

Impacts of nutrients and heavy metals in European agriculture

Current and critical inputs in relation to air, soil and water quality



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Contents

Preface and acknowledgments	5
Summary	7
1 Introduction	11
1.1 Adverse impacts of agricultural intensification in Europe	11
1.2 Indicators for acceptable agricultural intensification related to fertilizer management	12
1.3 Policies aiming to reduce adverse impacts of agriculture on the European environment	13
1.4 Aim and approach of this report.....	16
2 Current and critical inputs of nitrogen and phosphorus in European agriculture	17
2.1 Nitrogen and phosphorus use in view of food production and environmental impacts	17
2.1.1 Nitrogen and phosphorus requirements and use in view of food production	17
2.1.2 Nitrogen and phosphorus limits in view of environmental impacts.....	18
2.1.3 Approaches for increasing nitrogen and phosphorus use efficiencies	21
2.2 Nitrogen and phosphorus budgets in European agricultural soils	22
2.2.1 The fate of nitrogen and phosphorous inputs	22
2.2.2 Nitrogen and phosphorus budgets and use efficiencies at EU level.....	24
2.2.3 Spatial variation in nitrogen and phosphorus budgets and use efficiencies	25
2.3 Critical nitrogen losses and inputs in view of environmental impacts and crop growth	29
2.3.1 Assessing critical nitrogen losses to air and water and critical nitrogen inputs	29
2.3.2 Critical nitrogen losses to air and water and exceedances by current losses	30
2.3.3 Critical nitrogen inputs and exceedances by current inputs	33
2.3.4 Needed increase in nitrogen use efficiencies	35
2.4 Critical phosphorus inputs in view of environmental impacts and crop growth	36
3 Current and critical inputs of heavy metals in European agriculture	39
3.1 Heavy metals in soils: concentrations, benefits and risks	39
3.1.1 Sources and presence of metals in soils	39
3.1.2 Assessment of heavy metals in soils: towards risk-based quality standards.....	40
3.2 Heavy metals budgets in European agricultural soils	45
3.2.1 The fate of metal inputs.....	45
3.2.2 Metal budgets at EU level	46
3.2.3 Spatial variation in metal budgets	51
3.3 Current and critical metal inputs in view of environmental impact	53
3.3.1 General	53
3.3.2 Critical metal concentrations and inputs and exceedances in view of soil biodiversity.....	55
3.3.3 Critical cadmium concentrations and inputs and exceedances in view of food quality.....	59
4 Evaluation and key messages	63
References	67

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Summary

Abstract

Fertilizer and manure management is relevant for food production but simultaneously causes environmental impacts, due to adverse effects on air, soil and water quality. Impacts on air and water quality are specifically caused by losses of ammonia to air and nitrate and phosphate to water in response to enhanced nitrogen (N) and phosphorus (P) inputs, leading to eutrophication of terrestrial and aquatic ecosystems. Impacts on soil quality are mainly due to the addition of heavy metals, which may cause impacts on soil biodiversity and, in case of cadmium, on food quality

This report describes the spatial variation in critical concentrations/losses and inputs of N and P and of the four most abundant heavy metals, i.e. cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn), and their exceedance by current inputs. The results allow to identify spatial hot spots for environmental impacts of nitrogen and phosphorus inputs for the four most abundant heavy metals, and the possibility to protect the environment by improved management. More specifically, the spatial variation in the needed increase in N use efficiency to reconcile food production with environmental protection is presented. It is shown that in some intensive agricultural production regions, crop production and environmental thresholds cannot be reconciled by an increase in NUEs only and a reduction in crop or livestock production is inevitable to fully protect the environment. It thus informs policy processes important for planning and guiding sustainable nutrient and soil management, such as the farm to fork strategy and the zero pollution strategy.

Extended summary

Agricultural intensification and its impacts on the environment

Agriculture is vital to Europe's prosperity. However, the intensification of livestock and crop production, associated with an increased use of fertilizers and manure has caused enhanced soil accumulation and losses of nitrogen (N), phosphorus (P) and metals, such as cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn) to air (nitrogen compounds only) and water. This has caused pollution of groundwater by nitrate, affecting drinking water quality, eutrophication and acidification of terrestrial ecosystems by enhanced nitrogen deposition, eutrophication of aquatic ecosystems (rivers, lakes, and seas) by diffuse inputs of N and P, soil biodiversity impacts from metal accumulation and potential impacts on food quality by Cd accumulation

Assessment of current and critical inputs and losses of nitrogen, phosphorus and metals

In this study, the spatial variation in the fluxes of N, P, Cd, Cu, Pb and Zn were determined in EU agriculture by using a land balance approach. Both N and P are major nutrients but losses to air and water affect air and water quality, thus affecting ecosystems and human health. Both Cu and Zn are minor nutrients but at high inputs, they may cause adverse impacts on soil biodiversity, whereas Cd and Pb are toxic metals that may lead to soil degradation, by affecting soil biodiversity and food quality. The spatially explicit input and output budgets were calculated with the INTEGRATOR model for approximately 40,000 unique combinations of soil type, administrative region, slope class and altitude class. As inputs, various statistics (mainly Eurostat and FAO) were used on the use of fertilizers, livestock numbers, and biosolids (compost and sludge) and on crop yields and crop areas, as well as EMEP deposition data, which were downscaled to the required resolution. The N, P and metal surplus were calculated as the difference between N, P or metal input and crop N, P or metal removal. The fate of the N, P and metal surplus, i.e. emissions to air (N only), soil accumulation or release and losses to ground water and surface water, were calculated with the INTEGRATOR model, accounting for variations in soil and climate which affect the biological and chemical processes that determine this fate. The base year used in this study for the current N, P and metal inputs is 2010, a year with detailed information on fertilizer use, livestock numbers, crop areas, crop yields. Despite the fact that this is now more than 10 years ago, the base line of 2010 is indicative for the current situation, since the change in inputs by fertilizer and manure is limited since 2010.

By combining soil data and climate data with environmental protection targets (thresholds), critical input levels were derived (i.e. the inputs that imply losses below environmental thresholds). For nitrogen, the following thresholds were used: (i) critical N deposition on natural ecosystems, (ii) a critical nitrate (NO_3) concentration of $50 \text{ mg NO}_3 \text{ l}^{-1}$ in leachate to groundwater and (iii) a critical N concentration of 2.5 mg N l^{-1} in runoff to surface water. For phosphorus, the long-term critical P input by fertilizer and manure was set equal to the P removal by the crop. For metals, available critical limits for food, water and soil organisms, from different existing regulations and studies, were converted to soil property dependent critical metal concentrations (soil-based quality standards), which were then used to calculate critical metal inputs.

Current and critical nitrogen inputs and losses in view of air and water quality impacts

Results of the European average current N budget (base line year 2010) show a total N input to European agricultural soils of $145 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, mainly by fertilizer and fixation ($78 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), followed by manure and biosolids ($56 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), of which on average near 40% is lost (N use efficiency is near 60%), mainly in the form of ammonia emissions, N runoff and nitrate leaching. Current ammonia emissions and N runoff exceed critical ammonia emissions and critical N runoff on a large share of agricultural land. Highest exceedances occur in regions with intensive livestock systems and high N inputs, but also in regions with sensitive ecosystems (e.g., low critical N loads). European average critical N inputs not exceeding critical ammonia emissions are $100 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ and for surface water quality $83 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ being, respectively 31% and 43% lower than the current N inputs. For ground water, the exceedances are limited and the European average critical N input in view of ground water quality is comparable to the average current N input of $145 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.

Nitrogen is, however, an essential nutrient in agricultural production and reducing N inputs implies a reduction in crop yield at current nutrient management practices. The difference in current N inputs and critical N inputs in space thus illustrates the tension between food production and environmental protection. The critical N inputs were, however, calculated by using the current nitrogen use efficiencies (NUE), defined as the ratio between N offtake (crop N removal) compared to the N input to soil. The tension between food production and environmental protection is further illustrated by the fact that increasing crop yields to target levels (80% of water-limited yield potentials) requires an estimated 27% increase in N inputs (from 145 to $185 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) using the current NUE.

In addition we thus calculated the spatial variation on the needed increase in NUE to reconcile food production with environmental protection. In regions where current N inputs, or N inputs required to obtain target yields, exceed critical inputs, crop production and environmental protection can only be reconciled by increasing NUE of agricultural production (i.e., unit of agricultural output per unit of N input). To achieve surface water quality targets without crop production losses, the European average NUE needs to increase from 0.64 to 0.78, but the required increases in NUE shows large spatial variation in Europe. The largest increases in NUE are required in Benelux, Poland, the Po valley in Italy and certain regions in Spain and Greece. In some regions, crop production and environmental thresholds can only be reconciled at NUEs > 90%, which is not considered feasible given that N is highly mobile and some N losses are unavoidable. Here, a reduction in crop or livestock production is inevitable to fully protect the environment.

Current and critical phosphorus inputs in view of food production and aquatic eutrophication

Unlike N, P is adsorbed in the soil. However, at steady state, P soil adsorption can be neglected and the long-term critical P input can be calculated as the sum of P uptake and a critical P loss to groundwater and surface water. By assuming that P losses by runoff and leaching are equal to P deposition, the critical P input by fertilizer and manure can be set equal to the P removal by the crop (equilibrium P fertilization) assuming target crop yields. The spatial variation in critical P inputs shows a needed decrease in P inputs in western Europe including Ireland, UK, the Netherlands, Belgium and

Luxembourg and Bretagne in France, while increased P inputs are needed in the remaining parts of Europe in view of crop P demand.

Current and critical metal concentrations and inputs in view of soil biodiversity

Current levels of cadmium and lead show a distinct spatial relationship with the degree of industrialization whereas copper and zinc levels in soil are more related to variation in natural (geogenic) levels of metals in soil. At EU level, zinc, copper and lead accumulate in soil whereas for cadmium the overall balance is slightly negative at EU-level. The regional variation in the net accumulation rate, however, is large and related to a combination of management (choice of fertilisers), soil type and climatic conditions. Leaching losses are low in Mediterranean areas due to a low net water loss from soil and neutral to high pH levels. This however also leads, on average, to a positive balance for most metals considered here in Mediterranean areas. In areas characterized by moderate to high rainfall and acid soils on the other hand, leaching losses of cadmium and zinc are high. For cadmium this results in a net negative balance (depletion) in large parts of north western EU members states despite elevated input levels to soil. For both copper and zinc, inputs in areas with intensive animal husbandry such as the Netherlands, Denmark, Brittany (France) and northern Italy (Po area) are high leading to a substantial accumulation rate compared to other areas in the EU.

Current cadmium and lead levels in soil, however, are largely below the critical threshold levels in view of effects on biodiversity. On the other hand, exceedance of the critical threshold level as percentage of the total area considered at EU level in soil is high for copper (23%) and zinc (18%). Especially in Mediterranean countries current copper levels are particularly high when compared to the critical threshold level for soil biodiversity.

Current inputs to soil for cadmium are expected not to lead to an exceedance of the soil critical threshold level for biodiversity. For lead, copper and zinc current inputs exceed critical inputs in 48% and 70% of the total surface area. This implies that where current levels in soil do not exceed critical soil threshold levels, this will occur in the regions where critical inputs are exceeded if current inputs remain unchanged. For lead this is concentrated in Mediterranean countries and largely related to the relatively low mobility of lead in soils. For zinc and copper exceedance of the critical inputs occur across most member states. This is due to either high inputs (in areas with intensive animal husbandry in north-western parts of the EU) or low outputs from soil (in Mediterranean countries).

Current and critical cadmium threshold levels and inputs in view of food quality

For copper and zinc critical limits for human food products are not in place. Only for animal feed such limits have been set but largely related to additives in feed. For lead, quality standards for food and feed that can be used as critical limit are available (EU 1881/2006, amended by commission regulation (EU) 2021/1323). However, the relation between soil Pb levels and those in crops grown on that soil are mostly of poor quality and not suitable to calculate reliable critical threshold levels in soil.

For cadmium, current (2010) concentrations in soil are largely below the critical threshold level in soil in view of food quality. Due to the rather mobile character of cadmium in soil, leaching losses are pronounced which also implies that current inputs do not exceed critical inputs in 96% of the total surface area used for agricultural crop production. Only in Spain, Greece and small parts of southern Poland, current inputs are such that with time the soil cadmium concentration can exceed the critical threshold cadmium concentration in view of food safety. In Poland this is predominantly due to high inputs whereas in Spain and Greece this is due to very low leaching rates which facilitate cadmium retention in the (top) soil.

1 Introduction

1.1 Adverse impacts of agricultural intensification in Europe

Agriculture is vital to Europe's prosperity. In 2012 the EU export market for agricultural products was 116 Billion € (EU, 2012). Over the last 50 years the EU Common Agricultural Policy (CAP) 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. Despite CAP reform there is a legacy of environmental damage from farming across Europe, including effects on ecosystems and human health. The cost of this to European society, plus restoration is immense, i.e. 38 Billion € per year for soil degradation alone (EEA, 2010a). The related problems and impacts are summarized below.

Pollution of water bodies: Agriculture is a major cause of pollution of water bodies (rivers, lakes, groundwater and seas) and aquatic ecosystems in many parts of Europe, by diffuse and point source inputs of nutrients (especially nitrogen (N) and phosphorus (P) from mineral and organic fertilisers), pesticides, pathogenic micro-organisms excreted by livestock, and organic pollutants from manure washed into waterways or leached from soils (Stark & Richards, 2008; WWDR3, 2009). Key impacts include the proliferation of algal blooms due to eutrophication, and loss of aquatic life from excessive nutrient levels, genetic defects and reduced fertility in fish from endocrine-disrupting chemicals, and environmental toxicity and human health risks from pesticide residues and metals, such as cadmium (Cd) and lead (Pb). Nitrate pollution in groundwater mainly arises from agriculture, requiring expensive measures to ensure compliance with drinking water standards (WWDR4, 2012). In areas with relatively high (ground)water tables and intensive agriculture, emission of metals like Cu and Zn from soil is believed to contribute significantly to surface water concentrations. In the NL for example, Zn exceeds surface water standard as defined by the Water Framework Directive in 40% of the Dutch waterbodies (Oste et al., 2018) .

Damage to ecosystem biodiversity: Europe's biodiversity is closely linked to agricultural practices, creating important agroecosystems across the continent. This biodiversity is at risk from intensification of agricultural production. This includes the emission of ammonia and related nitrogen deposition, causing eutrophication and acidification of non-agricultural ecosystems. It has reduced the number and range of wildlife, plant and microorganism species in impacted habitats, (EEA, 2010a). The conservation and sustainable use of biodiversity for food and agricultural production is critical to ensure productive agroecosystems and their services. However, reporting for the EU Habitats Directive shows that only 7% of habitats in agroecosystems have good conservation status, versus 17% for habitat types not related to agroecosystems (EEA, 2010a).

Degradation of soil quality and subsequent loss of function: Due to intensification of the land use in large areas across the EU during the last century, both related to agriculture, industrialization and urbanization, the pressure on soil has grown. Such developments will, at some point affect the capacity of the soil to fulfil specific functions. Relevant functions considered here in relation to agriculture include the production of food, conservation of soil biodiversity and protection of water quality.

Adverse impacts of agricultural management practices include salinization from irrigation, soil compaction due to use of heavy machinery, long-term contamination due to inputs of metals and organic micro-pollutants by fertilizers and pesticides, wind and water erosion due to poor management, soil acidification due to excess N inputs and loss of organic carbon due to enhanced mineralisation by tillage practices. Such changes in soil quality can cause a decrease in soil biodiversity

and soil fertility but also can enhance desertification (EEA, 2000). This is critical in Mediterranean Europe (Spain, Portugal, S. France, Greece and S. Italy), where 23% of the area has moderate-high sensitivity to desertification.

Soil carbon sequestration and climate change: Agriculture contributes ca. 20% of global annual greenhouse gas (GHG) emissions and is the largest emitter of non-CO₂ GHGs (N₂O and CH₄). Soil processes are critical in regulating biogeochemical cycles, organic matter decomposition and N transformation, which affect fluxes and cycles of C and N at the global scale. Soil has an important role in slowing climate change, via sequestration of atmospheric CO₂ into soil organic carbon, stored as biomass and soil organic matter (SOM) by plant growth. The soils of EU-27 Member States store ~79 billion tonnes of carbon, but this capacity is sensitive to climatic conditions, with a high risk that global warming will turn soils into a major source of GHGs. About 45% of European agricultural soils have a low to very low SOM content (0–2% organic carbon) and 45% have a medium content (2–6% organic carbon). Several practices can cause a decrease in SOM, including excess N inputs from high fertiliser applications that can increase the mineralisation and loss of SOM (EEA, 2010b). Furthermore, high N fertilizer application causes soil acidification which can be counteracted by liming but this is not always done adequately. For example, data suggests that nearly 40% of arable soils and 57% of grassland soils in United Kingdom have a pH below the recommended level because insufficient lime is being applied (Goulding, 2016).

1.2 Indicators for acceptable agricultural intensification related to fertilizer management

Indicators for impacts of nutrients on the environment

Fertilizer management is essential to sustain food production but it is simultaneously an important cause for environmental impacts. Observed forms of fertilizer related impacts are damage to ecosystem biodiversity caused by eutrophication of terrestrial ecosystems and water bodies in response to enhanced nitrogen (N) and phosphorus (P) inputs. Here we are facing challenging dilemma's where the need to enhance food production requires an additional input of N and P (required N and P inputs), 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, there is a need for N and P indicators with related critical limits, 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 (NO₃) 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, being the inputs at which such limits are reached, either on the short term (N) or the long term (P).

In this report, we assess the spatial variation in current and critical N and P inputs to illustrate the tension between food production and environmental protection. These assessments are made while using the current nitrogen and phosphorus use efficiencies (NUE and PUE), defined as the ratio between N and P crop removal (net uptake) compared to the N and P input to soil. In addition we calculate the spatial variation in the needed increase in NUE and PUE to reconcile food production with environmental protection. The various concepts are discussed in more detail in Chapter 2.

Indicators for impacts of metals on soil quality

Fertilizer management also affects soil quality, both via the inputs of major nutrients but also due to input of metals such as copper (Cu), zinc (Zn), cadmium (Cd) and lead (Pb). In this context Cu and Zn are micro-nutrients and essential for crop growth and animal health but at high concentrations in soil, they can adversely affect soil biodiversity. Cadmium and Pb are toxic metals without any known

function for living organism that, if present at excess levels lead to soil degradation, both by affecting soil biodiversity and transfer into food and feed crops.

As with N and P, there is a need for thresholds of metals concentrations in soil that are related to critical limits for targets to be protected (water, food), above which adverse effects can be expected. Such thresholds of soil metal concentrations in turn can facilitate the derivation of the critical metal inputs. Relevant thresholds in this context are critical total metal concentrations in view of impacts on soil biodiversity and food quality. In this report, we assess the spatial variation in both current and critical soil metal concentrations and current and critical soil metal inputs to illustrate the potential environmental problems of elevated metal inputs. In this context, an exceedance of critical soil metal concentrations currently leads to adverse impacts whereas an exceedance of critical metal inputs may lead to adverse impacts in the long term. The various concepts are discussed in more detail in Chapter 3.

1.3 Policies aiming to reduce adverse impacts of agriculture on the European environment

Several legislative and policy frameworks aim to reduce adverse impacts of agriculture on air, soil and water quality. These include, amongst others:

Water quality

- Water Framework Directive (EC, 2000) in view of surface water. The Water Framework Directive was established to improve the quality of water resources across the EU. Its main objective is that all surface water and groundwater should hold good status by 2015, which means that certain standards for ecology, chemistry and morphology of waters are met.
- Nitrates Directive (EC, 1991) in view of groundwater and surface water. The Nitrates Directive (1991) aims to protect water quality across Europe by preventing nitrates from agricultural sources polluting ground- and surface waters and by promoting the use of good farming practices. It forms an integral part of the Water Framework Directive and is one of the key instruments in the protection of waters against agricultural pressures.

Air quality

- The National Emission Ceilings Directive (NEC), which sets national emission reduction commitments for Member States and the EU for five important air pollutants: nitrogen oxides (NO_x), non-methane volatile organic compounds (NMVOCs), sulphur dioxide (SO₂), ammonia (NH₃) and fine particulate matter (PM_{2.5}). These pollutants contribute to poor air quality, leading to significant negative impacts on human health and the environment. In view of agriculture, the emission of ammonia is most important, affecting terrestrial biodiversity and contributing to fine particulate matter.
- The Habitats Directive (EC, 1992) in view of protection of the Natura 2000 sites against N deposition. The provisions of the Habitats Directive require strict site protection measures to avoid deterioration. By introducing a precautionary approach “plans and projects” can only be permitted if they are shown to have no significant adverse effect on a Natura 2000 site.

Soil quality: general

- Soil Thematic Strategy. To fill the gap in European environmental legislation and to provide a more holistic approach to soil protection in the EU the European Commission presented in 2006 the “Thematic strategy for soil protection” [SEC(2006)620]. The main threats to soil were described, including erosion, decline in organic matter and biodiversity, contamination, sealing, compaction and salinization. This proposal however was not adopted. In 2012 a policy report (COM(2012) 46) was published on the implementation of the Strategy and ongoing activities but as of yet, the foreseen Soil Framework Directive has not been implemented.

- Roadmap to a Resource Efficient Europe. The vision of this road map is that “By 2050 the EU's economy has grown in a way that respects resource constraints and planetary boundaries..... and provides a high standard of living with much lower environmental impacts” with all resources being “sustainably managed, from raw materials to energy, water, air, land and soil”. It includes the sustainable consumption and production of food and increased waste recycling.

Soil quality: heavy metals

Aside from national legislation, various relevant EU Directives have been installed or are being revised to protect soil (and crops) and water against metal pollution, based on either acceptable levels of soil amendments, crops or water levels. In Table 1, several relevant directives are listed. At present various policies are being discussed that may have a substantial impact on the load of metals to soils. The most relevant Directives that affect the load of metals to agricultural land are:

- Fertilising Product Regulation (FPR, EU2019/1009) which replaces the former regulation on Fertilisers (EU2003/2003. Regulation 2019/1009 sets maximum levels for contaminants in a.o. inorganic and organic fertilisers as well as organic soil improvers including compost intended to be marketed as EU fertilisers. As such these limits are primarily meant to facilitate trade between countries but such limits also can result in an intentional reduction of emission of potentially harmful substances like Cd and Pb. Especially for Cd, changes in the levels of Cd in mineral P fertiliser can have a profound impact on the annual load of Cd to soil. The magnitude of this impact depends on both the actual application rates, largely to arable soils used for crop production and the variation in the quality of fertilisers used. At present the average Cd levels in mineral P fertilisers ranges from values close to zero mg Cd kg⁻¹ P₂O₅ in countries like Estonia, Sweden and Finland to levels between 40 and 60 mg Cd kg⁻¹ P₂O₅ in Poland, Portugal and Spain. Long term changes are especially relevant in view of food quality since there is a link between the Cd content in soil and that in most food products grown on soils including vegetables, grains and potato being relevant food products that affect intake of Cd via consumption. Other metals regulated in EU2019/1009 are arsenic (As), chromium (Cr, measured as Cr(VI)), mercury (Hg), nickel (Ni) and, not included previously in EU2003/2003 also for Cu and Zn. In addition a limited number of metals are included for specific components (CMC's) from with fertilisers (PFC's) are made. This is the case for vanadium (V) and thallium (Tl) in oxidation products such as wood ash
- Council Directive 86/278/EEC which sets standards for sewage sludge based on the maximum allowed concentration in sludge and the maximum amount of sludge to be used. Criteria for this are to be based on the principle of limited accumulation where it is not allowed that the use of sludge results in unwanted accumulation of metals in the soil beyond specific target levels.
- Waste Framework Directive 2008/98/EC. Within the context of the Waste Framework Directive, alternative materials to be used as fertilizer or source material for the production thereof are being discussed. The aim is to more effectively re-use specific waste materials as fertilizer provided these meet specific quality criteria.

These various directives (related to air and water quality) and initiatives (related to soil quality and food production) require Member States to adopt programmes of measures which protect, restore and ensure the long-term sustainable use of European soil and water resources (Table 1). A key obstacle for this is to ensure that agricultural intensification is sustainable regarding the use, impact on and conservation of soil and water resources, while ensuring biodiversity and addressing climate change. This is reflected by the statement of the EEA that *"to make our food systems more resilient, responsive and adaptive to future requirements we must become more resource efficient, placing sustainability at the heart of agricultural production and water management"* (EEA, 2012).

Table 1.1. Overview of selected legal frameworks at EU level and targets addressed. Unless specified otherwise the Directives listed refer to As, Cd, Cr, Cu, Hg, Ni, Pb and Zn

EU Directive	Addressing	Regulating principle
Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture	Levels of heavy metals in sewage sludge used on arable land	Setting maximum concentration in sludge, the total annual load of metals to be applied to soil and allowed increase in total metal content relative to background values to avoid unwanted accumulation of metals in soil
Regulation (EC) No 2003/2003 of the European Parliament and of the Council of 13 October 2003 relating to fertilisers. As of 2021 this regulation is replaced by FPR (EU2019/1009)	Acceptable upper limits for metals in inorganic and organic fertilisers as well as a selected types of components thereof	Setting maximum allowed levels for metals in a.o. inorganic and organic fertilisers as well as soil improvers to have the product listed as EC Fertiliser.
Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. For Cd and Pb this Regulation has been replaced by commission regulation (EU) 2021/1323	Quality of food to protect human intake of metals via food	Setting maximum levels for Cd, Pb and Hg in products for human consumption to avoid excess intake of metals
Directive 2002/32/EC of the European Parliament and of the Council of 7 May 2002 on undesirable substances in animal feed	Quality of fodder for animals	Setting maximum levels for Pb, Cd, As and Hg in fodder and food products for animals to reduce intake and transfer into food products for human consumption
Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy	Quality of surface waters to protect ecosystem health	Setting maximum levels for metals in surface waters based on ecological thresholds for aquatic organisms
DIRECTIVE 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration	Quality of groundwater	Obligation to set standards for priority elements (Cd, Pb, As, Hg as well as some other non-metallic contaminants) to protect the general quality of water. Part of the WFD.
COUNCIL DIRECTIVE 98/83/EC of 3 November 1998 on the quality of water intended for human consumption	Quality of drinking water intended for human consumption	Setting maximum levels for contaminants including metals (note: Zn not included)

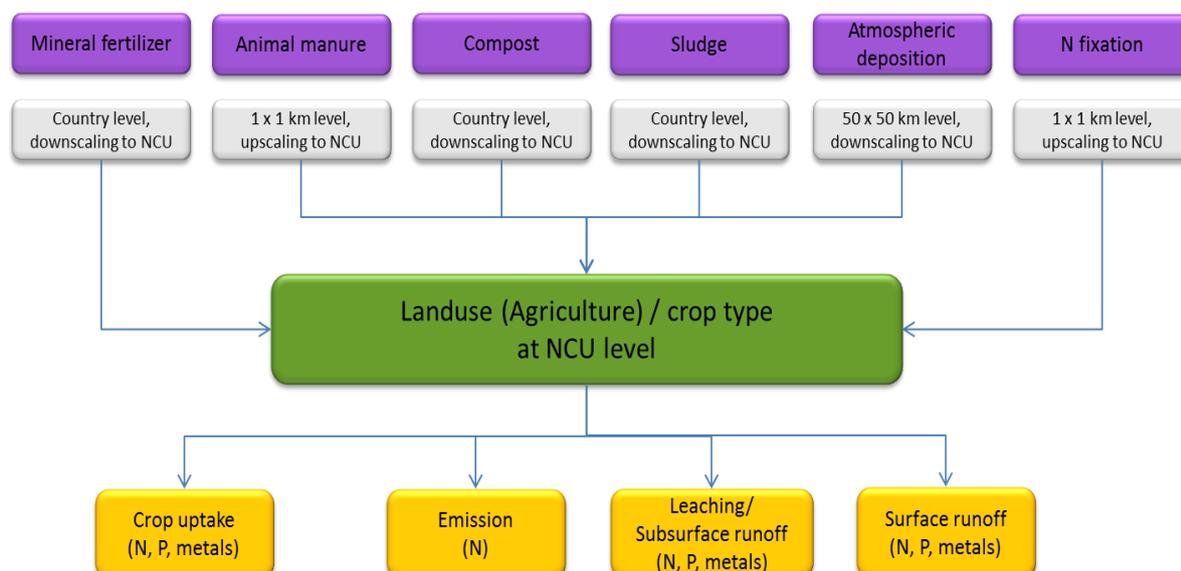
1.4 Aim and approach of this report

This report assesses current inputs of nutrients, i.e., nitrogen (N) and phosphorus (P) and of heavy metals, i.e., cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn), especially by mineral and organic fertilizers, and their fate in soil, air and water, in view of potential adverse impacts. This includes (i) emission of ammonia to air and leaching and runoff of nitrate and phosphate to groundwater and surface water, that may cause eutrophication of terrestrial and aquatic ecosystems (Chapter 2), and (ii) soil accumulation, uptake and leaching of metals with potential impacts on soil biodiversity, crop quality and aquatic biodiversity (Chapter 3). It further assesses critical inputs of nutrients (especially nitrogen) and metals in view of acceptable losses to air and water (N) and acceptable accumulation levels in soil (P and metals). In addition, this report assesses needed increases in nitrogen use efficiency (NUE) for sustainable crop production in terms of attaining current or target crop yields with acceptable N losses to air and water. In this context, the NUE has been defined as the ratio between N output and N input, according to $NUE = N_{output}/N_{input}$. Impacts of organic pollutants and pesticide inputs as well as industrial pollution are not part of this report.

The base year used in this study for the current N, P and metal inputs is 2010, a year with detailed information on fertilizer use, livestock numbers, crop areas, crop yields. Despite the fact that this is now more than 10 years ago, the base line of 2010 is indicative for the current situation, since the change in N inputs by fertilizer and manure, based on FAO stat data, is limited since 2010 as illustrated in Figure 2.1. Similar trends can be expected for P and metals.

A key approach is the assessment of the budgets and the fate of nutrients and heavy metals, including inputs and outputs for approximately 40,000 so-called NCUs, being unique combinations of soil type, administrative region, slope class and altitude class, using the model INTEGRATOR (De Vries et al., 2011b; De Vries et al., 2011c). The inputs and outputs included in the calculations are illustrated in Figure 1.1. The model has also been used to assess required and critical inputs.

Figure 1.1 Overview of inputs, outputs and scale of source data used to calculate nutrient and metal budgets at EU-level



The results have been aggregated at various spatial scales, varying from NCUs to NUTS3 regions, countries and the whole EU-27. Details on the various calculations are given in De Vries et al. (2014) for P and metals and in De Vries et al. (2021) for nitrogen. The approach to calculate required increases in the N use efficiency is given in detail in Schulte-Uebbing and de Vries (2021).

2 Current and critical inputs of nitrogen and phosphorus in European agriculture

2.1 Nitrogen and phosphorus use in view of food production and environmental impacts

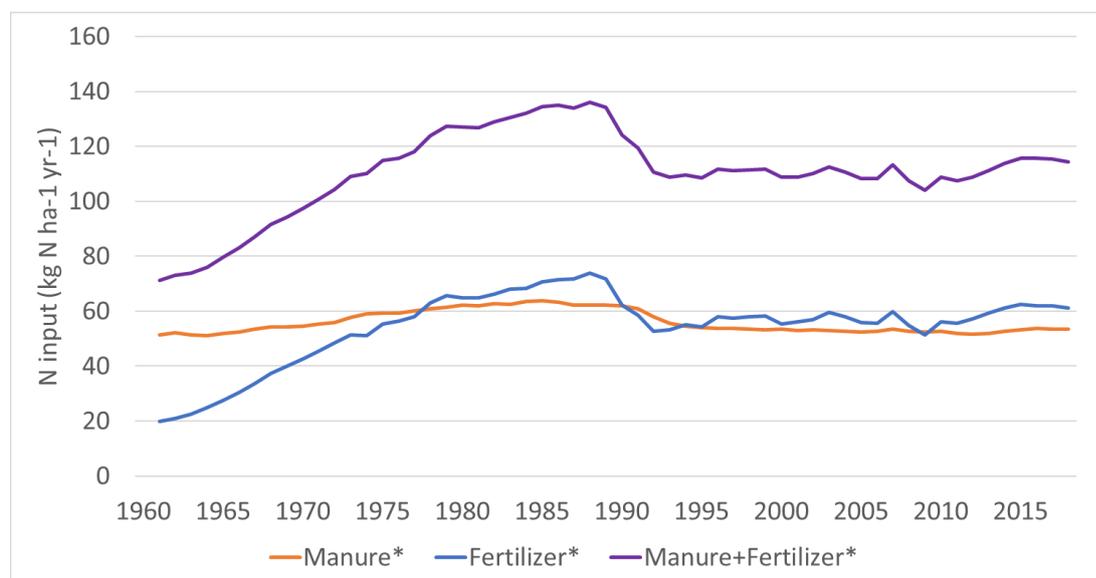
2.1.1 Nitrogen and phosphorus requirements and use in view of food production

Nitrogen (N) and phosphorus (P) fertilizers are applied to agricultural soils to increase crop growth. Phosphorus inputs specifically increase growth when soils are limited in P, due to a low soil P (soil fertility) status. Supply of N and P fertilizer has large socioeconomic benefits, and has become essential to raise crops and animals to feed an ever-increasing world population (Eickhout et al., 2006; Erismann et al., 2008; Robertson & Vitousek, 2009; Sutton et al., 2013). However, global phosphate ore reserves are limited, and studies indicate that the availability of P may limit the growth of agricultural production in the forthcoming decades (Cordell et al., 2009) or centuries (Scholz & Wellmer, 2013; Syers et al., 2011; Van Vuuren et al., 2010) although others found that past depletion concerns were refuted by means of new resource appraisals (Ulrich & Frossard, 2014).

Trends in nitrogen and phosphorus inputs in European agriculture

Agriculture and food production, as well as food consumption and waste along the food chain, play a central role in the environmental flows of N and P. Since the early 1960s, European agriculture has intensified greatly, resulting in large inputs of N and P to soil by mineral fertilizers and organic fertilizers (manure, compost and biosolids), especially up to 1985, followed by a decline since then. This is illustrated in Figure 2.1 for trends in N inputs by mineral fertilizers and organic fertilizers in Europe.

Figure 2.1 Trends in N inputs from fertilizers and livestock manure in the period 1961-2018 at EU-level (Data from FAOSTAT).



In the 1960s, the total N input was about 23 million tons. Manure-N applied to soils contributed 45 percent (about 10 million tons of N) and manure left on pasture 32 percent (about 7 million tons of N) of the total N input. During the period 1961-2018, the contribution of synthetic fertilizer-N increased from 23 percent (about 5 million tons of N) of the total N input in the 1960s to about 55 percent by 1988, followed by a marked decrease in the period 1989-1995 and being rather constant since then (Figure 2.1). The reason for this decrease is the EU Nitrates Directive that led to limitations in nutrient

use (Sutton et al., 2011), re-enforced by the economic collapse of the Soviet Union, including agricultural production. As a result, from 1985 to 1995, use of synthetic fertilizers dropped by nearly 50 percent, while applications and availability of manure-N decreased by over 30 percent (FAO, 2018). Between 1985 and 2015, there has also been a reduction in the number of dairy cattle by about 1% per year since the implementation of the EU milk quota in 1984 (Oenema et al., 2007), but started to increase again after the quota was abolished in 2015. Despite the decrease in livestock numbers, the total N excretion has hardly changed due to an increased production per animal, associated with higher N excretion rates, and similar trends can be expected for P and metals.

Required inputs of in nitrogen and phosphorus

Even though Europe is one of the most food secure regions worldwide (per capita food availability is only higher in North America; FAOSTAT), agricultural productivity in Europe will probably need to increase in the future to maintain or increase food self-sufficiency, to help meeting global food demands and to provide biomass resources for energy and other uses (De Wit et al., 2011). In recent years, Europe has shifted from food self-sufficiency to relying on trade to fulfil demands (Sadowski & Baer-Nawrocka, 2016). Europe is also a net importer of vegetal proteins, mainly of feed (Lassaletta et al., 2014). This makes Europe vulnerable in case a disruption leads to scarcity on global crop markets (Puma et al., 2015).

While Europe's food demand is only projected to increase by a few percent between now and 2050 (Bruinsma, 2012), global demands are projected to increase by 60% (FAO, 2017) to 100% (Tilman et al., 2011) by 2050, driven by an increasing global population and consumption of animal protein. There is strong concern that historical rates of crop yield growth are not sufficient to meet growing demands (Ray et al., 2013; Ray et al., 2012), especially in Sub-Saharan Africa (Van Ittersum et al., 2016) and some regions in Asia and South America (Fader et al., 2013). In today's globalized world it is likely that Europe will play a role in meeting global demands.

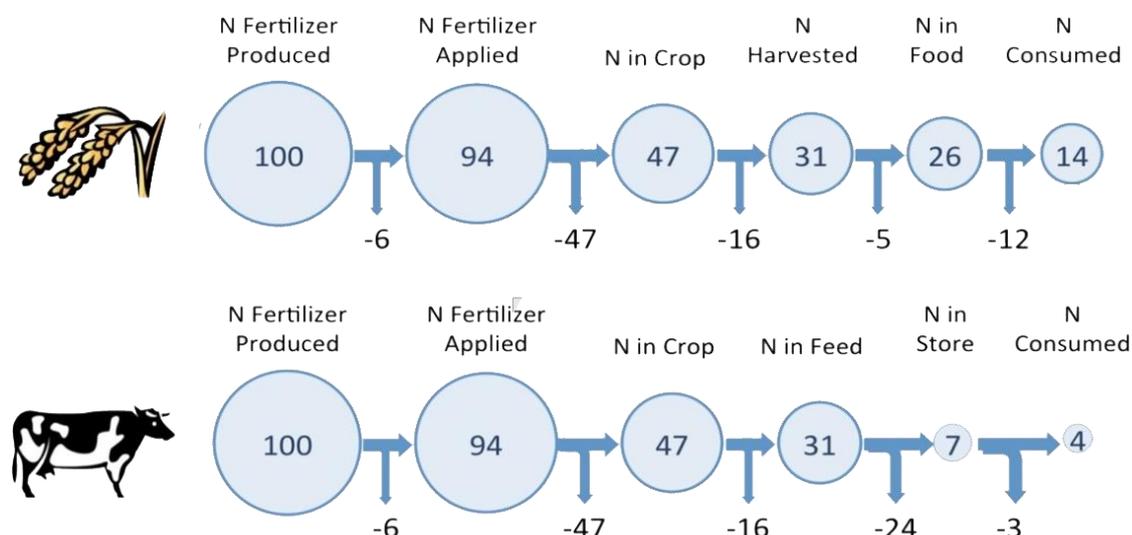
In several regions in Europe, there are still substantial crop yield gaps, that are induced by nutrient limitation, besides other yield limiting factors. The yield gap (Y_g) is the difference between the yield potential (Y_p) in case of irrigated crops or the water limited yield potential (Y_w) in case of rainfed crops and the actual yield (Y_a). For Europe, GYGA currently provides yield gap estimates for wheat and barley for all EU27 countries except Cyprus and Malta. 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, 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). But even when using this target value, there is large potential for increasing crop yields in Southern and Eastern Europe.

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. The extra required N or P can be derived as the extra amount of N or P that is removed from the field (being the crop yield times the N or P content in the crop) divided by the NUE or PUE.

2.1.2 Nitrogen and phosphorus limits in view of environmental impacts

Only a small part of the N and P applied in agriculture ends up in people's mouths. N and P losses occur at each step of the food chain. A first estimate of global food chain N losses has been made by (Galloway & Cowling, 2002a) as shown in Figure 2.2.

Figure 2.2. N flows through the global food production and consumption system, indicating use, loss and recovery at each key stage of the process (Adapted after Galloway & Cowling, 2002b). NB: data are slightly different for P but the numbers are also illustrative for P.



The figure illustrates the N flows and losses in every step of the food production chain, from the fixation of N from the atmosphere towards the N intake by humans. This includes N losses during transportation, application in the field, feeding animals with feed crops, and feeding humans with vegetal and animal, including food waste. Note that similar losses occur for P from the production of P, by mining from natural resources, towards P intake by humans.

From 100 kg N fertilizer produced approximately 6 kg (6%) is lost to the environment during fertilizer production, transportation and distribution. After that, only half of the applied fertilizer is estimated to be taken up by the crops. The other half is lost to the atmosphere and to ground- or surface water. After being taken up by crops, 16% of N produced as fertilizer is estimated to be lost during harvesting (e.g. N losses from crop residues). Finally, an estimated 17% is lost during food processing and due to food waste, both at the field to the shop (5%) and then from the shop to the ultimate human consumption (12%). Only 14% of the N ends up in human's mouth for a fully vegetarian diet. This situation is even worse when people consume meat. In that case harvested crops need to be used for animal production, resulting in a global N loss of 31%. Another 24% of N is lost due to animal metabolism and production preparation. From the remaining 7% N that is contained in the eventual animal product, roughly 3% is lost again due to spoilage and waste. Thus the full-chain N use efficiency of animal protein is only about 4% (Galloway & Cowling, 2002a).

Environmental impacts of nitrogen and phosphorus losses

Nitrogen and phosphorus losses to the environment resulting from (over-)use in agriculture have many negative impacts, as listed below. The first impact (freshwater eutrophication) is caused by both N and P losses, the other impacts are specifically caused by N. Nitrogen-related impacts are illustrated in Figure 2.3, which also includes the related policy directives to reduce its impacts (See Section 1.3).

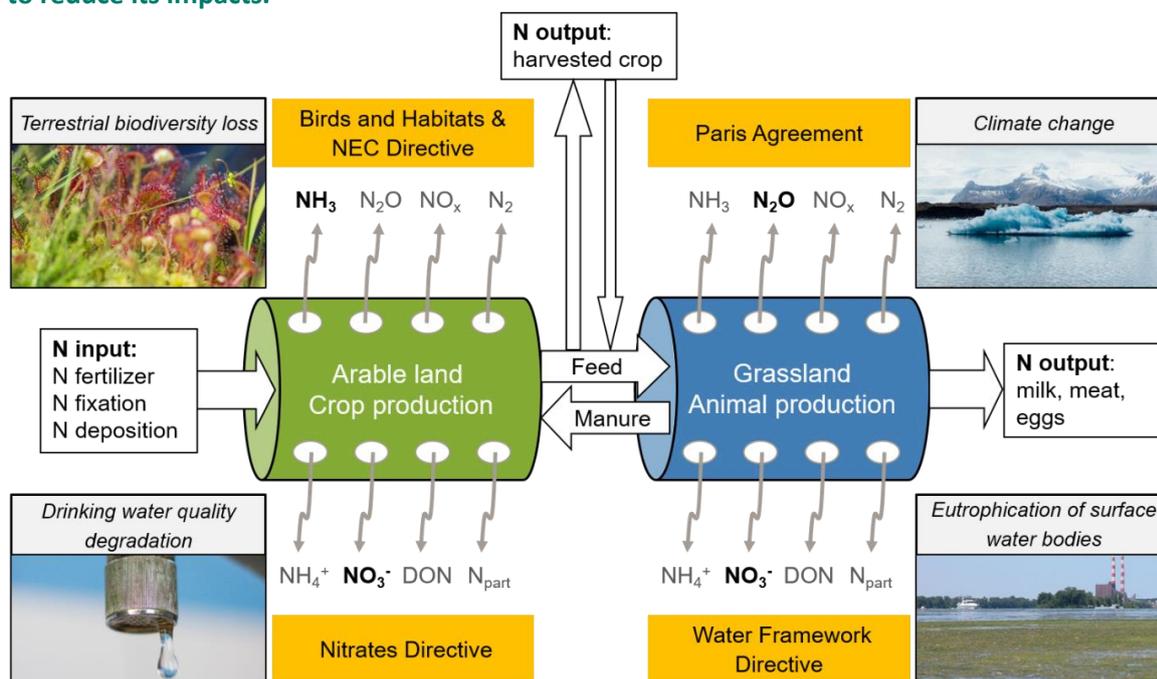
Impacts of N & P:

1. Eutrophication of freshwater and coastal marine ecosystems, resulting in loss of biodiversity, the development of hypoxic (oxygen-free) conditions in the coastal ocean causing fish dieback (Diaz & Rosenberg, 2008; Seitzinger et al., 2002) and harmful algal blooms (Glibert & Burford, 2017). This is mainly caused by runoff and leaching of N and P from agricultural land and by inputs from point sources, such as sewage, causing an exceedance of ecological limits and related eutrophication of the aquatic environment (Rabalais et al., 2002; Selman et al., 2008).

Impacts of N:

2. Eutrophication of terrestrial ecosystems, reducing their biodiversity and altering their functioning, (Bleeker et al., 2011; Bobbink et al., 2010; Clark & Tilman, 2008), due to N deposition induced by ammonia (NH_3) and nitrogen oxide (NO_x) emission, where the contribution of NH_3 is the dominating effect. The contribution of agriculture to NO_x emissions is very limited compared to transport and industry, and it can thus be ignored when assessing critical N inputs to agriculture.
3. Acidification of soils and fresh waters (Guo et al., 2010; Velthof et al., 2011), due to N inputs in agriculture not counteracted by liming (a threat occurring in China; (Guo et al., 2010; Velthof et al., 2011)) or N deposition on forests and nature, induced by NH_3 emission. NO_x emissions also cause acidification, but the contribution of agriculture to NO_x emissions is nearly negligible, as mentioned above.
4. Groundwater contamination by nitrate (NO_3^-), with negative impacts on human health (EEA, 2005; UNEP, 2007; Van Grinsven et al., 2006; Van Grinsven et al., 2010)
5. Negative impacts on human health due to secondary particles, induced by emission of NH_3 (Pozzer et al., 2017).
6. Contribution to the global greenhouse effect through the emission of nitrous oxide (N_2O), a potent greenhouse gas. N_2O is emitted either directly from agricultural soils; or indirectly from terrestrial and aquatic systems, including rivers, coastal zones and open oceans, following volatilization or leaching of applied N and re-deposition and processing downwind or downstream of the agricultural regions (De Vries et al., 2017).

Figure 2.3. Picture illustrating the losses of N to the environment and the related policy directives to reduce its impacts.



Assessment of critical inputs of nitrogen and phosphorus

The fact that increased nitrogen inputs drives multiple effects has led to the concept of a planetary N boundary, defined as global critical N input by fertilizer and biological N fixation (Rockström et al., 2009a; Rockström et al., 2009b). Similarly, a planetary P boundary has been derived. These planetary N and P boundaries have, however, been criticized, partly because they do not account for the spatial variability in impacts, both in terms of N and P limitations and N and P overuse, associated with areas where crop N and P removal is higher or (much) lower than N and P application (Lewis, 2012; Nordhaus et al., 2012). Global N and P fertilizer application is highly unevenly distributed. In many African

countries (Liu et al., 2010), as well large areas of Latin America and South East Asia (MacDonald et al., 2011), N and P inputs are insufficient to maintain soil fertility, posing risks of land degradation (Sutton et al., 2013). Many developed and rapidly growing economies, on the other hand, have large N surpluses (Vitousek et al., 2009). Hence, in many parts of the world an increase in N and P input is needed to avoid land degradation and increase crop yields, while in other parts N and P application can be reduced while simultaneously maintaining or even enhancing yields and reducing environmental impacts (Ju et al., 2009).

Considering this criticism, De Vries et al. (2013) calculated a global critical N input while accounting for spatial variation in N losses to air, especially as ammonia, and to surface water. They assessed the areas where N losses and related N inputs should be reduced as it caused the exceedance of a critical NH_3 concentration in air (in view of biodiversity decline in terrestrial ecosystems) and a critical N concentration in runoff (in view of eutrophication of aquatic ecosystems).

2.1.3 Approaches for increasing nitrogen and phosphorus use efficiencies

Required improvements of nutrient use efficiencies

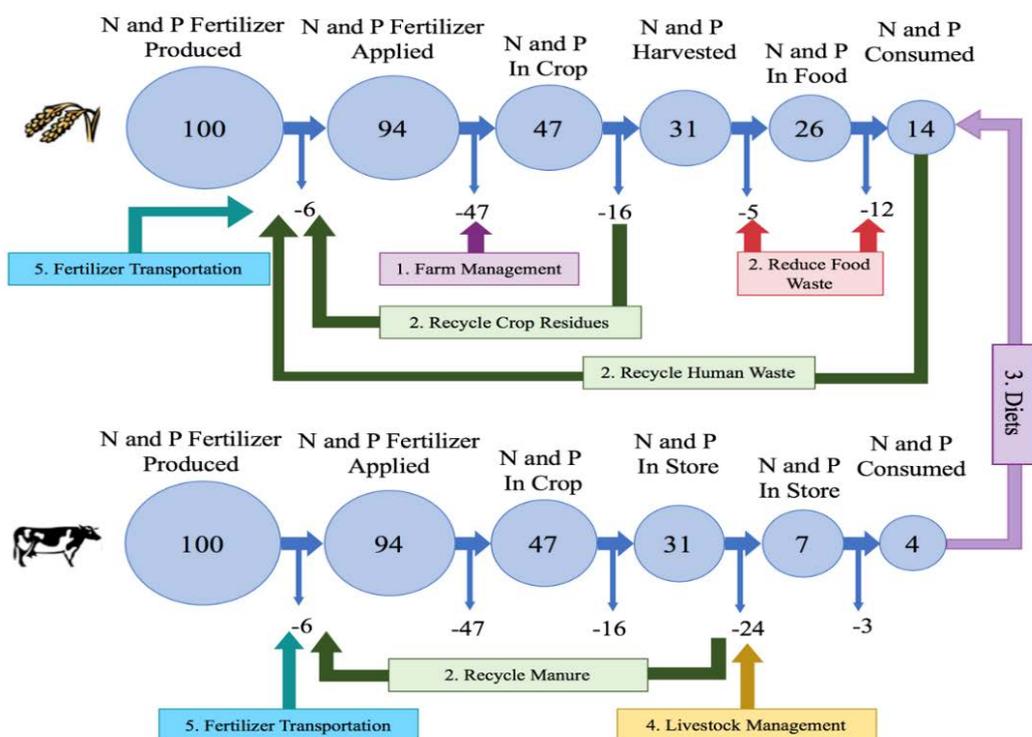
The nutrient use efficiency is here defined as the ratio between N and P uptake compared to the N and P input to soil. In case of N addition by fertilizer and/or manure, the nitrogen use efficiency (NUE) is nearly always below 1 because part of the added amount is inevitably lost to air and water, but NUE values above 1, implying mining of N from the soil, do occur in drained peat soil with high net N mineralization. The current European scale average NUE is near 0.6 (Leip et al., 2011). Losses of N to air and water are associated with adverse environmental effects when it exceeds certain limits, such as concentrations of NH_3 in air or of NO_3 in water (see Section 1.2). Unlike N, P losses are limited and a PUE value near 1.0 can be attained when fertilizing or above 1 when the soil P status is so high that mining of P is allowable.

Based on those limits, it is possible to calculate critical losses and then critical N inputs for a given NUE. At the current NUE, the required N input levels to attain target yields will be higher than the current N inputs in areas with a yield gap. Inversely, however, the critical N input, defined as the N input above which critical concentration levels for N in air and water are exceeded, may be lower than the current N input. In this situation, current or target crop yield can only be reached in a sustainable way (i.e., without exceeding critical losses) if NUE is increased. If NUE is increased, the required N input decreases, as a higher fraction of applied N is taken up by the crop and the current or target crop yield can thus be reached with less N fertilizer. Inversely, the critical N input increases since a lower fraction of N is lost to the environment. In this context, it is highly relevant to calculate the NUE at which the required N input equals the critical N input. By comparing these 'required' NUEs with the current NUEs, one can assess the needed improvement in NUE. Defining NUE targets that meet production and environmental goals at the same time indicates the way towards sustainable nutrient use.

Impacts of management actions for increasing N and P use efficiencies

The most practical way to improve NUE is by tuning the rate, timing, method and type of N application (Snyder, 2017), with the goal to improve matching of N supply with crop demand. Furthermore, NH_3 emissions can be reduced by manure management, both in stables, e.g. separating faeces and urine and during application, e.g. by injecting the manure. It should be noted however, that apart from increasing N and P use efficiencies in the field, losses can be strongly in other parts of the food chain. An overview of approaches to improve the food chain nutrient use efficiency and thus decrease N and P losses is given in Figure 2.4.

Figure 2.4. Approaches to improve the food chain nutrient use efficiency (losses are based on N data by Galloway & Cowling, 2002b), see also Figure 2.2.



The most important management strategies to keep the food system within environmental limits are Springmann et al. (2018):

- improvements in nutrient use efficiency along the whole food chain, through on-farm technological and management-related interventions, especially from soil to crop and from feed crop to animals,
- improved recycling of crop, animal and human waste and residues,
- reductions in food loss/waste, and
- dietary changes towards more plant-based diets

Priorities include mapping “hotspot” areas where the most severe environmental impacts coincide with the most intensifying social drivers and identifying steps and targets along pathways to reverse the undesirable trends.

2.2 Nitrogen and phosphorus budgets in European agricultural soils

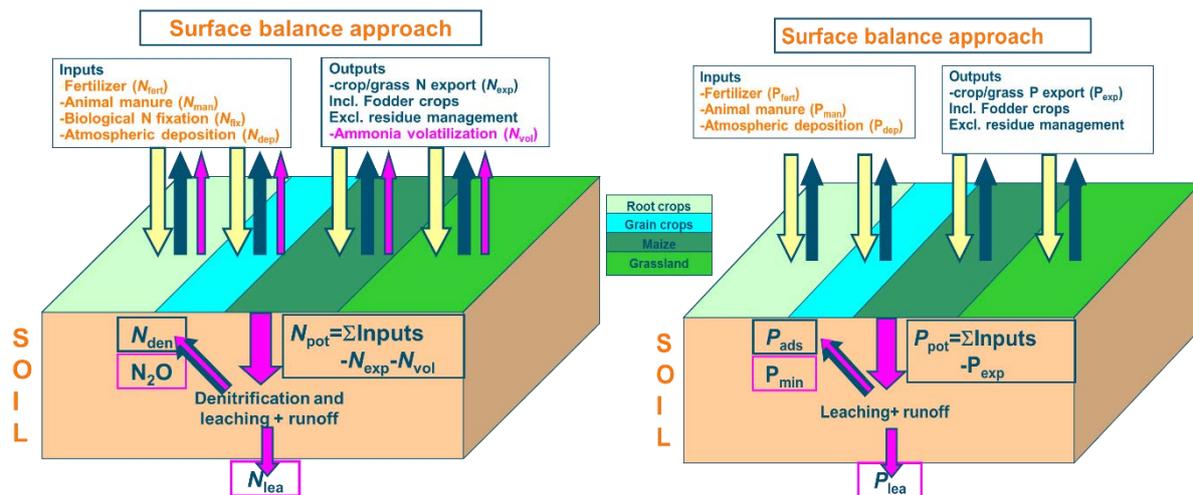
2.2.1 The fate of nitrogen and phosphorous inputs

Differences in nitrogen and phosphorous behaviour and consequences for their fate

A simplified overview of N and P fluxes in a soil balance approach is given in Figure 2.5. N and P are added to the soil as fertilizer, manure, biosolids (compost and sludge) and deposition. N is also added through biological N fixation. The main N and P output is N and P removal by harvesting crops. The difference between N or P input and crop N or P removal is called the N or P surplus. Nitrogen and P surpluses are important indicators for environmental risk, as surpluses will lead to soil accumulation and/or losses to air (N only) and water.

N is hardly adsorbed in soil and consequently, N must be added yearly to avoid crop yield losses. To the contrary, P can be adsorbed or desorbed, and P taken up by plants from soil solution is buffered by a pool of plant-available P. With sufficient plant-available P, there is fast buffering of P in solution and crop yields hardly respond to additional P application (Jungk et al., 1993). An effect of P application on crop yield only occurs when the soil P status drops below a critical plant-available P level. The linkage between crop yield and soil P status and related critical soil P values has been derived in many studies (e.g. Bai et al., 2013). These differences in N and P behaviour also affect the fate of the N or P surplus, as described below.

Figure 2.5. Simplified overview of nitrogen (left) and phosphorus (right) fluxes in an agricultural soil



Fate of N: The N surplus can be accumulated in the soil, but there are no N minerals in (agricultural) soils and N does not precipitate or dissolve and it also hardly adsorbed to or desorbed from clay or soil organic matter. Changes in soil N pools occur mainly by mineralization/immobilization of N in soil organic matter. These biological processes affect the N availability to crops, but unlike P there is no chemical equilibrium between N concentrations in soil and in soil solution. On the longer term (decades) the change in organic soil N pool is very limited and the N surplus is either emitted to air as ammonia (NH₃), nitrogen oxides (NO_x), nitrous oxide (N₂O) or dinitrogen (N₂) or to water (groundwater and surface water), mainly as nitrate (NO₃). N emissions to air and water are mediated by biological N transformation processes. This includes: (i) mineralization, which leads to the formation of ammonium (NH₄) that can be lost to air through ammonia volatilization, (ii) nitrification, i.e. the conversion of NH₄ to NO₃, which is mobile in solution and partly lost to water by leaching or runoff and (iii) denitrification, i.e. the transfer of NO₃ to the gases N₂O, NO_x and N₂. Losses of applied N through ammonia volatilization, leaching or runoff and denitrification are thus inevitable.

Fate of P: Unlike N, there are P containing minerals in (agricultural) soils and P can dissolve or precipitate and it also 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 leads to increase of P in soil organic matter (an organic P pool) which in turn decreases by mineralization of P. Plants take up P from 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 (Fig. 2.5). 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.

Differences in nitrogen and phosphorous use efficiency

There are various definitions for N and P use efficiency. The definition of nitrogen use efficiency (NUE) used in this report is the fraction of applied N that is taken up by a crop, calculated as $NUE = N_{\text{output}}/N_{\text{input}}$. P use efficiency (PUE) is calculated in the same way, i.e. as $PUE = P_{\text{output}}/P_{\text{input}}$. The NUE indicator also provides the opportunity to estimate the possible gap between actual NUE and an attainable (or target) NUE, based on for example best management practices or the average values obtained in field experiments.

As mentioned above, N is hardly or not adsorbed in soil and thus a part of applied N is always lost through ammonia volatilization, leaching or runoff and denitrification. Therefore, NUE is at most 90%. NUEs above 90% likely are the result of soil mining, i.e. a decrease in soil N content and thus in soil N supply by mineralization of organic matter. At the European scale, only 60% of the nitrogen applied to agricultural land as manures, fertilisers, deposition and from fixation is taken up by crops while the remainder increases the N-fluxes to the environment, herein termed 'lost' to the environment (Leip et al., 2011). Since the 1990s, the NUE of European agriculture has increased (Van Grinsven et al., 2014) but not enough to reduce N losses sufficiently to meet environmental targets at all regions in the EU.

Unlike for N, a target P use efficiency (PUE) can be 100%, under the condition that the soil P status is at a sufficient high level. As described above, P is mainly taken up from soil and above a critical plant available soil P level, crop yields hardly respond to P application. The only reason for adding P is to avoid P mining the soil and the amount of applied P in both fertilizer and manure can be set equal to the amount of P removed from the field by crop harvesting. This approach is now used in manure legislation in the Netherlands: P input should not exceed P removal. Only in situations where the P status is insufficient, some extra P is allowed, while the reverse is true for P saturated soils (less P input than P removal).

2.2.2 Nitrogen and phosphorus budgets and use efficiencies at EU level

Average N and P budgets for the year 2010 for all agricultural land, further subdivided in arable land, and grassland (including fodder) are given in Table 2.1 and Table 2.2, respectively. Results show that on average total N and P inputs are ca. 50% higher on grassland than on arable land due to much (more than twice) higher manure N and P inputs, while fertilizer N and P inputs are slightly (ca 20%) higher on arable land as compared to grassland. The higher N manure inputs on grassland are reflected by higher ammonia (NH₃-N) emissions. Despite the higher N inputs in grassland, the N surplus, equal to total N input minus crop N removal, is slightly higher for arable land due to much higher N uptake by grass compared to crops. Overall, the N surplus is about 40% of the N input in arable land (NUE near 0.6) and about 30% in grassland (NUE near 0.7). Considering the lower NH₃ emissions from arable land, the calculated N leaching rates from arable land are on average higher than from grassland (Table 2.1).

Compared to the inputs, the P surplus is near 10% of the P input in arable land (PUE near 0.9) and about 20% in grassland (PUE near 0.8). Unlike N, the relative P surplus is thus larger on grassland than on arable land, since the crop P uptake is not so much higher in grassland. On average, about 30% of the surplus is calculated to be lost to ground- and surface water while the remainder is accumulated in the soil (Table 2.2).

Table 2.1 Average actual (2010) annual N inputs, N uptake and N losses for total agricultural land, arable land and grassland (including fodder) in EU-27 calculated by INTEGRATOR

	N budget EU-27 (kg N ha ⁻¹ yr ⁻¹)		
	Total	Arable land	Grassland /fodder
Input to land			
Fertilizer + fixation	78.3	83.3	70.4
Manure and biosolids	55.7	36.5	85.9
N deposition	10.5	10.1	11.2
Net N mineralisation	0.8	0.2	1.7
Total input	145.3	130.1	169.2
Output from land			
Crop removal (offtake)	92.3	76.7	116.8
Total surplus	53.0	53.2	52.4
• NH ₃ emission	16.0	13.6	19.8
• N ₂ O + NO _x emissions	3.2	2.2	4.7
• N ₂ emissions (Denitrification)	17.4	18.8	15.1
• Leaching + runoff	16.5	18.9	12.8
Total output	145.3	130.1	169.2

Table 2.2 Average actual (2010) annual P inputs, P uptake and P losses for total agricultural land, arable land and grassland (including fodder) in EU-27 calculated by INTEGRATOR

	P budget EU-27 (kg P ha ⁻¹ yr ⁻¹)		
	Total	Arable land	Grassland /fodder
Input to land			
Fertilizer	7.3	7.6	6.7
Manure and biosolids	9.1	6.5	13.4
P deposition	0.5	0.5	0.5
Total input	16.9	14.6	20.6
Output from land			
Crop removal (offtake)	14.6	13.2	16.7
Total surplus	2.3	1.3	3.8
• Accumulation	1.6	0.9	2.7
• Leaching + runoff	0.7	0.4	1.1
Total output	16.9	14.6	20.6

2.2.3 Spatial variation in nitrogen and phosphorus budgets and use efficiencies

Spatial variation in current N and P surpluses and use efficiencies

Maps of the spatial variation in N surplus and P surplus, and of NUE and PUE for the year 2010 are given in Figure 2.6 and 2.7, respectively. Results show that N surpluses occur in all countries, being highest in areas with intensive animal husbandry, such as the Netherlands, Belgium, Brittany in France and the Po region in Italy (Figure 2.6 left). Since N surpluses occur in nearly all regions, the NUE is nearly always below 1 (Figure 2.7, left), whereas P surpluses can be negative, associated with regions where the PUE is higher than 1 (compare Figure 2.6 and 2.7 right). The calculated current NUE is generally between 50 and 90% but at some places, it is above 100% (not shown in legend) implying that N is mined from the soil. This is generally the case in drained peat soils where large amounts of N are mineralized. Unlike N, P is adsorbed in the soil and the availability of P for plant uptake is mainly determined by the solubility of P from the soil P pool and not so much by the added P. Values of the PUE can thus be higher than 1 in areas where crop uptake exceeds P input and the soil is mined of P, which occurs in large parts of France and central and Eastern Europe (figure 2.7).

Figure 2.6. Calculated spatial variation in N surplus and P surplus for the year 2010 in EU27.

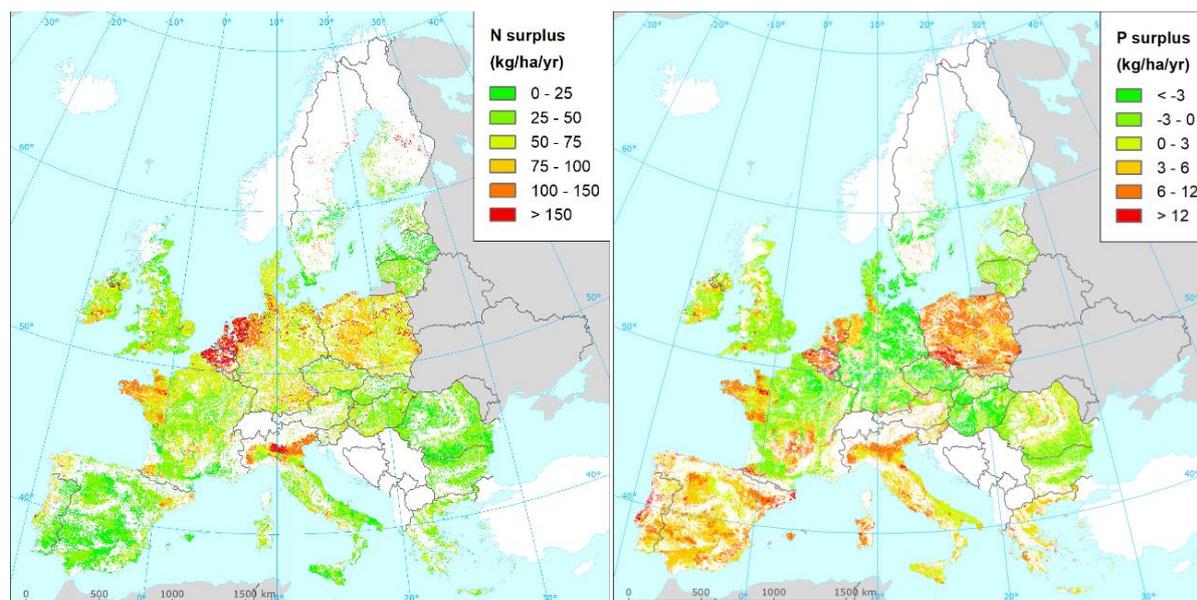
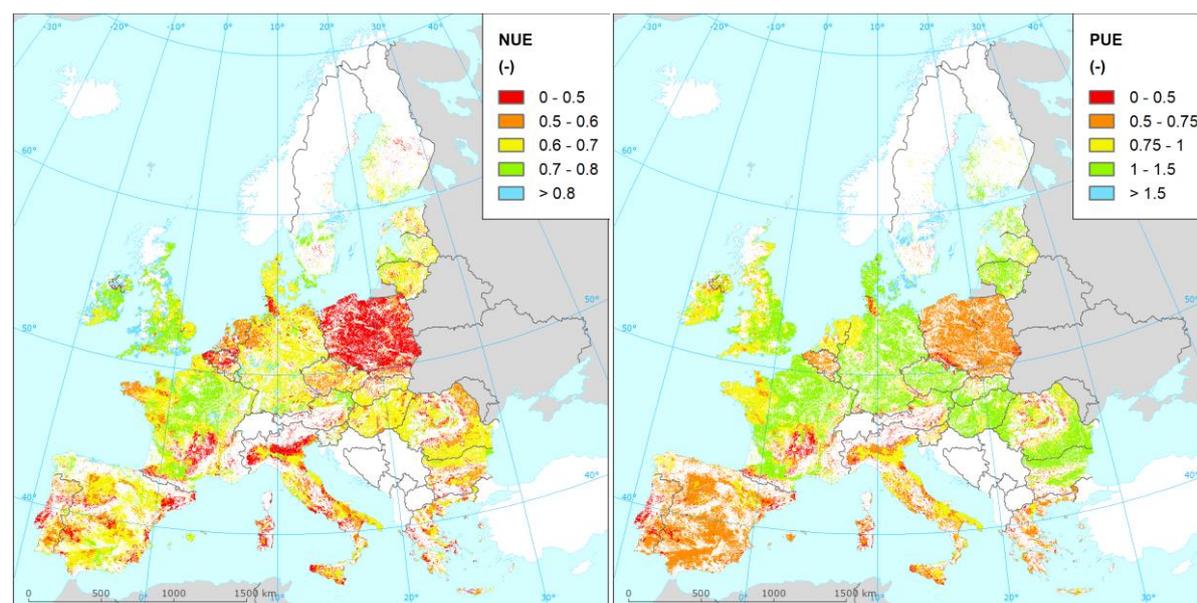


Figure 2.7. Calculated spatial variation in N use efficiency (NUE, Left) and P use efficiency (PUE, right) for the year 2010 in EU27.

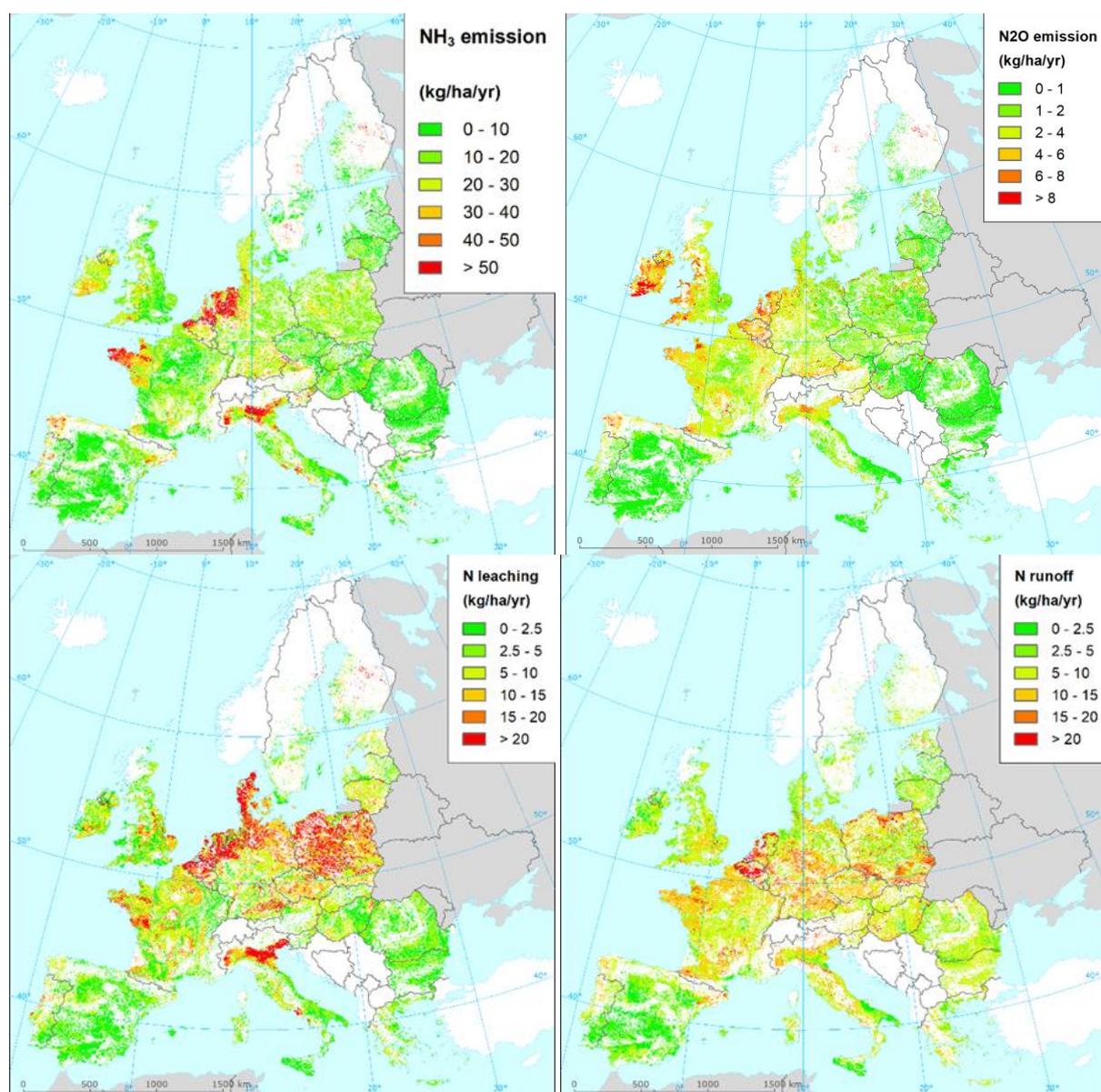


There is a reasonable correlation between NUE and PUE but there are regions where NUE is higher than PUE, such as most of Spain. Regions with an NUE or PUE <0.5 mainly occur in Poland. P surpluses are negative in many regions. Positive P surpluses mostly occur in 11 countries with on average a positive surplus, i.e. Austria, Belgium, Spain, Greece, Ireland, Italy, Netherlands, Poland, Portugal, Slovenia and United Kingdom, whereas there is on average a negative surplus in the remaining 16 countries. The positive P surplus is always related to soil P accumulation except for France where P leaching is higher than the very limited P surplus. This is positive for soil fertility when the soil P status is low, affecting crop yield, but negative when it is enough as it then does not improve soil fertility whereas it increases the risk for leaching and runoff to surface water. In all countries with a negative surplus there is a negative P accumulation (soil P mining). This is negative in countries where the soil P status is low and crop yield may thus be limited by P deficiency, whereas it is positive when the soil P status is (too) high, thus decreasing the runoff risk.

Spatial variation in current nitrogen losses to air and water

Similar to N surpluses, losses of N compounds to air and water also show large variations across the EU27. Maps showing emissions of ammonia (NH_3) and nitrous oxide (N_2O) to air, leaching of nitrogen (mainly as nitrate) to groundwater and runoff of nitrogen to surface water in EU27 for the year 2010 are given in Figure 2.8.

Figure 2.8. Calculated spatial variation in current (year 2010) emissions of ammonia (top left) and nitrous oxide (top right) to air, leaching of nitrate to groundwater (bottom left) and runoff of nitrogen to surface water (bottom right) in EU 27.



The spatial variation illustrates the difference in behaviour of the various N compounds in relation to differences in climate, land use and soil properties. NH_3 emissions are especially high in areas with intensive livestock and high manure applications whereas N_2O emissions and nitrogen losses by leaching to groundwater and runoff to surface water are also largely affected by soil type and climate. Whether these losses exceed critical values is discussed further in Section 2.3.

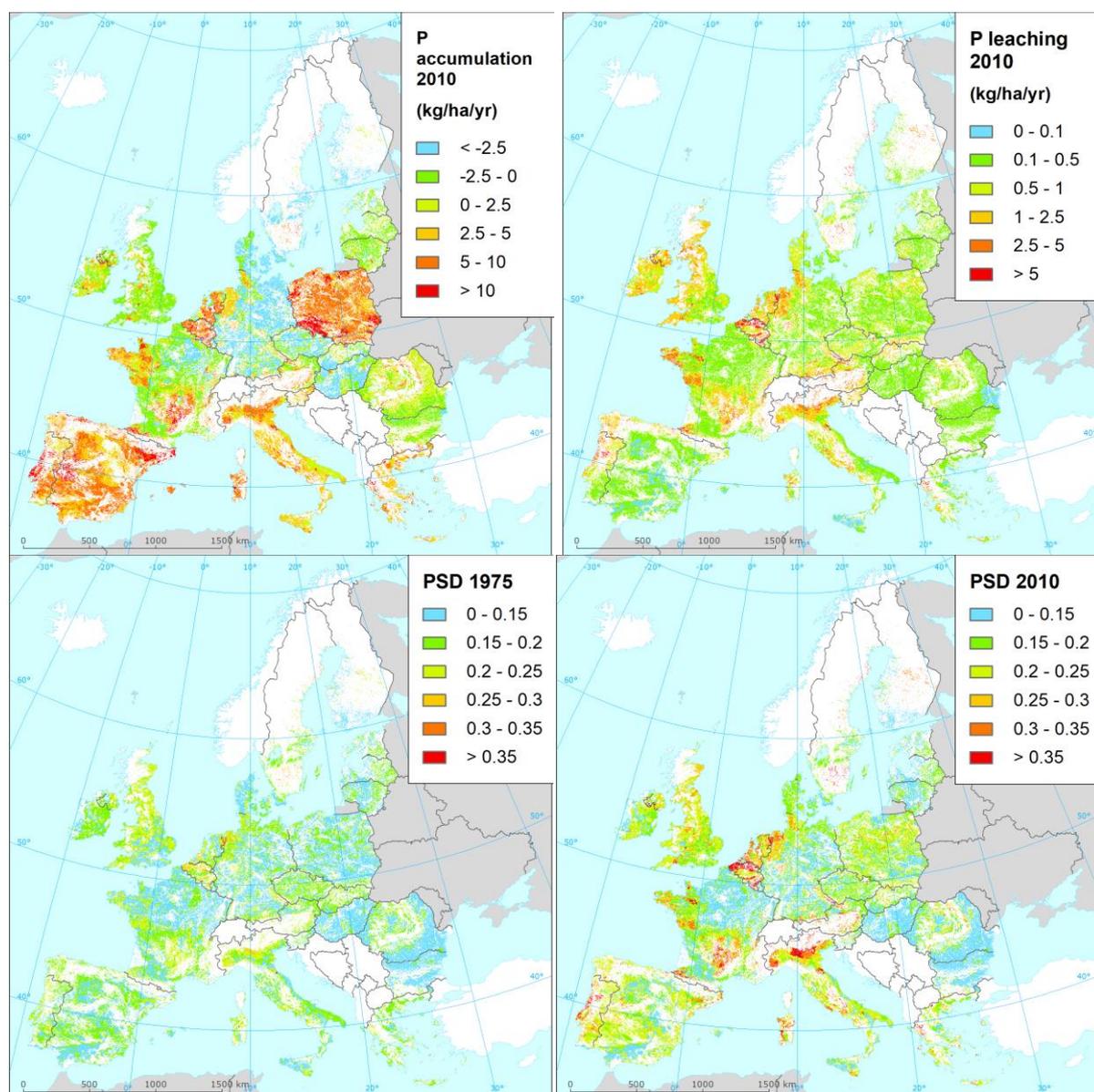
Spatial variation in current phosphorus accumulation and losses to water

Insight in the P accumulation and P leaching in response to current fertilizer and manure practices in EU-27 is relevant to gain insight in needed changes in management practices in view of sustainable agriculture, leading to sustained crop production and avoiding surface water eutrophication. A P surplus that is causing ongoing P accumulation may be good for soil quality in areas where P is deficient in view of crop growth (increasing P means an improved soil quality and thereby improved crop growth) and bad for environmental quality in areas where P is (more than) sufficient in view of crop growth (plant uptake) and where it leads to enhanced P losses to surface water, causing eutrophication. Unfortunately, we do not have good data on the current plant-available P content in the EU27.

A dynamic P mass balance model was used to assess soil P accumulation and P leaching plus runoff from the topsoil (0-30 cm) in response to current (year 2010) P inputs starting with the simulation in 1960. We also calculated the P Saturation degree (PSD), being the ratio of the current amount and the maximum amount of P that is adsorbed on aluminium and iron hydroxides. Critical PSD values have been derived based on an acceptable natural background concentration of 0.1 mg l⁻¹ ortho-P at the mean highest groundwater level. For non-calcareous sandy soils and clay soils, the critical PSD varies mostly between 0.25 and 0.30 (Schoumans & Chardon, 2015). The idea is that eutrophication risks, which are related to P runoff, and specifically the P concentration in runoff to surface waters, are low below a critical PSD.

The calculated spatial variation of P accumulation and P leaching in 2010 is large (Figure 2.9). The figure also shows the PSD in 1975 (starting year of the simulation) and 2010. The PSD pattern reflects the P accumulation areas. Where P accumulation is negative, implying P mining, the PSD is low (< 0.15) and where P accumulation is positive, it varies from approximately 0.15-0.4 (Figure 2.9). Using e.g. a PSD of 0.30 as a risk indicator (see above), this ratio is exceeded in (large) parts of the UK, Denmark, the Netherlands, Belgium, France, Portugal, Spain, Italy and Greece. The assessment of this risk indicator value for the P concentration in (surface) runoff to surface water needs, however, further evaluation. The reliability of the predictions is strongly determined by the accuracy of the calculated P surplus, the initial soil P status and the contents of Al and Fe hydroxides, determining soil P adsorption. In the future, improved soil analysis data on soil P status and contents of Al and Fe hydroxides may become available, so that the analysis can be improved.

Figure 2.9 Calculated spatial variation in P accumulation (top left) and P leaching (top right) in 2010 and the P Saturation degree (PSD) in 1975 (bottom left) and 2010 (bottom right) in EU27.



2.3 Critical nitrogen losses and inputs in view of environmental impacts and crop growth

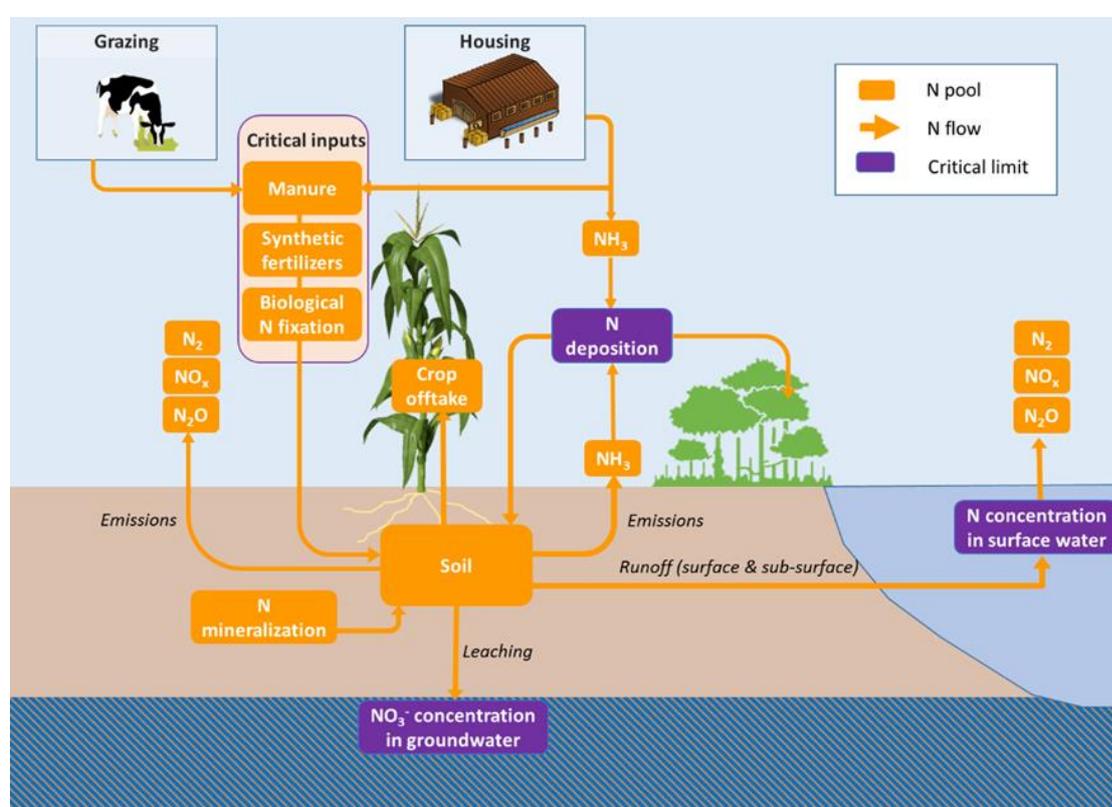
2.3.1 Assessing critical nitrogen inputs and nitrogen losses to air and water

The increased application of fertilizers and manure in the period 1960-2010 (see chapter 1.1) have increase crop production but also induced unwanted environmental effects including (see chapter 1.2): (i) elevated runoff of N and P, causing an exceedance of ecological limits resulting in eutrophication of nearby surface waters, (ii) increased leaching rates of nitrate to groundwater affecting drinking water quality, (iii) increased emission of ammonia (NH₃) and deposition on nearby terrestrial ecosystems causing their eutrophication and (iv) elevated emissions of nitrous oxide (N₂O) causing global warming. As shown before, N surpluses and related losses of N to air and water vary across the EU27 (See Section 2.2). We thus calculated the spatial variation in critical N inputs and their exceedances (current N inputs minus critical N inputs) for agricultural soils in this region. Nitrogen

inputs refer to the sources that can be managed by the farmer, including manure, fertilizer and biological N fixation, and additional N inputs by net mineralization and deposition.

Critical N inputs were calculated for three N indicators: (i) critical N deposition on natural ecosystems, (ii) critical nitrate (NO_3) concentration in leachate to groundwater and (iii) critical nitrogen concentration in runoff to surface water (Figure 2.10). The values used were: (i) critical N deposition levels in view of biodiversity impacts as presented in Hettelingh et al. (2014), (ii) a critical NO_3 concentration in groundwater of $50 \text{ mg NO}_3 \text{ l}^{-1}$, the WHO drinking water limit based on epidemiological evidence for methemoglobinemia in infants (WHO, 2011) and a critical N concentration in runoff of 2.5 mg N l^{-1} , based on an extensive study on the ecological and toxicological effects of inorganic N pollution (Camargo & Alonso, 2006).

Figure 2.10. Simplified scheme of the N flows and N indicators considered in the calculations of critical nitrogen inputs. Nitrogen indicators for which critical limits are set are shown as purple boxes.



N inputs also cause N_2O emissions, but this aspect was not included in the assessment since NH_3 emissions due to agricultural N inputs cause an enhanced CO_2 sequestration, largely compensating for the warming impact of N_2O emissions, such that the overall effect of N use in agriculture in Europe on climate is near neutral (De Vries et al., 2011a).

2.3.2 Critical nitrogen losses to air and water and their exceedances by current losses

The calculated spatial variation in critical N losses and its exceedance by current (year 2010) N losses in EU 27 are given in Figure 2.11. A distinction is made in: (i) critical ammonia emissions, in view of terrestrial eutrophication/acidification caused by the related enhanced NH_3 deposition, (ii) critical nitrate leaching in view of groundwater contamination and (iii) critical N runoff in view of surface water eutrophication.

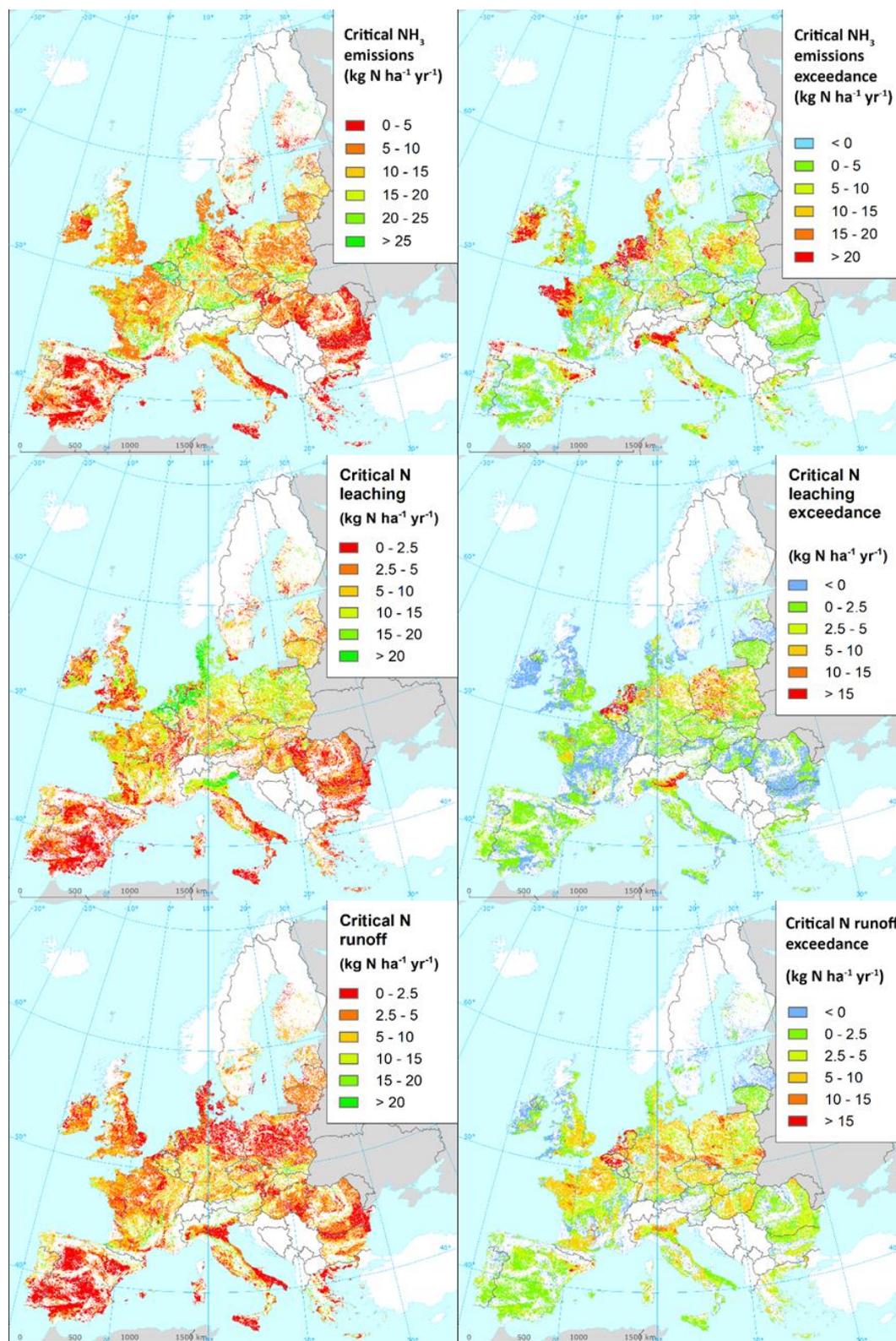
Ammonia emissions: The spatial variation of critical values for NH_3 -N emissions (Figure 2.11 top left) in view of and terrestrial acidification and eutrophication shows less variation than in current values

(Figure 2.8 top left) and in exceedances (Fig 2.11 top right). Current values are determined by spatial variation in N inputs by fertilizer and manure, in combination with variations in ammonia (NH₃-N) emission factors emissions, whereas critical values are determined by variations in critical N deposition levels (critical loads, i.e., ecosystems' sensitivity to N deposition), and, to a smaller extent, by variation in the share of agricultural area and the contribution of NO_x to total N deposition. Critical NH₃ emissions are lowest in Spain, Italy, Romania, Bulgaria and Greece (0–5 kg N ha⁻¹ yr⁻¹, Figure 2.11 top left). In Spain and Greece, this is mainly due to low N deposition thresholds (< 7 kg N ha⁻¹ yr⁻¹, data not shown), while a high contribution of NO_x to total N emissions (especially in Italy where NO_x contributes >50%, data not shown) and a high share of agricultural land (especially Romania) further reduce critical NH₃ emissions.

Nitrate leaching and nitrogen runoff: The calculated spatial variation in critical nitrate leaching in view of groundwater contamination (Figure 2.11 middle left) and of N runoff in view of eutrophication of surface waters: (Figure 2.11 bottom left) shows less variation than those in current values (Figure 2.8 bottom left and bottom right). This is not so surprising since current values are determined by spatial variation in N inputs, in combination with variations in N uptake and N leaching and runoff factors, affected by land use, soil type and slope, whereas critical nitrate leaching and critical N runoff values are determined by variations in precipitation surplus and share of agricultural land (for runoff). Considering the large variation in current nitrate leaching and N runoff, the variation in exceedances of critical nitrate leaching and N runoff is also large (Fig 2.11 middle right and bottom right).

Critical N runoff rates and N leaching rates are low (<2.5 kg N ha⁻¹ yr⁻¹) in Spain, Portugal, Italy and Greece (Figure 5d,g), because low runoff volumes in these areas mean that critical concentrations are exceeded even at low absolute losses. Due to the more stringent critical N concentration for runoff to surface water (2.5 mg N l⁻¹) compared to leachate to groundwater (11.3 mg N l⁻¹), the surface water criterion is almost always more stringent, i.e., critical N runoff rates are lower than critical N leaching rates (Compare Figure 2.11 middle left and bottom left). This difference is especially pronounced in Denmark, the Netherlands and North-West Germany, where critical leaching rates often exceed critical runoff rates by a factor 10. This is due to two reasons: first, shares of agricultural land in these regions are high, leading to limited dilution of agricultural with non-agricultural runoff and thus a lower critical N runoff. Second, in these flat and low-lying areas a larger share of precipitation surplus is allocated to base flow to groundwater rather than interflow to surface water (i.e., surface and sub-surface runoff fractions are low, leading to more dilution of N leaching below the rooting zone and thus higher critical N leaching).

Figure 2.11 Calculated spatial variation in critical losses of N compounds (left) and their exceedance by the current (year 2010) N losses (right) in EU 27 for ammonia emissions (top), nitrate leaching (middle) and nitrogen runoff (bottom).



2.3.3 Critical nitrogen inputs and their exceedances by current inputs

Spatial variation: Based on the critical N losses in view of eutrophication of terrestrial and aquatic ecosystems presented in 2.3.2, we calculated corresponding critical agricultural nitrogen inputs. Critical inputs are defined as inputs that lead to critical losses at current NUEs and emission fractions. Maps of the spatial variation in critical nitrogen inputs in view of eutrophication of terrestrial ecosystems and of surface waters and its exceedance by current N inputs are given in Figure 2.12 and 2.13, respectively. Note that ground water is not added, since the limits for surface water are nearly always much more stringent, implying that ground water is nearly always protected when surface water is protected.

Figure 2.12. Spatial variation in critical N inputs in view of eutrophication of terrestrial ecosystem by ammonia emissions (left) and its exceedance by current (year 2010) N inputs (right) in EU 27.

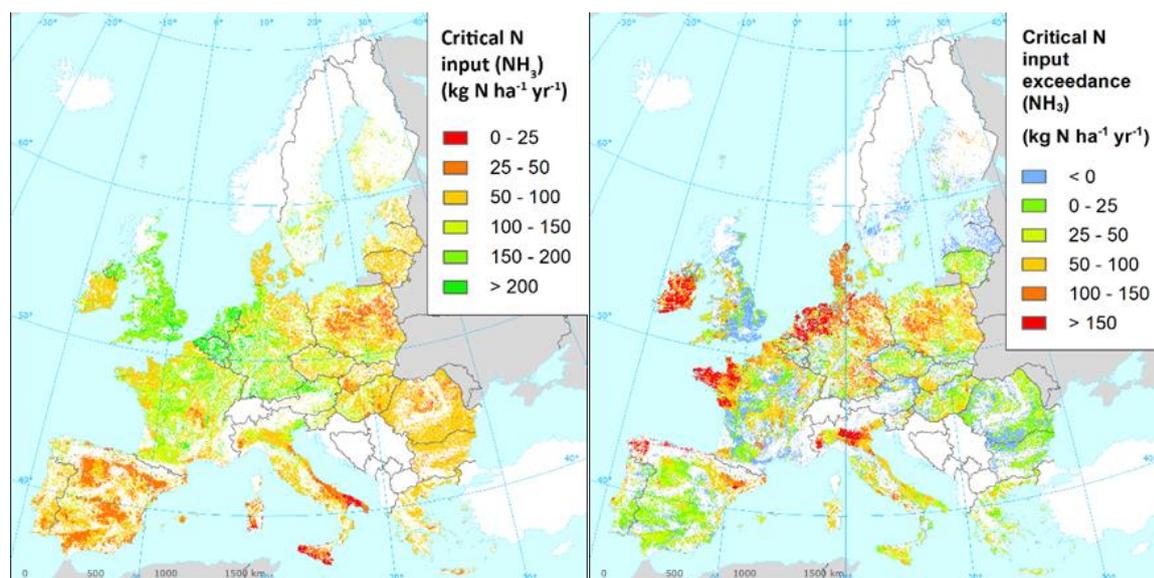
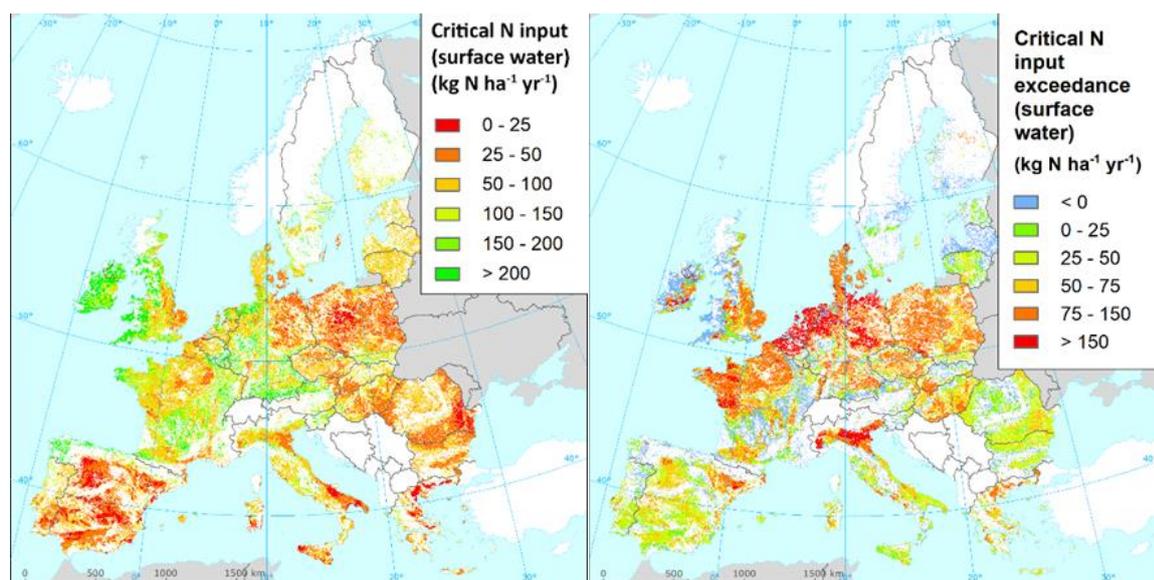


Figure 2.13. Spatial variation in critical N inputs in view of surface water eutrophication (left) and its exceedance by current (year 2010) N inputs (right) in EU 27.



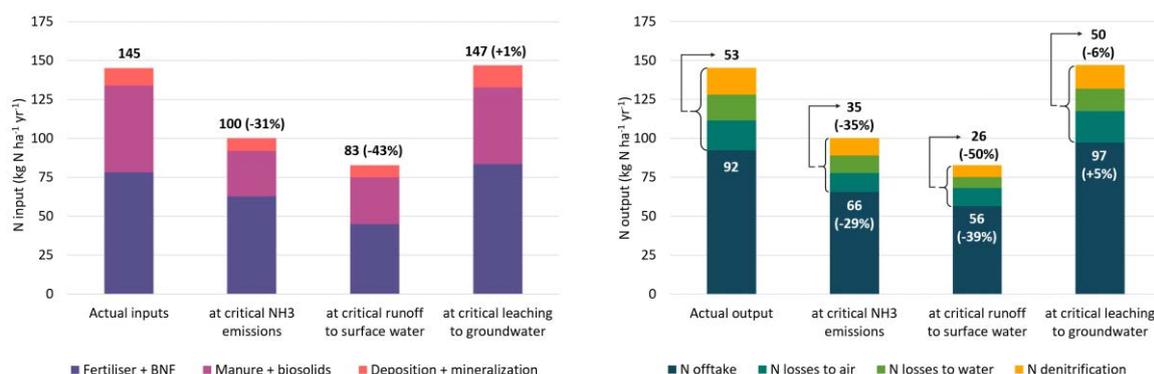
Some regions, such as the UK and Central/Northern France, display relatively high critical N inputs (Figure 2.12 left) despite relatively low critical NH₃ emissions (Figure 2.11 top left), due to low average NH₃ emission fractions in these regions. The largest exceedances of critical N inputs by actual inputs

generally occur in regions with high N manure inputs, including Ireland and western UK, the Netherlands, Belgium and Luxembourg, Bretagne in France and the Po valley in Italy (Figure 2.12 right). Despite relatively high critical N inputs in the Netherlands, Belgium and Eastern Germany (mostly 150–200 kg N ha⁻¹ yr⁻¹), owing to relatively high critical loads, high actual inputs in these regions still lead to exceedances critical inputs by > 75-150 kg N ha⁻¹ yr⁻¹ (Figure 2.12 right).

Critical N inputs for the surface water criterion (Figure 2.13 left) show a decreasing trend from Northern to Southern Europe, driven by a decrease in precipitation surplus. Low critical N runoff (Figure 2.11 left) generally also imply low critical N inputs for the surface water criterion (Figure 2.13 left), but not always. For example, across large areas of Ireland critical N input rates exceed 200 kg N ha⁻¹ yr⁻¹ despite low critical runoff rates of 0–10 kg N ha⁻¹ yr⁻¹. This can be explained by a high share of grasslands in Ireland, with higher denitrification and lower N losses to water compared to arable land. While critical N inputs for the surface water criterion are lowest in South-Eastern Europe (Figure 2.13 left), the highest exceedances of critical N inputs by actual N inputs occur in North-Western Europe (Figure 2.13 right), because actual inputs in these regions are much higher (Figure 4a). or the current N use efficiency is low, such as in Poland (Figure 2.7 left).

European average values: Based on the derived spatially explicit critical N inputs, boundaries for N inputs and losses were derived at country- and EU-level. Figure 2.14 shows mean actual and critical N inputs to agriculture as well as mean actual and critical crop N offtake and N losses from agriculture for the EU. Complying with thresholds for N runoff to surface water requires the highest reductions in N inputs (-43%), followed by thresholds for NH₃ emissions (31%, Figure 2.14a). Average critical N inputs related to thresholds for N leaching to groundwater (147 kg N ha⁻¹ yr⁻¹) are 1% higher than actual N inputs, which means that on average, increases in N inputs in areas where thresholds are not exceeded are higher than needed reductions in areas where thresholds are exceeded.

Figure 2.14 Average actual (year 2010) and critical N inputs (left) and N outputs (right) for all agricultural land in the EU. In the left figure, the numbers above bars show total N inputs in kg N ha⁻¹ yr⁻¹, in the right figure, the numbers above bars show total N losses, while numbers for N offtake are shown within bars (both in kg N ha⁻¹ yr⁻¹). Percentages show difference between actual and critical N inputs, losses or offtake.



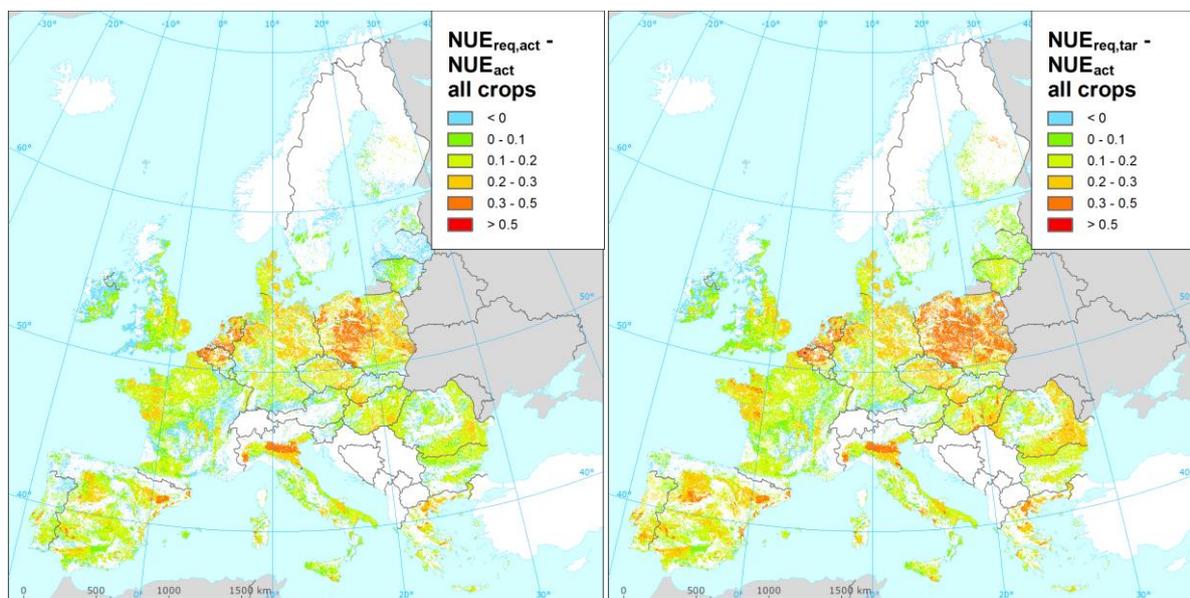
For all impacts, relative reductions needed to respect thresholds are higher for N losses than for N inputs (Figure 6). In order to respect thresholds for N runoff to surface water, for example, N runoff needs to decrease by 50% while N inputs need to decrease by 43% (from 145 to 83 kg N ha⁻¹ yr⁻¹, Figure 6). This implies that on average, higher reductions in N losses are required in areas with lower NUEs. For all thresholds, required reductions in manure inputs are higher than required reductions in fertilizer inputs, with the difference being especially pronounced for the deposition threshold (requiring a 48% reduction in manure inputs compared to a 20% reduction in fertilizer inputs, see Figures 6), due to the higher NH₃ emission fraction of manure compared to fertilizer.

2.3.4 Needed increase in nitrogen use efficiencies in view of crop production

To meet the world's food production needs, N fertilizers are needed, but at the same time the environmental footprint of agricultural N use on water quality, biodiversity and climate has to be reduced (Foley et al., 2005; Foley et al., 2011). Until now, we only focused only on the required reductions in N inputs (critical N inputs) considering the ecological consequences of elevated N inputs in EU27, while neglecting the need of N in view of crop production. As described in Section 2.1.3, when critical N inputs are below current N inputs, environmental objectives can only be reached at a lower N input, causing a loss in crop production, unless NUE is increased. With an enhanced NUE, a lower fraction of N is lost to the environment. It has been shown that there are large opportunities to reduce environmental impact of agriculture by reducing nitrogen inputs, while still allowing an increase in production of major cereals (Mueller et al., 2012).

The NUE increase that is needed to attain the current crop yield and a target crop yield (80% of the yield potential; see section 2.1.1) while N losses are acceptable is given in Figure 2.15. On a European scale, a 27% increase in N inputs (from 145 to 185 kg N yr⁻¹) is required to reach the target crop yields at current NUE (not shown). The figure presents the difference between the required NUE to stay within environmental limits for groundwater and surface water as compared to the current NUE. The required NUE to stay within environmental boundaries is nearly always between 50 and 90% and in many cases lower than the current NUE. These are areas where the current production does not lead to N losses to either air and/or water exceeding critical limits. % . Notable regions with relatively large (> 0.3) difference between required and current NUE are the Netherlands, the Western part of Belgium, large parts of Poland, the Po valley in Italy, Eastern Spain and parts of Greece. In Northern and Central parts of the EU this difference is most likely related to the high fertilizer application rates, whereas in the Mediterranean areas this is due to the low precipitation surpluses. Both effects are causing high N concentrations in runoff and nitrate concentrations in groundwater.

Figure 2.15 Calculated spatial variation in the difference between the required NUE to attain the current crop yield (left) or target crop yield (right), while nitrogen losses to surface water and nitrate losses to groundwater are acceptable.



To achieve surface water quality targets without crop production losses, the European average NUE needs to increase from 0.64 to 0.78, whereas achieving groundwater targets requires a modest increase from 0.64 to 0.67. The needed NUE increase can up to a certain level be attained by precision fertilization management such as the 4 R strategy (placement of fertilizers at the right rate, with the right type at the right time and the right place). In hotspot areas, however, crop production and N

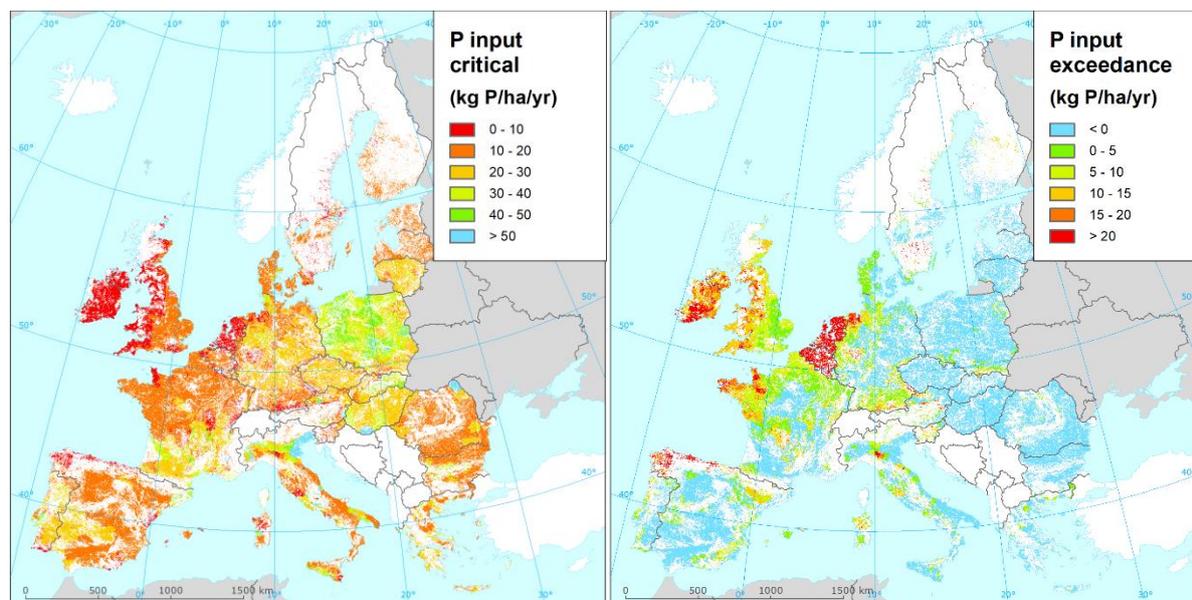
thresholds can only be reconciled at NUEs of > 0.90, which is not feasible. The best approach reduce NH₃ emissions below critical limits is to reduce NH₃ emission fractions from manure by e.g. low-emission housing systems and manure application methods such as injection. More details on the options to reconcile food production with air and water quality by improved management are given in Schulte-Uebbing and de Vries (2021).

2.4 Critical phosphorus inputs in view of environmental impacts and crop growth

Unlike N, P is adsorbed in the soil. However, at steady state, dynamic aspects and thus P soil adsorption can be neglected and the long-term critical 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 runoff. The critical P runoff from the soil can be derived by multiplying the current runoff rate of water with the critical P concentration in runoff to surface water, being equal to 0.15 mg/l of total P.

A more simple approach is to calculate the critical P input by fertilizer and manure on the basis of equilibrium P fertilization, implying that the critical P application rate is equal to the P removal by the crop, while assuming that P losses by runoff and leaching are equal to P deposition, as these are both in the same order of magnitude. This approach is used in countries like the Netherlands to assess the acceptable P input to comply with manure legislation, based on an available soil P concentration that does not limit crop growth. Maps of the spatial variation in critical P inputs, in view of equilibrium fertilization while assuming target crop yields, and the exceedance by current P inputs (current P application minus target P removal) are given in Figure 2.16.

Figure 2.16 Spatial variation in critical P inputs, considering target crop yields (left) and their exceedance by the current (year 2010) P inputs (right) in EU 27, in an equilibrium situation where the soil P status does not limit crop growth.



It shows a needed decrease in P inputs in western Europe including Ireland, UK, the Netherlands, Belgium and Luxembourg and Bretagne in France. Inversely, increased P inputs are needed in the remaining parts of Europe in view of crop P demand.

A crucial question for the critical P load calculation given above is whether the available soil P concentration that does not limit crop growth is not causing an exceedance of a critical P concentration in runoff in view of impacts on surface water quality. This is not considered in our assessments since information is lacking to make such an analysis. The assumption is that long-term equilibrium P fertilization, keeping the soil P status constant, also avoids environmental problems.

3 Current and critical inputs of heavy metals in European agriculture

3.1 Heavy metals in soils: concentrations, benefits and risks

3.1.1 Sources and presence of metals in soils

Presence and sources

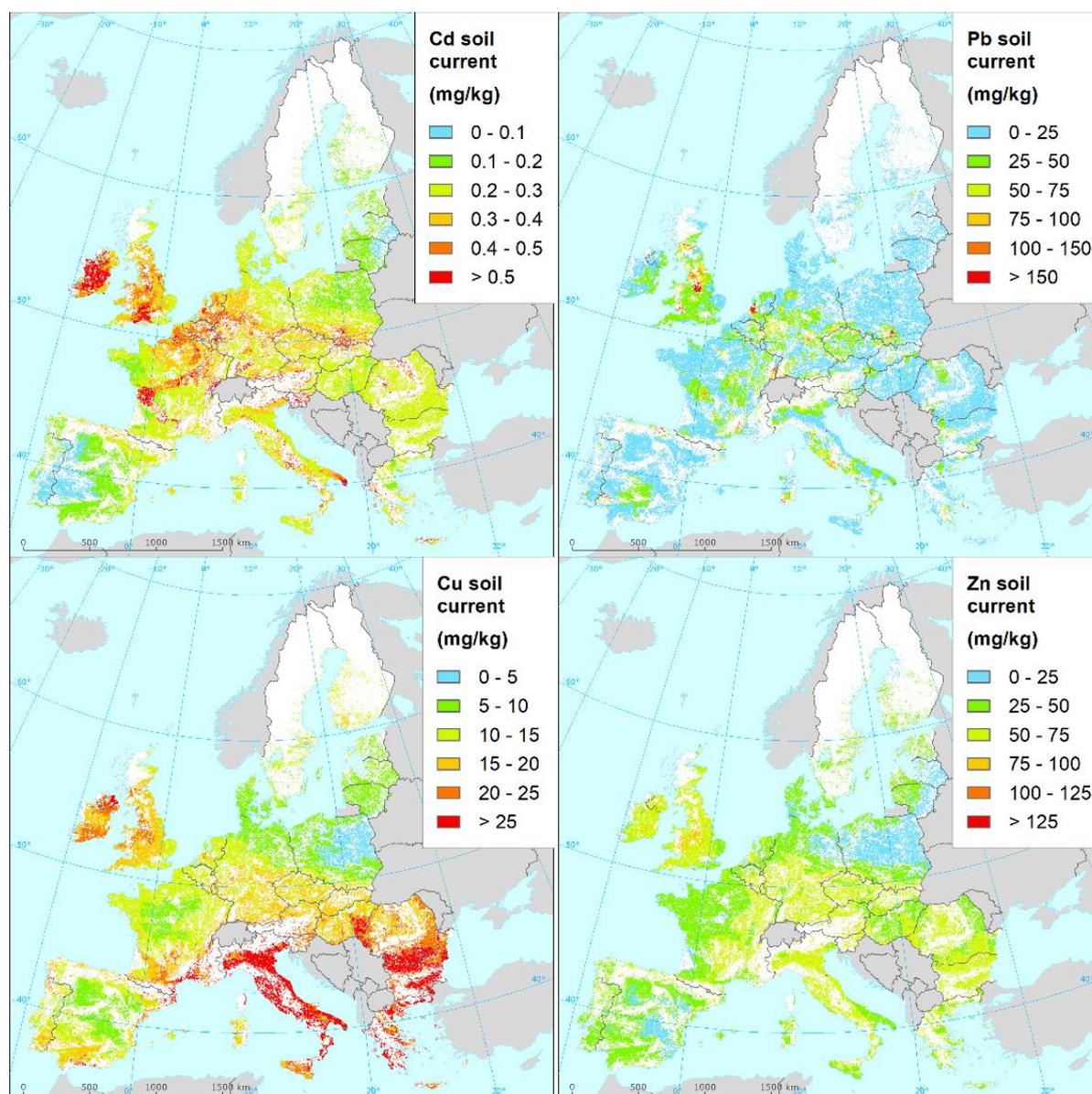
Most heavy metals, including the ones included here, i.e. cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn), have always been present in soil due to natural causes, either because of their presence in the parent material from which the soil has been developed or via other, usually less abundant, natural inputs including atmospheric deposition (insofar related to natural emission like eruptions from volcanoes) or deposition from sediments along rivers. Differences in background levels can be substantial due to differences in the composition of the parent material and/or diffuse natural inputs both at national level (e.g. Mol et al., 2012 illustrating differences between soils in the NL) as well as at EU level (Reiman et al., 2014). In most soils however, background levels are such that the main functions of the soil are not affected. As a result of industrialization and intensification of the use of (arable) land especially during the 20th century, a substantial increase in the metal load to soil for most metals occurred. On a national scale this was largely due to increased levels of atmospheric deposition -from industrial emissions and traffic- as well as increased inputs from agriculture (manure, biosolids, fertilisers). Regionally local sources include deposition of polluted sediments along rivers, the regionally elevated inputs from atmosphere near industrial complexes.

For the metals considered here, inputs from atmospheric deposition (lead, cadmium) and inputs from agriculture (copper, zinc and to a lesser extent cadmium) have been dominant. Both for lead and cadmium however, EU-wide inputs from industrial emissions have decreased substantially during the last 3 decades due to emission control at the source (industry, traffic). Major contributors to the inputs in agriculture include those via animal manure (copper, zinc), sewage sludge (lead, and to a lesser extent cadmium and zinc), compost (zinc) and inorganic fertilisers (cadmium). Inputs in agricultural systems however are highly variable both on a regional scale as well as on a national scale. Inputs via biosolids are still substantial in countries where sewage sludge is allowed to be used in agriculture (e.g. in the UK) but negligible in those where sewage sludge is legally banned from use in agriculture or technically banned due to very strict standards imposed on sludge (e.g. in the Netherlands). Even though the use of compost and other organic soil improvers is growing in agriculture, inputs of metals from compost usually are low compared to those from other sources. The use of animal manure in countries with intensive animal husbandry (e.g. the Netherlands, Belgium, France and Denmark) is the main source for inputs of copper and zinc. The use of mineral P fertilisers is the main source for cadmium even though there is a substantial variation between countries depending on the amount of fertilizer used and the quality thereof (Römkens et al., 2018).

Spatial variation in current concentrations

Due to variation in natural levels in soil forming minerals and regionally variable additional inputs from both local and diffuse sources, levels of cadmium, lead, copper and zinc vary across Europe as illustrated in Figure 3.1. The maps reveal that for cadmium there is a clear relation between the degree of industrialization with higher levels in countries like the UK, NL, B and parts of PL, and lower levels in rural areas of Spain and several Baltic countries. For lead and to a lesser extent zinc the spatial variation in soil concentrations is less pronounced with only small areas with high concentrations reflecting historical high inputs (NL, UK). Nevertheless, both for zinc and lead, levels in both NE parts of the EU and Spain are lower compared to central Europe which is a combination of differences in parent material and industrial (atmospheric) and agricultural inputs (Reiman et al., 2014).

Figure 3.1. Current concentrations of cadmium, lead, copper and zinc in the topsoil across EU-27 member states, based on the combined data from the GEMAS database (Reiman et al., 2014) and the FOREGS data base (Lado et al., 2008).



Results for copper clearly reveal a difference in the background levels with higher levels in south-eastern parts of the EU and low levels in the north-eastern parts of the EU. It should be noted that the source data underlying these maps are largely based on samples collected from arable soils and grassland soils and therefore do not or only in a limited way show variation in levels near or in urbanized areas. At present, the LUCAS database is being extended with data for heavy metals (ESDAC, 2019; Orgiazzi et al., 2018). However, at the time of writing these data are not yet available to be included in this report. Only for copper an updated map from the LUCAS database has been prepared (Ballabio et al., 2018.) based on LUCAS data samples upscaled to a resolution 500 x 500 m.

3.1.2 Assessment of heavy metals in soils: towards risk-based quality standards

Risks of heavy metals in soils: protection of specific environmental targets

The key issue to be addressed is to assess to what extent metals present in soil or those added to soil will affect levels in soil and whether such changes in the level in soil results in unwanted effects on

selected environmental compartments. The following compartments (or targets) that need to be protected from adverse effects due to the presence of heavy metals in soil can be distinguished:

- Soil organisms (soil ecosystem),
- Crop production (yield) and quality of crops
- Quality of animal products for human consumption,
- Animal health (e.g. grazing cattle)
- Quality of water leaching from the soil including drinking water, groundwater or nearby surface water.

To avoid such effects, the quality of soil is currently regulated both via controls of *inputs* to soil as well as legal *limits or thresholds in soil*. Such limits in soil (soil quality standards) aim to minimize effects on selected environmental compartments (crops, water, ecosystem). In addition, several compartments (like food or water) also have additional quality standards like maximum levels in food or (drinking) water. This is summarized in Table 3.1, whereas figure 3.2 shows examples of different kind of regulations related to heavy metals in soil.

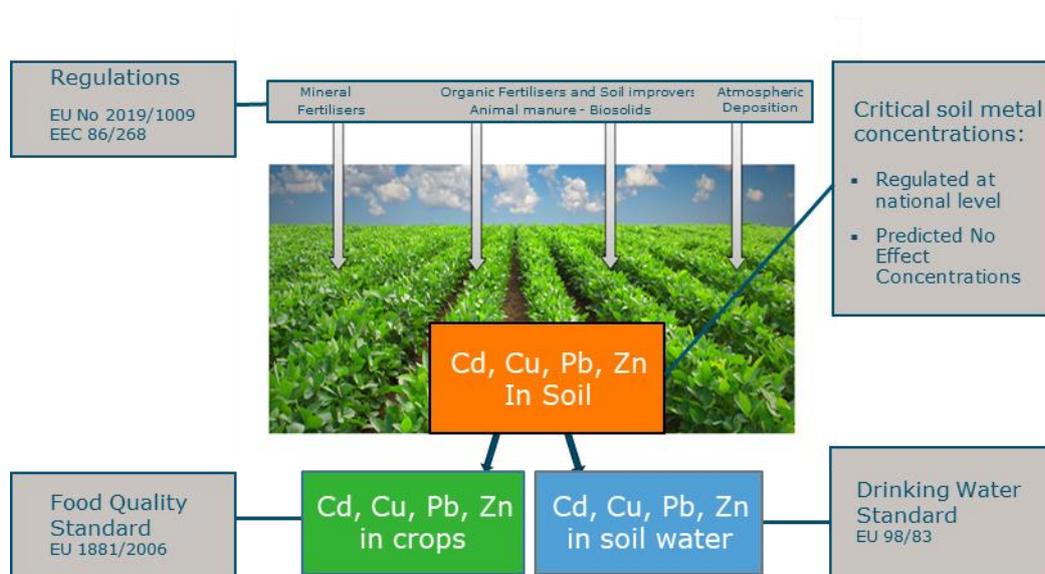
The specific regulations that have been set in place with respect critical limits in soil, food or water aim to protect:

- The local terrestrial soil ecosystem including soil microorganisms, but also health of above ground animals
- The quality of food, fodder and animal products (meat, milk etc) grown on the soil and health of (farm) animals feeding on the receiving soil
- The quality of ground- or nearby surface waters used for drinking water or habitat for aquatic organisms

Table 3.1. Regulation to limit pollutant (metal) inputs to soil and set standards for pollutant (metal) concentrations in soil, food and water

Regulation to limit Inputs to soil		Regulation to set standards for concentrations in soil		Regulation to set standards for concentrations in food and water	
Relevant directive	Mechanism	Relevant policy	mechanism	Relevant directive	mechanism
Council Directive 86/278/EEC on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture	Maximum levels in sludge, maximum application rate, control of soil quality and changes therein	National soil quality standards (MS level)	Usually including standards for identification of clean soils (background levels) or soils in need of remediation	Commission Regulation (EC) No 1881/2006 and commission regulation (EU) 2021/1323 setting maximum levels for certain contaminants in foodstuffs	Prevention of excess intake via food (product quality)
Regulation (EU) 2019/1009 laying down rules on the making available on the market of EU fertilising products	Maximum levels in mineral and organic fertilisers/soil amendments			Council Directive 98/83/EC on the quality of water intended for human consumption	Prevention of exposure via drinking water

Figure 3.2. Schematic overview of inputs and outputs of metals to soils including a number of legal frameworks that regulate inputs and quality of outputs.



Limitations of current soil quality standards used in policy making

Current soil quality standards, including background-, threshold- and critical values as defined for policy-making purposes, use criteria which are not necessarily related to the specific environmental targets listed here. Often, target values set in many countries are derived from inventories of actual concentrations in soil thus not being related to any type of risk. If such inventories are based on non-contaminated sites only, the resulting target or background values usually are below levels at which risks occur. For actual soil clean-up values currently in use, the rationale behind the 'risk' principle used is often not explicitly described or explained or vary widely between countries resulting in a marked range of soil quality guidelines across for example for cadmium in most EU member states (Carlon et al., 2007). Such differences in approaches to calculate soil quality standards also has resulted in a sometimes confusing array of indicators of these standards including but not limited to 'quality values' (UK), 'target values' (Netherlands), 'maximum allowable values' (Switzerland), 'soil orientation values' (Germany) or 'interim assessment criteria' (Canada), as listed by De Vries and Bakker (1998).

Information on how these critical limits were derived was not given in these overviews. It is interesting to note that, as of now, there are still no EU-wide standards for soil. Despite attempts to harmonize standards for soil (Carlon et al., 2007), the concepts and principles used by various members states vary widely resulting in substantially variable soil quality guidelines or standards between countries. At present most member states do have national soil protection frameworks, including standards for soil.

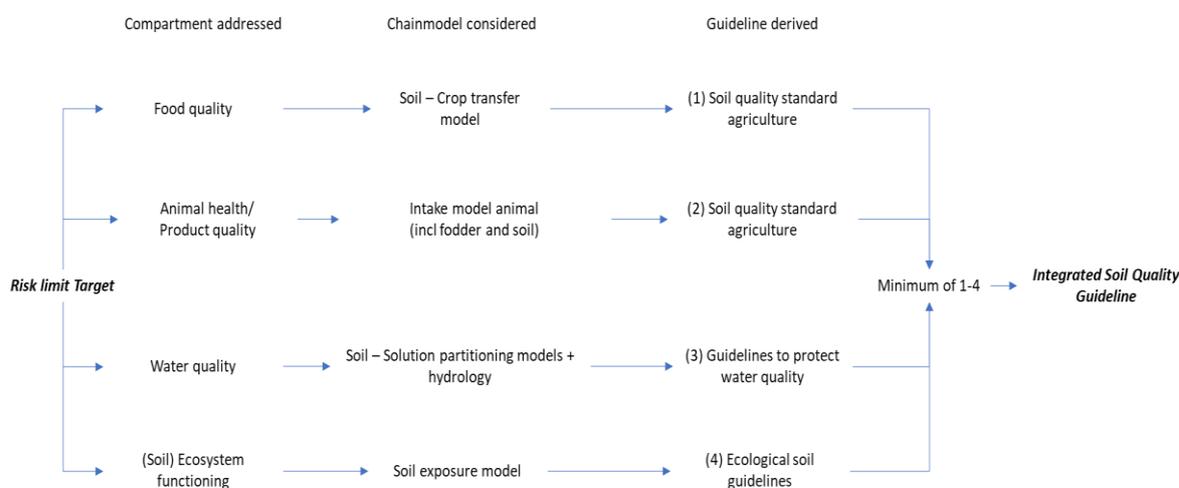
A complicating factor for some metals, notably copper and zinc, is that they are also essential micronutrients which means that at *low* levels deficiencies for crops and animals can be a growth limiting factor. Clearly, excess levels of both copper and zinc are known to cause adverse impacts on soil organisms and plants especially in surface water systems. This implies that for these elements both low level limits (deficiency level) and high-level limit (toxicity levels limits) should be considered in the evaluation of soil quality. Lower critical levels are those below which crop performance is reduced due to a lack of micronutrients whereas upper levels include levels beyond which adverse effect occur either in soil or in adjacent environmental compartments (e.g. surface water). It is interesting to note that upper levels of critical limits (soil quality standards) are common and in use across the EU for most metals even though differences in the value of most standards between MS

are common. Information on lower critical limits for copper and zinc on the other hand is limited and as of now not implemented in any national system for the evaluation of soil quality. For cadmium and lead beneficial effects have not been identified as of now and for these elements therefore only upper level limits are in place.

Approaches to relate targets for water and food to soil standards

It is important to realise that limits in Table 3.1 with respect to food, fodder and water or dissolved critical concentrations in view of impacts on soil organisms need to be converted to corresponding levels in soil. To do this, relationships between the risk limit in the target (water, food etc) and corresponding levels in soil are essential. This is schematically shown in Figure 3.3.

Figure 3.3. Conceptual approach to link risk limits in environmental target to integrated soil quality guidelines.



At present, there are quite a few risk limits for specific targets at EU MS level. Relevant ones in this respect include quality criteria for drinking water, surface water and food. As illustrated in Figure 3.3, the conversion of such quality criteria in targets to soil guidelines requires the use of models. At present, however, there is no consensus at EU level on what model to use to obtain generic standards or models to be applied across the various member states. As a result, soil guidelines derived according to this risk-based principle are -sometimes highly- variable as was demonstrated by Carlon et al. (2007) for cadmium.

In this report we will illustrate the potential of such model-based approach to link specific targets (water, food) to environmental guidelines (soil standards) as was done by De Vries et al. (2007a; 2007b). One key aspect is that the underlying model concepts to translate guidelines for quality of food or water to soil consider differences in soil type to correct for pH and soil properties like organic matter or clay.

The soil-based quality standards or thresholds thus derived are considered risk-based since they are specifically related to a corresponding adverse effect in one or more environmental targets. Such risk-based soil quality standards therefore differ from traditional soil quality guidelines in that the latter are not necessarily risk-based. Each pathway (e.g. from soil to crops) results in a chain specific guideline. These can be used on their own as is the case for example in agriculture in the Netherlands where specific guidelines to protect food quality are in use based on soil to crop transfer models (Römken et al., 2007). Ultimately these -in this case four – target specific thresholds can be integrated to an

overall soil protection threshold. If levels in soil remain below these what is called in Figure 3.3 integrated soil quality guidelines, no effects on either target are to be expected.

A key aspect related to the risk any chemical present in soil (or water) can pose is the fact that the effect of such contaminants is largely related to the *actual availability* of a contaminant, rather than the total concentration (in soil or water). Here we define availability as the degree to which metals in soils can be taken up by crops, soil organisms or leach to the ground- and surface water. For most metals this strongly depends on a combination of type of input and soil properties.

The capacity of the soil to retain or release metals to water for leaching or uptake by crops or soil organisms is related to soil properties like acidity (pH) and the amount – and quality- of organic matter and clay. Knowledge of both the inputs to soil and the processes that control the fluxes of metals in soil (uptake, leaching) is crucial to assess short- and long-term changes of levels of metals in soils. Depending on the inputs to soil and losses from soil (uptake/leaching) metal levels in soil can either increase or decrease. This then means that acceptable levels in soil, or even guidelines will range from low levels in ‘sensitive’ (e.g. low pH for metals) to higher levels in ‘non-sensitive’ soils (e.g. calcareous clay soils). Correction for soil type thus required is already common in current soil quality standards among others in the Netherlands where first tier standards for soil are corrected for organic matter and clay (<https://wetten.overheid.nl/BWBR0023085/2018-11-30/#BijlageL>).

Aim and content of this section

In paragraph 3.2 and 3.3 of this report we first document the current status of inputs and outputs of 4 selected metals (cadmium, copper, lead and zinc) and calculate, at MS level, the resulting balance for each of these metals. In addition, maps are presented at NCU level to provide insight in the regional distribution of such balances (paragraph 3.2). In paragraph 3.3 examples are given of the risk-based approach for ecological risks (3.3.2) and food safety (3.3.3). In 3.3.2, we present data on current levels of cadmium, copper, lead and zinc in soil in combination with regional (NCU) variable ecological soil critical thresholds to assess where, at present, effects can be expected. In addition, critical metals inputs are presented to assess where long-term soil ecological impacts could be expected. In 3.3.3 we focus on the assessment of the status of soils and inputs to soil in view of food safety. This is done for cadmium only for two reasons: *i.* for copper and zinc, which are also considered micronutrients there are, at present no relevant standards in food for human consumption, and *ii.* for lead, for which food quality guidelines are in place (Reg. commission regulation (EU) 2021/1323) models to relate levels in food to corresponding levels in soil considering differences in soil properties are still very poor in their performance. For lead uptake by crops is less if at all related to lead in soil and/or soil properties like pH and organic matter.

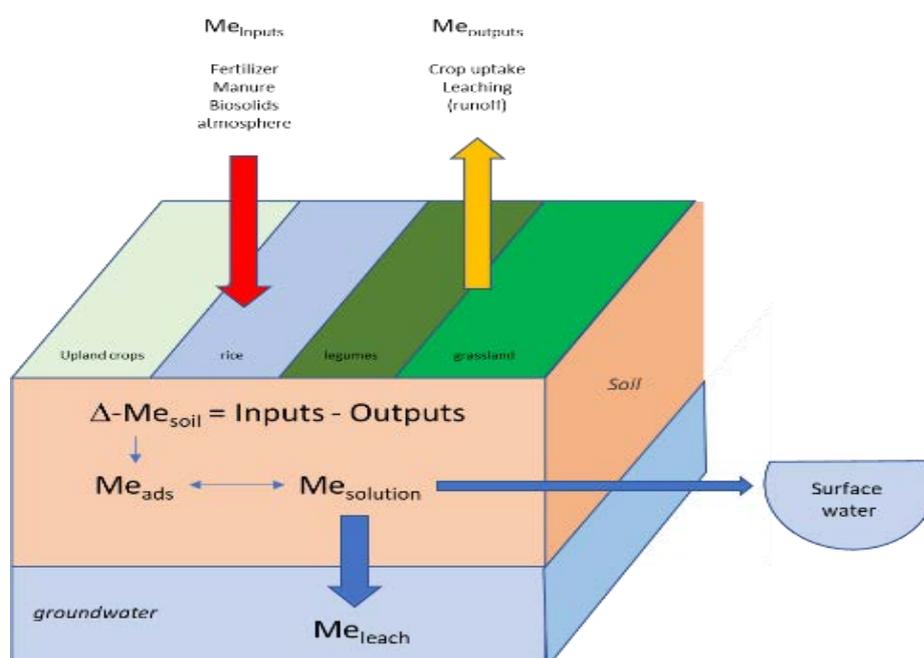
The derivation of soil critical limits in view of protection of surface water or groundwater is not included in this report, since we focus on the quality of topsoil of arable (and grassland) soil. Concentrations in groundwater and surface water, however, are only partly if at all related to the concentration of metals in the topsoil since water percolates through the entire soil profile and, in many area in the EU will pass underlying aquifers to reach surface waters or groundwater used for extraction of drinking water. Hence the quality of the topsoil in most arable cropping systems is not or only marginally related to the ultimate quality of water. Imposing a direct relation between (top)soil quality and that of drinking water therefore would possibly results in overprotective soil quality guidelines especially in soils with a low buffering capacity such as acid sandy soils. On the other hand, such direct relationship could lead to extremely high acceptable levels for elements with a low mobility due to a high sorption affinity for organic matter or clay. In any case the soil to groundwater or surface water pathway requires far more information on both soil (notably the properties and contamination levels in the subsoil) and hydrology.

3.2 Heavy metals budgets in European agricultural soils

3.2.1 The fate of metal inputs

Metals present in, and added to, arable cropping systems either remain in the soil adsorbed to reactive surfaces like organic matter or clay, are taken up by crops or are transported to lower soil layers with the percolating soil water. The degree to which metals accumulate or are removed from soil depends on both the magnitude of the input and output fluxes as well as the geochemical and biological processes that take place in soil, notably the adsorption – desorption processes and plant uptake. Hence, to calculate metal balances for arable cropping systems both inputs and outputs need to be quantified. The distinguished inputs and outputs are given in Figure 3.4.

Figure 3.4. Simplified overview of metal fluxes in an agricultural soil considered in this approach



In the approach used in this report the following fluxes are included in the calculations:

- Inputs:
 - Mineral fertilisers (Fert): here all inputs from mineral fertilisers (largely N, P and K) fertilisers as well as lime are accounted for. In this respect especially inputs for cadmium are affected by the use (and quality) of mineral P fertilisers.
 - Animal manure (AM): especially for copper and zinc, inputs via animal manure are considerable and in many cases form the largest part of the load of both metals to arable soils (Eckel et al., 2005).
 - Biosolids (BS, here compost + sludge): even though biosolids are not used in all countries, inputs for lead can be substantial in those countries where biosolids are applied at larger scale e.g. in the UK. In other countries, a.o. the NL, inputs from biosolids are virtually absent due to legal limitations.
 - Atmospheric Deposition (AD): historically inputs from atmosphere were substantial for metals including lead and cadmium, and to a lesser extent zinc. Due to emission control, deposition of most metals has decreased substantially to levels that, on average, are less relevant compared to most other sources.

- Outputs:
 - Crop uptake (Upt.): removal of metal via crops is relevant but in most cases levels of metals in crops are low and the resulting removal rates limited compared to e.g. losses via leaching. Differences in uptake patterns between crops causes a further marked range in removal rates with higher removal rates in high biomass crops like potato, rice or maize compared to a.o. most legumes like beans and peas or fruit that do not or only in a limited amount accumulate metals in the aboveground biomass.
 - Leaching (Le.): for most metals removal of metals via leaching is the dominant output flux. Here both the net water flux (precipitation minus evapotranspiration) and the geochemical conditions in soil (pH, content of organic matter and clay) determine the net removal rate. Due to climatic conditions and regionally variable soil properties conditions are such that the net removal via leaching is substantially higher in NW parts of the EU compared to most Mediterranean MS.

In the calculations, we focus on a topsoil of 0 to 30cm and the outflow at 30 cm is based on leaching and sub-surface runoff. Metal removal by erosion and/or surface runoff is not considered. For each spatial scale level, the balance for cadmium, copper, lead and zinc is calculated as the difference between inputs and outputs (in gram per hectare) according to:

$$Me_{\text{Balance}} = Me_{\text{Fert.}} + Me_{\text{AM}} + Me_{\text{BS}} + Me_{\text{AD}} - Me_{\text{Upt.}} - Me_{\text{Le}}$$

Using these inputs and outputs includes the majority of diffuse sources from agriculture and atmosphere. Local sources are not included. More information on relevant inputs and outputs can be found in De Vries et al. (2014). Results in the following paragraphs are presented both as the total metal budget at EU level (in ton year⁻¹, par. 3.2.2) and at the most detailed spatially explicit output level currently used (NCU, in gram ha⁻¹ yr⁻¹, par. 3.2.3).

3.2.2 Metal budgets at EU level

Metal budgets at EU-27 level for copper, zinc, cadmium and lead as calculated by INTEGRATOR are shown in Figure 3.5. Data at EU-27 level for all arable land (arable crops and grassland) confirm the dominant contribution of manure to the total load for copper and zinc (77-78%). For cadmium, inputs from mineral fertiliser are approx. 43% of the total inputs followed by atmospheric deposition (31%) and manure (21%). In Figure 3.6 the relative contributions are shown for arable land and grassland separately.

For lead atmospheric deposition is still the dominant source (47%), despite the considerable reduction in lead emission from traffic since the 1980's following the introduction of lead-free fuel. The contribution of biosolids is limited and ranges from approximately 4 to 7% for cadmium and zinc to approximately 10 to 12% for copper and lead. For copper and zinc outputs from leaching and crop uptake are approximately 50% for each whereas for cadmium and lead leaching is the dominant process (> 85%) that leads on average to removal of both elements from soil. For grassland and arable soils combined, inputs of zinc, copper and lead still exceed output levels across all of EU-27, even though, for zinc balances in grassland soils are negative (average) due to leaching losses mainly. For cadmium, a net loss of approx. 119 tons per year was calculated (Römken et al., 2018) being the total loss for arable and grassland soils (Table 3.2). The input-output balance differs between cropland and grassland soils. Soil properties as well as kinds of fertilizers applied differ between the two land use systems. Differences between metal inputs and outputs are most pronounced for cadmium and lead.

Figure 3.5. Inputs, outputs and net balance for cadmium, copper, lead and zinc at EU-27 level (in ton year⁻¹). Numbers above the bars of individual fluxes indicate the relative contribution to either inputs or outputs. The number above balance is the absolute value in ton yr⁻¹

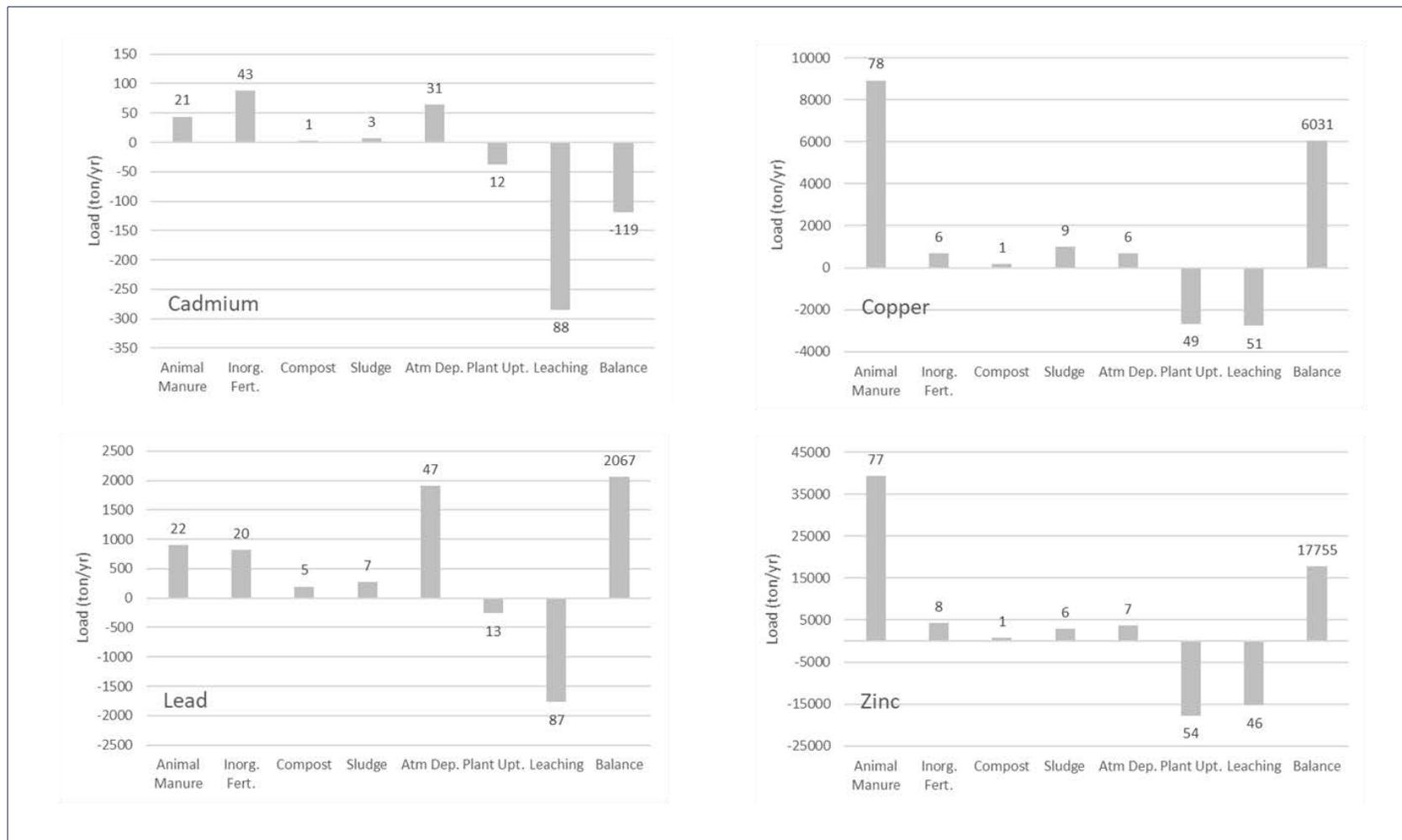


Figure 3.6 Relative contribution of inputs and outputs at EU-27 level for arable land (light grey) and grassland (dark grey).

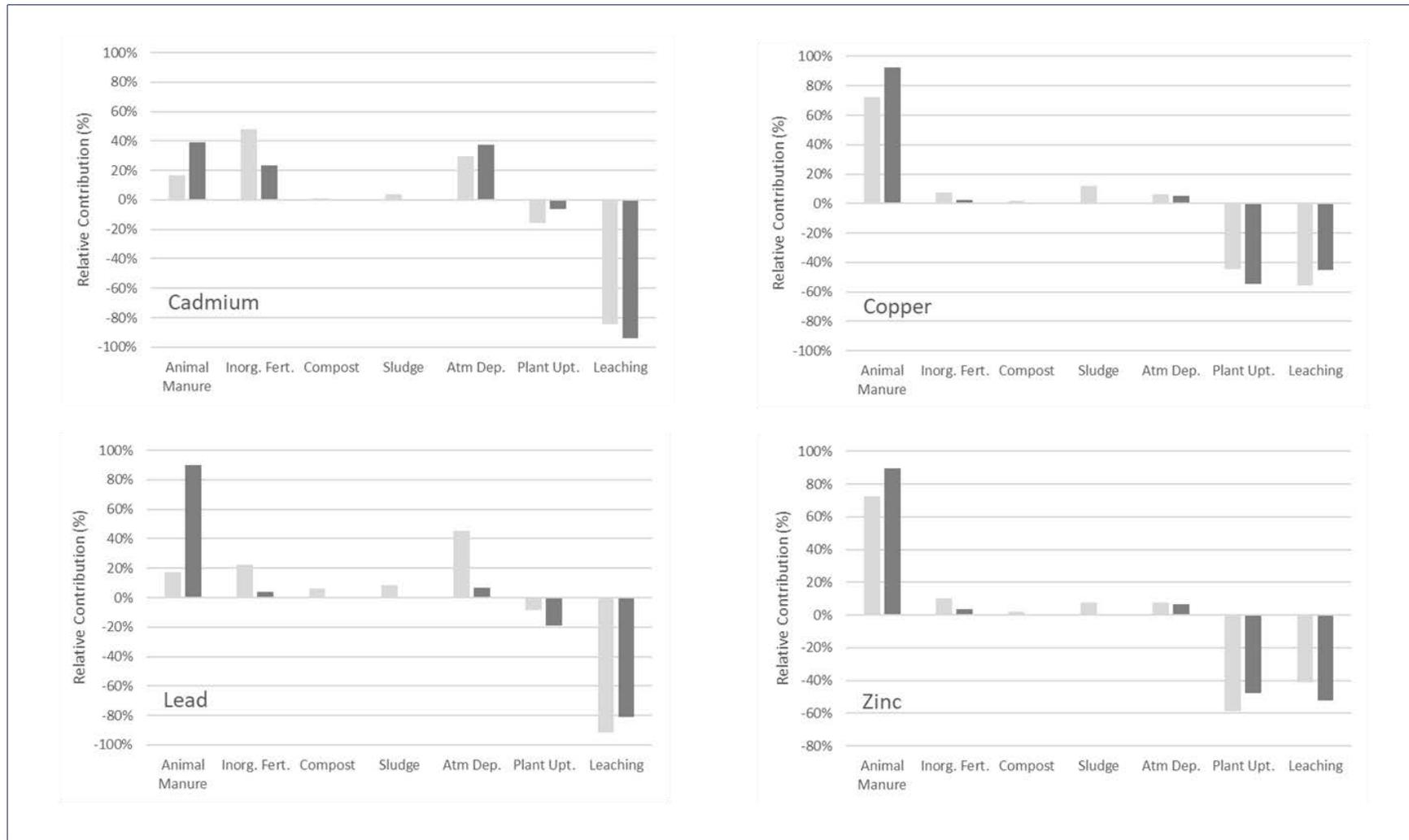


Table 3.2. Overview of metal balances for arable and grassland soils (in ton yr⁻¹)

Land	Inputs				Outputs		Balance	
<i>Arable Land</i>	Animal Manure	Inorganic Fertiliser	Biosolids		Atm. Dep.	Plant uptake	Leaching	
			Compost	Sludge				
cadmium	27	77	2	6	48	30	160	-30
copper	6041	607	169	1000	521	1312	1650	5375
lead	551	704	198	268	1422	96	1073	1973
zinc	26464	3783	695	2851	2746	10543	7442	18554
<i>Grassland</i>								
cadmium	17	10	0	0	16	8	125	-89
copper	2883	79	0	0	168	1352	1121	656
lead	351	119	0	0	483	162	697	94
zinc	12845	527	0	0	941	7246	7865	-798
<i>Total Land</i>								
cadmium	44	88	2	6	64	38	285	-119
copper	8924	686	169	1000	688	2664	2771	6031
lead	902	823	198	268	1905	258	1770	2067
zinc	39309	4310	695	2851	3687	17789	15307	17755

A few key findings summarizing the results presented for each metal include:

Cadmium:

- In arable soils cadmium is mainly introduced via mineral fertilisers (48%), followed by atmospheric deposition (30%).
- In grassland soils there is an equal share from animal manure (39%) and atmospheric deposition (37%) followed by mineral fertilisers (24%)
- Outputs largely occur via leaching (84% – 92%)
- Both in grassland and arable soils Cd has a negative balance, the one for grassland being more pronounced due to lower inputs and, on average, lower pH levels compared to arable soils.

Lead:

- In arable soils inputs from atmospheric deposition (45%) are slightly higher than those from mineral fertilisers (22%)
- In total 15% from all inputs to arable soils stem from biosolids
- In grassland soils inputs via manure (37%) are approx. equal to those from atmospheric deposition (51%).
- Outputs largely occur via leaching (81 – 92%)
- In grassland soils, the balance for lead is close to neutral, in arable soils, the balance is positive due to larger inputs from biosolids and atmospheric deposition.

Copper and zinc

- For both copper and zinc animal manure is the prime source in both arable (approximately 80% of all inputs) and grassland soils (90-92% of all inputs)
- In arable soils there is a small additional input from mineral fertilisers ranging from 7% for copper to 10% for zinc
- Outputs for both copper and zinc are equally distributed between leaching and crop uptake
- At EU-27 level zinc is removed from grassland soils even though variation between countries is large. In countries with intensive animal husbandry, such as the Netherlands and Belgium, balances for

zinc in grassland soils are positive due to large inputs via animal manure. In Italy and Spain balances for zinc in grassland soils are also positive due to low leaching rates.

To provide more insight in regional differences in metal fluxes, inputs and outputs are converted to a flux in gram ha⁻¹ year⁻¹ at member state level (Table 3.3).

Table 3.3 Minimum, median and maximum metal input and output fluxes at member state level in gram ha⁻¹ yr⁻¹ (grassland and arable land combined)

Metal	Inputs				Outputs		Balance	
	Animal Manure	Inorganic Fertiliser	Biosolids Compost	Sludge	Atmospheric Deposition	Plant uptake		Leaching
Cadmium								
Minimum	0.12	0.01	0.00	0.00	0.16	0.05	0.36	-6.28
Median	0.27	0.40	0.00	0.01	0.42	0.18	2.12	-1.15
Maximum	0.86	1.21	0.15	0.41	0.91	1.10	7.77	1.03
Copper								
Minimum	22.6	0.4	0.0	0.0	1.4	4.6	6.3	-9.4
Median	51.4	3.6	0.2	2.5	4.0	12.7	20.8	23.3
Maximum	181.1	7.1	11.3	26.1	9.1	56.3	67.0	137.0
Lead								
Minimum	2.4	0.2	0.0	0.0	5.4	0.1	3.3	-0.3
Median	5.4	4.3	0.3	0.7	12.4	0.7	12.0	12.7
Maximum	17.5	15.0	13.3	9.9	20.7	9.3	34.1	36.2
Zinc								
Minimum	100.0	6.5	0.0	0.0	5.0	27.4	11.8	-233.9
Median	241.2	24.4	0.9	6.8	19.4	78.4	132.7	93.9
Maximum	800.1	48.3	46.8	101.6	50.0	364.0	349.5	491.7

The data on the mass balances as presented in this chapter (Table 3.2) reveal that except for cadmium, the overall balances for copper, lead and zinc are positive which implies that levels in soil will increase. However, the net balance itself has little meaning in terms of risks and impact in either soil or adjacent media like water and crops since it does not relate to the magnitude of the absolute change of the metal content in soil. Hence, current balances do not indicate whether critical thresholds in soil will be exceeded with time. A key aspect to consider when looking at the balance data at EU-27 level is that such EU average balance data do not reveal the large regional variability of calculated metal balances. Differences in both land use, the type and amount of fertilisers or manure applied, soil type and climate have a profound impact on the distribution of metals to and in soils as well as the distribution between leaching rates as is reflected by the ranges in each of the fluxes in Table 3.3

Data in Table 3.3 suggest that even for metals like copper and zinc that, at EU level accumulate in soil, differences between MS are large resulting in some MS where zinc and to a lesser extent copper is removed from the topsoil due to the net negative balance. On the other hand, cadmium, which at EU-27 level shows a slightly negative balance will continue to accumulate in several member states (for more detail on cadmium balances we refer to Römken et al., 2018). A few striking differences between MS include:

- The contribution of biosolids for copper, zinc and lead varies considerably since sludge is not used in some MS like the Netherlands but are commonly used in for example the UK.

- Differences in the use of animal manure result in a striking range in loads for copper and zinc. Especially in countries like the NL and B application rates of animal manure are high and the main reason for the pronounced accumulation of both metals in soil,
- Leaching losses are highly related to the combination of climatic conditions (net water leaching rate) and the dominant soil type. This causes high leaching losses of zinc in the North-Western parts of the EU where the precipitation surplus is high in combination with soils characterized by acidic pH values. On the other hand, the net water loss in the Mediterranean is low to almost zero and soils typically are calcareous, leading to very low predicted concentrations in solution for all metals considered here.

3.2.3 Spatial variation in metal budgets

Differences in land use (e.g. crop type), soil type, climate and farm management related to fertilisation result in a large range of inputs, outputs and resulting balances. The spatial distribution at regional scale of both inputs and outputs (crop removal and leaching) and the balances at EU levels are shown in Figure 3.7 through Figure 3.10 for cadmium, copper, lead and zinc.

Both at country level and regionally, the modelling results show a rather pronounced range in accumulation or depletion rates for cadmium. Even in cases of accumulation, however, the absolute levels of accumulation rates are relatively low. A net accumulation rate of 1 g ha^{-1} for example corresponds to an approximate annual increase in the soil cadmium content of less than 0.001 mg kg^{-1} . Nevertheless, a positive balance still implies that cadmium levels in soil, on the long term, will increase which is an unwanted development in view of potential effects on food quality and leaching to ground- and surface waters. The fact that current cadmium inputs result in low accumulation rates, or even a net decrease of soil cadmium concentrations reflects the substantial decrease in cadmium inputs during the last decades. These are mostly due to lower application rates of inorganic P fertilizers and policy induced reductions of atmospheric inputs. Such lower inputs to soil in combination with ongoing leaching of cadmium from the topsoil leads to the calculated net decrease of cadmium in soil at EU-27 level.

Copper inputs are highest in areas with intensive animal husbandry including the Netherlands, Belgium and parts of France and Denmark. Calculated ranges of copper accumulation are directly related to regional differences in copper inputs, which are highest in areas with animal husbandry as is the case in among others the Netherlands and parts of France (Brittany). Differences in most other fluxes are less pronounced such as losses via leaching, which are distributed more evenly across the EU. To some extent, higher initial copper levels in soil and/or high precipitation surpluses play a role in the variability of calculated leaching rates as well. Copper accumulation prevails in other areas than those with intensive animal husbandry as well despite moderate copper inputs since both crop uptake and leaching rates are low. For lead, the regional variation of inputs and accumulation is lower than that of the other metals. Deviations are observed in areas with intensive animal husbandry (lead in manure), notably the Netherlands and Belgium, and may also be related to the application of compost.

Figure 3.7 Spatial variation in current (2010) cadmium input (left), output (middle) and accumulation (right) in EU-27.

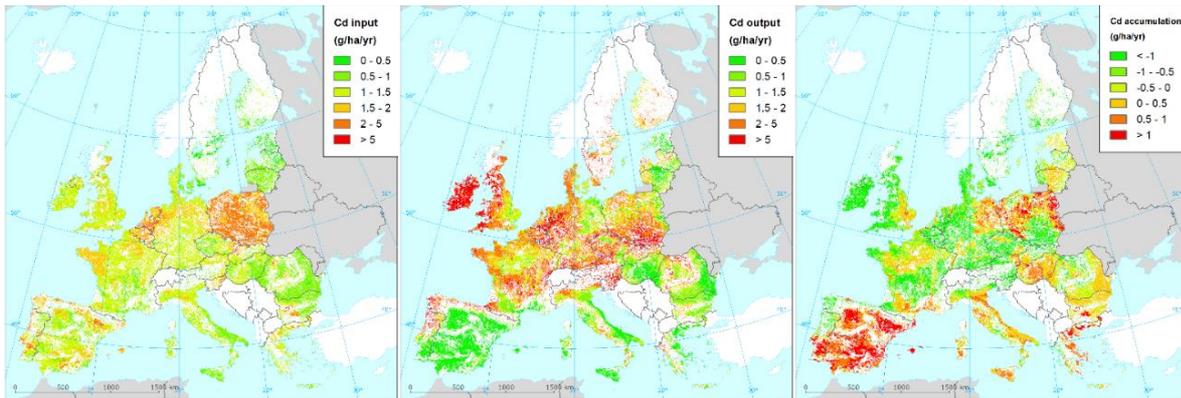


Figure 3.8 Spatial variation in current (2010) copper input (left), output (middle) and accumulation (right) in EU-27.

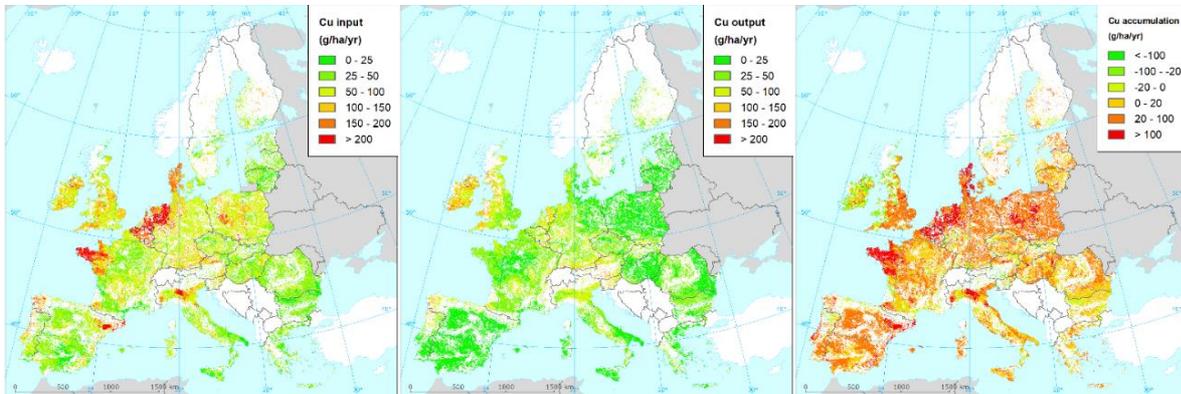


Figure 3.9 Spatial variation in current (2010) lead input (left), output (middle) and accumulation (right) in EU-27.

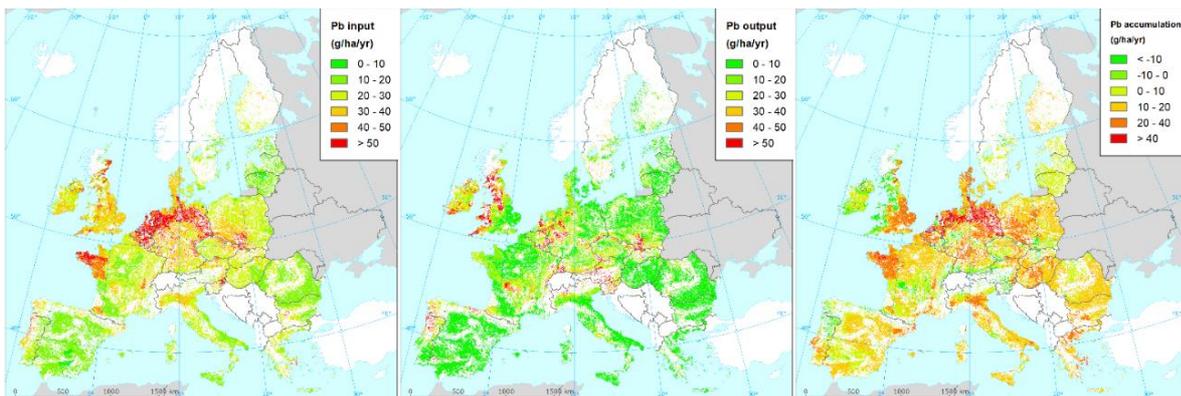
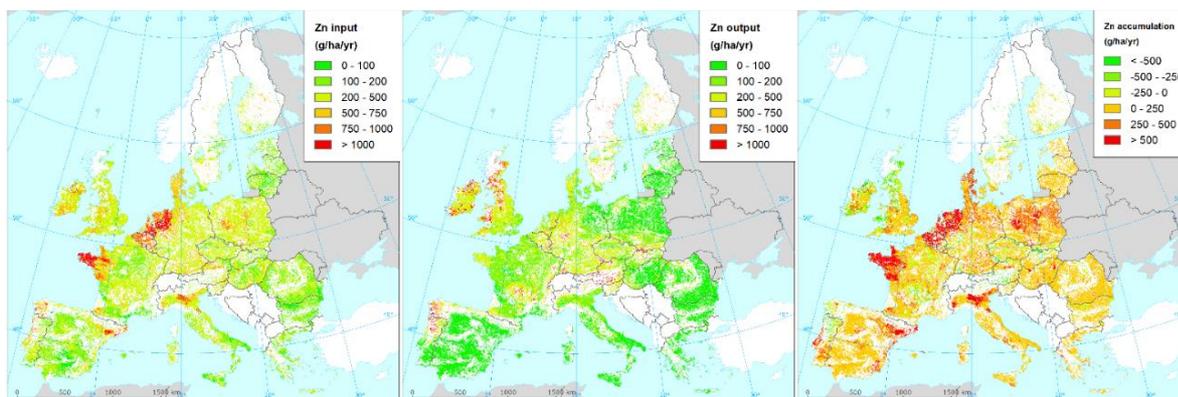


Figure 3.10 Spatial variation in current (2010) zinc input (left), output (middle) and accumulation (right) in EU-27.



Maps for zinc show an almost similar pattern as for copper; which is related to the similarity in inputs, occurring mostly via manure. For zinc, however, leaching losses, are quantitatively more important due to the higher relative mobility of zinc compared to copper. High leaching losses in combination with high uptake fluxes even can result in a negative balance (i.e. leaching + crop uptake > inputs) for zinc, both at country level but also at regional level, in areas with relatively zinc inputs. Examples of such regions include central parts of Germany, Portugal, Belgium and Luxemburg, where a net accumulation is observed at country level while depletion occurs regionally. This illustrates the added value of a regional balance approach in addition to a country balance approach, as zinc depletion is unwanted in view of sustainable crop quality and production.

Differences in leaching are an important cause for the variation in metal outputs as illustrated in Figure 3.11a (cadmium and copper) and 3.11b (lead and zinc). High cadmium leaching losses correspond with areas where soil pH generally is low (pH < 6) in combination with a low to moderate clay content (sandy soils) and high initial soil cadmium levels. Examples of such areas are former industrial areas in Belgium and the Netherlands (Kempen), and the border area between the Czech Republic and Poland. On the other hand, low leaching rates are observed in areas dominated by calcareous soils and/or low precipitation surpluses, like Spain and other Mediterranean areas.

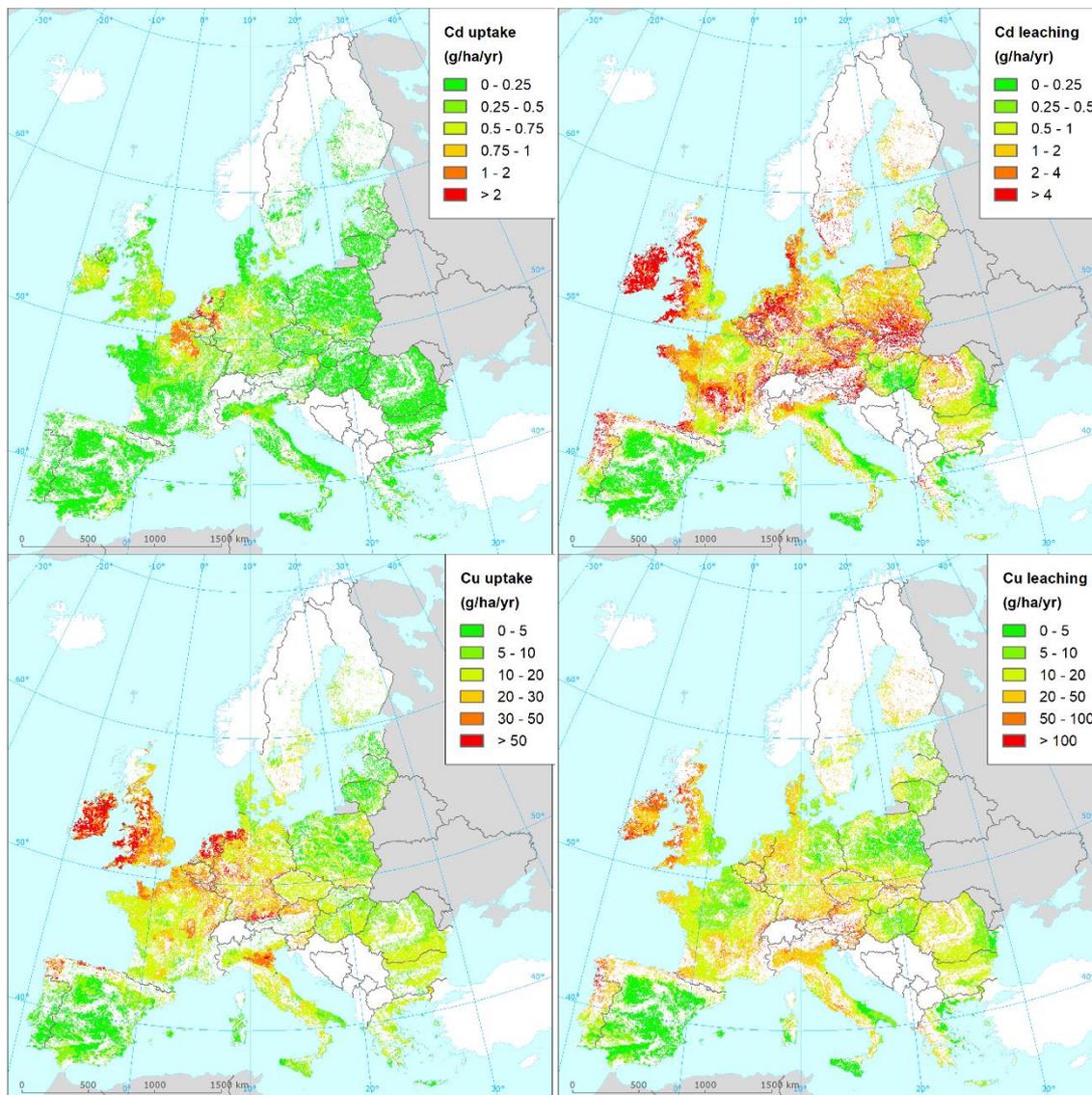
Differences in leaching are mostly due to differences in pH, and negative accumulation rates (release) for lead are mostly confined to areas with high rainfall and low pH. In many areas, the size of the leaching component is comparable to that of inputs, implying a relatively low accumulation in those areas.

3.3 Current and critical metal inputs in view of environmental impact

3.3.1 General

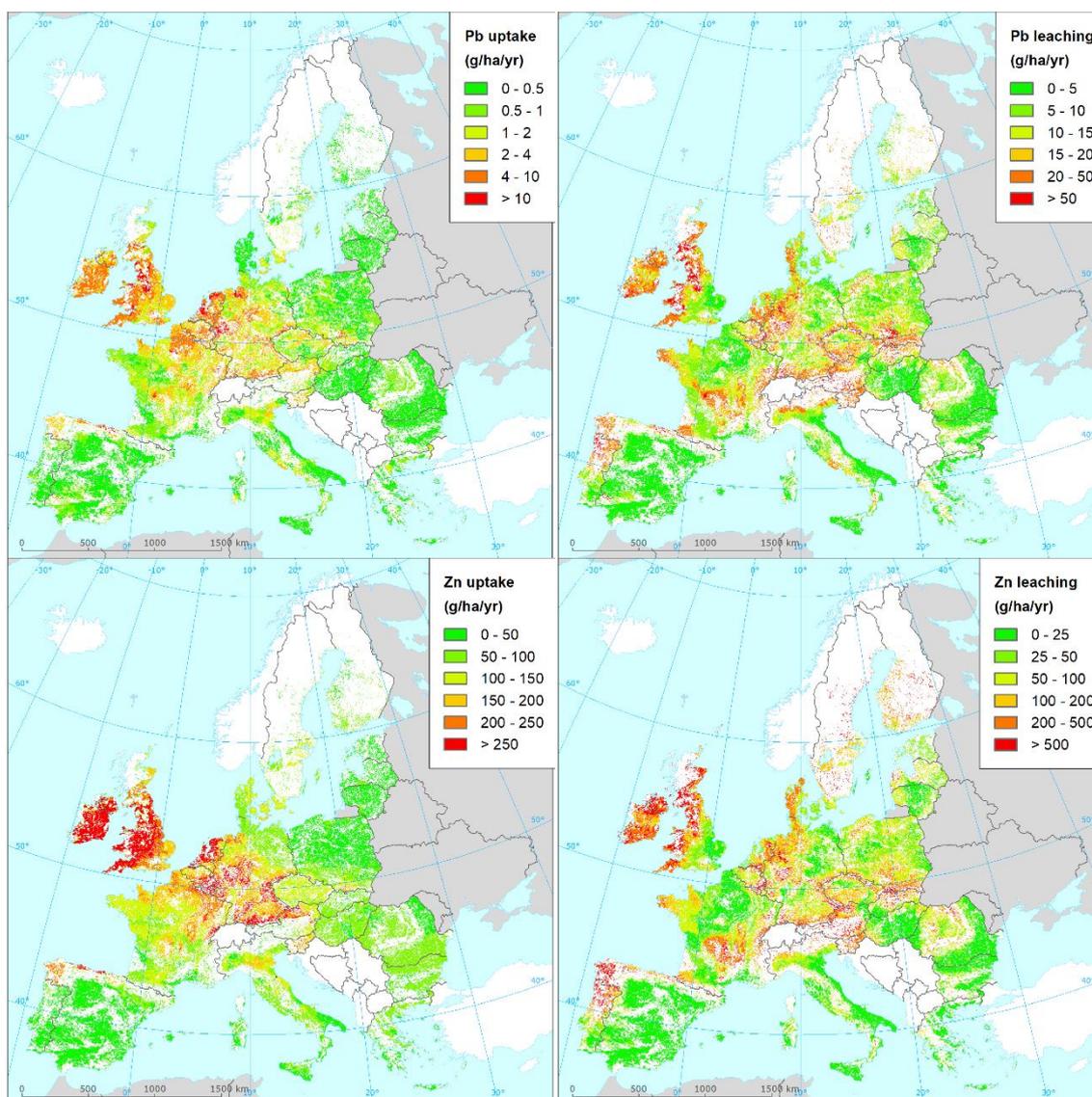
Concerns about input of metals to agricultural ecosystems are related to the potential impact on soil ecology, food quality and water quality. Here we focus on ecology and food safety as explained earlier. Ecological impacts include effects on micro-organisms, fungi and soil meso-fauna including nematodes and earthworms. Effects of metals in soil on these organisms may result in a reduction of species diversity, abundance and biomass and affect microbe-mediated processes. In this chapter we present the spatially explicit critical soil concentrations (thresholds) and compare these with current metal concentrations in soil for cadmium, copper, lead and zinc (par. 3.3.2). In addition, critical inputs for cadmium, copper, lead and zinc to agricultural soils in EU-27 in view of ecotoxicological impacts on soil organisms are calculated and compared to current inputs to assess where, at steady state, such inputs exceed critical inputs.

Figure 3.11a. Spatial variation in current (2010) cadmium and copper uptake (left) and leaching (right) in EU-27.



Risks for food safety are limited to an assessment for cadmium and include the same approach as for soil ecology. First critical levels in soil are calculated using a soil to crop transfer model and using the current food quality criteria in wheat, being a sensitive crop. These predicted critical levels for cadmium in soil are subsequently compared to current cadmium concentrations in soil (par. 3.3.3). Secondly a critical load is calculated at steady state where cadmium levels in soil are assumed to be equal to the critical soil concentration (threshold) in view of food safety for cadmium. This critical load is then compared with the current load.

Figure 3.11b. Spatial variation in current (2010) lead and zinc uptake (left) and leaching (right) in EU-27.



3.3.2 Critical metal concentrations and inputs and exceedances in view of soil biodiversity

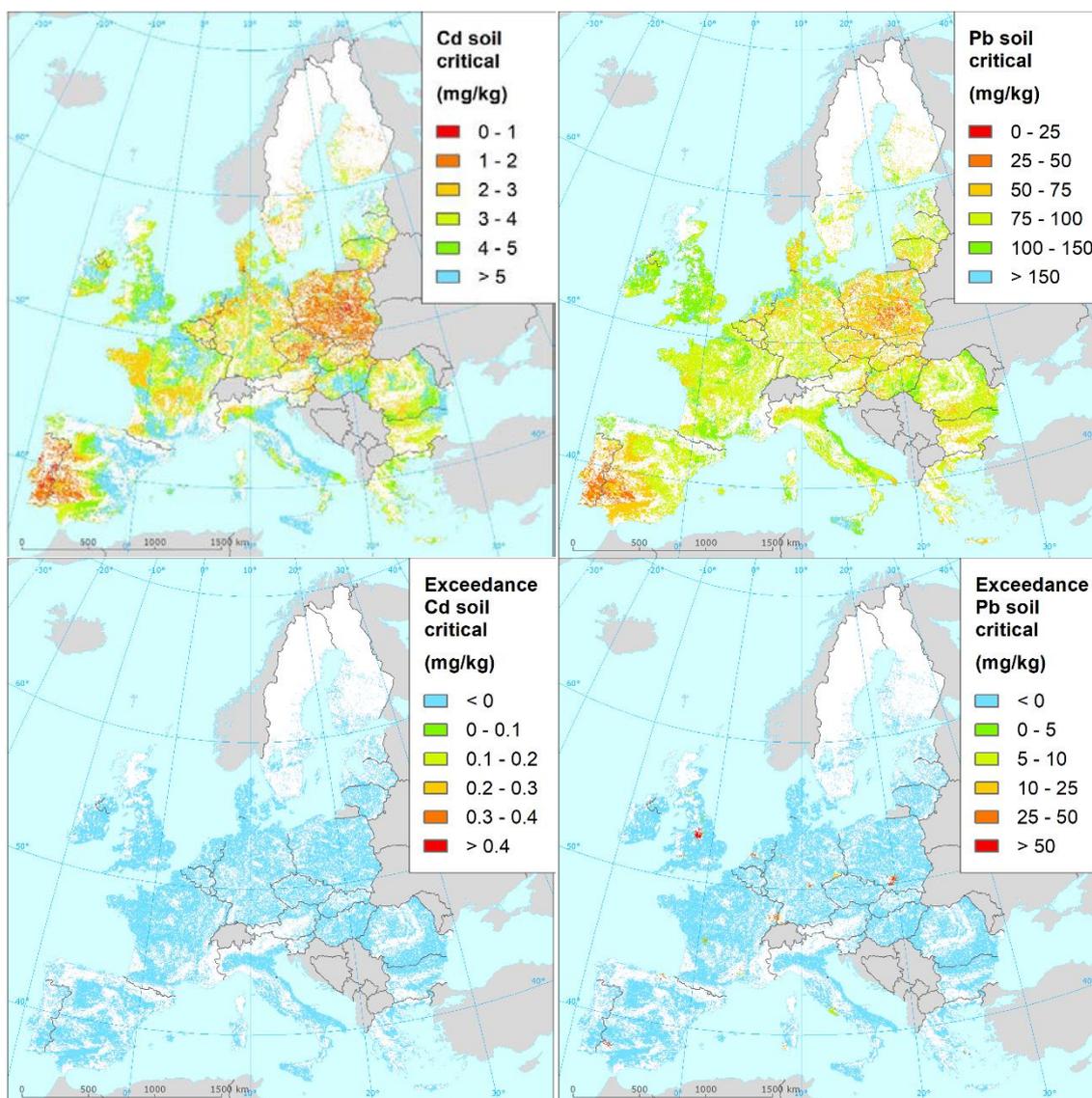
Critical metal concentrations and their exceedances

Critical reactive soil metal concentrations in view of ecotoxicological effects on soil organisms were based on No Observed Effect Concentrations (NOECs) for the concentration of added metal in laboratory experiments. These added metal concentrations were assumed to be in a reactive form and were related to soil organic matter content (SOM) and pH_{H_2O} using a procedure described in Lofts et al. (2004) and De Vries et al. (2007a).

As a threshold for ecotoxicological effects, critical reactive metal concentrations as a function of soil organic matter content (SOM) and pH were used, as derived by Lofts et al. (2004) and De Vries et al. (2007a). 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. In those cases where the actual soil content exceeds the calculated critical soil content the functioning of the soil ecosystem is considered at risk.

Ranges in current (2010) and calculated critical total metal concentrations in agricultural soils in EU-27 are given in Table 3.4. The spatial distribution of critical soil concentrations as well as the difference between current and critical concentration at EU-27 level are shown in Figure 3.12 for cadmium and lead and in Figure 3.13 for copper and zinc.

Figure 3.12. Spatial variation in critical concentrations (thresholds) in view of impacts on soil biodiversity for cadmium (top left) and lead (top right) and their exceedance by current (year 2010) soil concentrations (bottom) in EU-27) for cadmium (left) and lead (right).

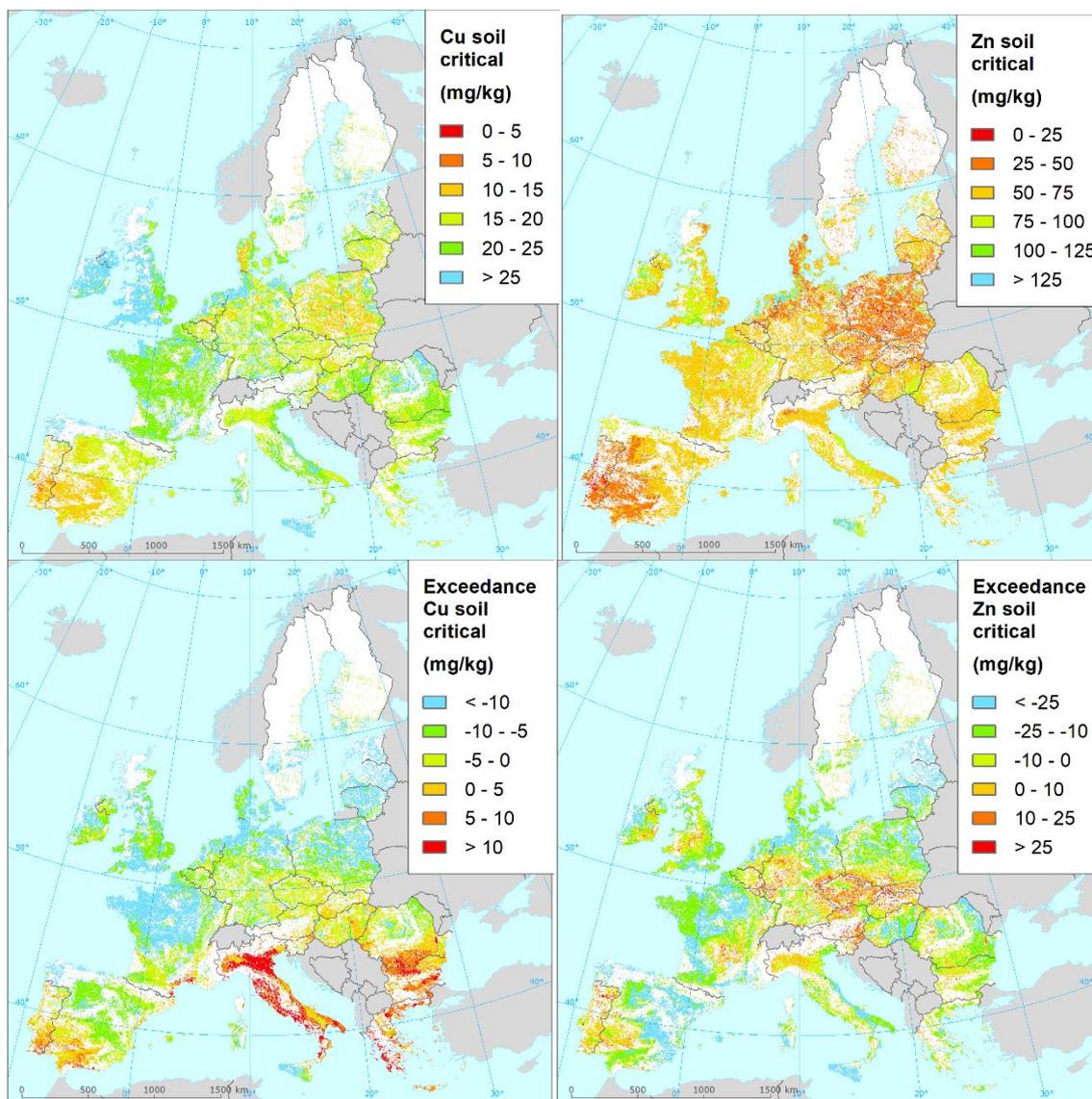


Results in Table 3.4 indicate that median current total soil cadmium, copper, lead and zinc concentrations at EU-27 level, do not exceed the calculated critical metal concentration for each of the metals considered.

Table 3.4. Current and critical median (P50) metal concentrations in agricultural soils in EU-27. Between brackets the P05-P95 values

Metal	Current concentration (mg kg ⁻¹)	Critical concentration (mg kg ⁻¹)
cadmium	0.24 (0.1 - 0.48)	3.7 (1.5 - 8.6)
lead	19 (8 - 44)	86 (53 - 141)
copper	15 (5.3 - 30)	21 (13 - 36)
zinc	45 (21 - 68)	60 (32 - 98)

Figure 3.13. Spatial variation in critical concentrations (thresholds) in view of impacts on soil biodiversity for copper (top left) and zinc (top right) and their exceedances by current (year 2010) soil concentrations (bottom) for copper (left) and zinc (right) in EU-27 .



For copper and zinc, the ranges of the P5-P95 predicted intervals (Table 3.4) of current levels in soil partly overlap with the P5 – P95 interval of the critical predicted concentration ranges. This suggests that there can be areas where the actual concentration in soil exceeds the critical concentration. After correction for surface area for all NCU’s considered, current copper concentrations exceed the calculated critical concentrations in 23% of the area included in the model at present (Figure 3.13 bottom left), for zinc this was 18% (Figure 3.13 bottom right). For cadmium and lead there is no such overlap between the P5-P95 current and critical concentrations in soil which indicates that, within the P5-P95 range, current soil cadmium and lead concentrations do not exceed the critical concentrations. Surface corrected data indeed reveal that for cadmium, current levels in soil never exceeded the critical concentrations, but for lead this still occurs in 1% of the soils (Figure 3.12).

Critical metal inputs and their exceedances

The critical metal input, being the input that, at steady state maintains the ecological critical soil concentrations (ECSC according to the method developed by Lofts et al. (2004) and applied by De Vries et al. (2007a)), was calculated as the sum of the uptake rate and a leaching rate that would occur in equilibrium with the ECSC. This is called the steady state critical input since if net removal rates (uptake

plus leaching) equal the sum of all inputs, no change in the ECSC will occur. Crop uptake at the ECSC was derived via multiplication of crop yield and the calculated crop metal concentration. This crop metal concentration was calculated from the ECSC using a fixed bioconcentration factor (BCF) per crop type, BCF being the ratio between metal concentration in crops and soil. Values for the BCF were based on literature data and sometimes a fixed BCF was attributed to a range of (similar) crops if data are lacking for specific crops.

Leaching rates for the four metals from the topsoil (0-30 cm) were derived by multiplication of the net water flux (precipitation minus evapotranspiration) through the soil and the calculated dissolved metal concentrations. The dissolved metal concentration was derived from the ECSC using non-linear Freundlich-type transfer functions accounting for differences in organic matter content, clay content, pH and dissolved organic carbon (DOC) concentration, according to Römken et al. (2004).

Ranges in current (2010) and calculated critical metal inputs in agricultural soils in EU-27 are given in Table 3.5. Results show that at EU-27 level, median current metal inputs exceed critical metal inputs for zinc and copper whereas current inputs for lead are close to the critical input. For cadmium on the other hand, the predicted median critical input (P50 = 17 g Cd ha⁻¹ yr⁻¹) is far larger than the average current input (1.2 g Cd ha⁻¹ yr⁻¹) (Table 3.5).

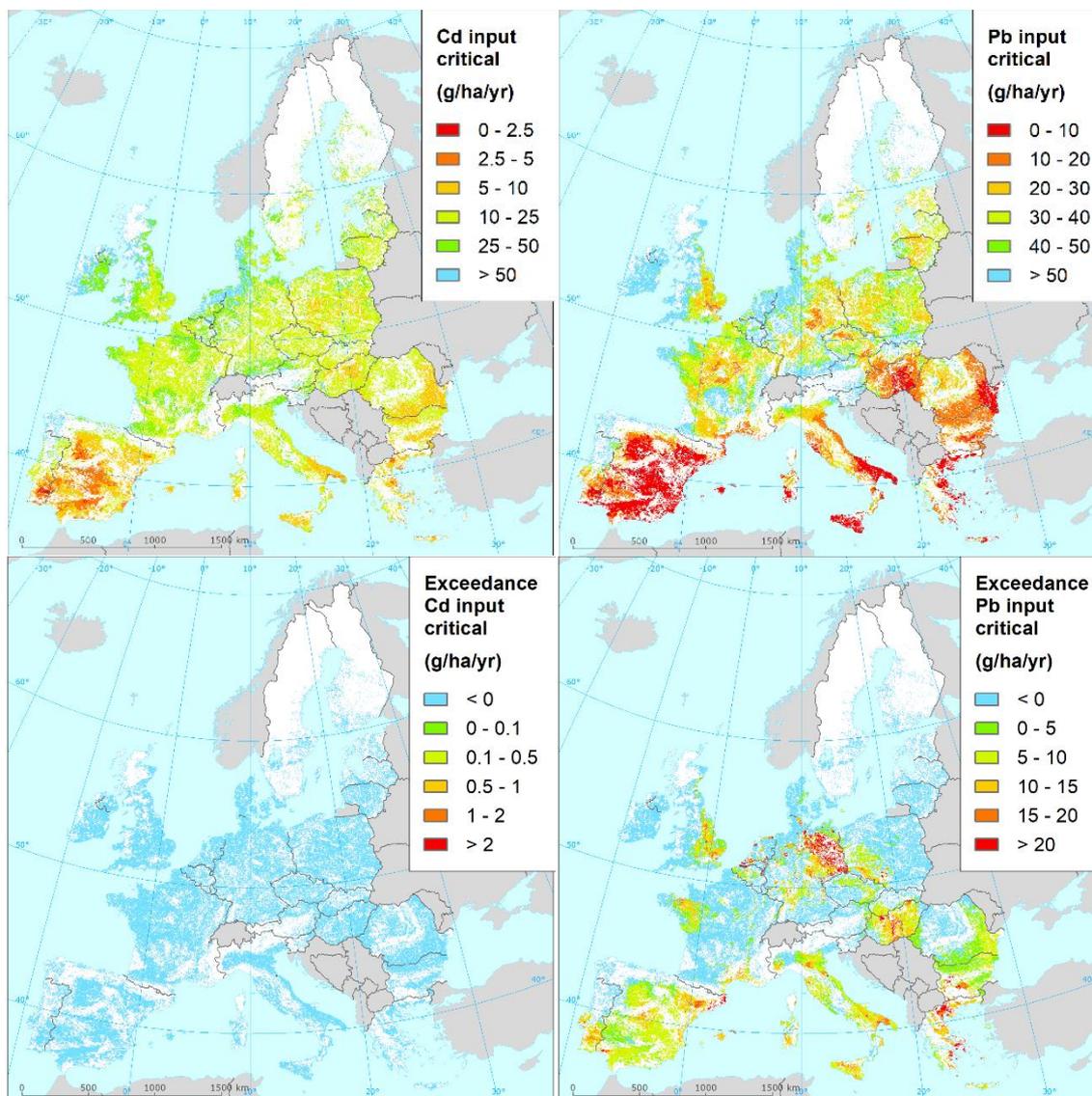
Table 3.5. Current (year 2010) and critical median (P50) metal inputs in agricultural soils in EU-27. Between brackets the P05-P95 values

Metal	Current input (g metal ha ⁻¹ yr ⁻¹)	Critical input (g metal ha ⁻¹ yr ⁻¹)
Cadmium	1.2 (0.5 - 2.8)	17 (4 - 62)
Lead	24 (11 - 60)	27 (4.8 - 106)
Copper	57 (15 - 198)	37 (8 - 141)
Zinc	253 (71 - 887)	199 (23 - 935)

Maps of critical metal inputs and the difference between current and critical metal inputs are given in Figure 3.14 for level for cadmium and lead and in Figure 3.15 for copper and zinc. Similar to results for the exceedance of soil critical concentrations in view of soil biodiversity (Figures 3.12 and 3.13) the area where inputs for cadmium exceed of the critical input is small (0.4 % of the total area, Figure 3.14 bottom left). For lead (Figure 3.14) a somewhat different result was obtained based on the critical input assessment. Unlike for the critical lead concentrations that at present hardly exceeded critical soil concentrations (Figure 3.12 bottom right), model predictions indicate that in 48% of the total area current lead inputs exceed critical inputs (Fig 3.14 bottom right). Compared to cadmium this reflects the rather low mobility of lead in soil causing a slow but steady accumulation even under current inputs. For cadmium, losses via leaching and plant uptake are far larger which results in a relatively much higher critical input to maintain the critical soil concentration.

The area where critical inputs of copper and zinc are exceeded (Fig 3.15) is, however, much larger than the area in which critical concentrations of copper and zinc are exceeded (Fig 3.13). For copper, critical inputs at equilibrium exceed current inputs in 70% of the total area compared to 23% (data Figure 3.13) where current concentrations exceed critical concentrations. For zinc the current input exceed the critical input in 60% of the area (Figure 3.15) compared to 18% where current concentrations exceed critical concentrations (data Figure 3.13). This difference between the size of the area where current concentrations exceed critical soil concentrations versus that where, at equilibrium, current inputs exceed critical inputs suggests that there are relatively large areas in the EU where, if current inputs of copper and zinc are to be maintained, soil concentrations will reach levels beyond the critical concentration. The area where adverse impacts on soil biodiversity related to copper and zinc concentrations in soil will occur is thus expected to substantially increase compared to the area depicted in Figure 3.15.

Figure 3.14. Spatial variation in critical input in view of impacts on soil biodiversity (top) and exceedances of critical inputs by current (year 2010) inputs (bottom) for cadmium (left) and lead (right) in EU 27.

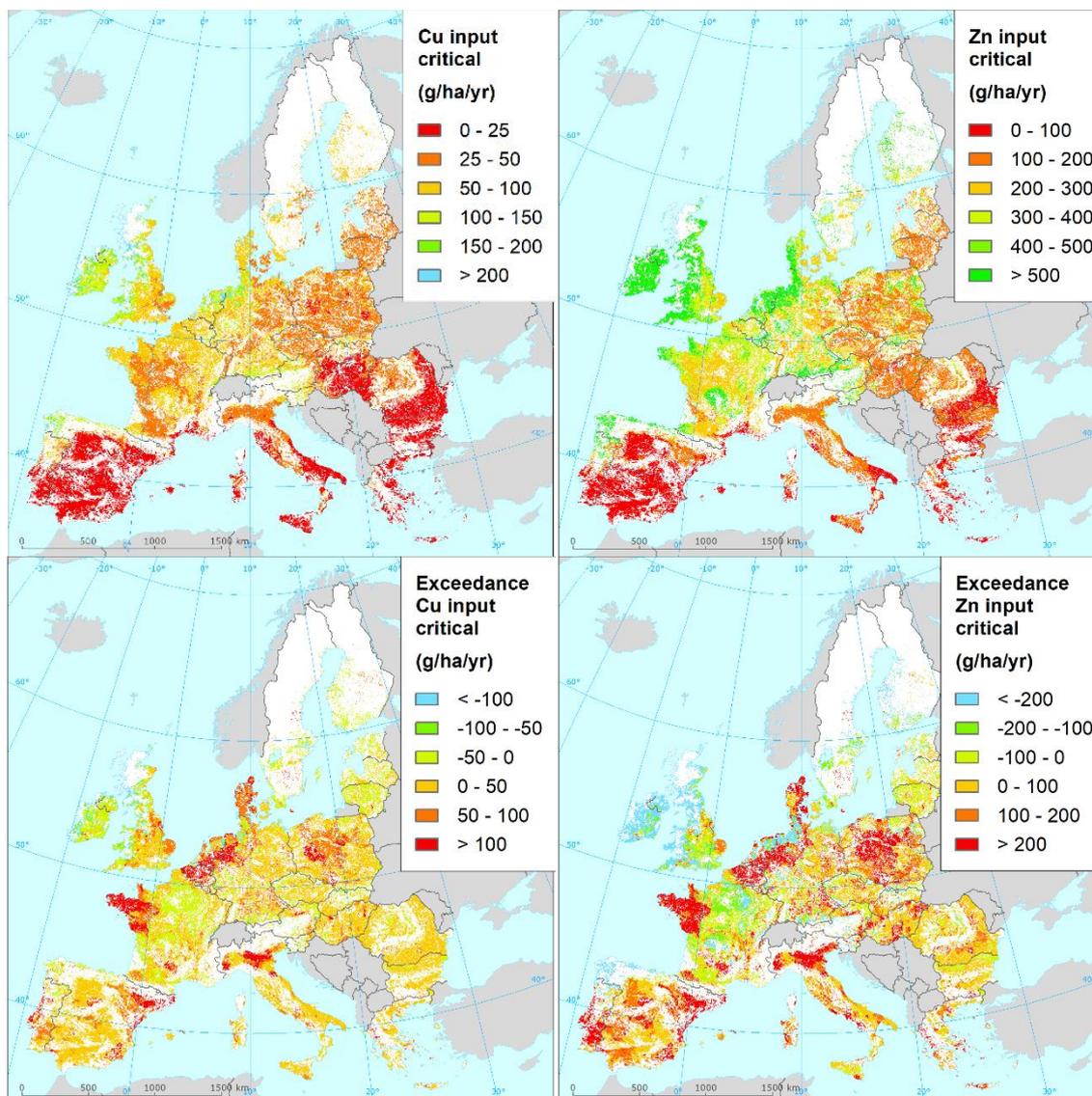


3.3.3 Critical cadmium concentrations and inputs and their exceedances in view of food quality

Prediction of the critical cadmium soil concentrations in soil

Critical soil cadmium concentrations (thresholds) in view of food quality were calculated using the food quality criterion for wheat as guideline. This was done for several reasons. First, wheat is one of the staple food crops grown in most countries across the EU and rather sensitive in view of uptake of cadmium. Second, the soil to plant relationship for wheat is of reasonable to good quality which allows to calculate rather reliable cadmium concentrations in wheat for a range of soils present at EU level. Uptake of cadmium, as well as other metals, depends on soil properties like pH and organic carbon. For most crops data to derive such relationships are, however, scarce and soil to plant relationships for many crops are of poor quality leading to rather large uncertainties in the predicted levels of cadmium in crops. The standard approach to predict metal concentrations in food crops using the transfer factor (or bioconcentration factor, BCF) defined simply as the ratio between the metal concentration in crops and that in soil.

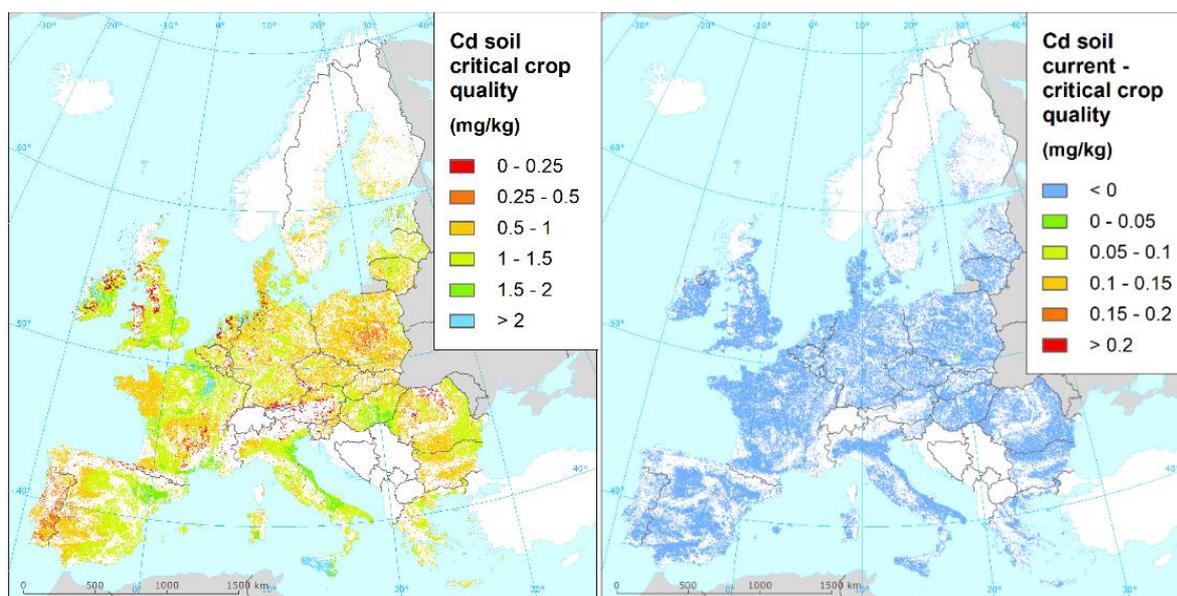
Figure 3.15. Spatial variation in critical inputs in view of impacts on soil biodiversity (top) and exceedances of critical inputs by current (year 2010) inputs (bottom) for copper (left) and zinc (right) in EU 27.



This approach was used when calculating metal budgets, but it yields poor predictions of the range of cadmium in crops and also is not able to account for differences in soil pH and organic matter or clay (see Römken et al., 2007). The current EU food quality standards for wheat ($0.2 \text{ mg Cd kg}^{-1}$ fresh weight¹) was converted to dry weight using an average dry matter content in grain of 88%. Current soil properties (pH, organic matter and clay) were used to calculate the corresponding critical cadmium concentration in soil at which the critical level of cadmium in wheat would be exceeded. Maps of critical soil cadmium concentrations thus calculated as well as the differences between the critical and the current soil cadmium concentrations are given in Figure 3.16. Areas in red mark those areas where the actual soil cadmium concentration is below the critical concentration and crop cadmium concentrations thus are predicted to exceed the food quality standard. This area however is very small and corresponds to 0.02% of the agricultural area within EU27.

¹ The latest revision of EU 1881/2006 by commission regulation (EU) 2021/1323 has resulted in a decrease of the maximum allowed concentration of Cd and Pb in a number food products. Only for wheat germs the standard remains 0.2 mg kg^{-1} . For durum wheat the thresholds was reduced to 0.18 mg kg^{-1} . For grains in general the maximum limit was reduced to 0.1 mg kg^{-1} . For barley and rye the limit is even further reduced to 0.05 mg kg^{-1} . These changes have not been taken into account in this model approach

Figure 3.16. Spatial variation in critical soil cadmium concentrations in view of food safety for wheat (left) and the differences between current (year 2010) and critical soil cadmium concentrations (right).



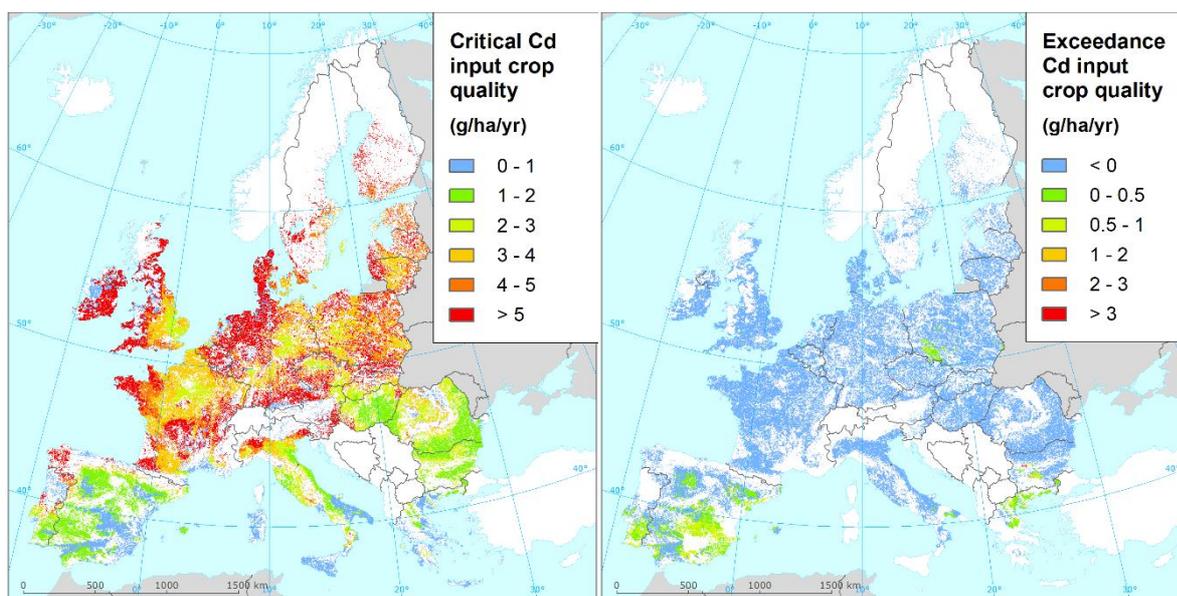
For lead, for which food quality criteria also exist it was decided not to apply this approach due to the notoriously poor predictions of lead levels in crops using a BCF approach or a soil to crop transfer model. And lastly, for zinc and copper there are simply no food quality standards for products meant for human consumption. Uptake of these elements is highly regulated by most crops since both elements are essential elements. At levels at which both copper and zinc would become toxic for humans, most crops would suffer already from extreme toxic effects.

Critical cadmium inputs and their exceedances

Similar to the calculation of the critical load in view of soil biodiversity, a critical input at equilibrium for cadmium in wheat was calculated. Model calculations as presented in Figure 3.17 reveal that current soil cadmium inputs across the EU will not cause wide-spread issues with cadmium in wheat in the long-term in most countries outside the Mediterranean area. Exceptions to this is a small area in southern Poland where cadmium inputs are relatively high compared to the EU average. In substantial parts of the Mediterranean area cadmium inputs can exceed the critical input. Notably in Spain and Greece as well as smaller areas in Italy, the combination of very low leaching losses of cadmium result in ongoing accumulation (Figure 3.17) and, if input levels remain at current levels, exceedance of the soil critical concentration.

In most areas across the EU, however, leaching losses of cadmium from soil are substantial and current inputs will not lead to excess uptake of cadmium by wheat at equilibrium. It should be noted however that models used to predict leaching of cadmium from soil still are a matter of debate and predicted leaching losses can vary substantially based on the type of model used. This affects the balance especially when considering changes in soil cadmium in the long term (Römken et al., 2018; Arcadis, 2021).

Figure 3.17. Spatial variation in current (year 2010) input rates of cadmium (left) and the difference between current inputs and critical inputs (right) in view of food safety focusing on wheat.



The variation in current cadmium inputs is mainly related to variation in the quantity and quality of mineral P fertilizer applied by farmers and, to a lesser extent due to variation in atmospheric deposition. The critical cadmium inputs are determined by the precipitation surplus, the crop considered (in our calculation wheat, being a sensitive crop), its food quality criterion and the soil properties affecting cadmium crop uptake, i.e. pH and to a lesser extent, clay content and organic matter contents.

At EU level, current cadmium loads are quite evenly distributed over atmospheric deposition ($0.41 \text{ g Cd ha}^{-1} \text{ yr}^{-1}$), inputs of mineral fertilisers ($0.39 \text{ g Cd ha}^{-1} \text{ yr}^{-1}$) and the sum of manure and biosolids ($0.32 \text{ g Cd ha}^{-1} \text{ yr}^{-1}$), but this is not necessarily the case at country level. In Bulgaria, for example, the contributions from atmospheric deposition, fertilizers, manure and biosolids are 0.86 , 0.36 and $0.09 \text{ g Cd ha}^{-1} \text{ yr}^{-1}$, respectively. At country level and EU27 level, critical cadmium inputs are always much higher than current inputs when using a food quality criterion. However, in approximately 4% of the agricultural area within EU27, crops are grown on soils where current cadmium inputs would in the long-term lead to soil cadmium concentrations that do not meet crop-specific food quality criteria (right panel in Fig 3.17).

Results in Figure 3.16 (differences in critical and current soil cadmium concentrations) and those in Figure 3.17 (difference between critical and current inputs) show minor differences. Based on the current soil cadmium concentrations in soil (Figure 3.16) there are, at the scale level presented here, no areas where the food quality criteria (for wheat) are not met. This suggests that under present conditions, soil cadmium concentrations in combination with soil pH and organic matter content pose no threat to the quality of wheat (grain) insofar related to the cadmium content in grain. Results in Figure 3.17 (right hand side) are largely similar to those in Figure 3.16 (right hand side) with the exception of areas in the Mediterranean area and southern Poland where, at equilibrium, which is reached at timescales ranging from decades to hundreds of years from now, current cadmium inputs can lead to an exceedance of the food quality standards for wheat. These areas in Figure 3.17 correspond to areas with relatively high cadmium inputs but are also characterized by conditions that favour accumulation of metals in soil. Notably in Mediterranean areas, soils are largely neutral to alkaline and leaching losses are small, both conditions leading to accumulation in soil more so than in areas in north-western part of the EU with similar cadmium inputs (e.g. Belgium or the Netherlands). Here (in Belgium or the Netherlands) leaching losses are such that the cadmium balance is largely negative which, at equilibrium, will lead to lower cadmium concentrations in soil compared to current levels.

4 Evaluation and key messages

Model approaches and their potential in risk assessment and policy making

In this study, the spatial variation in the current (year 2010) fluxes (inputs, uptake, accumulation and losses to air and water of nitrogen (N), phosphorus (P) and metals cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn) were determined in EU agriculture by using a land balance approach. In addition, the critical losses and critical input levels were derived (i.e. the input that is possible without harm to the environment) by using thresholds/critical limits for N, P and metal concentrations in food (Cd), air (ammonia, linked to critical N deposition on natural ecosystems), (ii) a critical nitrate (NO_3) concentration of $50 \text{ mg NO}_3 \text{ l}^{-1}$ in leachate to groundwater and (iii) a critical N concentration of 2.5 mg N l^{-1} in runoff to surface water. For metals, available critical limits for food, water and soil organisms, from different existing regulations and studies, were converted to soil property dependent critical metal concentrations (soil-based quality standards), which were then used to calculate critical metal inputs.

Reducing nitrogen inputs with critical inputs, to minimize nutrient losses from soils to acceptable levels, will reduce crop yields unless the efficiency of nitrogen application is increased. In order to advice policy and management, it is important to know the required increase in N use efficiencies (NUE) to optimize nutrient provision and availability to crops, to combine target crop yields with acceptable N losses to air and water. For that the spatial variation in required NUEs were determined and compared with the current NUEs. Any gap between actual NUE and an attainable (or target) NUE indicates the potential for meaningful management response. If the required NUE is still higher, the production level needs to be reduced when full protection of the environment is required

The results of the N, P and metal balances as well as the examples of risk assessment in view of impacts on the plant species diversity of terrestrial ecosystems (N), eutrophication of aquatic ecosystems (N and P), soil biodiversity (metals) and food safety (cd) illustrate the potential of spatially explicit models. Risks of N, P and metals in soil and added to the soil strongly depend on several spatially variable factors including soil properties, farm management and climatic conditions. Only by combining such data in a model environment allow to evaluate regional differences in risks. This also enables the user to evaluate measures to reduce effects which also need to be adjusted according to the area of interest.

Current and critical nitrogen inputs in view of air and water quality impacts

Nitrogen is an essential nutrient in agricultural production. Nitrogen is highly mobile and thus part of the N applied is not taken up by the crop and the N surplus is mainly lost in the form of ammonia emissions, N runoff and nitrate leaching. Ammonia emissions and subsequent re-deposition on natural ecosystems causes biodiversity loss and acidification, whereas N runoff and leaching negatively affect water quality and aquatic ecosystems. Reducing N inputs, however, implies a reduction in crop yield at current nutrient management practices. The difference in current N inputs, required N inputs to increase crop yields to target levels and critical N inputs thus illustrates the tension between food production and environmental protection.

Key findings reported on current, required and critical nitrogen inputs and ways to reconcile the tension between food production and environmental protection are:

- Average nitrogen inputs to European agricultural soils were $145 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in 2010. Most N was applied by fertilizer and fixation ($78 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), followed by manure and biosolids ($56 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). Average inputs to grasslands were higher than inputs to arable land, mainly due to higher manure inputs on grasslands. Nitrogen inputs vary substantially both across and within countries, reflecting differences in the intensity of agricultural production across Europe.
- Overall, the N surplus is about 40% of the N input in arable land (NUE near 0.6) and about 30% in grassland (NUE near 0.7) with NUE defined as the ratio between crop N removal and N input. Nitrogen

surpluses occur in all countries, being highest in areas with intensive animal husbandry, such as the Netherlands, Belgium, Brittany in France and the Po region in Italy.

- Current N losses from agriculture exceed critical limits in view of these impacts on a large share of agricultural land. Ammonia emissions are especially high in areas with intensive livestock and high manure applications and highest exceedances occur in these regions, but also in regions with sensitive ecosystems (low critical nitrogen loads). The spatial variation in Exceedances of critical nitrogen losses by leaching to groundwater and runoff to surface water is not only also largely affected by nitrogen inputs but also by soil type and climate.
- European average critical N inputs not exceeding critical ammonia emissions are $100 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ and for surface water quality $83 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ being, respectively, 31% and 43% lower than the current N inputs. Relative reductions needed to respect thresholds are higher for N losses than for N inputs. In order to respect thresholds for N runoff to surface water, for example, N runoff needs to decrease by 50% while N inputs need to decrease by 43%
- Increasing crop yields to target levels (80% of water-limited yield potentials) requires an estimated 27% increase in N inputs (from 145 to $185 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) at the current NUE.
- In regions where current N inputs, or N inputs required to obtain target yields, exceed critical inputs (inputs that respect environmental thresholds), crop production and environmental protection can only be reconciled by increasing the NUE of agricultural production (i.e., unit of agricultural output per unit of N input). To achieve surface water quality targets without crop production losses, the European average NUE needs to increase from 0.64 to 0.78, being an average increase of 22%.
- Required increases in NUE to respect thresholds related to water quality while maintaining crop production show, however, large spatial variation in Europe. The largest increases in NUE are required in Benelux, Poland, the Po valley in Italy and certain regions in Spain and Greece. In some regions, crop production and environmental thresholds can only be reconciled at NUEs > 90%, which is not considered feasible given that N is highly mobile, and some N losses are unavoidable.
- Furthermore, N inputs and associated losses may be reduced by increasing NUE along other parts of the food chain, e.g. during fertilizer production, or by avoiding losses that occur at the production and consumption stage. In addition, the higher NUE of plant-based food compared to animal-based food means that reducing consumption of animal products is an effective strategy to mitigate N-related impacts.

Current and critical phosphorus inputs and risks in view of food production and water quality

As with nitrogen, phosphorus is an essential nutrient in agricultural production. Unlike N, P is strongly adsorbed in the soil and this less mobile. Phosphorus adsorbs to particles in the soil, which means that substantial reserves of phosphate can build up in the soil if application rates exceed uptake and losses for extended periods of time. Phosphorus benefits for yields and environmental risks thus do not only depend on inputs in a given year, but also on the soil P status (both how much P is contained in soils and to what extent it is available for crops). On P-saturated soils, additional P inputs hardly lead to increases in crop yields, while those inputs substantially increase the risks for P runoff, whereas in soils with a low P status, sorption may be strong and thus little of the added P may become available for crop growth. On the long term, sustainable P management should aim for application rates that equal crop P removal rates.

Key findings reported on current P inputs and P uptake and ways to reconcile the tension between food production and environmental protection are:

- Average phosphorus inputs to European agricultural soils were $17 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ in 2010. Most P was applied in the form of manure ($9 \text{ kg P ha}^{-1} \text{ yr}^{-1}$), followed by fertilizer ($7 \text{ kg P ha}^{-1} \text{ yr}^{-1}$). Average P inputs to grasslands were higher than P inputs to arable land, mainly due to higher manure P inputs on grasslands.
- The current (year 2010) average crop P removal from European agricultural soils was near $15 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ while average crop P removal at target yields values were ca 25% higher, thus exceeding the average current soil P inputs.

- A comparison of current P inputs and P removals at current (and target) crop yields shows a needed decrease in P inputs in western Europe including Ireland, UK, the Netherlands, Belgium and Luxembourg and Bretagne in France, while increased P inputs are needed in the remaining parts of Europe in view of crop P demand.

Current and critical metal budgets and risks in view of soil biodiversity and food quality

To assess risks of metals in soil, it is imperative to relate the concentration in soil, and changes therein to relevant targets or receptors. Soil biodiversity and food quality are key issues that can be affected by the presence of excess levels of metals like cadmium, copper, lead or zinc in soils. In this report EU-27 wide data are presented that provide insight in the degree to which current levels of the four metals can lead to issues with biodiversity and, for cadmium also for food quality. In addition, current input levels to soils are evaluated to assess to what extent the so-called critical levels in soil can be exceeded in the future. Critical levels correspond to levels at which effects on either biodiversity or food quality are expected to occur. To do so, both data on current levels in soil are presented as well as input output balances that provide insight in the degree of accumulation or depletion that can exist in different climatic zones across the EU.

Key findings reported on concentrations of metals in soils and accumulation are:

- Levels of cadmium and lead show a distinct spatial relationship with the degree of industrialization whereas copper and zinc levels in soil are still more related to variation in natural (geogenic) levels of metals in soil.
- At EU level, zinc (+18000 tons yr⁻¹), copper (+ 6000 tons yr⁻¹) and lead (+2000 tons yr⁻¹) accumulate in soil whereas for cadmium the overall balance (at EU-level) is slightly negative with an annual estimated net loss of cadmium of 119 tons yr⁻¹.
- The regional variation in the net accumulation rate however is large and related to a combination of management (choice of fertilisers), soil type and climatic conditions.
- Mineral fertilisers (43%) and atmospheric deposition (31%) are the dominant inputs of cadmium to agricultural soils, whereas leaching (88%) is the main output.
- For copper (78%) and zinc (77%) animal manure is the dominant source; outputs via crop uptake and leaching are approximately equal for both elements.
- Atmospheric deposition is, for lead (47%) still the dominant source followed by animal manure (22%) and inorganic fertiliser (20%)
- Leaching losses are low in Mediterranean areas due to both a low net water loss from soil and high pH levels which leads, on average to a positive balance for most metals considered here
- In areas characterized by moderate to high rainfall and acid soils on the other hand, leaching losses of cadmium and zinc are high. For cadmium this results in a net negative balance (depletion) in large parts of north western EU members states despite elevated input levels to soil
- For both copper and zinc inputs in areas with intensive animal husbandry such as the Netherlands, Denmark, Brittany (France) and northern Italy (Po area) are high leading to substantial accumulation compared to other areas in the EU.

Key findings related to the evaluation of risks in view of soil biodiversity and food quality are:

- Current cadmium and lead levels in soil are largely below the critical threshold levels in view of effects on biodiversity
- Exceedance of the critical threshold concentration for copper (23%) and zinc (18%) is however substantial with copper levels being particularly high (compared to the threshold concentration) in Mediterranean countries.
- Current inputs to soil for cadmium will not lead to an exceedance of the soil threshold concentrations. For lead, copper and zinc current inputs exceed critical inputs in 48% and 70% of the total surface area. This implies that where current levels in soil do not exceed the critical soil

concentrations, this will occur in the regions where critical inputs are exceeded if current inputs remain unchanged.

- For lead this is concentrated in Mediterranean countries and largely related to the relatively low mobility of lead in soils. For zinc and copper exceedance of the critical inputs occur across most member states and are due to both elevated inputs in areas with intensive animal husbandry in north-western parts of the EU and low outputs from soil in Mediterranean countries.
- Current levels of cadmium largely hardly ever exceed the critical concentration in soil in view of food quality (>99% of the surface area) do not as represented by the uptake by wheat.
- Due to the rather mobile character of cadmium in soil leaching losses are pronounced which also leads to the observation that current inputs do not exceed critical inputs in 96% of the total surface area used for agricultural crop production.
- Only in Spain, Greece and small parts of southern Poland, current inputs are such that with time the soil cadmium concentration can exceed the critical cadmium concentration in view of food safety. In Poland this is predominantly due to high inputs whereas in Spain and Greece this is due to very low leaching rates which facilitate cadmium retention in the (top) soil

References

- Arcadis. 2021. Contaminants in fertilisers: Assessment of the risks from their presence and socio-economic impacts of a possible restriction under REACH Final Report under contract 070201/2019/817112/SER/ENV.B2. Arcadis, July 2021
- Bai, Z., Li, H., Yang, X., Zhou, B., Shi, X., Wang, B., . . . Zhang, F.S. (2013). The critical soil P levels for crop yield, soil fertility and environmental safety in different soil types. *Plant and Soil*, 372, 27-37. doi: 10.1007/s11104-013-1696-y
- Ballabio, C., Panagos, P., Lugato, E., Huang, J.-H., Orgiazzi, A., Jones, A., . . . Montanarella, L. (2018.). Copper distribution in European topsoils: An assessment based on LUCAS soil survey. *Science of the Total Environment*, 636, 282-298. doi: 10.1016/j.scitotenv.2018.04.268
- Bleeker, A., Hicks, W.K., Dentener, F., Galloway, J., & Erisman, J.W. (2011). N deposition as a threat to the World's protected areas under the Convention on Biological Diversity. *Environmental Pollution*, 159(10), 2280-2288. doi: 10.1016/j.envpol.2010.10.036
- Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., . . . de Vries, W. (2010). Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications*, 20(1), 30-59. doi: 10.1890/08-1140.1
- Bruinsma, J. (2012). *European and Central Asian Agriculture: Towards 2030 and 2050*, Budapest, Hungary.
- Camargo, J.A., & Alonso, A. (2006). Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environment International*, 32(6), 831-849. doi: 10.1016/j.envint.2006.05.002
- Carlou, C., D'Alessandro, M., & Swartjes, F. (2007). Derivation methods of soil screening values in Europe: a review of national procedures towards harmonisation. Office for Official Publications of the European Communities.
- Clark, C.M., & Tilman, D. (2008). Loss of plant species after chronic low-level nitrogen deposition to prairie grasslands. *Nature*, 451, 712-715. doi: 10.1038/nature06503
- Cordell, D., Drangert, J., & White, S. (2009). The story of phosphorus: Global food security and food for thought. *Global Environmental Change*, 19, 292-305.
- Council of the European Union. (2018). Proposal for a REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL laying down rules on the making available on the market of CE marked fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009.
- De Vries, W., & Bakker, D.J. (1998). Manual for calculating critical loads of heavy metals for terrestrial ecosystems. Guidelines for critical limits, calculation methods and input data *Report* (pp. 144). Wageningen, the Netherlands: DLO Winand Staring Centre.
- De Vries, W., Du, E., Bahl, K.B., Schulte-Uebbing, L., & Dentener, F. (2017). Global-scale impact of human nitrogen fixation on greenhouse gas emissions. *Oxford Research Encyclopedia of Environmental Science*, DOI: 10.1093/acrefore/9780199389414.9780199389013.9780199389413.
- De Vries, W., Kros, J., Kroeze, C., & Seitzinger, S.P. (2013). Assessing planetary and regional nitrogen boundaries related to food security and adverse environmental impacts. *Current Opinion in Environmental Sustainability*, 5(3-4), 392-402. doi: 10.1016/j.cosust.2013.07.004
- De Vries, W., Kros, J., Reinds, G.J., & Butterbach-Bahl, K. (2011a). Quantifying impacts of nitrogen use in European agriculture on global warming potential. *Current Opinion in Environmental Sustainability*, 3(5), 291-302. doi: 10.1016/j.cosust.2011.08.009
- De Vries, W., Leip, A., Reinds, G.J., Kros, J., Lesschen, J.P., & Bouwman, A.F. (2011b). Comparison of land nitrogen budgets for European agriculture by various modeling approaches. *Environmental Pollution*, 159(11), 3254-3268. doi: 10.1016/j.envpol.2011.03.038
- De Vries, W., Leip, A., Reinds, G.J., Kros, J., Lesschen, J.P., Bouwman, A.F., . . . Winiwarter, W. (2011c). Geographic variation in terrestrial nitrogen budgets across Europe. In M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. van Grinsven & B. Grizzetti (Eds.), *The European Nitrogen Assessment* (pp. 317-344). Cambridge, UK: Cambridge University Press, Chapter 15.
- De Vries, W., Lofts, S., Tipping, E., Meili, M., Groenenberg, B.J., & Schütze, G. (2007a). Impact of soil properties on critical concentrations of cadmium, lead, copper, zinc and mercury in soil and soil solution in view of ecotoxicological effects. *Reviews of Environmental Contamination and Toxicology*, 191, 47-89. doi: 10.1007/978-0-387-69163-3
- De Vries, W., Römkens, P., Kros, J., Voogd, J.C., Schröder, C., & Aksoy, E. (2014). Field-scale balances of nitrogen, phosphorus, copper, zinc, cadmium and lead, and the impacts of policies on those balances: European Environment Agency technical report, ETC-SIA. Available under the EIONET webpage: http://forum.eionet.europa.eu/etc-sia-consortium/library/2014-subvention/182_2-impact-policy-changes/deliverables.

- De Vries, W., Römken, P.F.A.M., & Schütze, G. (2007b). Critical soil concentrations of cadmium, lead and mercury in view of health effects on humans and animals. *Reviews of Environmental Contamination and Toxicology*, 191, 91-130. doi: 10.1007/978-0-387-69163-3
- De Vries, W., Schulte-Uebbing, L., Kros, H., Voogd, J.C., & Louwagie, G. (2021). Spatially explicit boundaries for agricultural nitrogen inputs in the European Union to meet air and water quality targets. *Science of The Total Environment*, 147283. doi: <https://doi.org/10.1016/j.scitotenv.2021.147283>
- De Wit, M., Londo, M., & Faaij, A. (2011). Productivity developments in European agriculture: Relations to and opportunities for biomass production. *Renewable and Sustainable Energy Reviews*, 15, 2397-2412. doi: 10.1016/J.RSER.2011.02.022
- Diaz, R.J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, 321(5891), 926-929. doi: 10.1126/science.1156401
- EC. (1991). *Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources*. Brussel: European Commission.
- EC. (1992). *Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora*. Brussel: European Commission.
- EC. (2000). *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy* Brussel: European Commission.
- Eckel, H., Roth, U., Döhler, H., Nicholson, F., & Unwin, R. (Eds.). (2005). *Assessment and reduction of heavy metal input into agro-ecosystems. Final report of the EU-concerted action Aromis. KTBL-Schrift 432*. Darmstadt: KTBL.
- EEA. (2000). *Down to earth: Soil degradation and sustainable development in Europe-A challenge for the 21st century*.
- EEA. (2005). *The European environment – State and outlook 2005*. Copenhagen: European Environment Agency.
- EEA. (2010a). *10 messages for 2010 - Agricultural ecosystems*. Copenhagen: European Environment Agency.
- EEA. (2010b). *The European Environment : State and outlook 2010 – Soil*. Copenhagen: European Environment Agency.
- EEA. (2012). *Water for agriculture*. Copenhagen: European Environment Agency.
- Eickhout, B., Bouwman, A.F., & van Zeijts, H. (2006). The role of nitrogen in world food production and environmental sustainability. *Agriculture, Ecosystems & Environment*, 116(1-2), 4-14. doi: 10.1016/j.agee.2006.03.009
- Erismann, J.W., Sutton, M.A., Galloway, J., Klimont, Z., & Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience*, 1(10), 636-639. doi: 10.1038/ngeo325
- ESDAC. (2019). European Soil Data Centre (ESDAC), esdac.jrc.ec.europa.eu, European Commission, Joint Research Centre.
- EU. (2012). *Agriculture in the EU: Statistical & economic information report*.: EU Directorate-General for Agriculture and Rural Development.
- Fader, M., Gerten, D., Krause, M., Lucht, W., & Cramer, W. (2013). Spatial decoupling of agricultural production and consumption: Quantifying dependences of countries on food imports due to domestic land and water constraints. *Environmental Research Letters*, 8. doi: 10.1088/1748-9326/8/1/014046
- FAO. (2017). *The future of food and agriculture: Trends and challenges*, Rome.
- FAO. (2018). *Nitrogen inputs to agricultural soils from livestock manure New statistics*. Rome: Food and agriculture organization of the United Nations.
- Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., . . . Snyder, P.K. (2005). Global consequences of land use. *Science*, 309(5734), 570-574. doi: 10.1126/science.1111772
- Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., . . . Zaks, D.P.M. (2011). Solutions for a cultivated planet. *Nature*, 478(7369), 337-342. doi: 10.1038/nature10452
- Galloway, J.N., & Cowling, E.B. (2002a). Reactive Nitrogen and The World: 200 Years of Change. *AMBIO: A Journal of the Human Environment*, 31(2), 64-71. doi: 10.1579/0044-7447-31.2.64
- Galloway, J.N., & Cowling, E.B. (2002b). Reactive nitrogen and the world: 200 years of change. *Ambio*, 31(2), 64-71.
- Glibert, P.M., & Burford, M.A. (2017). Globally changing nutrient loads and harmful algal blooms: Recent advances, new paradigms, and continuing challenges. *Oceanography*, 30(1), 58–69. doi: 10.5670/oceanog.2017.110

- Goulding, K.W.T. (2016). Soil acidification and the importance of liming agricultural soils with particular reference to the United Kingdom. *Soil Use and Management*, 32(3), 390-399. doi: 10.1111/sum.12270
- Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., . . . Zhang, F.S. (2010). Significant acidification in major Chinese croplands. *Science*, 327(5968), 1008-1010. doi: 10.1126/science.1182570
- Hettelingh, J.-P., Posch, M., Slootweg, J., & Mathijssen, L. (2014). Exposure of ecosystems to acidification and eutrophication in Europe: an update of EEA-Core Set Indicator 005*. In J. Slootweg, M. Posch, J. P. Hettelingh & L. Mathijssen (Eds.), *Modelling and Mapping the impacts of atmospheric deposition on plant species diversity in Europe, CCCE status report 2014* (pp. 13-27).
- Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., . . . Zhang, F.S. (2009). Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proceedings of the National Academy of Sciences of the United States of America*, 106(9), 3041-3046. doi: 10.1073/pnas.0813417106
- Jungk, A., Claassen, N., Schulz, V., & Wendt, J. (1993). Availability of Phosphate Reserves in Arable Soils. *Zeitschrift Fur Pflanzenernahrung Und Bodenkunde*, 156(5), 397-406. doi: 10.1002/jpln.19931560504
- Lado, L.R., Hengl, T., & Reuter, H.I. (2008). Heavy metals in European soils: A geostatistical analysis of the FOREGS Geochemical database. *Geoderma*, 148, 189-199.
- Lassaletta, L., Billen, G., Grizzetti, B., Garnier, J., Leach, A.M., & Galloway, J.N. (2014). Food and feed trade as a driver in the global nitrogen cycle: 50-year trends. *Biogeochemistry*, 118(1-3), 225-241. doi: 10.1007/s10533-013-9923-4
- Leip, A., Britz, W., Weiss, F., & de Vries, W. (2011). Farm, land, and soil nitrogen budgets for agriculture in Europe calculated with CAPRI. *Environmental Pollution*, 159(11), 3243-3253. doi: 10.1016/j.envpol.2011.01.040
- Lewis, S.L. (2012). We must set planetary boundaries wisely. *Nature*, 485(7399), 417. doi: 10.1038/485417a
- Liu, J., You, L., Amini, M., Obersteiner, M., Herrero, M., Zehnder, A.J.B., & Yang, H. (2010). A high resolution assessment on global nitrogen flows in cropland. *Proceedings of the National Academy of Sciences of the United States of America*, 107(17), 8035-8040.
- Lobell, D.B., Cassman, K.G., & Field, C.B. (2009). Crop yield gaps: their importance, magnitudes, and causes. *Annual Review of Environment and Resources*, 34, 179-204. doi: 10.1146/annurev.enviro.041008.093740
- Lofts, S., Spurgeon, D.J., Svendsen, C., & Tipping, E. (2004). Deriving soil critical limits for Cu, Zn, Cd and Pb: a method based on free ion concentrations. *Environmental Science and Technology*, 38(13), 3623-3631.
- MacDonald, G.K., Bennett, E.M., Potter, P.A., & Ramankutty, N. (2011). Agronomic phosphorus imbalances across the world's croplands. *Proceedings of the National Academy of Sciences of the United States of America*, 108(7), 3086-3091. doi: 10.1073/pnas.1010808108
- Mol, G., Spijker, J., van Gaans, P., & Römkens, P. (Eds.). (2012). *Geochemische bodematlas van Nederland*. Wageningen, the Netherlands. : Wageningen Academic Publishers.
- Mueller, N.D., Gerber, J.S., Johnston, M., Ray, D.K., Ramankutty, N., & Foley, J.A. (2012). Closing yield gaps through nutrient and water management. *Nature*, 490, 254-257. doi: 10.1038/nature11420
- Nordhaus, T., Shellenberger, M., & Blomqvist, L. (2012). *The planetary boundaries hypothesis a review of the evidence*. Oakland, Ca.: Breakthrough Institute.
- Oenema O, Oudendag D, Velthof GL. (2007) Nutrient losses from manure management in the European Union. *Livest. Sci.* 112: 261-272.
- Orgiazzi, A., Ballabio, C., Panagos, P., Jones, A., & Fernández-Ugalde, O. (2018). LUCAS Soil, the largest expandable soil dataset for Europe: A review. *European Journal of Soil Science*, 69(1), 140-153. doi: 10.1111/ejss.12499
- Oste, L.A., Postma, J.F., Roskam, G.D., Keijzers, R., Van Duijnhoven, N., 2018. Basisdocumentatie probleemstoffen. Rapport bij de basisdocumenten 2018. Deltares Rapport 11202236-001-BGS-0001.
- Pozzer, A., Tsimpidi, A.P., Karydis, V.A., de Meij, A., & Lelieveld, J. (2017). Impact of agricultural emission reductions on fine-particulate matter and public health. *Atmospheric Chemistry and Physics*, 17, 12813–12826. doi: 10.5194/acp-17-12813-2017
- Puma, M.J., Bose, S., Chon, S.Y., & Cook, B.I. (2015). Assessing the evolving fragility of the global food system. *Environmental Research Letters*, 10, 024007. doi: 10.1088/1748-9326/10/2/024007

- Rabalais, N.N., Turner, R.E., & Scavia, D. (2002). Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. *Bioscience*, 52, 129-142.
- Ray, D.K., Mueller, N.D., West, P.C., & Foley, J.A. (2013). Yield Trends Are Insufficient to Double Global Crop Production by 2050. *PLOS ONE*, 8, e66428.
- Ray, D.K., Ramankutty, N., Mueller, N.D., West, P.C., & Foley, J.A. (2012). Recent patterns of crop yield growth and stagnation. *Nature Communications*, 3, 1293. doi: 10.1038/ncomms2296
- Reiman, C., Birke, M., Demetriades, A., Filzmoser, P., & O'Connor, P. (2014). *Chemistry of Europe's Agricultural Soils. Part A Methodology and Interpretation of the GEMAS Data Set. Geologisches Jahrbuch Reihe B Regionale Geologie Ausland und Angewandte Geowissenschaften. Heft 102*.
- Robertson, G.P., & Vitousek, P.M. (2009). Nitrogen in Agriculture: Balancing the Cost of an Essential Resource. *Annual Review of Environment and Resources*, 34(1), 97-125. doi: 10.1146/annurev.enviro.032108.105046
- Rockström, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., III, Lambin, E., . . . Foley, J. (2009a). Planetary Boundaries: Exploring the Safe Operating Space for Humanity. *Ecology and Society*, 14(2).
- Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., . . . Foley, J.A. (2009b). A safe operating space for humanity. *Nature*, 461, 472-475. doi: 10.1038/461472a
- Römken, P.F.A.M., Groenenberg, J.E., Bonten, L.T.C., de Vries, W., & Bril, J. (2004). Derivation of partition relationships to calculate Cd, Cu, Ni, Pb and Zn solubility and activity in soil solutions *Alterra Rapport* (pp. 72). Wageningen, the Netherlands: Alterra.
- Römken, P.F.A.M., Groenenberg, J.E., Rietra, R.P.J.J., & De Vries, W. (2007). Onderbouwing LAC2006-waarden en overzicht van bodem-plant relaties ten behoeve van de Risicotoolbox; een overzicht van gebruikte data en toegepaste methoden *Alterra Rapport 1442* (pp. 103). Wageningen, the Netherlands: Alterra.
- Römken, P.F.A.M., Rietra, R.P.J.J., Kros, J., Voogd, J.C., & de Vries, W. (2018). Impact of cadmium levels in fertilisers on cadmium accumulation in soil and uptake by food crops (pp. 115). Wageningen, the Netherlands: Wageningen Environmental Research.
- Sadowski, A., & Baer-Nawrocka, A. (2016). Food self-sufficiency of the European Union countries - energetic approach. *Journal of Agribusiness and Rural Development*, 10. doi: 10.17306/JARD.2016.44
- Scholz, R., & Wellmer, F.W. (2013). Approaching a dynamic view on the availability of mineral resources: what we may learn from the case of phosphorus? *Global Environmental Change*, 23, 11-27.
- Schoumans, O.F., & Chardon, W.J. (2015). Phosphate saturation degree and accumulation of phosphate in various soil types in The Netherlands. *Geoderma*, 237, 325-335. doi: 10.1016/j.geoderma.2014.08.015
- Schulte-Uebbing, L., & de Vries, W. (2021). Reconciling food production and environmental boundaries for nitrogen in the European Union. *Science of The Total Environment*, 786, 147427. doi: <https://doi.org/10.1016/j.scitotenv.2021.147427>
- Seitzinger, S.P., Kroeze, C., Bouwman, A.F., Caraco, N., Dentener, F., & Styles, R.V. (2002). Global patterns of dissolved inorganic and particulate nitrogen inputs to coastal systems: recent conditions and future projections. *Estuaries*, 25(4), 640-655. doi: 10.1007/BF02804897
- Selman, M., Sugg, Z., Greenhalgh, S., & Diaz, R. (2008). Eutrophication and Hypoxia in Coastal Areas: A Global Assessment of the State of Knowledge.
- Snyder, C.S. (2017). Enhanced nitrogen fertiliser technologies support the '4R' concept to optimise crop production and minimise environmental losses. *Soil Research*, 55(6), 463-472. doi: <https://doi.org/10.1071/SR16335>
- Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B.L., Lassaletta, L., . . . Willett, W. (2018). Changes in food management, technology and diets to stay within planetary boundaries. *Nature*, (Accepted).
- Stark, C., & Richards, K. (2008). The continuing challenge of N loss to the environment: Environmental consequences and mitigation measures. *Dynamic Soil, Dynamic Plant*, 2, 41-55.
- Sutton, M.A., Bleeker, A., Howard, C.M., Bekunda, M., Grizzetti, B., de Vries, W., . . . Zhang, F.S. (2013). *Our Nutrient World: The challenge to produce more food and energy with less pollution. Global Overview of Nutrient Management*. Nairobi: Centre for Ecology and Hydrology, Edinburgh & United Nations Environment Programme.
- Sutton, M.A., Howard, C.M., Erismann, J.W., Billen, G., Bleeker, A., Grennfelt, P., . . . Grizzetti, B. (2011). *The European Nitrogen Assessment*. Cambridge, UK: Cambridge University Press.
- Syers, K., Bekunda, M., Cordell, D., Corman, J., Johnston, J., Rosemarin, A., . . . Loughheed, T. (2011). Phosphorus and food production *UNEP Yearbook 2011* (pp. 34-45). Nairobi: UNEP.

- Tilman, D., Balzer, C., Hill, J., & Befort, B.L. (2011). Global food demand and the sustainable intensification of agriculture. *Proceedings of the National Academy of Sciences of the United States of America*, 108, 20260-20264. doi: 10.1073/pnas.1116437108
- Ulrich, A.E., & Frossard, E. (2014). On the history of a reoccurring concept: Phosphorus scarcity. *Science of the Total Environment*, 490, 694-707. doi: 10.1016/j.scitotenv.2014.04.050
- UNEP, United Nations Environment Programme. (2007). *Global Environmental Outlook 4. (GEO-4) United Nations Environment Program*. Nairobi, Kenya.
- Van Grinsven, H., Ward, M., Benjamin, N., & de Kok, T. (2006). Does the evidence about health risks associated with nitrate ingestion warrant an increase of the nitrate standard for drinking water? *Environmental Health*, 5(1), 26. doi: 10.1186/1476-069X-5-26
- Van Grinsven, H.J.M., Spiertz, J.H.J., Westhoek, H.J., Bouwman, A.F., & Erisman, J.W. (2014). Nitrogen use and food production in European regions from a global perspective. *Journal of Agricultural Science: Cambridge University Press*.
- Van Grinsven, J.J.M., Rabl, A., & de Kok, T.M. (2010). Estimation of incidence and social cost of colon cancer due to nitrate in drinking water in the EU: a tentative cost-benefit assessment. *Environmental Health*, 9, 58. doi: 10.1186/1476-069X-9-58
- Van Ittersum, M.K., Cassman, K.G., Grassini, P., Wolf, J., Tittonell, P., & Hochman, Z. (2013). Yield gap analysis with local to global relevance—A review. *Field Crops Research*, 143, 4-17. doi: 10.1016/j.FCR.2012.09.009
- Van Ittersum, M.K., van Bussel, L.G.J., Wolf, J., Grassini, P., van Wart, J., Guilpart, N., . . . Cassman, K.G. (2016). Can sub-Saharan Africa feed itself? *Proceedings of the National Academy of Sciences of the United States of America*, 113, 14964-14969. doi: 10.1073/pnas.1610359113
- Van Vuuren, D.P., Bouwman, A.F., & Beusen, A.H.W. (2010). Phosphorus demand for the 1970-2100 period: A scenario analysis of resource depletion. *Global Environmental Change*, 20(3), 428-439. doi: 10.1016/j.gloenvcha.2010.04.004
- Velthof, G.L., Barot, S., Bloem, J., Butterbach-Bahl, K., de Vries, W., Kros, J., . . . Oenema, O. (2011). Nitrogen as a threat to European soil quality. In M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. van Grinsven & B. Grizzetti (Eds.), *The European Nitrogen Assessment* (pp. 494-509). Cambridge, UK: Cambridge University Press, Chapter 21.
- Vitousek, P.M., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., . . . Zhang, F.S. (2009). Nutrient imbalances in agricultural development. *Science*, 324, 1519-1520.
- WHO. (2011). Nitrate and Nitrite in Drinking-water Background document for development of WHO Guidelines for Drinking-water. WHO/SDE/WSH/07.01/16/Rev/1.
- WWDR3. (2009). Water in a changing world. United Nations World Water Development Report, 3rd edition.
- WWDR4. (2012). Managing water under uncertainty and risk. United Nations World Water Development Report, 4th edition.



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