixation, excreted manure, biosolids and deposition, N losses to air due to NH_3 emissions and denitrification (losses of N_2, N_2O and NO_x) and N losses to water due to N leaching and N runoff. These relationships are used to underpin the critical values for the various N losses.

We assumed that the NUE of N fixation is equal to that of N fertilizer, and that the NUE of N biosolids is equal to that of excreted N manure. The NH_3-N emissions and N deposition are indicated by dotted lines (i.e. they cancel one another out), since we assume that the average total N deposition on agricultural land equals the average NH_3-N emissions from agriculture. This assumption holds for larger spatial regions but differences between N deposition and emission flux will increase at spatial geometries with areas lower than 5 km^2 as well when other land uses (e.g. forests) dominate in the area.

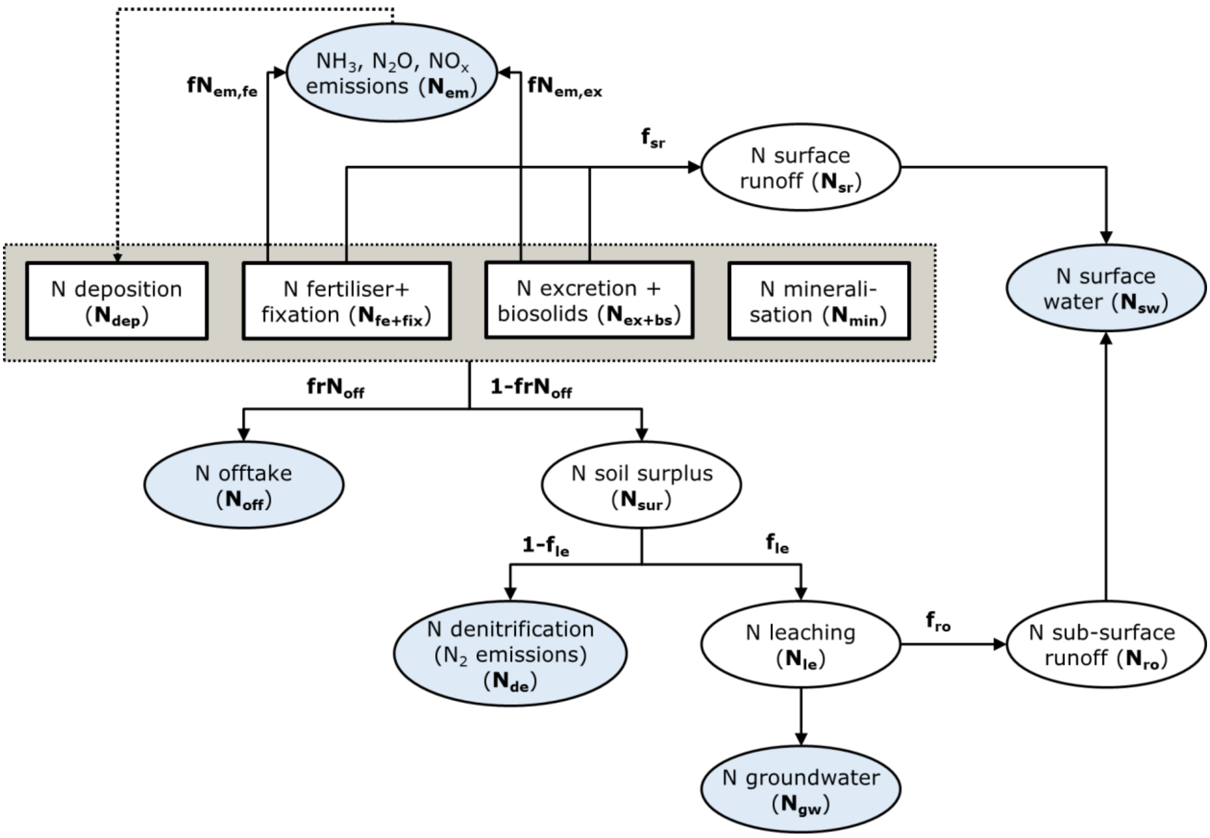

Figure 1 Linkage between N inputs, N losses to air by NH_3 emissions and denitrification (losses of N_2, N_2O and NO_x) and N losses to water due to N leaching and N runoff at steady state (Source: De Vries et al., 2022a).

Critical N inputs are back-calculated from N surpluses which need to be constrained by thresholds for NH_3-N emissions, N concentrations in and related runoff to surface water and for critical concentrations in and related NO3-N leaching to groundwater (blue boxes). Nitrogen inputs also cause N_2O emissions, but since there are no clear limits for N_2O emissions, apart from a required reduction target, this aspect is not included in the derivation of critical thresholds for the N surplus as Key Performance Indicator. Another argument for not including N_2O emissions is the fact that emissions due to agricultural N inputs cause an enhanced CO_2 sequestration in response to elevated NH_3 deposition, largely compensating for the global warming potential caused by N_2O emissions (De Vries et al., 2011, 2017).

2.2.1 Critical surpluses for groundwater quality

The critical surplus for protecting groundwater quality is here defined as the N surplus that should not be exceeded to avoid nitrate concentrations in groundwater exceeding the 50 mg NO_3 L^{-1}.

The critical N leaching flux to groundwater (N_{gw_crit} in kg N ha^{-1} yr^{-1}) is derived as:

$$N_{gw\_crit} = [NO3]_{gw\_crit} * Q_{gw} * cF_{NO3} \qquad (2.1)$$

where $[NO_3]_{gw\_crit}$ is the critical nitrate concentration in leaching flux towards groundwater (set to 50 mg NO_3 L^{-1}), Q_{gw} is the water flux leaching towards groundwater (m^3 m^{-2} yr^{-1}), and cF_{NO3} is the conversion factor from (mg NO_3 L^{-1})* (m^3 m^{-2}) to kg N ha^{-1}, i.e. (14/62)*10.

The critical N leaching flux towards groundwater is related to a critical N surplus ($N_{sp\_gw\_crit}$ in kg N ha^{-1} yr^{-1}) as follows:

$$N_{sp\_gw\_crit} = \frac{N_{gw\_crit}}{f_{le} * (1 - f_{ssro})} \qquad (2.2)$$

where f_{le} represents the nitrogen leaching out of the root zone and f_{ssro} represents the nitrogen fraction lost towards the surface water via subsurface runoff, being estimated as function of lithology, groundwater depth and the occurrence of natural surface waters (Keuskamp et al., 2012). The fraction percolating to the groundwater (1-fssro) ranges from 0 to 1, varying from 1.0 in unconsolidated sediments with coarse textures to 0.15 in alluvial deposits with medium-fine textures, and down to 0.067 in very fine textured soils.

The value of f_{le} is calculated from a maximal leaching fraction for nitrogen and a set of reduction fractions (Velthof et al., 2009), where the fractions for land use and precipitation surplus have (slightly) been adapted:

$$f_{le} = f_{le.max} * f_{lu} * min(f_p, f_t, f_c) \qquad (2.3)$$

where $f_{le.max}$ represents the maximum leaching fraction for nitrogen being dependent on soil type, f_{lu} represents a reduction fraction for land use, f_p a reduction fraction for precipitation surplus, f_t a reduction fraction for temperature and f_c a reduction fraction for soil organic carbon content. The following soil type dependent maximum leaching fractions ($f_{le,max}$) are used: 1.0 for sand, 0.75 for loam, 0.5 for clay and 0.2 for peat. The reduction fraction for land use, f_{lu}, is 0.85 for grassland and 1.0 for cropland.

Denitrification increases and thus leaching decreases at lower precipitation surplus (PS) due to longer residence times allowing enhanced denitrification. The reduction fraction for precipitation surplus, f_p, can be calculated as a continuous function based on soil type.

For sandy and loamy soils:

- PS < 50 mm: f_p = 0.25
- 50 mm ≤ PS < 300 mm: f_p = 1+(PS-300)*0.003
- PS ≥ 300 mm: f_p = 1

For peat and clay soils:

- PS < 50 mm: f_p = 0.25
- 50 mm ≤ PS < 100 mm: f_p = 1+(PS-100)*0.015
- 100 mm ≤ PS < 300 mm: f_p = 1
- 300 mm ≤ PS < 400 mm: f_p = 1-(PS-300)*0.005
- PS ≥ 400 mm: f_p = 0.5

Denitrification increases with increasing temperature and thus leaching decreases. The following reduction fractions for temperature (f_t) are used (assuming that denitrification at 15°C is twice as high as at 5°C; a general effect of temperature on microbial activity), using air temperature as a proxy for the soil temperature:

- < 5 °C: $f_t = 1$
- 5-15 °C: $f_t = 0.75$
- > 15 °C: $f_t = 0.50$

Denitrification increases with increasing total soil organic carbon (SOC) content and thus leaching decreases. The following reduction fractions for SOC content (f_c) are used:

- SOC < 1%: $f_c = 1$
- SOC 1-2%: $f_c = 0.90$
- SOC 2-5%: $f_c = 0.75$
- SOC > 5%: $f_c = 0.50$

This also implies that the N leaching of peat soils often used as grassland due to high groundwater levels are rather low due to the combined impact of land use, soil type and SOC levels.

2.2.2 Critical surpluses for surface water quality

The critical surplus for protecting surface water quality is here defined as the N surplus that should not be exceeded to avoid nitrogen concentrations in surface water exceeding the 2.5 mg N L^{-1}.

Nitrogen inputs from fertiliser contribute to runoff through two pathways: (i) surface runoff (or direct runoff) (N_{sr}), being a fraction of N inputs and (ii) sub-surface runoff (N_{ro}), being a fraction of N leaching below the root zone. Similarly, total precipitation surplus (Q_{tot}) is distributed over surface runoff (Q_{sr}), sub-surface runoff (Q_{ro}) and leaching to groundwater (Q_{gw}).

The critical N runoff flux is the sum of surface and sub-surface runoff and can be derived by multiplying the critical N concentration for surface waters $[N]_{crit}$ (2.5 mg N L^{-1}) with the total water runoff volume and a unit correction factor.

$$N_{sw\_crit} = [N]_{crit} * (Q_{ssro} + Q_{ro}) * cF_{cN} \qquad (2.4)$$

where Q_{ssro} is the sub-surface runoff water flux (m^3 m^{-2} yr^{-1}), Q_{ro} is the water flux via surface runoff (m^3 m^{-2} yr^{-1}) and cF_{cN} is a conversion factor from mg N L^{-1} to kg N ha^{-1} / (m^3 m^{-2}), i.e. 10.

Thresholds for N concentrations in runoff can range from1 to 2.5 mg N L^{-1}, a limit that holds for surface waters, where a higher limit value in runoff water could be acceptable due to denitrification or N retention in surface water. Likewise, a stricter limit value for runoff water could be used because of the mixing of runoff water with point loads of N into surface water. To avoid complex interactions between agricultural and non-agricultural N sources to surface water, and the focus on agricultural activities in the NutriBudget project, we selected the threshold of 2.5 mg N L^{-1} as a generic threshold to optimise the nitrogen budgets for farming activities.

N leaching plus runoff and N denitrification is assumed to be directly and linearly related to the N surplus (being the difference between N inputs and N uptake by crop removal). Hence, the critical N runoff flux via surface and subsurface runoff can be transformed into a critical N surplus via:

$$N_{sp\_sw\_crit} = N_{sw\_crit}/(f_{ro} + f_{le} * f_{ssro}) \qquad (2.5)$$

The surface runoff fraction f_{ro} is calculated as a function of the slope, being corrected for the precipitation surplus, land use and depth to rock. Similar to the leaching fraction, the value of f_{ro} is calculated from a maximal runoff fraction and a set of reduction fractions according to Velthof et al. (2009):

$$f_{ro} = f_{ro.max} * f_{lu} * min(f_p, f_s, f_{rc}) \qquad (2.6)$$

where $f_{ro.max}$ represents the maximum surface runoff fraction being dependent on slope, f_{lu} represents a reduction fraction for land use (-), f_p a reduction fraction for precipitation surplus, f_s a reduction fraction for soil type (-) and f_{rc} a reduction fraction for depth to rock (-). The maximum surface runoff fraction ($f_{ro,max}$) is determined as a function of slope, where $f_{ro.max}$ is 10% at slopes ranging between 0 and 8% (limited slope), $f_{ro.max}$ is 20% when slopes range between 8 and 15% (weak slope), $f_{ro.max}$ is 35% when slopes range between 15 and 25% (moderate steep) and $f_{ro.max}$ is 50% at steep areas with slopes exceeding the 25%. The reduction fraction for land use, f_{lu}, is 0.25 for grassland and 1.0 for cropland.

The reduction fraction for precipitation, f_p, is included as a function of precipitation surplus (PS):

- PS > 300mm: $f_p = 1.00$
- PS 100–300 mm: $f_p = 0.75$
- PS 50–100 mm: $f_p = 0.50$
- PS < 50 mm: $f_p = 0.25$

The reduction fraction for soil type, f_s, is included as a function of texture:

- Very fine (clay ≥ 60 %): $f_s = 1.00$
- Fine (35% ≤ clay < 60%): $f_s = 0.90$
- Medium (18% ≤ clay < 35%): $f_s = 0.75$
- Coarse (18% < clay): $f_s = 0.25$
- Peat: $f_s = 0.25$

The reduction fraction for depth to rock f_{rc} is included as:

- For a depth of less than 25 cm: $f_{rc} = 1.0$
- For a depth > 25 cm: $f_{rc} = 0.8$

2.2.3 Target surpluses for soil health and crop production

The target surplus for soil health and crop production is here defined as the desired N surplus to ensure a N supplying capacity of soils of 70 kg N ha^{-1} yr^{-1} for cropland and 100 kg N ha^{-1} yr^{-1} for grassland.

Cycling of N in soil is vital to maintaining soil health and agricultural productivity. While N research related to optimal fertilization rates and nutrient management has been conducted for decades, traditional soil fertility tests do not usually provide a robust quantification of the capacity of soil to supply N over the long term. In the context of soil health, N indicators focus on the biological processes that determine a soil's ability to cycle and supply N over time, rather than measuring the nitrogen available to plants in a given soil at a single time point. Various soil health indicators that measure a chemically defined fraction of N or a process related to N cycling have been proposed to quantify the potential to supply N to crops. The meta-analysis by Ros et al. (2011) showed that none of the existing soil health indicators is better than the total N content of soil in explaining the variability in the capacity of soils to supply N.

Soil N often limits productivity in agroecosystems, prompting fertilizer applications that increase crop yields. According to the Soil Health Index (Moebius-Clune et al., 2016) and the Open Soil Index (Ros et al., 2022) frameworks, the capacity of soils to supply N is evaluated in view of an averaged crop demand, being optimum around 50 to 100 kg N ha^{-1} yr^{-1} for croplands and around 100 to 140 kg N ha^{-1} yr^{-1} for grassland fields. Though this optimum might vary across European countries, we propose to use a more generic threshold following the averaged N uptake in crop rotation systems across Europe, equalling to 70 kg N ha^{-1} yr^{-1} for cropland and 100 kg N ha^{-1} yr^{-1} for grassland. This implies that the critical threshold for the N inputs from manure, compost and organic residues (as these organic N sources builds up the soil N pool thereby increasing the capacity of soils to supply N whereas inorganic N does not build up the soil N pool) should be higher than zero when the soil N supply is lower than these two thresholds.

This implies that the target surplus to increase the capacity of soils to supply N ($N_{sp\_org\_sq}$, kg ha^{-1} yr^{-1}) is only related to the organic N fraction of the supplied organic N products that remain in the soil after one year, often estimated via so-called humification coefficients (expressing the fraction of organic nitrogen that is mineralized in one year). Following this reasoning, the target surplus for organic N inputs can be estimated as:

$$N_{sp\_org\_sq} = max\left(0, N_{ss.target} - N_{ss.act}\right)/(T * f_{hc}) \quad (2.7)$$

where $N_{ss.target}$ refers to the desired N supply for agricultural soils (i.e. 70 kg N ha^{-1} yr^{-1} for cropland and 100 kg N ha^{-1} yr^{-1} for grassland), $N_{ss.act}$ refers to the actual N supply (being estimated as function of soil organic carbon, C-to-N ratio and the temperature), and f_{hc} the humification coefficient, being the fraction of organic N that is left in the field one year after its application, and T the time (in years) to achieve the desired increase in the soil N supply. Note that the total N input via manure is not allowed to exceed the 170 kg N ha^{-1} given the European Nitrate Directive, and T can be adapted to ensure that the annual dose remains below this limit.

2.2.4 Critical surpluses for ammonia emissions and nature quality

The critical surplus for ammonia emissions and nature quality is here defined as the N surplus that should not be exceeded to avoid nitrogen depositions exceeding the critical N load on ecosystems in nature areas (in the same region where the farms and fields are located).

Critical NH_3 emissions and associated critical N inputs and N surpluses can be derived from critical levels of N deposition. NH_3 emissions from agriculture to air are diluted by emissions from non-agricultural land in the area. Therefore we accounted for differences in the fraction of agricultural land when assessing critical NH_3 emissions for a given spatial geometry. Critical levels of NH_3 emission from agricultural land can be calculated as:

$$NH3_{em\_crit} = Ndep_{tot\_crit} * \frac{f_{NH3}}{f_{ag}} \tag{2.8}$$

where $Ndep_{tot\_crit}$ is the critical N deposition on non-agricultural terrestrial ecosystems, calculated as the area-weighted average critical N load for those ecosystems in nature areas (kg N ha^{-1} yr^{-1}) following the fraction agricultural land (fag) and the fraction NH_3 in the total (NO_x+NH_3) deposition. The calculated NH_3 emissions are based on the assumptions that within the spatial geometry evaluated (i) the average N deposition rate on agricultural land equals the average N deposition rate on non-agricultural land (both in kg N ha^{-1}), (ii) the amount of NH_3 that is emitted (coming from agriculture) is deposited in the same region, but then on all (agricultural land and non-agricultural) land (both in kg N) and (iii) the current shares of NH_3 and NO_x in total N deposition stay constant (and thus NOx emissions/deposition increase or decrease in the same proportion as NH_3 emissions/deposition). The derivation of Eq. 2.8, based on these assumptions is given in de Vries et al. (2022). We assume that all NH_3 originates from agricultural sources.

Critical NH_3 emissions determine critical N inputs from fertilizer (+ fixation) and from excretion (+ biosolids), and (constant) emission fractions, whereas NH_3 emissions are calculated as a fraction of these inputs according to:

$$NH3_{em\_crit} = N_{fe\_fix\_crit} * fNH3_{em,fe} + N_{ex\_bs\_crit} * fNH3_{em,ex} \tag{2.9}$$

Eq. 2.9 has two unknowns, i.e. $N_{fe\_fix\_crit}$ and $N_{ex\_bs\_crit}$. By assuming constant relative contributions to total NH_3 emissions from N fertilizer and N fixation (N_{fe_fix}) and N excretion by animals and N biosolids (N_{ex_bs}), we can express $N_{fe\_fix\_crit}$ as a function of $N_{ex\_bs\_crit}$:

$$N_{fe\_fix\_crit} = \frac{fN_{fe}}{1-fN_{fe}} * N_{ex\_bs\_crit} \tag{2.10}$$

The fraction of N inputs from fertilizer and fixation in total critical N inputs (fN_{fe}) is calculated by dividing actual N inputs from fertilizer and fixation by actual N inputs from fertilizer, fixation, manure and biosolids for each crop combination:

$$fN_{fe} = \frac{N_{fe}+N_{fix}}{N_{fe}+N_{fix}+N_{am}+N_{bs}} \tag{2.11}$$

Replacing $N_{fe\_fix\_crit}$ in Eq. 2.10 with the right-hand side of Eq. 2.11 and solving the equation for $N_{ex\_bs\_crit}$, critical N inputs from excretion in view of critical NH_3 emission can be derived as

$$N_{ex\_bs\_crit\_NH3} = \frac{NH3em_{crit}}{fNH3_{em,ex} + fNH3_{em,fe} * \frac{fN_{fe}}{1 - fN_{fe}}} \tag{2.12}$$

Where $N_{ex\_bs\_crit\_NH3}$ is the critical N inputs from excretion and biosolids in view of a critical ammonia emission rate (kg N ha^{-1} yr^{-1}). The related critical N input from fertilizer and fixation can be calculated with equation 2.10. Equation 2.10 and 2.12 implies that the critical nitrogen input to the soil via fertiliser ($N_{fe\_fix\_crit\_NH3}$) and excretion ($N_{ex\_bs\_crit\_NH3}$) in view of critical NH_3 emissions depends on (i) the NH_3 emission fraction for fertiliser applied to land ($fNH3_{em,fe}$), (ii) the NH_3 emission fraction for excretion ($fNH3_{em,ex}$) and the share of N fertilizer + fixation in total critical inputs (fN_{fe}).

2.2.5 Spatial variation in critical surpluses for nitrogen

The spatial variation in the critical N surplus in view of ground water and surface water as well the critical ammonia emission is illustrated below using former data published in 2010 (de Vries et al., 2016), to be updated when new model results from MITERRA-Europe are becoming available. We do not expect huge differences in these critical N surpluses since the underlying mechanisms are the same, but the actual variation might differ due to updated source data reflecting the spatial variation in farming systems, soil properties and geohydrological conditions.

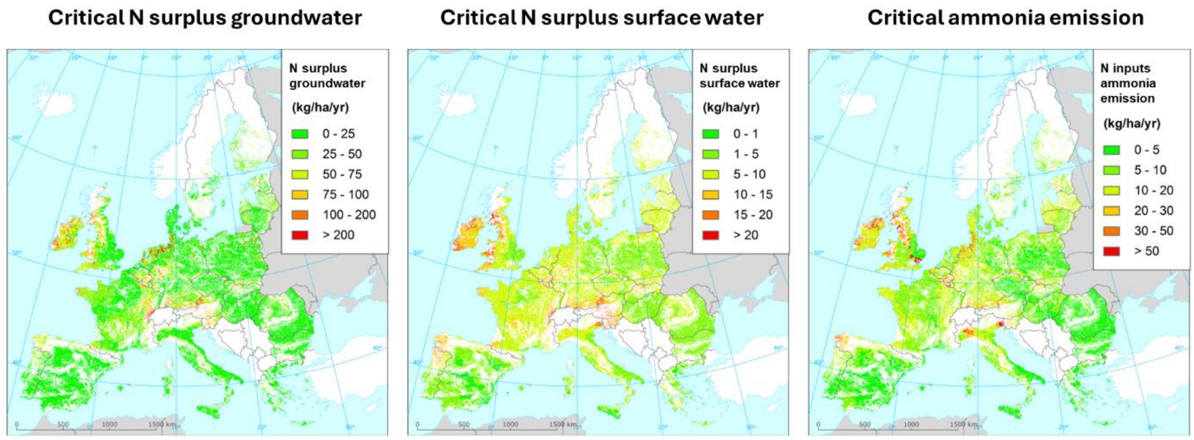

Figure 2 Maps of critical values for the N surplus in view of groundwater (left), for N runoff to surface water (middle), and ammonia emission (right).

2.3 Critical limits and targets for phosphorus surpluses

Unlike N, there are P containing minerals in (agricultural) soils and P can dissolve, precipitate and be adsorbed or desorbed, especially on aluminium and iron hydroxides. The P surplus is generally adsorbed to soil in a readily available (adsorbed) P pool. A small part is retained in a poorly available stable inorganic P pool. Furthermore, organic P added by crop residues and manure increases P in soil organic matter (an organic P pool), which is the substrate for P mineralization. Plants take up P from the soil solution but the concentration in solution is buffered by the pool of adsorbed soil P, which is thus termed readily available or plant-available P. A small part of the dissolved P is also leached from the soil. Since P is adsorbed in soil and P concentrations in soil solution are governed by soil P contents, changes in soil P contents drive changes in P leaching and runoff to water. P losses to surface water thus react with a large delay time to changes in P input, and P behaviour should be modelled by a dynamic approach.

Understanding P accumulation and P leaching in response to current fertilizer and manure practices in EU-27 is necessary to inform the required changes in management practices in view of sustainable agriculture aiming to sustain crop production and avoid surface water eutrophication. P surplus that is causing P accumulation may be good for soil quality in areas where P is deficient regarding crop growth, but bad for environmental quality in areas where P availability is not limiting crop growth and where it leads to enhanced P losses to surface water, causing eutrophication.

2.3.1 Critical surpluses for surface water quality

The critical surplus for surface water quality is here defined as the P surplus that should not be exceeded to avoid P concentrations in surface waters exceeding the 0.10 mg P L^{-1}.

At steady state, net P exchange can be neglected and the long-term critical net P input can be calculated as the sum of a critical P uptake and a critical P loss to groundwater and surface water due to runoff and leaching, based on a critical P concentration in soil water. The critical P runoff from the soil can be derived by multiplying the current runoff rate (surface runoff and sub-surface runoff) of water with the critical P concentration in runoff to surface water, being equal to 0.15 mg L^{-1} of total P (De Vries et al., 2022). Water runoff is derived from precipitation surplus and is assumed to be constant, and can be calculated similarly as done for N. For critical P concentrations, Carpenter and Bennet (2011) use Carlson's index of 24 mg P m^{-3} (total P) as a boundary between mesotrophy and eutrophy and a pre-industrial P concentration in rivers of 160 mg P m^{-3}. In most countries critical P values vary mostly between 50-150 mg P m^{-3} depending on the type of water and we used a value of 100 mg P m^{-3} or 0.10 mg P L^{-1}.

The critical P surplus, being the difference between inputs and uptake, can be calculated as:

$$P_{sp\_crit\_sw} = P_{in\_crit} - P_{up\_crit} = P_{erosion\_crit} + P_{ro\_crit} \quad (2.13)$$

where P_{in_crit} refers to the critical total P input to the soil via fertiliser, manure, biosolids and deposition (kg P ha^{-1} yr^{-1}), P_{up_crit} refers to the critical phosphorus uptake, related to a soil P status in equilibrium with runoff and leaching from the soil (kg P ha^{-1} yr^{-1}), and the combination of $P_{erosion\_crit}$ and P_{ro_crit} refer to critical phosphorus loss to surface water by erosion and runoff from the soil (kg P ha^{-1} yr^{-1}). The P leaching to groundwater is assumed to be negligible given the high sorption of P to soil matrix.

The Eq 2.13 can be simplified by ignoring the differences between erosion and runoff. In that case the total critical P loss to surface water can be estimated from the critical P concentration in surface water (P_{ss_crit}, being 0.15 mg L^{-1}) and the total water flux (units) via:

$$P_{sp\_crit\_sw} = Q * P_{ss\_crit} \quad (2.14)$$

2.3.2 Target surplus for soil health and crop production

The target surplus for soil health and crop production is here defined as the desired P surplus to bring soils up to a desired P soil content where P is not deficient and limiting crop growth.

Sustainable P management in view of soil health and crop production can be based on the "Build-up or mining and Maintenance" approach. Depending on whether soil P availability (as being assessed by a soil P test) is high, low, or optimal, P fertilizer inputs can be less than, more than, or equal to crop P removal, respectively (Li et al., 2011a). The objective is to move from the environmental risk level (high P-status) or P deficient level (low P-status) to the level of ensuring stable crop yield (optimum P-status).

The required P input to sustain crop production and soil health can be calculated as a function of the soil P status, thereby accounting for extra P to be added to improve soil P status and thereby crop yields (in case of P-deficient soils) or mined (in case of P-saturated soils) to bring the soil P level to an adequate status within a predefined target time horizon according to the *build-up and maintenance* approach in addition to the target crop P uptake. This input can be assessed as:

$$P_{in\_target} = P_{up\_target} + P_{loss} + (\rho \times D \times (P\text{-}Av_{target} - P\text{-}Av_{current}) \times P_{total}/P\text{-}Av \times 0.01)/T \quad (2.15)$$

where P_{in_target} is the required P input in kg P ha^{-1} yr^{-1}, P_{total} is the total content of P in the soil solid phase (mg kg^{-1}), ρ is the bulk density of the soil (kg m^{-3}), D the thickness of the soil layer (m), and P-Av the plant available content of P in the soil solid phase (mg kg^{-1}) for the current situation ($P\text{-}Av_{current}$) as well the desired situation ($P\text{-}Av_{target}$), T is the defined target time in which the soil P level should come to acceptable status and 0.01 is used to convert mg P m^{-2} to kg P ha^{-1}. $P\text{-}Av_{target}$ is the critical plant available-P level above which crop yields hardly respond to an increase in soil plant available P. Note that this approach assumes that part of the added P transforms to stable P forms (given the ratio between P_{total} and P-Av) by diffusion-precipitation, making the P unavailable for uptake. However, the ratio P_{total}/P-Av most likely decreases when adding P, as added P is more available for uptake than soil-available P. For a fast calculation, the integral average P_{total}/P-Av ratio going from the current to the critical P status should be used.

Instantaneous P losses due to surface runoff are calculated as a surface runoff faction (f_{ro}) of the P applications in the form of fertilizer and manure with f_{ro} being a function of slope, texture and land use (Beusen et al., 2015). Estimates of soil P loss by rainfall erosion, including a P memory effect, are based on a large database of measurements by Cerdan et al. (2010), relating erosion rates to slope, soil texture and land cover type.

Subtracting the P uptake from the P input leads to the following target for the soil P surplus:

$$P_{sp\_target} = P_{loss} + (\rho \times D \times (P\text{-}Av_{target} - P\text{-}Av_{current}) \times P_{total}/P\text{-}Av \times 0.01)/T \quad (2.16)$$

Assuming that mitigating measures are implemented to reduce erosion and the fact that leaching fluxes generally are in the same order of magnitude to the P deposition, one can simplify this target for the soil P surplus as follows:

$$P_{sp\_target} = (\rho \times D \times (P\text{-}Av_{target} - P\text{-}Av_{current}) \times P_{total}/P\text{-}Av \times 0.01)/T \quad (2.17)$$

2.3.3 Spatial variation in critical targets for P

The current P management was evaluated by comparing the actual P balance, defined as inputs minus outputs (Muntwyler et al., 2024), with the soil available P concentrations (P-Olsen) (Ballabio et al., 2019). In 22 or 33% of EU+UK agricultural land, there is still build-up of soil P despite P-Olsen concentrations being already above 15 or 30 mg kg^{-1} respectively, while in 2 or 13% of EU+UK agricultural land there is mining of soil P despite P-Olsen concentrations being below 15 or 30 mg kg^{-1}, respectively (Jordan-Meille et al., 2012; Steinfurth et al., 2022).

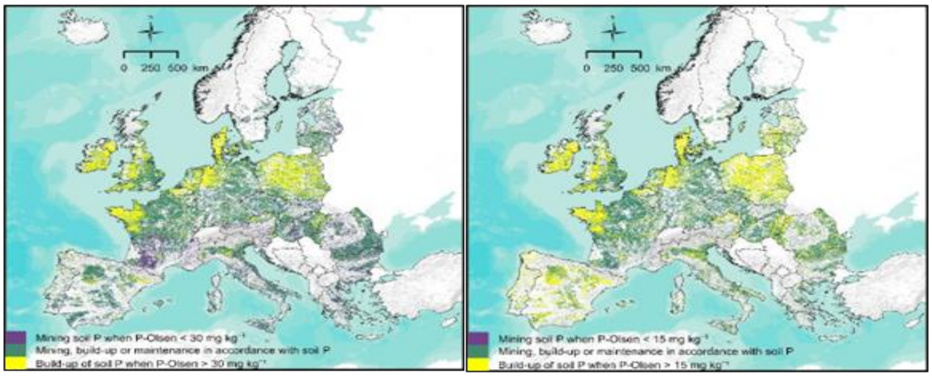

Figure 3 The evaluation of current P inputs when using P-Olsen of 30 (left) or 15 (right) mg kg^{-1} as critical value for soil health and crop production. When P-Olsen concentrations are below this threshold value, negative P balances imply unsustainable mining of soil P. When P-Olsen concentrations are above this threshold, positive P balances imply unsustainable build-up of soil P (Van Eynde et al., 2024).

3. Critical values and target values for the surpluses of other elements

3.1 Target values for carbon surpluses

Soil organic matter is key for healthy and high-quality agricultural soil and drives soil processes controlling both crop yield and environmental losses. An increase in soil organic matter (SOM) or carbon (SOC) levels is seen, both by many conventional farmers and policy makers, as a desirable objective. Better plant nutrition due to retention of nutrients, ease of cultivation, penetration and seedbed preparation, greater aggregate stability, reduced bulk density, improved water holding capacity and enhanced porosity have all been associated with increased amounts of SOC. By increasing SOC, the impacts of climate change can be mitigated (Lessmann et al., 2021). The sequestration of carbon in soils, as promoted by the “4 per mille Soils for Food Security and Climate” initiative in 2015, is claimed to be a ‘no regret’ or a ‘win-win’ option because increasing SOC in soils provides numerous co-benefits and few trade-offs (IPCC WG1, 2021).

3.1.1 Target surpluses for soil health and crop production

In 2003, Loveland & Webb reviewed existing literature on critical levels of SOM in temperate regions and concluded that the quantitative evidence for critical thresholds is scarce, whereas others mainly foresee positive benefits on soil physical functionality (Murphy, 2015). Given the complex interactions among soil physical, chemical and biological processes, one might wonder whether a single critical threshold can be defined. Nevertheless, a major threshold of 2% soil organic carbon (ca. 3.4% SOM) has been defined earlier (Loveland & Web, 2003; Oldfield et al., 2019) below which a potentially serious decline in soil quality will occur.

Using this threshold for SOC, C_{crit}, we can derive a target value for the carbon surplus as follows:

$$C_{sp\_crit\_yield} = C_{sp\_crit\_soilhealth} = max(0, C_{crit} - C_{act}) * D * \rho * 0.01/T \quad (3.1)$$

where C_{act} refers to the actual soil organic carbon content (in mg kg^{-1}), C_{crit} refers to the target value for SOC in view of soil health and crop production (mg kg^{-1}) in soil, being estimated at 2% (equal to 20 g kg^{-1}), D refers to the depth of the top soil (m), ρ refers to the bulk density of the soil (kg m^{-3}), and 0.01 is a unit correction from mg m^{-2} to kg ha^{-1}, and T refers to the time period (in years) to reach the desired target.

In soils having an actual SOC content higher than the target value for soil health and crop production, there is no requirement to increase the carbon content in the soil. In that case, the desired C surplus equals zero.

Soils vary in their potential to sequester carbon given the variation in climate, soil texture, land use, and available technologies as well as current yield levels. Multiple approaches have been used to assess the potential carbon storage in soil from local to regional scale:

- experimental studies using sorption experiments or long-term field experiments with high organic matter inputs,
- process based models like CENTURY or ROTHC are used to simulate the response of C pools to changes in management while accounting for weather conditions and main farming practices (Lugato et al., 2014, 2015; Lesschen et al., 2021).
- statistical models used to unravel the impact of agroecosystem properties on actual and potential SOC levels in agricultural soils (Weismeier et al., 2019). Statistical models vary from simple linear regression (Hassink, 1997; Chen et al., 2018) to geostatistical (McBratney et al., 2003) and machine learning models such as quantile regression neural networks and random forest regression models. For example, Hassink (1997) found a strong correlation between the content of fine soil particles and the fine fraction SOC in a wide range of soils and proposed to use this relationship to estimate the degree of stable SOC storage potential.

In 1997, Hassink proposed to estimate the C sequestration potential of arable and grassland soils as a function of the fine earth fraction (< 20 µm) using an empirical function calibrated on soils from temperate and tropical regions. This approach was recently applied to 2092 French top soils covering different soil, climate, relief and land cover conditions (Chen et al., 2018). Using the same approach here, the C_{crit} in view of maximum carbon storage capacity for climate mitigation can be estimated as follows:

$$C_{crit\_climate} = 4.09 + 0.37 * clay \qquad (3.2)$$

where $C_{crit\_climate}$ is the maximum C saturation (g kg^{-1}) and *clay* refers to the fraction (%) of soil particles with a particle-size smaller than 20 μm.

3.1.2 Critical surpluses for methane

Since the majority of the European soils are not organic and function as methane sinks, we excluded the potential carbon surplus in view of the climate effect of methane emissions.

3.1.3 Spatial variation in target values for carbon surpluses

As an illustration we plot here the spatial variation in current (reference) and target values for SOC for the 81 country and farm-type combinations as being used in the INTEGRATOR model (De Vries et al., 2011). Current clay and SOC values in the topsoil were generated from WISE, SPADE1 and EFSDB databases, which jointly contain about 3,600 soil profiles, irregularly distributed over Europe (Heuvelink et al., 2016). Data for clay and soil organic matter contents at the NCU level were derived with a multivariate regression kriging model accounting for the spatial structure of soil properties and their dependence on explanatory variables such as soil type and land cover (Heuvelink et al., 2016). Target limits for SOC were assessed as a function of clay content. Target SOC ranges from 1 to 1.5%, with slightly higher targets for clay and loam compared to sand. Reference SOC ranges from 0.5 to 10%, with a clear general increase in the order of clay, loam, sand.

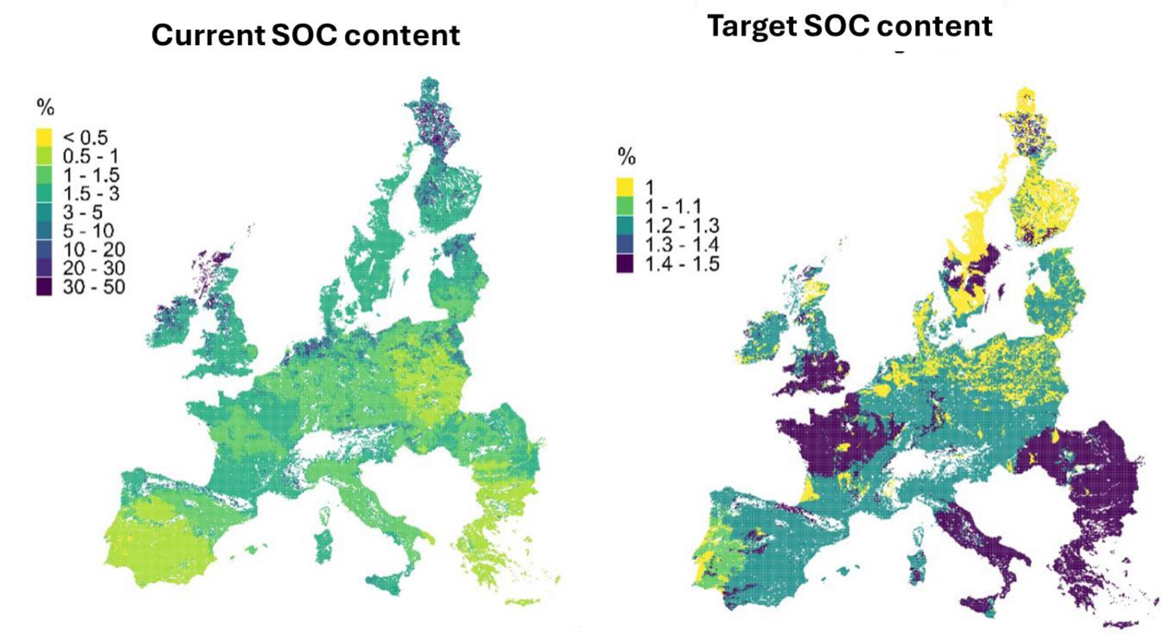

Figure 4 Spatial variation in the current SOC content (left) and the target value for the SOC content (right) across Europe.

3.2 Target surpluses for the base cations: Ca, Mg and K

Base cations play an important role in soil by providing essential plant nutrients (in particular Ca, Mg and K), but also through their buffering potential moderating soil acidification from natural processes and anthropogenic inputs. Of the various cations needed by plants, K is often limited whereas deficiencies of Mg and Ca can occur at very low pH and situations with long-term negative surpluses, i.e. where the soil has been mined or leaching losses resulted in a loss of cations from the topsoil.

Soil acidification is generally indicated by a decrease in soil pH, the negative logarithm of proton (H^+) concentration in soil solution. Soil pH serves as a key predictor of soil biological, chemical and physical processes, directly affecting plant growth and development. Generally, a soil pH between 6.0-7.5 leads to an optimal availability of most nutrients for plant uptake (Läuchli and Grattan, 2012; FAO, 2021b). Studies have revealed that the yield of common field crops declined by 5-10% when soil pH decreases to 5.0 or lower (Holland et al., 2019; Zhu et al., 2020). Soil pH is a measure of only the intensity of H^+ activity. The decrease in soil pH is much smaller than the amount of external H^+ input due to the presence of acid buffering substances within the soil. The amount of acid buffering substances in the soil is defined as the acid neutralizing capacity (ANC). Both depletion of basic components (e.g. neutralized by external H^+ input or removed by crop uptake and leaching) and accumulation of acidic components can deplete the ANC and lead to soil acidification.

Acid buffering processes and the buffering capacity of soils differ as pH changes (van Breemen et al., 1983; Chadwick and Chorover, 2001). Apart from rate limited silicate weathering, which plays a role in all pH ranges, different buffer reactions play a role in specific pH ranges as defined by Ulrich (1981). When soil pH drops from 8 to around 3.5, the buffering system generally goes through a carbonate buffer system, a base cation exchangeable buffer system, and a hydroxy aluminium and a hydroxy iron buffer system. The carbonate buffer system in a calcareous soil has a strong buffering capacity, being about 700 keq H^+ ha^{-1} for 50 cm soil depth per 1% $CaCO_3$ content, assuming a bulk density of 1500 kg m^{-3}. As an example, it takes about 50 years to consume 1% calcium carbonate under natural conditions up to this depth (De Vries and Breeuwsma, 1986). Therefore, calcareous soil can generally maintain a high soil pH of 7.0–8.5 for a long period despite elevated H^+ inputs. However, when the carbonate in the soil is exhausted, the soil pH drops generally below 7 and soils enter the base cation exchange buffer system. In this pH range, H^+ production is partially neutralized by base cation weathering from primary silicate minerals and mainly by exchange of protons against base cations adsorbed on clay and organic matter (called the exchange complex). The acid buffer capacity of the exchangeable base cation pool depends on the base saturation level of the cation exchange capacity, which in turn is determined by the clay and organic matter content, which varies mostly between 50-200 meq kg^{-1}. Assuming a bulk density of 1500 kg m-3, this implies a total buffer capacity of 375-1500 keq ha^{-1} for 50 cm soil depth at full base saturation. When this pool is depleted the pH gradually drops from ca 7.0 to ca 4.5 (Ulrich, 1991). Below pH 4.5, the soil enters the buffer system of aluminium hydroxide, where the buffering capacity depends on the amount of Al oxides which varies mostly from 50-300 meq kg^{-1}, being equal to 375–2250 keq ha^{-1} for 50 cm soil depth. This indicates that the main acid buffering systems of both calcareous soil (pH > 7.0) and very strongly acidic soil (pH < 4.5) generally have a strong buffering capacity, where the soil pH is insensitive to H^+ input, whereas the pH of soils in the base cation exchange system (pH 4.5–7.0), being common for agricultural soils, is more sensitive to H^+ addition.

3.2.1 Target surpluses for soil health and crop production

To maintain the capacity of soils to buffer acidity and to supply sufficient cations for crop production, the target values for the soil surplus of Ca, Mg and K can be derived from a given target pH, and the associated base saturation. We use an empirical relationship between pH and base saturation (BS) derived from various field experiments being applicable in the pH range between 4 and 6.5, being in line with previous studies.

$$BS_{crit} = min(100, (pH_{crit} - 4)/0.025) \quad (3.3)$$

Where BS_{crit} is the critical base saturation (%) at a desired critical pH value required for crop production, and pH_{crit} refers to the pH in soil solution, being set at 5.5 for most agricultural crops.

The desired BC surplus can therefore be calculated as:

$$BC_{sp\_crit} = (BS_{crit} - BS_{act})/(CEC * \rho * D * 10) \qquad (3.4)$$

where CEC is cation exchange capacity (mmol.kg^{-1}), ρ is bulk density of the soil (kg m^{-3}) and D is soil thickness (m), and 10 is the conversion factor from mmol m^{-2} yr^{-1} to mol ha^{-1} yr^{-1}.

Given the positive contribution of base cations to the buffering capacity of the soil, we recommend to avoid a decline during the duration of a crop rotation plan. That implies that for situations where BS_{act} is higher than BS_{crit}, this situation is acceptable on the annual scale but should be avoided on the duration of the crop rotation.

The fraction of Ca, Mg, K in the release from or accumulation of base cations on the adsorption complex is set equal to the fraction of these elements (Ca/BC, Mg/BC, and K/BC) on the adsorption complex. The target value for the surplus of the base cations is thus proportioned over Ca, Mg, and K with the initial respective fractions on the adsorption complex being derived from data and assumed to stay equal over time. When the cation occupation of the CEC is unknown, then the cation fractions are set at 0.7 for Ca, 0.2 for Mg, and 0.1 for K. This leads to the following critical surpluses for Ca, Mg and K:

$$Ca_{sp\_crit} = BC_{sp\_crit} * Ca_{cec} \qquad (3.5)$$

$$Mg_{sp\_crit} = BC_{sp\_crit} * Mg_{cec} \qquad (3.6)$$

$$K_{sp\_crit} = BC_{sp\_crit} * K_{cec} \qquad (3.7)$$

where Ca_{cec}, Mg_{cec} and K_{cec} represent the occupation of the CEC (being a fraction, -) for Ca, Mg and K, respectively. Critical targets for sodium (Na) can be quantified similarly, but this cation is not implemented in the Nutribuget project.

In calcareous soils, base saturation is set equal to 100% and the change in base saturation is assumed to be negligible since the acid production rate is fully counteracted by the dissolution of $CaCO_3$. In these soils the desired surplus is set equal to zero.

3.2.2 Critical surpluses for water quality

Given the fact that the loss of Ca, Mg or K to ground or surface water is not leading to adverse environmental impacts, there is no need to derive critical surpluses in view of water quality.

3.2.3 Spatial variation in CEC and soil pH

As the soil pH and the CEC strongly control the base saturation and the pool size of the cations Ca, Mg and K. There is considerable variation across Europe in these properties as illustrated below.

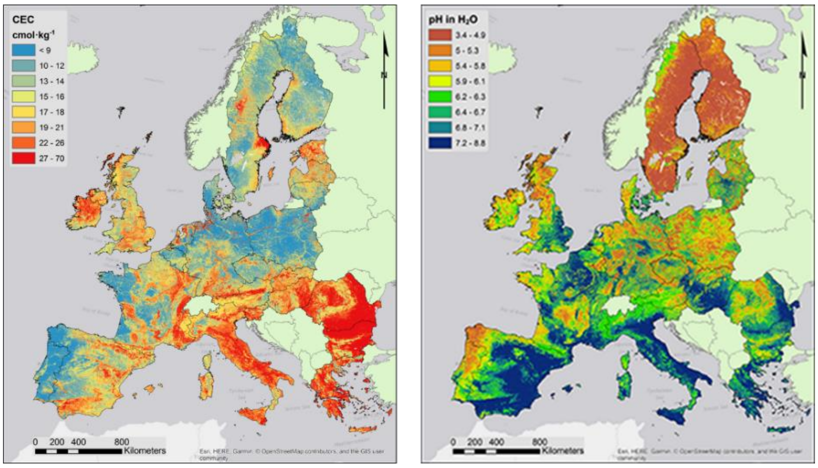

Figure 5 Spatial variation in CEC and soil pH determining the critical surpluses of Ca, Mg and K (Ballabio et al., 2019).

3.3 Target surpluses for sulphur

Sulphur plays a crucial role in synthesizing proteins, chlorophyll, enzymes, and vitamins, as well as influencing general metabolic and photosynthetic mechanisms. In the early 1950s, S deficiency was only noted in specific soils, but now it is becoming universally deficient (Shamra et al., 2021). The amount of plant-available S in the soil has decreased by 34–86 % between 2000 and 2020, leaving crop production at risk. The reasons for widespread S deficiency include lower industrial atmospheric deposition, stricter environmental laws, prevailing management practices such as selection of high-yielding varieties, increased use of low S fertilizers, decreased use of S containing fungicides and insecticides, and in some cases reduced tillage intensity.

The ability of the soil to supply S is based on biological, chemical, and physical transformations. Sulphur exists in both organic and inorganic forms in the soil. Organic S refers to S that is integrated into organic compounds while inorganic S exists as simple ions or small inorganic molecules. Usually, organic S concentrations in soils are positively correlated with organic matter.

3.3.1 Target values for soil health and crop production

Sulphur bioavailability in grasslands and croplands can be calculated from carbon decomposition and the C-to-S ratios. Currently there are no guidelines regarding optimum S levels in soil or the capacity of soils to supply them. Best agronomic practices for S would imply that crop S removal is replaced by fertilizers while also accounting for the unavoidable S leaching losses. In that case the desired target surplus for S in view of crop production and soil health can be defined as.

$$S_{sp\_crit} = S_{leaching} = 0.001 * Q_{gw} * S_{ss} \quad (3.8)$$

where S_{sp_crit} refers to the target S surplus in view of soil health and crop production (kg S ha^{-1}), Q refers to the water flux leaching out of the topsoil (m^3 year^{-1}) via leaching to groundwater and S_{ss} is the S concentration in soil solution (mg L^{-1}) being derived from an empirical Freundlich equation accounting for the impact of soil pH, and the factor 0.001 is the conversion from g ha^{-1} to kg ha^{-1}.

3.3.2 Critical surpluses for water quality

Given the fact that leaching of S does not lead to environmental problems, there is no need for critical values for S surplus in view of surface water and groundwater.

3.4 Critical values and target values for surpluses of Cu, Zn and Cd

The NutriBudget project focuses on three metals: copper (Cu), zinc (Zn) and cadmium (Cd). These metals have always been present in soil given the parent material and natural inputs from atmospheric deposition or sedimentation along rivers. Differences in background levels can be substantial at the national level (e.g. Mol et al., 2012) as well as at EU level (Reiman et al., 2014) and can be linked to negative effects on main ecosystem functions. During the 20th century, metal inputs increased due to industrialisation and intensification of manure, fertilizer and biosolid use in agriculture. The dominant inputs for Cd came from atmospheric deposition and for Zn and Cu came from agriculture. Major contributors to the inputs in agriculture include animal manure (Cu and Zn), sewage sludge (particularly lead, and to a lesser extent Zn), compost (Zn) and inorganic fertilizers (Cd). Inputs in agricultural systems via manure and fertilizers are however highly variable over space whereas inputs from sludge strongly depend on the existing regulations and are reflected in clear geographical differences in soil concentrations of the three metals (Figure 6).

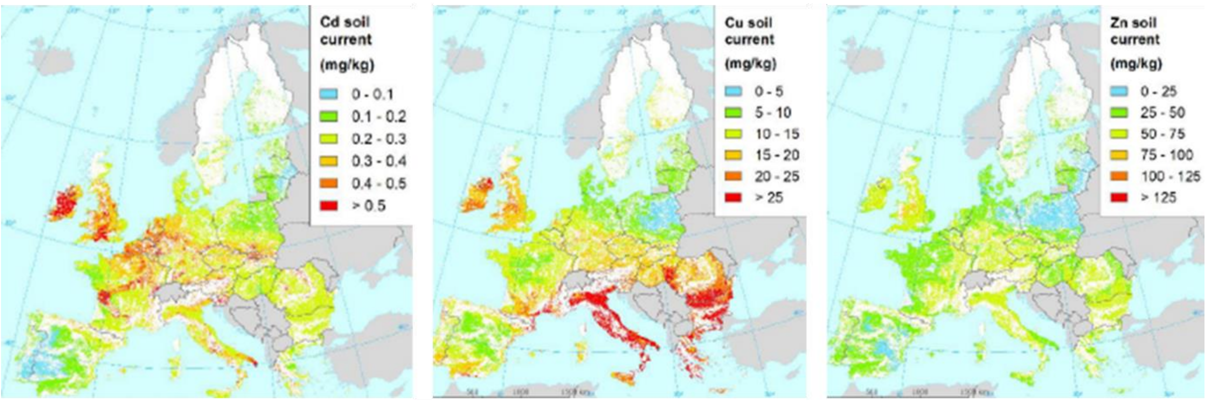

Figure 6 Current concentration of cadmium (Cd), copper (Cu) and zinc (Zn) in the topsoil across EU-27 member states, based on the combined data from the GEMAS and FOREGS database (Lado et al., 2008; Reiman et al., 2014).

The maps in Figure 6 reveal for example that Cd in agricultural soils strongly reflects levels of industrialization with higher concentrations in West European countries and parts of Poland, and lower levels in rural areas of Spain and several Baltic countries. For Zn the spatial variation is less pronounced with only small areas with high concentrations reflecting historical high inputs. Higher levels of Cu were found in south-eastern part of Europe.

Adverse impacts of high metal inputs and associated metal concentrations in the soil occur in view of soil health (i.e. abundance and activity of soil organisms), crop production and crop quality, and water quality in view of its use for drink water or aquatic biodiversity. To avoid adverse impacts of metals on these compartments, the quality of soil is currently regulated via controls on the inputs as well via legal limits or thresholds in the soil. A description of the existing regulations and associated frameworks is described comprehensively in De Vries et al. (2022). However, given that Zn and Cu are essential nutrients and may limit plant growth if deficient, reducing the inputs of all three metals is not always desirable, except for Cd. This implies that for these metals we not only consider critical limits that should not be exceeded (in view of toxicity) but also critical limits below which serious crop decline will occur. While upper critical limits are quite common across the EU (though the actual values might differ between countries), information on lower critical limits is rather limited.

The critical metal surplus in view of soil quality (biodiversity), water quality and (healthy) crop production can be estimated via:

$$X_{sp\_crit} = X_{le\_gwsw\_crit} = \left(Q_{gw} + Q_{ssro}\right) * X_{crit} \qquad (3.9)$$

where $X_{le\_gwsw\_crit}$ is the total leaching given critical metal concentrations, X_{crit} represents the metal concentration (Cd, Zn, Cu) in soil solution (ug L^{-1}) and Q the total water flux leaching via leaching and subsurface runoff out the soil profile (m^3 $year^{-1}$). Note that this calculation is applied after calculation of the critical concentration in view of soil health (section 2.4.1), water quality (section 2.4.3) and crop quality (for Cd, section 2.4.2).

The target metal surplus in view of soil fertility (Cu and Zn, being minor nutrients) can be estimated via:

$$X_{sp\_crit} = (X_{act} - X_{crit}) * D * \rho * 0.01/T \qquad (3.10)$$

where X_{act} refers to the actual metal content in the soil (in mg kg^{-1}, with X being either Cd, Zn or Cu), X_{crit} refers to the critical metal content in the soil (mg kg^{-1}) in soil, D refers to the depth of the top soil (m), ρ refers to the bulk density of the soil (kg m^{-3}), and 0.01 is a unit correction from mg kg^{-1} to kg ha^{-1}, and T refers to the time period (in years) to reach the desired target. Note that this calculation is applied after calculation of the target soil concentration for Cu and Zn (see section 2.4.2).

A key aspect related to the risk of Cd, Cu and Zn present in soil (or water) can pose is the fact that their impact is largely related to their actual availability for crop uptake or leaching rather than the total concentration (in soil or water). Here we define availability as the degree to which these metals can be taken up by crops, soil organisms or leached to the ground- and surface water. For most metals the availability strongly depends on a combination of type of input and soil properties like pH and the amount (and quality) of organic matter and clay. This implies that the critical surplus for metals will vary depending on soil texture and soil acidity.

3.4.1 Critical surpluses for soil health

Critical reactive soil metal concentrations in view of ecotoxicological effects on soil organisms were based on No Observed Effect Concentrations in laboratory experiments. In the approach, it is assumed that apart from the hard-bodied invertebrates, where soil ingestion is the major intake route, soil solution is the major pathway for metal impacts on all soil organisms and plants. This assumption is certainly valid for plants and microorganisms and for invertebrates living in soil water, such as nematodes, but is also a reliable assumption for soft-bodied invertebrates living in soil, such as earthworms.

As a threshold for ecotoxicological effects on soil biota, critical reactive metal concentrations as a function of soil organic matter (SOM) and pH were used, as derived by Lofts et al. (2004) and De Vries et al. (2007). Their analysis resulted in the following critical limits for three metal ion concentrations:

$$Cd_{crit\_soilhealth} = 1e6 * M_{Cd} * 10^{-0.43*pH_{ss}-5.66} \qquad (3.11)$$

$$Cu_{crit\_soilhealth} = 1e6 * M_{Cu} * 10^{-1.21*pH_{ss}-2.57} \qquad (3.12)$$

$$Zn_{crit\_soilhealth} = 1e6 * M_{Zn} * 10^{-0.34*pH_{ss}-4.66} \qquad (3.13)$$

where $Cd_{crit\_soilhealth}$, $Cu_{crit\_soilhealth}$ and $Zn_{crit\_soilhealth}$ are the reactive metal concentrations in soil solution being simulated with the Nutrifarm models (in mg L^{-1}), and M_{Cd} , M_{Zn}, M_{Cu} are the molar mass (112.414 g mol^{-1} for Cd, 65.35 g mol^{-1} for Zn and 63.55 g mol^{-1} for Cu), pH_{ss} represents the pH in soil solution (and can be estimated from soil pH measurements via extract methods, see below) and SOM the soil organic matter content (%). The multiplication with 1e6 is done to convert the mass unit from nanomolar (nM) to mg L^{-1}.

When the pH is measured in a soil extract such as water, 0.01M $CaCl_2$ or 2M KCL, then the pH in soil solution can be derived via simple linear transformations (for details, see De Vries et al., 2007), and vice versa via:

$$pH_{ss} = 0.88 * pH_{cacl2} + 1.32 \qquad (3.14)$$

$$pH_{ss} = 1.05 * pH_{h20} - 0.28 \qquad (3.15)$$

$$pH_{ss} = 0.97 * pH_{kcl} + 0.62 \qquad (3.16)$$

3.4.2 Critical surpluses for Cd for crop quality and target Cu and Zn surpluses for crop production

Crop quality: For Cd no biological function is known, and adverse health impacts of exposure to Cd are well established, both for humans and for animals. In general, Cd intake through food is the most important route of exposure to Cd for non-smokers. Food quality criteria, combined with soil–plant relationships, can be used to derive critical soil limits, thus avoiding the use of a detailed human exposure model while still using the concept of acceptable daily intakes. Critical limits for Cd in food and drinking water have been set at 0.2 mg kg^{-1} and 3 µg L^{-1} respectively, following European food quality criteria and standards of the WHO. These limits are implemented in Eq.3.9 to derive the critical Cd surplus in view of healthy crop production.

Because metal uptake by crops is plant specific, the kind of crop influences the derived limit for soil. It is thus necessary to derive relationships for the most sensitive crops to assess critical soil metal concentrations. For Cd, these functions have been derived for grass, maize, wheat and lettuce, and the critical metal concentration in soil (mg kg^{-1}) can be estimated as a function of the critical limit for the crop quality and the associated soil properties controlling the metal availability. As such the critical *total* metal content in soil (mg kg^{-1}) can be derived as:

$$Cd_{crit\_soil\_grass} = \left(10^{0.17-0.12*pH_{kcl}-0.28*\log SOM-\log Cd_{grass\_crit}}\right)/-0.49 \quad (3.17)$$

$$Cd_{crit\_soil\_maize} = \left(10^{0.9-0.21*pH_{kcl}-0.32*\log clay-\log Cd_{maize\_crit}}\right)/-1.08 \quad (3.18)$$

$$Cd_{crit\_soil\_wheat} = \left(10^{0.35-0.15*pH_{kcl}-0.39*\log SOM-\log Cd_{wheat\_crit}}\right)/-0.76 \quad (3.19)$$

$$Cd_{crit\_soil\_lettuce} = \left(10^{2.55-0.33*pH_{kcl}-0.19*\log clay-0.39*\log SOM-\log Cd_{lettuce\_crit}}\right)/-0.85 \quad (3.20)$$

where pH is determined in a KCl extraction, and the clay and soil organic matter content are in units of percentage. The critical limit for the harvested crop content is expressed in the unit mg kg^{-1} and can be set at a value of 0.2. Note that the critical Cd content for grass should be determined for animal health in view of the Cd uptake via fresh feed intake by animals (see e.g. De Vries et al., 2007).

Similarly, critical *total* metal contents (in mg kg^{-1}) can be derived for zinc per crop as follows (de Vries et al., 2008) when critical values for the plant content are known.

$$Zn_{crit\_soil\_grass} = \left(10^{2.06-0.09*pH_{kcl}-1.05*\log clay+1.09*\log SOM-\log Zn_{grass\_crit}}\right)/-0.41 \quad (3.21)$$

$$Zn_{crit\_soil\_maize} = \left(10^{3.05-0.31*pH_{kcl}-0.61*\log clay-\log Zn_{maize\_crit}}\right)/-0.64 \quad (3.22)$$

$$Zn_{crit\_soil\_wheat} = \left(10^{1.32-0.06*pH_{kcl}-0.24*\log clay-\log Zn_{wheat\_crit}}\right)/-0.45 \quad (3.23)$$

$$Zn_{crit\_soil\_lettuce} = \left(10^{2.76-0.21*pH_{kcl}-0.26*\log clay-\log Zn_{lettuce\_crit}}\right)/-0.34 \quad (3.24)$$

where pH is determined in a KCl extraction, and the clay and soil organic matter content are in units of percentage. The critical limit for the harvested crop content is expressed in the unit mg kg^{-1} and can for Zn be set at a value of 100 mg kg^{-1}. For animal health the maximum acceptable Zn content has been set at 500 mg kg^{-1}. In ration recommendations for dairy cows an optimum range of 25 to 50 mg kg^{-1} is usually recommended. When no critical Zn values for crops can be derived it is also possible to follow the build-up-and-maintenance approach where critical soil levels have been derived from field trials, where critical levels might vary between 1.8 and 13 mg kg^{-1} for Mehlich extractable zinc.

Similarly, critical *total* metal contents (in mg kg^{-1}) can be derived for copper per crop as follows (de Vries et al., 2008):

$$Cu_{crit\_soil\_grass} = \left(10^{1.41-0.18*pH_{kcl}-0.65*\log SOM-\log Cu_{grass\_crit}}\right)/-0.83 \quad (3.25)$$

$$Cu_{crit\_soil\_maize} = \left(10^{0.0+0.07*pH_{kcl}-0.11*\log clay-\log Cu_{maize\_crit}}\right)/-0.20 \quad (3.26)$$

$$Cu_{crit\_soil\_wheat} = \left(10^{0.65-0.03*pH_{kcl}-\log Cu_{wheat\_crit}}\right)/-0.16 \quad (3.27)$$

$$Cu_{crit\_soil\_lettuce} = \left(10^{0.7-0.06*pH_{kcl}-\log Cu_{lettuce\_crit}}\right)/-0.42 \quad (3.28)$$

where pH is determined in a KCl extraction, and the clay and soil organic matter content are in units of percentage. The critical limit for the harvested crop content is expressed in the unit mg kg^{-1} and can be set at a value of 24 mg kg^{-1}. For grassland the critical Cu content depends also on animal category and the ratios between copper with zinc and molybdenum. In the ration assessment the optimum range varies between 12 and 15 mg kg^{-1} whereas the maximum may increase up to 40 mg kg^{-1} for cows.

The negative coefficients in these functions for Cd, Zn and Cu imply that an increase in pH, clay or organic matter content leads to a lower metal content in the soil. This result is in agreement with the impact of the aforementioned soil properties on the availability of metals in soil.

The critical surpluses following these critical soil contents for Cd, Zn and Cu can be estimated via Eq. 3.10 introduced at the start of this section.

3.4.3 Critical surpluses for water quality

The critical limit for the metal concentration in groundwater and surface water has been set at 3 µg Cd L^{-1}; 75 µg Cu L^{-1} and 800 µg Zn L^{-1}. The associated critical surpluses to protect the quality of water systems can be calculated by multiplying the volume of water leaching out of the soil with the critical concentration, using Eq 3.9.

3.4.4 Spatial variation in critical Cd, Cu and Zn contents

The spatial variation of critical Cd, Cu and Zn contents in the soil in view of soil health can be illustrated via earlier work published by De Vries et al. (2023).

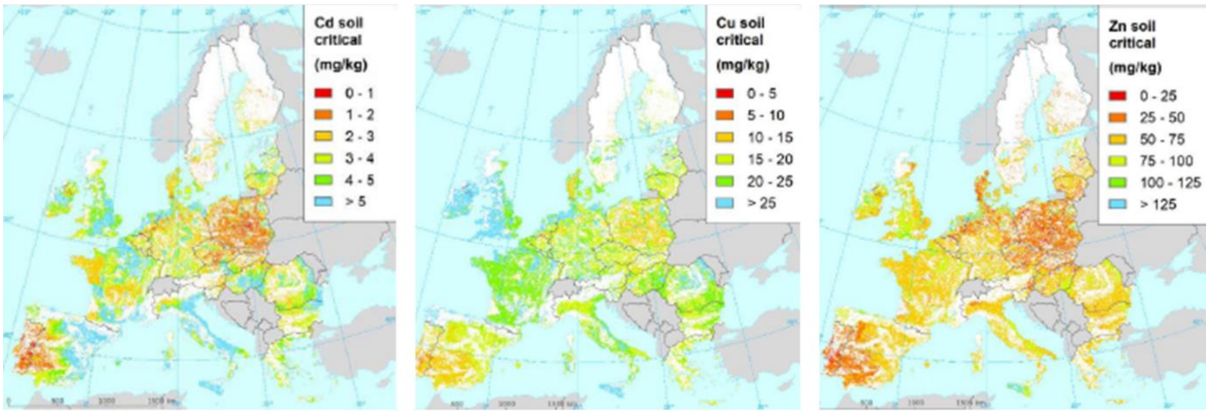

Figure 7 Spatial variation in critical concentrations (thresholds) in view of impacts on soil biodiversity for cadmium (left), copper (mid) and zinc (right) for the EU-17 countries. (Source: De Vries et al., 2022).

4. Outlook

4.1 Setting targets

In Europe, N deposition hampers the achievement of a favourable conservation status within regions being defined as Natura 2000 sites in view of the Birds and Habitats Directive. In terrestrial ecosystems, excessive levels of reactive forms of atmospheric N leads to the loss of biodiversity by favouring nutrient-demanding species. High levels of reactive atmospheric N originate from the emissions of ammonia and nitrogen oxide, and policies have been implemented to reduce these emissions to improve the ecological condition of nature areas. Even though the word nitrogen is not explicitly mentioned in the Birds and Habitats Directive, the current ecological condition is strongly controlled by N deposition, implying that a reduction in N deposition is needed for all terrestrial ecosystems where current N deposition exceeds a critical N deposition level (critical load). This in turn requires a strong reduction in NH_3 and NO_X emissions. Despite a general reduction in ammonia emissions by 24% since 1990, predictions up to 2020 indicate that the risk of exceeding critical loads remains high, irrespective of the implementation of current policies and measures to reduce N emissions (Eurostat, 2023). In the current study we describe a spatially explicit N balance approach to define critical limits and target values for the N surplus to protect water quality and aquatic and terrestrial biodiversity. This implies that for each unique combination of soil type, land use and geohydrological settings a target is available for the acceptable N surplus.

Increased N and P concentrations in surface water lead to eutrophication, characterized by excessive plant and algal growth and oxygen depletion, which negatively affects aquatic biodiversity. Critical concentrations for dissolved total nitrogen, indicating a risk for eutrophication, lie between 1.0 and 2.5 mg N L^{-1}, though this range goes as low as 0.3 mg N L^{-1}, following the guidelines of the Water Framework Directive (WFD) to achieve good water quality. Critical N runoff rates from agricultural soils can subsequently be calculated by multiplying the critical N concentration in runoff with the precipitation surplus, multiplied with runoff fraction as done by de Vries et al. (2023). With respect to P runoff, the P concentrations in surface waters should stay below the upper limit of good ecological status (EC, 2000), ranging from 0.04 to 0.53 mg L^{-1}. These criteria are here used to derive spatially explicit critical thresholds for both N and P budgets for all agricultural systems in Europe.

The critical nitrate concentration in groundwater was set to the WHO drinking water limit of 50 mg NO_3 L^{-1}, which is also the threshold stated in the EU Nitrates Directive (EC, 1991). As with runoff, critical N leaching rates from agriculture can be calculated by multiplying the critical N concentration in leachate to groundwater with the share of precipitation surplus leached to groundwater (see De Vries et al., 2021). Also here, these critical thresholds can be translated into acceptable N surpluses and N inputs (see above), as illustrated in chapter 2. The leaching of carbon via dissolved organic carbon or other nutrients needs to be minimized as well, but no critical concentrations have been defined yet given that dissolved organic carbon has limited effects on human or environmental quality.

Regarding agronomic soil health, target values have been identified for phosphorus, base cations (Ca, Mg and K), sulphur and three metals (Cu, Zn, Cd). No explicit thresholds for soil N are given due to its dynamic nature, though we add an optional target value for the organic N inputs to improve the capacity of soils to supply N throughout the growing season. In the classic agronomic concept of “build-up-and-maintenance” the desired surplus (or budget or accumulation) of phosphorus, potassium, calcium, magnesium, sulphur, copper and zinc depends on the size of the plant available nutrient pool in soil. Agronomic thresholds do not exist for pollutants like Cd, whereas critical inputs or surpluses can be defined based on its concentration in soil solution and the associated crop uptake on the one hand, and on the impact on soil organisms on the other. For Cu and Zn, using long-term field experiments, critical soil concentrations have been determined for various crops to derive the optimum soil nutrient concentration where the crop yield is not limited by deficiencies. When the soil concentration is in the optimum range, then it is sufficient to replace the crop nutrient uptake by fertilization (so the surplus is zero). In the case that the soil is deficient, fertilizer input need to exceed plant uptake in order to create a build-up of the soil nutrient pool. In the case that the soil is enriched with nutrients above the agronomic optimum, then the nutrient input should be lower than crop removal, implying a net mining of the soil nutrient pool. The optimum soil nutrient level might deviate across soil types, climatic conditions and crops, as shown in chapters 2 and 3. The methodology to assess this optimum soil nutrient level also varies across countries.

Targets for SOC have been derived as a function of clay content for mineral soils (Körschens et al., 1998). Using long-term experiments starting in 1902, they proposed critical limits for SOC for optimum crop production, ranging from 0.5% SOC at 4% clay to 1.75% SOC at 38% clay, where the beneficial impacts of SOC become less for soils being high in clay. This approach takes into account the fact that one uniform threshold for SOC is likely not appropriate (Goulding et al., 2013; Loveland and Webb, 2003; Oldfield et al., 2019) and that clay particles stabilizes and protect organic matter in soil from decomposition (Goulding et al., 2013). Note that SOC targets for peat soils and drained organic soils used for agriculture might be derived differently since SOC is already at maximum but the losses are highly dependent on drainage levels. Due to the current focus on mineral soils, no thresholds for SOC surpluses have been defined yet. On the other hand one can discuss whether this high SOC as such is needed for agronomic soil quality or mainly to mitigate climate change. This implies separate SOC targets in view of agronomy and C sequestration for climate mitigation.

4.2 Defining KPI thresholds

To assess the actual farm performance in view of agronomic and environmental targets, an integrative KPI framework has been designed to monitor the transition from the current to the desired status to have optimised farming systems in equilibrium with maximum agricultural performance and minimal environmental pressure (Ros et al., 2023a). As such, this framework will guide the actual decision support as well the identification of appropriate roadmaps to reach the desired status for soil surpluses of carbon and nutrients in view of targets for soil quality, water quality, climate, biodiversity and crop production. The KPIs selected include site specific thresholds for carbon and nutrient budgets in view of agronomic and environmental targets.

As introduced and explained in the report "*Overview of existing indicators used in national and European policies and market initiatives in relation to agronomic and environmental aims*" (D3.1) the NutriKPI framework distinguishes between pressure, effect and performance indicators. **Pressure indicators** are indicators related to human activities/external factors that influence the agro-ecosystem, such as nutrient inputs and management measures, including crop, soil, nutrient (and manure) management. These pressure indicators affect the associated nutrient flows and properties of the agro-ecosystem. **Effect indicators** (including both state and impact indicators) are the agro-ecosystem properties that change due to the impact of altering nutrient inputs/management measures, such as nutrient uptake such as nutrient uptake, surpluses, losses and pools. **Performance indicators** reflect the performance of the agro-ecosystem for the associated nutrient uptake, surpluses, losses and pools in view of the agronomic and environmental goals that need to be achieved.

The critical limits and targets developed in this study can be used to transform the effect indicators into KPIs by simply quantifying the distance to target. In the chapters 2 and 3, spatial explicit functions are developed for the carbon and nutrient (N, P, K, Ca, Mg, S, Cu, Zn and Cd) budgets in view of targets for soil health, crop production, air quality, and water quality. Via these targets we aim to reduce the nutrient emissions from agriculture with spatially explicit targets for groundwater quality (originating from the Nitrates Directive), surface water quality (originating from the Water Framework Directive), ammonia emissions (originating from the Birds and Habitats Directive), and greenhouse gas emissions (originating from the ambitions to reduce the emissions from agriculture and to mitigate part of the greenhouse gases (GHG) emissions by carbon sequestration, the Paris Agreement). At the same time there is the objective to produce sufficient food (an agronomic and economic objective) and to maintain soil health whereas the latter objectives are less quantitative in policy documents but clearly defined in agronomic recommendation systems.

This results for each region in five farm and field specific targets:

1. a critical N surplus in view of critical nitrate concentrations in groundwater
2. a critical N and P surplus in view of critical N and P concentrations in runoff to surface water

3. a critical N surplus in view of maximum NH_3 emissions from soil, storages and stables
4. a target surplus for N, P, K, S, Ca, Mg, Cu and Zn in view of soil quality and crop production
5. a critical target C surplus in view of soil health and the desired C sequestration for mitigating climate change

In WP2 this proposal will be implemented in the process-based models so that for each field and site across Europe, specific targets are defined for the desired carbon and nutrient budgets.

5. Conclusions

This report describes the derivation of thresholds (being critical limit values or target values) for carbon and nutrient (N, P, K, Ca, Mg, S, Cu, Zn and Cd) surpluses in European agriculture, thereby defining the local and regional targets to be used in the NutriKPI framework of the NutriBudget project. This results for each region in five specific targets:

6. a critical N surplus in view of critical nitrate concentrations in groundwater
7. a critical N and P surplus in view of critical N and P concentrations in runoff to surface water
8. a critical N surplus in view of maximum NH_3 emissions from soil, storages and stables
9. a target surplus for N, P, K, S, Ca, Mg, Cu and Zn in view of soil quality and crop production
10. a critical target C surplus in view of soil health and the desired C sequestration for mitigating climate change

Via these targets we aim to reduce the nutrient emissions from agriculture with spatially explicit targets for groundwater quality (originating from the Nitrates Directive), surface water quality (originating from the Water Framework Directive), ammonia emissions (originating from the Birds and Habitats Directive), and greenhouse gas emissions (originating from the ambitions to reduce the emissions from agriculture and to mitigate part of the greenhouse gases (GHG) emissions by carbon sequestration, the Paris Agreement). At the same time there it allows one to apply nutrients appropriately in order to produce sufficient food (an agronomic and economic objective) and to maintain soil health.

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Annexes

Annex 1 Critical soil contents

Below a series of equations are described for the metals linking critical thresholds to associated (reactive or total) metal contents in the soil.

Critical soil contents in view of soil biodiversity

As a threshold for ecotoxicological effects on soil biota, critical reactive metal concentrations as a function of soil organic matter (SOM) and pH were used, as derived by Lofts et al. (2004) and De Vries et al. (2007). Using empirical relationships that relate the added metal content to the total metal content, a critical total metal content was calculated that subsequently can be compared with the actual soil content. The critical reactive metal concentration can be derived as a function of pH in soil solution and the SOM content as follows (for the derivation, see De Vries et al., (2007):

$$Cd_{crit} = 1e6 * M_{Cd} * 10^{0.33*pH_{ss}+1.0*\log SOM-10.32} \qquad \text{(A1.1)}$$

$$Cu_{crit} = 1e6 * M_{Cu} * 10^{0.02*pH_{ss}+0.68*\log SOM-7.54} \qquad \text{(A1.2)}$$

$$Zn_{crit} = 1e6 * M_{Zn} * 10^{0.14*pH_{ss}+1.07*\log SOM-8.56} \qquad \text{(A1.3)}$$

where Cd_{crit}, Cu_{crit} and Zn_{crit} are the reactive metal contents determined by a HNO_3 soil extraction method and being simulated with the Nutrifarm models (in mg kg^{-1}), and M_{Cd} , M_{Zn}, M_{Cu} are the molar mass (112.414 g mol^{-1} for Cd, 65.35 g mol^{-1} for Zn and 63.55 g mol^{-1} for Cu), pH_{ss} represents the pH in soil solution and SOM the soil organic matter content (%). The multiplication with 1e6 is done to convert the mass unit from g g^{-1} to mg kg^{-1}.

Critical soil contents in view of critical concentrations for water quality

The critical limit for the reactive Cd, Zn and Cu content in soil (mg kg^{-1}) in view of the critical limit in drinking water (Cd_{gw}, mmol L^{-1}, equalling to 3 µg Cd L^{-1}; 75 µg Cu L^{-1} and 800 µg Zn L^{-1}) quality can be estimated as a function of the soil properties controlling its availability, via:

$$Cd_{crit\_gw} = 1e3 * M_{Cd} * \left(10^{-4.85+0.27*pH_{h2o}+0.58*\log SOM+0.28*\log clay}\right) * Cd_{gw} \qquad \text{(A1.4)}$$

$$Zn_{crit\_gw} = 1e3 * M_{zn} * \left(10^{-4.51+0.45*pH_{h2o}+0.39*\log SOM+0.35*\log clay}\right) * Zn_{gw} \qquad \text{(A1.5)}$$

$$Cu_{crit\_gw} = 1e3 * M_{cu} * \left(10^{-3.55+0.16*pH_{h2o}+0.48*\log SOM+0.18*\log clay}\right) * Cu_{gw} \qquad \text{(A1.6)}$$

where pH is determined in a KCl extraction, and the clay and soil organic matter content are in units of percentage.

Optimisation of nutrient budget in agriculture

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