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REQUIRED CHANGES IN NITROGEN INPUTS AND NITROGEN USE EFFICIENCIES TO RECONCILE AGRICULTURAL PRODUCTIVITY WITH WATER AND AIR QUALITY OBJECTIVES IN THE EU-27

By

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SUMMARY.

Nitrogen (N) losses to air, ground water and surface water in response to agricultural N inputs affect air and water quality. Agricultural N inputs in this article are defined as mineral N fertilisers, N manure and biosolids and biological N fixation. Using a spatially explicit N balance model, we assessed where agricultural N losses within THE EU-27 currently lead to an exceedance of critical ammonia (NH_3) emissions in relation to adverse impacts on terrestrial ecosystems, critical N concentrations in runoff to surface water in relation to eutrophication impacts and critical nitrate (NO_3) concentrations in groundwater in relation to drinking water quality. We then calculated the N inputs at which critical N emissions or concentrations are just not exceeded ('critical' N inputs). We also assessed required N inputs in order to achieve target yields, defined as 80% of the water-limited yield potential at actual N use efficiency. Actual, critical and required N inputs were calculated for c.40,000 unique soil-slope-climate combinations throughout the European Union. When actual or required N inputs exceeded critical inputs, we calculated the necessary reduction in ammonia emission fractions and necessary increase in NUE to attain actual or target yield while simultaneously reaching air and water quality goals. The ammonia emission fraction referred to the ratio of the total NH_3-N emissions, divided by the total N excretion by livestock.

Results show that required N inputs at the EU-27 level are on average 27% higher than actual inputs. Average critical N inputs are 31% and 43% lower than actual N inputs in relation to critical NH_3 emissions and critical N runoff to surface water, respectively, but 1% higher in relation to critical NO_3 leaching to groundwater. The risk for surface water is, however, likely overestimated, since calculated N concentrations in runoff to surface water appear to be higher than concentrations in surface water. An overall reduction in N inputs of 30% to protect air and water quality seems a reasonable average estimate. Critical inputs are most strongly exceeded in regions with high actual N inputs, such as Ireland, the Netherlands, Belgium and Luxembourg, Brittany in France and the Po valley in Italy.

The actual N use efficiency (NUE) for all agricultural land, averaged over the EU-27 is 61%. This value has to increase on average to 72% to protect surface water quality at actual crop yields and to 74% at target crop yields. Opportunities thus exist to reduce the environmental impact of agriculture by increasing the NUE, while still allowing an increase in crop production in the EU. However, in c.15-20% of the agricultural land area, it is not feasible to achieve the surface water criterion at actual crop yield and this area increases to 25% at target crop yield, because it would require an NUE over 90%. In these areas, an additional reduction of N inputs is necessary, but this comes at the expense of crop yield reductions.

CONTENTS

Keywords: agricultural management, nitrogen use, nitrogen use efficiencies, nitrogen boundaries, water quality, air quality.

1. INTRODUCTION.

1.1. Enhanced food production and acceleration of the nitrogen cycle.

Nitrogen (N) is an essential nutrient for the growth and functioning of plants, animals and humans. The Earth's atmosphere consists of 78% di-nitrogen (N_2), but in this form N is unavailable to most living organisms. Instead, reactive forms of nitrogen (Nr), which include all forms of N except N_2, are crucial for life on earth. Since the late nineteenth century, human activities have approximately doubled Nr inputs to the environment (Galloway *et al.*, 2004). This increase has mainly been driven by increased N fertiliser use, an increase in the cultivated area of N-fixing crops (Smil, 2001; Erisman *et al.*, 2008; Fowler *et al.*, 2013) and increased use of fossil fuels, to fulfil the food and energy demand of a growing world population (Galloway *et al.*, 2008). Supply of N fertiliser, produced by industrial synthesis of ammonia (NH_3) from atmospheric N_2 and hydrogen, is essential to feed an ever increasing world population (Eickhout *et al.*, 2006; Erisman *et al.*, 2008; Robertson and Vitousek, 2009; Sutton *et al.*, 2013). The approximately 105 million tonnes actually applied (FAO, 2010) are estimated to feed about 50% of the human population (Smil, 2002; Erisman *et al.*, 2008; Smil, 2011). Especially in Europe, agricultural production has increased rapidly since the early 1940s, associated with a large increase in fertiliser and manure inputs. N inputs to EU agriculture reached a maximum around 1985 and have decreased since then (e.g., Sutton *et al.*, 2011). This is partly due to reduced fertiliser application and partly because the number of dairy cattle has decreased by about 1% per year since the implementation of the milk quota system in the EU-15 in 1984 (Oenema *et al.*, 2007).

1.2. Unwanted side-effects of nitrogen application and planetary boundaries.

At the European scale, only 60% of the nitrogen applied to agricultural land is taken up by crops, while the remainder is lost to the environment (Leip *et al.*, 2011a). Since the 1990s, the nitrogen use efficiency (NUE) of European agriculture has increased (Van Grinsven *et al.*, 2014) but by not nearly enough to reduce N losses sufficiently to meet environmental targets. Nitrogen that is lost to the environment leads to substantial unwanted side-effects including (see also Figure 1): an increase in: (i) runoff of N, causing eutrophication of surface waters (e.g. Camargo and Alonso, 2006), (ii) ammonia (NH_3) emissions (e.g. Webb *et al.*, 2005; Oenema *et al.*, 2007) and deposition on nearby terrestrial ecosystems, causing nutrient enrichment and decreases in plant species diversity (Clark and Tilman, 2008; Bobbink *et al.*, 2010; De Vries *et al.*, 2010; Bleeker *et al.*, 2011), (iii) leaching of nitrate (NO_3^-) to groundwater, causing degradation in drinking water quality (e.g. Van Grinsven *et al.*, 2006; Powlson *et al.*, 2008) and (iv) emissions of nitrous oxide (N_2O), a greenhouse gas, causing climate change (Freibauer, 2003). Following the same trend as N inputs, N losses to air by NH_3 and N_2O emissions, and to water by NO_3 leaching and N runoff increased in Europe up to 1985 and decreased thereafter, also because of improved manure management in grasslands (Sutton *et al.*, 2011).

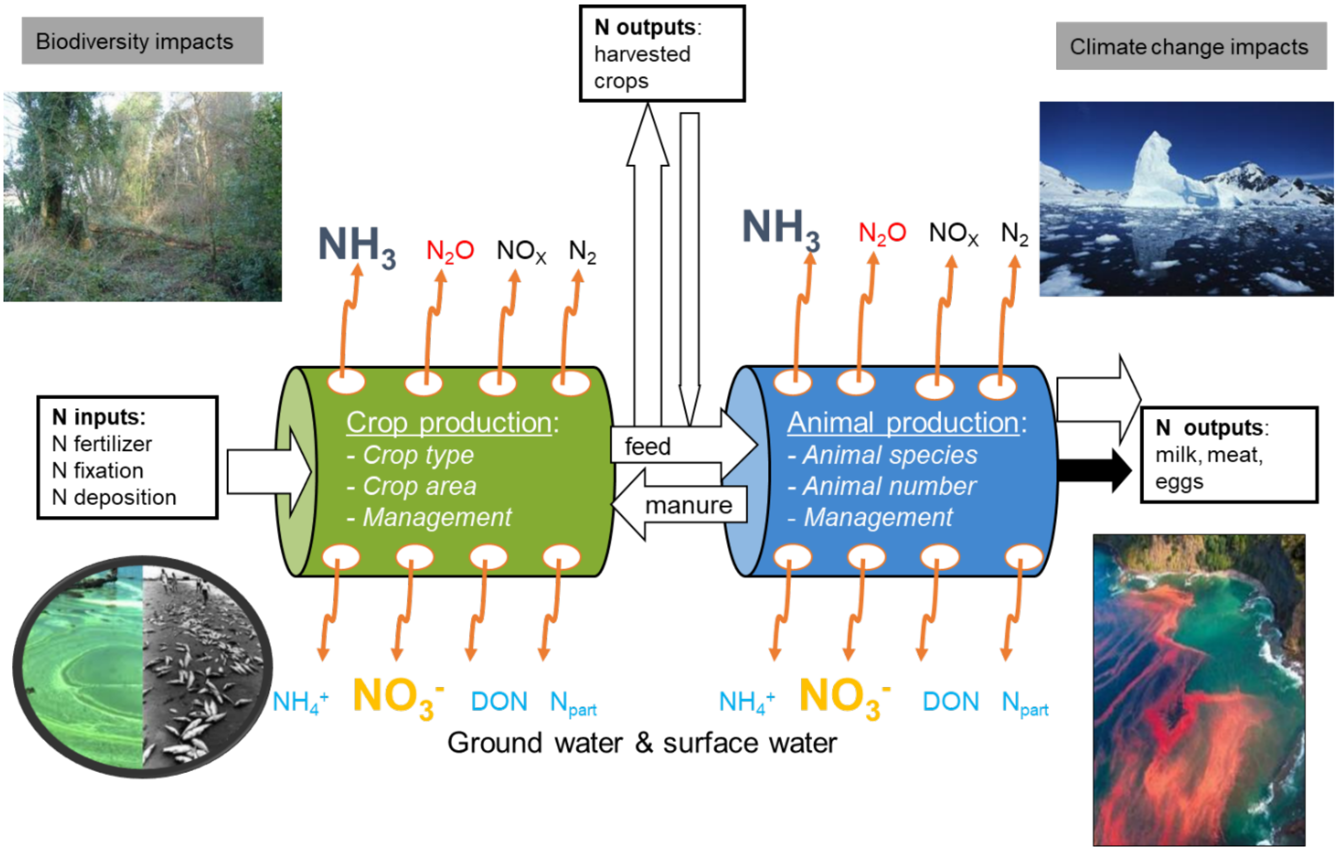

Figure 1: *Impacts of nitrogen inputs on losses of various N compounds to air and water, thus affecting biodiversity, water quality and climate.*

The fact that the expanded nitrogen cycle drives multiple, interacting global-scale effects has led to the concept of a 'planetary N boundary'. Planetary boundaries have been defined for several other environmental issues and can be seen as planet-wide environmental 'tipping points' beyond which humanity is at risk (Rockström *et al.*, 2009b; Rockström *et al.*, 2009a). The planetary N boundary has, however, been criticised for focussing only on the ecological consequences of the disturbance of the N cycle, while neglecting the need for N to feed the actual world population (Nordhaus *et al.*, 2012). Opportunities exist to reduce the environmental impact of agriculture by eliminating nutrient overuse, while still allowing an increase in production of major cereals (Mueller *et al.*, 2012). Furthermore, it is arbitrary to define a global threshold for the N boundary, due to the spatial variability in impacts both in terms of N limitation and N overuse (Lewis, 2012; Nordhaus *et al.*, 2012). Especially on a global scale, N fertiliser application is distributed very unevenly. In many African countries (Liu *et al.*, 2010), as well as in large areas of Latin America and South East Asia (MacDonald *et al.*, 2011), N 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 input is needed to avoid land degradation and increase crop yields, while in other parts N application can be reduced while simultaneously maintaining, or even enhancing, yields and reducing environmental impacts (Ju *et al.*, 2009).

To meet the world's food production needs, N fertilisers, including mineral fertilisers and manure, are thus 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). Considering these aspects, De Vries *et al.* (2013) calculated both required global N inputs in relation to food security and planetary N boundaries for adverse environmental impacts. The assessment of the planetary N boundary was based on a critical NH_3 concentration in air, in relation to biodiversity decline in terrestrial ecosystems, and a critical N concentration in runoff in relation to eutrophication of aquatic ecosystems. In De Vries *et al.* (2013), spatial variations were accounted for in a very approximate way. The present study addresses the need for a more sophisticated approach to assess the critical N inputs.

1.3. Aim of this study.

In this study we calculated critical N inputs and their exceedances by actual N inputs (inputs in the year 2010) and required N inputs, in order to achieve target yields, for agricultural soils in the EU-27 region, excluding Croatia. Critical inputs were derived in view of environmental effects on terrestrial and aquatic ecosystems and human health due to NH_3 emissions to air, N runoff to surface water and NO_3 leaching to ground water, respectively. Target yields were defined as 80% of the water-limited yield potential and required N inputs were assessed at actual N use efficiency (NUE).

When actual or required N inputs exceeded critical inputs, we calculated the increase in NUE that is necessary to attain actual or target yield while simultaneously reaching water quality goals by avoiding exceedance of critical N concentrations in groundwater and surface waters. In addition, the necessary reduction in ammonia emission fractions were calculated to avoid the exceedance of critical N deposition levels on nature.

2. ASSESSMENT METHODS AND DATA.

The model INTEGRATOR (De Vries *et al.*, 2011b) was used to calculate: (i) actual N inputs and associated N losses for the year 2010, (ii) required N inputs, i.e. inputs that are required to obtain target crop yields, set to 80% of the water limited yield potential, and associated N losses, and (iii) critical N inputs, i.e. N inputs to the soil at which critical N losses to air and water calculated from critical values for defined N indicators are not exceeded. The model INTEGRATOR (De Vries *et al.*, 2011b) was used to calculate actual N inputs and N losses (reference year 2010) and to back-calculate critical N inputs to the soil on the basis of calculated critical N losses to air and water from critical values for defined N indicators. In this study, we calculated critical and required N inputs using the actual N use efficiency (NUE_{act}) as derived for the year 2010, as illustrated in Figure 2.

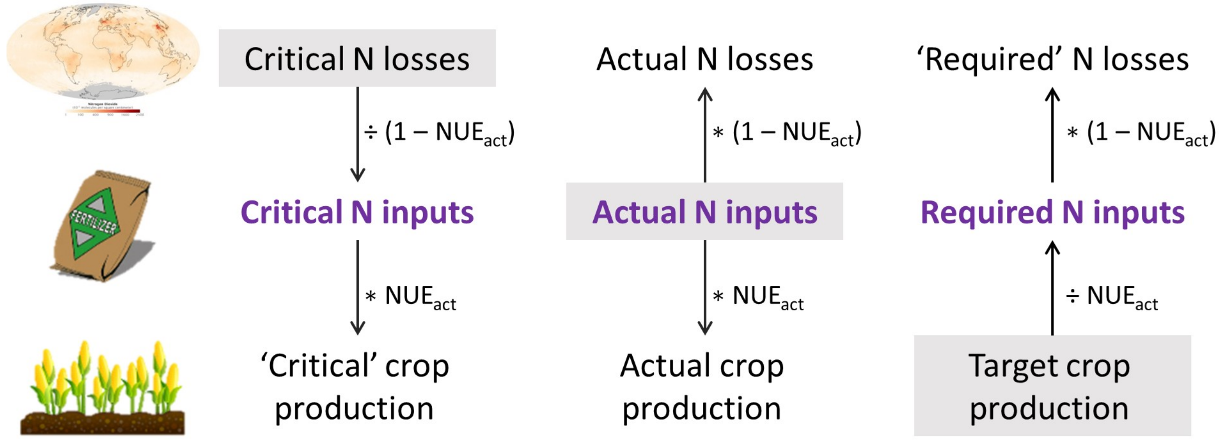

Figure 2: *The drivers behind the calculation of (i) actual N inputs, based on actual fertiliser consumption and livestock numbers, (ii) required N inputs, based on target crop yields and (iii) critical N inputs, based on critical N losses to air and water based on critical values for defined N indicators.*

INTEGRATOR calculates the various N flows by empirical linear relationships between the different N fluxes. In line with the need for spatial accuracy, calculations were carried out at NCU level, where NCUs stand for Nitrogen Calculation Units (NCUs), being approximately 40,000 unique combinations of soil type, administrative region, slope class and altitude class. NCUs are comprised of polygons that are a cluster of 1 km x 1 km pixels and are subdivisions of NUTS3 regions, where NUTS (Nomenclature of territorial units for statistics) stands for a hierarchical system for dividing up the economic territory of the EU for the purpose of socio-economic analyses of the regions (De Vries *et al.*, 2011b; De Vries *et al.*, 2011c). The results thus obtained were aggregated at the NUTS3 level, country level and the EU-27 level. The various calculations and input data used are given below, while details of the calculations are given in De Vries et al., (2020).

2.1. Actual inputs and losses of nitrogen.

2.1.1. Overall approach.

The flows of nitrogen (N) in agricultural soils are calculated by an empirical linear model predicting (i) N (NH_3, N_2O, NO_x and N_2) emissions from housing systems in response to N excretion, followed by (ii) N uptake, N (NH_3, N_2O, NO_x and N_2) emissions from soils and N runoff and leaching to ground water and surface water losses in response to inputs by fertilisers, animal manure, atmospheric deposition and biological N fixation. A schematic overview of this approach is presented in Figure 3, an elaboration of the MITERRA-Europe approach (Velthof *et al.*, 2009).

The figure specifically illustrates the approach to calculate N (NH_3, N_2O, NO_x and N_2) from housing systems and soils and leaching/runoff of N to ground water and surface water, using linear relationships. The empirical fractions are a function of combinations of land use (grassland, arable land), climate (precipitation, temperature), soil type and/or slope, depending on the type of fraction. More details on the calculation of inputs, uptake and losses to air and water are given below.

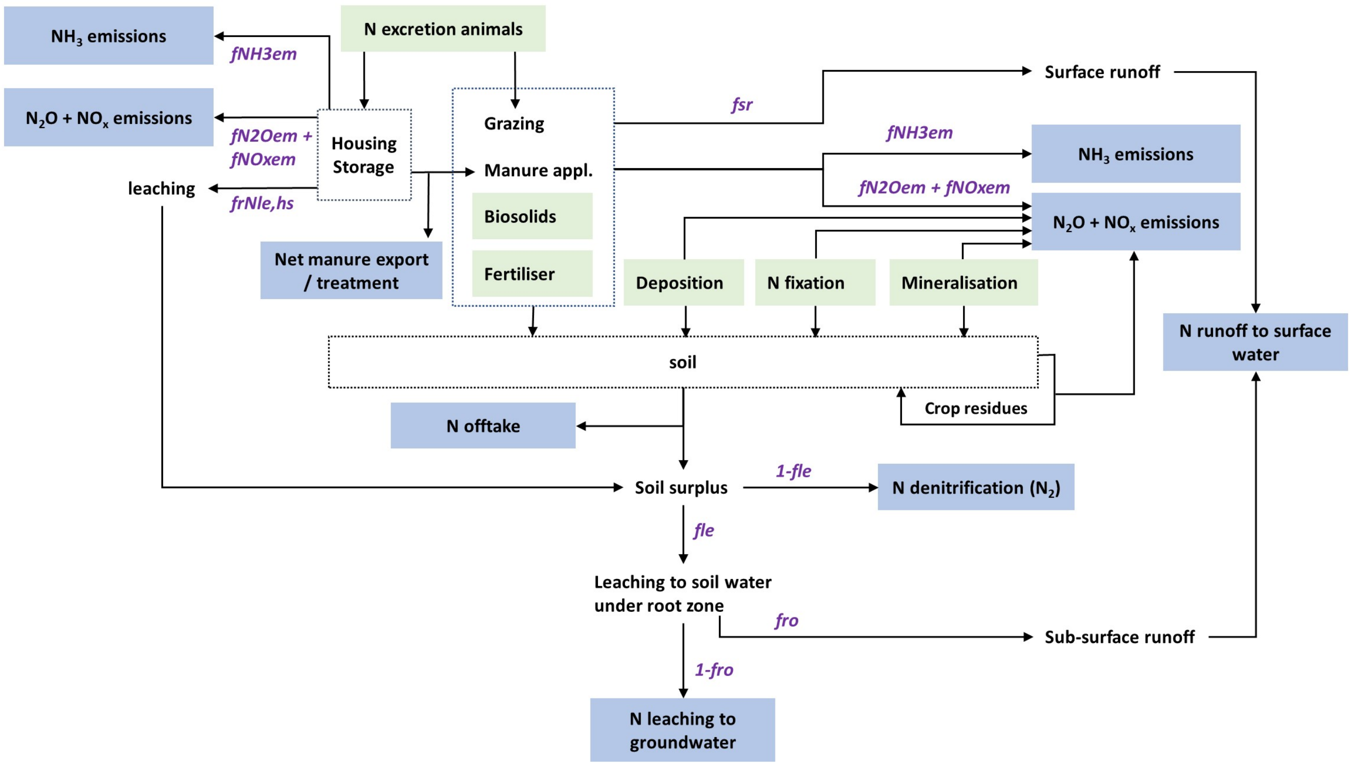

Figure 3: *Schematic presentation of the calculated N flows in INTEGRATOR (adapted after the MITERRA-EUROPE model by Velthof et al., 2009; see text for description of calculation methods). At the NCU level, the sum of total gross N inputs (green boxes) equals the sum of all N outputs (blue boxes). The sum of all green boxes equals the sum of all blue boxes.*

2.1.2. Assessment of spatially explicit inputs, uptake and surpluses of nitrogen.

The total N input to agricultural soils is calculated as the sum of inputs via fertiliser, animal manure, biosolids, atmospheric deposition and biological N fixation, while N offtake consists of the N that is removed from the soil by harvest. N fertiliser inputs at the NCU level are calculated on the basis of a crop N demand, accounting for all non-fertiliser N inputs, and using a balanced N fertilisation approach. The fertiliser inputs thus derived are scaled to FAO statistical data at country level, by multiplying the values with the ratio of FAO country data and the aggregated country level data. Non-fertiliser N inputs from manure, biosolids (compost, sludge), N fixation and deposition at the NCU level are derived by downscaling data (mostly at national level) to the required NCU level. Manure N inputs are calculated by multiplying animal numbers by N excretion rates. Biosolids (compost, sludge) N inputs are calculated by multiplying application rates of biosolids with the N content of biosolids (dry matter basis). For compost, national data (when available) have been used; for sludge, generic data (median value at the EU-27 level) have been used. Atmospheric N deposition is based on EMEP model estimates (Simpson *et al.*, 2003; EMEP, 2009) at 50 km x 50 km resolution, downscaled to the NCU level. N crop uptake is calculated by multiplying statistical data on crop yield for specified major crops (approximately 30) with plant-specific N content of harvested products. Details on the spatially explicit assessment of N excretion and related N manure input and the N fertiliser application are given in De Vries *et al.* (2020).

2.1.3. Assessment of spatially explicit losses of nitrogen to air and water.

Emissions of gaseous N compounds (NH_3, N_2O, NO_x and N_2) and leaching and runoff of N to surface water are due to N inputs from faeces and urine during storage in manure storage systems, by grazing of free ranging animals, after application of manure and fertilisers to agricultural land, through atmospheric deposition, N fixation and crop residue input. These emission losses are calculated in INTEGRATOR by multiplying N inputs by emission, leaching or runoff factors.

The fate of N in the agricultural system was calculated as a sequence of occurrences: ammonia (NH_3) and nitrous oxide (N_2O) emissions, followed by N uptake, leaching and surface runoff and resulting denitrification in the soil. The total leaching losses of N from the terrestrial system were calculated by subtracting all N outputs from the system (gaseous emissions, uptake, runoff, denitrification from the N inputs to the soil and net release/mineralisation in case of peat soils). All N transformation processes were assumed to linearly relate to N inputs. This implies that NH_3, N_2O and NO_x emissions depend linearly on the N input into the soil, N uptake on the N input minus the N (NH_3, N_2O and NO_x) emissions and N losses to water (leaching plus runoff) on N surplus (input minus N emissions minus N uptake), plus the change in soil N pool, by either net mineralisation or immobilisation. Leaching and subsurface runoff was partitioned to groundwater and to surface water by multiplying the total loss with a leaching fraction and a runoff fraction (1 – leaching fraction), respectively. Nitrogen loss by denitrification (practically equal to N_2 emissions) was calculated as the residual flux, being equal to N input minus N emissions minus N uptake plus the change in soil N pool minus N runoff and leaching.

The linear transformation constants (emission fractions, uptake fractions, leaching fractions and runoff fractions) are a function of the type of fertiliser or manure, land use, soil type, application method and/or hydrological regime. An overview of the assessment of the N loss fractions, determining the N losses to air (NH_3, N_2O, NO_x and N_2 emissions) and water (nitrate leaching and runoff) is given in De Vries *et al.* (2020). Details on the derivation of all fractions are given in Velthof *et al.*, (2009) and De Vries *et al.* (2014). The change in soil N pool by net mineralisation or immobilisation was calculated by dividing the C mineralisation or immobilisation by the soil C/N ratio. For peat soils, C mineralisation is derived by a relationship with groundwater level.

The parameterisation of the equations for estimating the N (NH_3, N_2O, NO_x and N_2) emissions was done in such a way that it includes all losses, including those from animal housing and manure storage systems and from the application of animal manure, fertilisers and dung and urine from grazing animals to the soil. The approach implicitly assumes that manure applied to the soil in a given grid cell (external data) comes from the farms in the same grid cell.

2.2. Required inputs of nitrogen and related N losses.

2.2.1. Assessment of yield gaps.

The global yield gap atlas (GYGA[1]) provides estimates of yield potentials for a variety of staple crops and countries. Estimates of yield potentials are obtained from crop models for specific weather stations that are scaled up to the country level by using zones of similar climate (Grassini *et al.*, 2015; Van Bussel *et al.*, 2015). These bottom-up estimates are more accurate than estimates of yield potentials that use global datasets on weather, soil and crop management (Van Ittersum *et al.*, 2013); however, they are currently only available for selected countries and crops. For Europe, GYGA currently provides yield gap estimates for wheat for all EU-27 countries except Cyprus and Malta, for barley for all countries except Cyprus, Malta, Italy and Portugal, and for maize for several important maize-growing countries.

In order to obtain estimates for yield potentials for other crops than wheat (for which data is available from GYGA for almost all EU-27 countries), we used a scaling approach. In this approach, we multiplied actual yields per climate zone with yield gaps per climate zone for wheat and then multiplied this with the ratio of the maximum country-level yield for this crop in Europe and the actual crop yield in the respective climate zone (for details, see De Vries *et al.*, 2020).

2.2.2. Calculation of required N inputs and related N losses.

In order to calculate N inputs that are needed to achieve target yields (so called 'required N inputs'), we multiplied actual N inputs with the ratio of target yield and the actual yield. We assumed that all required additional N is mineral N fertiliser and all other inputs stay constant. N uptake and N losses at required N inputs were derived by scaling actual uptake and losses by the ratio of target yield and the actual yield. This calculation of required N inputs and associated losses implicitly assumes that the NUE of the required N inputs is the same as the NUE of the actual N inputs. On the one hand, this may represent an overestimation of required N inputs, since all additional inputs are from fertiliser and fertiliser N inputs have a higher NUE than N inputs from other sources (mainly manure). On the other hand, the yield response to higher N inputs is lower at high yields (law of diminishing returns), which implies a decrease in NUE at higher N inputs, which compensates for this underestimation of the NUE of added N fertiliser.

2.3. Critical losses and inputs of nitrogen.

Critical N inputs in view of adverse environmental impacts were derived in three consecutive steps, i.e.: (i) Identification of critical values for defined N indicators in air and water, (ii) Assessment (back-calculation) of critical N

[1] www.yieldgap.org

losses to air and water that correspond to the critical values of the identified N indicators and (iii) Assessment (back-calculation) of critical N inputs and related N fertilisation and N excretion rates from critical N losses. Critical N inputs refer to the sources that can be managed by the farmer, including manure, fertiliser and biological N fixation (the latter being determined by the sown crops), whereas the defined N indicators are critical ammonia emissions determined by critical N deposition on natural ecosystems, nitrate (NO_3) concentration in leachate to ground water and nitrogen concentration in runoff to surface water (Figure 4).

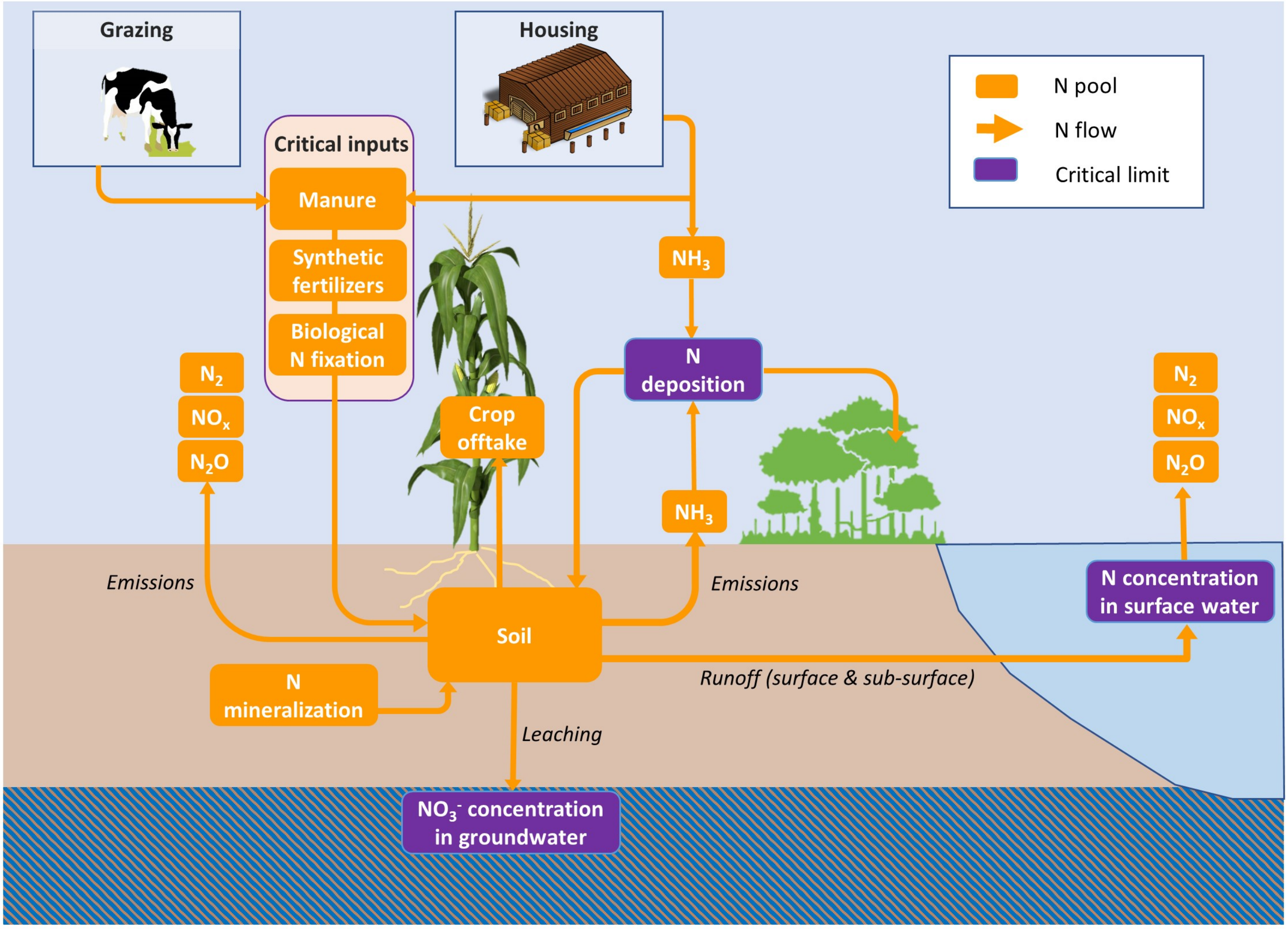

Figure 4*: Simplified scheme of the N flows and N indicators considered in the calculations of critical nitrogen inputs. Nitrogen indicators for which critical limits are set are shown as purple boxes.*

The identification of critical values, the back-calculation approaches and the input date that were used to perform the calculations, are discussed below.

2.3.1. Critical values and critical losses for nitrogen indicators.

Figure 5 gives a summary of the used nitrogen indicators for which critical values / thresholds were assessed and their use in assessing critical N losses and, ultimately, critical N inputs.

2.3.1.1. Critical ammonia emissions in view of impacts on biodiversity.

As an N-indicator in relation to eutrophication, and related biodiversity (especially plant species diversity) decline, of terrestrial ecosystems, we used the so-called critical N loads, being critical atmospheric N deposition levels for terrestrial ecosystem functioning/health. More specifically, area-weighted mean critical N loads based on NUTS3-level resolution were used as a basis to assess critical NH_3 emission rates.

In assessing the critical NH_3 emissions, we took account of differences in the fraction of agricultural land in a region, as atmospheric NH_3 emissions are diluted by emissions from non-agricultural land in the area. More details on the derivation of the critical N loads and the assumptions that were used to assess critical levels of NH_3–N emission are given in De Vries *et al.* (2020).

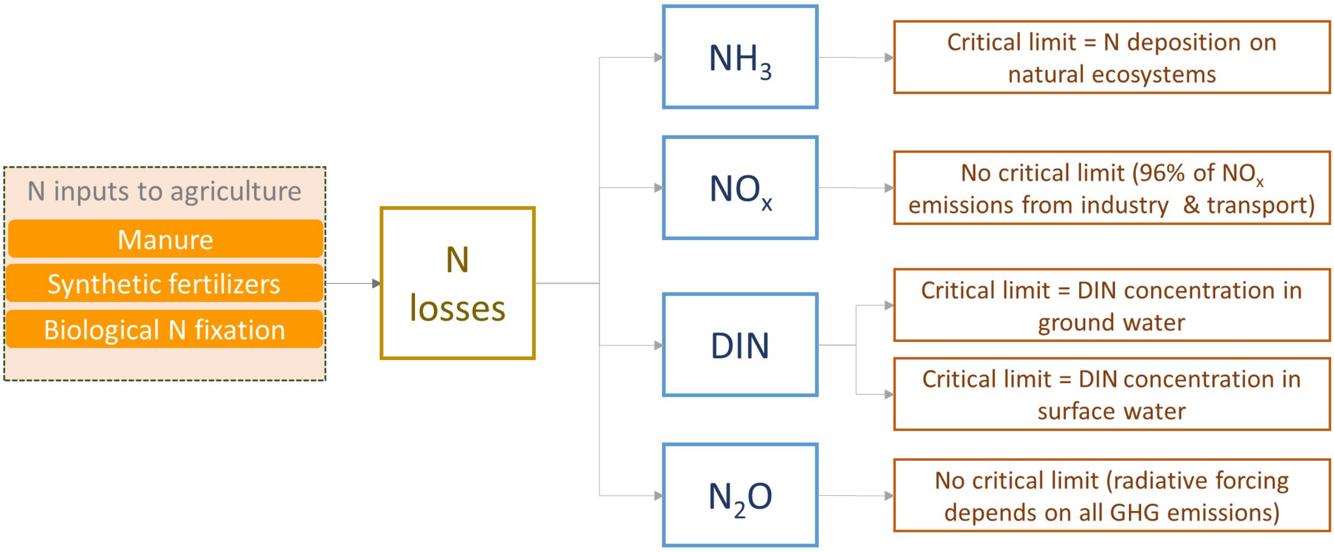

Figure 5: *Schematic overview of the nitrogen indicators used with critical values (thresholds) and their relationship with calculated N losses.*

2.3.1.2. Critical nitrogen runoff in relation to impacts on surface water quality.

As an N-indicator for the eutrophication of aquatic ecosystems, we identified critical concentrations of dissolved total N in surface water in the range of 1.0-2.5 mg N l^{-1}. This range is based on (i) an extensive study on the ecological and toxicological effects of inorganic N pollution (Camargo and Alonso, 2006), (ii) an overview of maximum allowable N concentrations in surface waters in national surface water quality standards (Liu *et al.*, 2011) and (iii) different European objectives for N compounds (Laane, 2005). For this study, we used the upper limit of 2.5 mg N l^{-1}.

Runoff to surface water comes both from agricultural and non-agricultural land. Runoff from non-agricultural land usually has a lower N concentration and thus dilutes the N concentration in runoff from agricultural land. As with the critical NH_3 emissions, we thus account for differences in the fraction of agricultural land in a region, as described in detail in De Vries *et al.* (2020). We used an average value of 0.5 mg N l^{-1} for the N concentration in runoff from non-agricultural land. A value of 0.5 mg N l^{-1} for streams/catchments in forests has been suggested by Gundersen *et al.* (2006) as the level at which a forest ecosystem can be considered 'leaky'. Critical N runoff rates from agriculture were calculated by multiplying the precipitation surplus with the critical N concentration, assuming that the precipitation surplus is constant (i.e. no impacts of climate change).

2.3.1.3. Nitrogen leaching to ground water in relation to health impacts.

The critical NO_3- concentration in groundwater was set to the WHO drinking water limit of 50 mg NO_3 l^{-1} or 11.3 mg NO_3-N l^{-1}. This limit is based on epidemiological evidence for methemoglobinemia in infants (WHO, 2011). As with runoff, critical N leaching rates from agriculture were calculated by multiplying the precipitation surplus with a critical N concentration, using 11.3 mg NO_3-N l^{-1}.

Note that 11.3 mg NO_3-N l^{-1} is higher than the limit for N in surface water, but this does not necessarily mean that drinking water quality is protected when using the surface water quality criterion, since dilution by runoff from non-agricultural land may allow high NO_3-N concentrations from agricultural land in areas with a low share of agriculture. Using a critical N concentration in surface water of 2.5 mg N l^{-1} and in runoff from non- agricultural land of 0.5 mg N l^{-1}, this dilution leads to allowable values near 11.3 mg N l^{-1} in areas where agriculture is only 20% of the land fraction.

2.3.2. Model approach to back-calculate critical nitrogen inputs.

2.3.2.1. Overall approach.

Total critical N inputs, being the sum of the N inputs from fertiliser, animal manure, biosolids, biological fixation, deposition and mineralisation, were calculated as the sum of N uptake, critical N emissions to air (NH_3, N_2O, NO_x and N_2 emissions) and critical N losses to water (leaching and runoff), using criteria related to either NH_3 emissions N leaching or N runoff. Based on a critical limit for either NH_3 emission, $NH_{3em(crit)}$, N concentration in runoff to surface water (determining $N_{sw(crit)}$) or NO_3 in leaching to groundwater (determining $N_{le(crit)}$), the critical N input is calculated by a back-calculation approach. The back-calculation approach is based on a slightly simplified version of the forward calculations in INTEGRATOR. Figure 6 illustrates the linkage between N inputs, N offtake and N losses. The grey box shows total N inputs, the blue circles show total N outputs. All N flows are expressed in kg N ha^{-1} yr^{-1}. Total N inputs are equal to total N outputs.

In the back-calculations we combined N fixation and N fertiliser, and also combined N biosolids and N excretion. Inputs from N deposition are assumed to be a function NH_3 emissions, as indicated by the dashed arrow in Figure 5, and inputs from N mineralisation (only on peat soils) are considered constant. We further assumed that (i) the N offtake fraction (frN_{off}), calculated as N offtake divided by total inputs minus N emissions and N surface runoff, is constant and equal to the 2010 value, and that the relative contribution of fertiliser plus fixation from total farmer-managed inputs (i.e., the sum of fertiliser, fixation, manure and biosolids) is constant and equal to the 2010 value. Details on the calculations of critical N inputs related to critical NH_3 emissions, critical N runoff to surface water and critical N leaching to groundwater are given in De Vries *et al.* (2020).

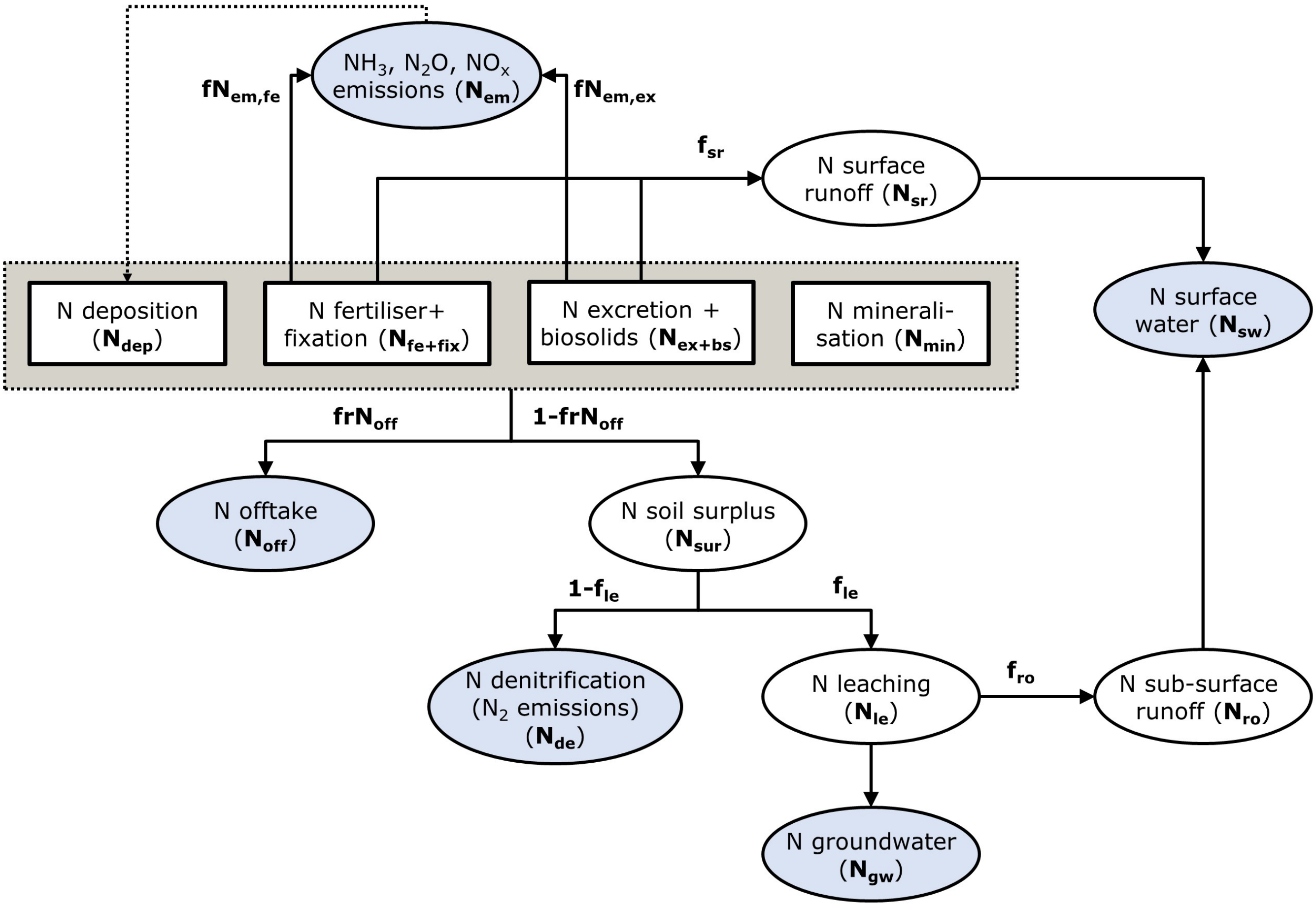

Figure 6: *Illustration of approach used for back-calculating critical N inputs. The schema shows links between N inputs (fertiliser, biological fixation, excreted manure, biosolids, deposition and mineralisation; grey box with dashed outline). and N outputs (N offtake, N losses to air due to N emissions (NH_3, N_2O and NO_x) and denitrification (N_2 emissions) and N losses to groundwater and surface water due to N leaching and N runoff; blue circles).*

Input data for the calculation of actual, required and critical N inputs include (i) areas, actual yields and target yields of cropland and grassland, (ii) manure allocation data and manure availability fractions, (iii) N uptake and N loss fractions, (iv) water fluxes and (v) soil data. Information on the various datasets and model parameters used is given in De Vries *et al.* (2020).

2.4. Necessary ammonia emission fractions and nitrogen use efficiencies.

The calculation of critical and required N inputs, described above, were all based on the assumption that the nitrogen use efficiency (NUE, defined as the ratio of N taken up by crops and the total N inputs, in line with the EU N expert panel[2] as well as NH_3 emission fractions from fertiliser and manure are equal to the actual (year 2010) values. If critical N inputs are below actual N inputs, this implies that environmental objectives can only be reached at a lower N input, which would be likely to cause a loss in crop production, unless: (i) the NH_3 emission fractions are reduced (this allows a higher N input in the context of NH_3 emissions and related N loads to ecosystems) or (ii) the nitrogen use efficiency (NUE) is increased (this allows a higher N input in relation to water quality, since a lower fraction of the N input is lost to ground water and surface water). This holds even more strongly when target yields above actual yields are aimed for. We thus calculated necessary reductions in NH_3 emission reduction fractions and necessary increases in NUE to reconcile food production and environmental N losses, as described below.

2.4.1. Calculation of necessary (reductions in) ammonia emission fractions.

Necessary NH_3 emission fractions, $fNH_3em,ex(nec)$, are defined as emission fractions at which we can achieve actual offtake (or target offtake) without exceeding critical NH_3 emission. We calculated those fractions as the ratio of the critical NH_3 emissions and the total actual N inputs (year 2010) and required N inputs by fertilisers, excretion (in housing systems and by grazing animals), biological fixation and biosolids for the year 2010.

We also included the necessary NH_3 emission fraction for N excretion (including emissions from housing systems, manure application and grazing), assuming constant NH_3 emission fractions for fertiliser. We assumed a minimum NH_3 emission fraction for excretion of 0.05. The average necessary NH_3 emission fraction for N excretion was thus calculated by excluding all NCUs where this fraction was smaller than 0.05. Finally, we calculated the necessary NH_3 emission fraction for N excretion, assuming that all fertiliser is replaced by nitrate fertilisers with an emission fraction of 2%. For all NCUs where the necessary emission fraction was larger than the actual emission fraction (those NCUs where critical NH_3 emissions exceed actual NH_3 emissions), we set the necessary NH_3 emission fraction to the actual NH_3 emission fraction before calculating the average necessary emission fraction. Details of the calculations are given in De Vries *et al.* (2020).

2.4.2. Calculation of necessary (increases in) nitrogen use efficiencies.

Increasing the NUE enables the reduction of losses to the environment. If the NUE is increased, the actual or target crop yields can thus be reached at lower N inputs, due to an enhanced N offtake fraction, while simultaneously the critical N input increases, since a lower fraction of N is lost to the environment (see Figure 7).

For all cases where actual N runoff to surface water and/or actual N leaching to groundwater exceeded critical N values, we calculated the necessary NUE at which actual yields or target yields are attained without exceeding

[2] http://www.eunep.com/wp-content/uploads/2017/03/N-ExpertPanel-NUE-Session-1.pdf

environmental limits. The necessary NUE was calculated using an iterative approach as described in De Vries *et al.* (2020). We calculated necessary NUEs both for actual and for target yields. We considered a maximum plausible NUE that farmers can achieve to be 0.9. NCUs where the necessary NUE was larger than 0.9 were excluded when calculating the mean necessary NUE at country-level or the EU-27 level. For all NCUs where the necessary NUE was lower than the actual NUE, we set the necessary NUE to the actual NUE before calculating the average necessary NUE. We also calculated the average necessary NUE by including all plots, but setting the maximum value for necessary NUE to 0.9.

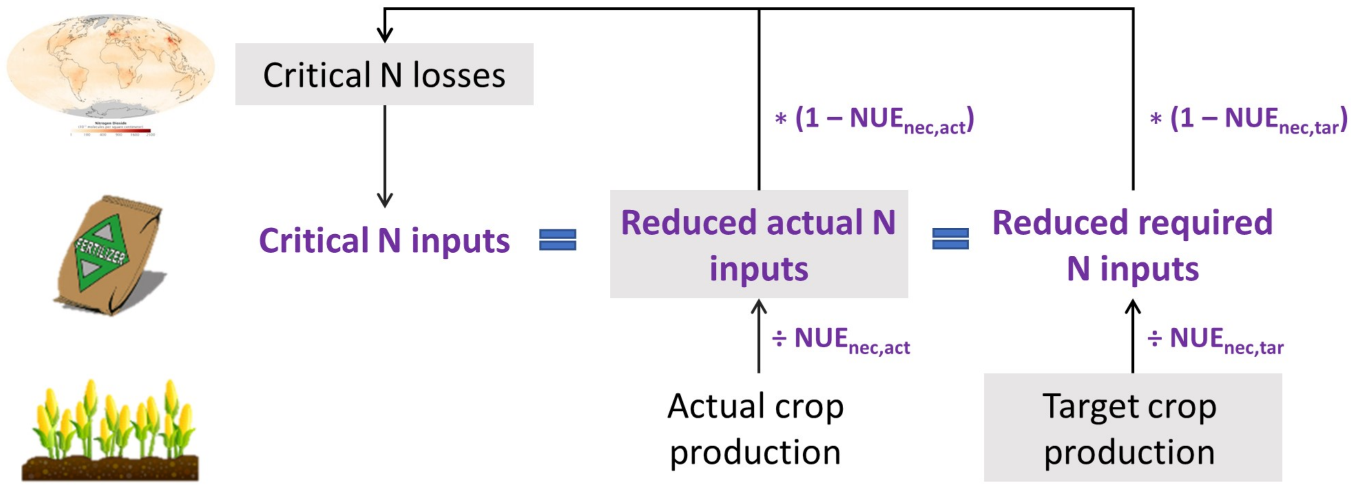

Figure 7: *Illustration of required NUE changes to reconcile crop production and environmental targets.*

3. RESULTS: IDENTIFYING PATHWAYS FOR RECONCILING NITROGEN INPUTS AND AGRICULTURAL PRODUCTIVITY.

3.1. Required, actual and critical nitrogen inputs and budgets at the EU-27 level.

The results show that N inputs that are required to obtain target crop yield are on average 27% higher than actual N inputs (185 vs 145 kg N ha^{-1} yr^{-1}; Figure 8). This implies that there is still an average yield gap of 27% between the actual crop yield and the target yield, which in turn is 80% of the potential yield. The difference between required and actual N inputs is higher for arable land (33%) than for grassland and fodder (21%), which is mainly because we assumed that target crop yields will not be 'targeted' for extensive grasslands but only for intensive grasslands. Average critical N inputs at the EU-27 level are approximately 43% lower than actual N inputs in relation to critical N concentrations in surface water, and c.31% lower than actual N inputs in relation to critical NH_3 emissions (100 and 83 vs 145 kg N ha^{-1} yr^{-1}; Figure 8). Critical N inputs for groundwater quality, however, are on average nearly equal to actual N inputs (Figure 8). The results imply that a greater than 30% reduction in N input by fertiliser and manure would be needed to fully protect air and water quality at current values for NH_3 emission fractions and N use efficiencies.

On arable land, critical N inputs in relation to surface water quality are lower than those in relation to critical NH_3 emissions, whereas the reverse is true for grassland (not shown in Figure 8). This is to be expected in view of the higher NH_3 emission fractions of manure, which is mostly applied to grassland, as compared to fertilisers, which are mostly applied to arable land.

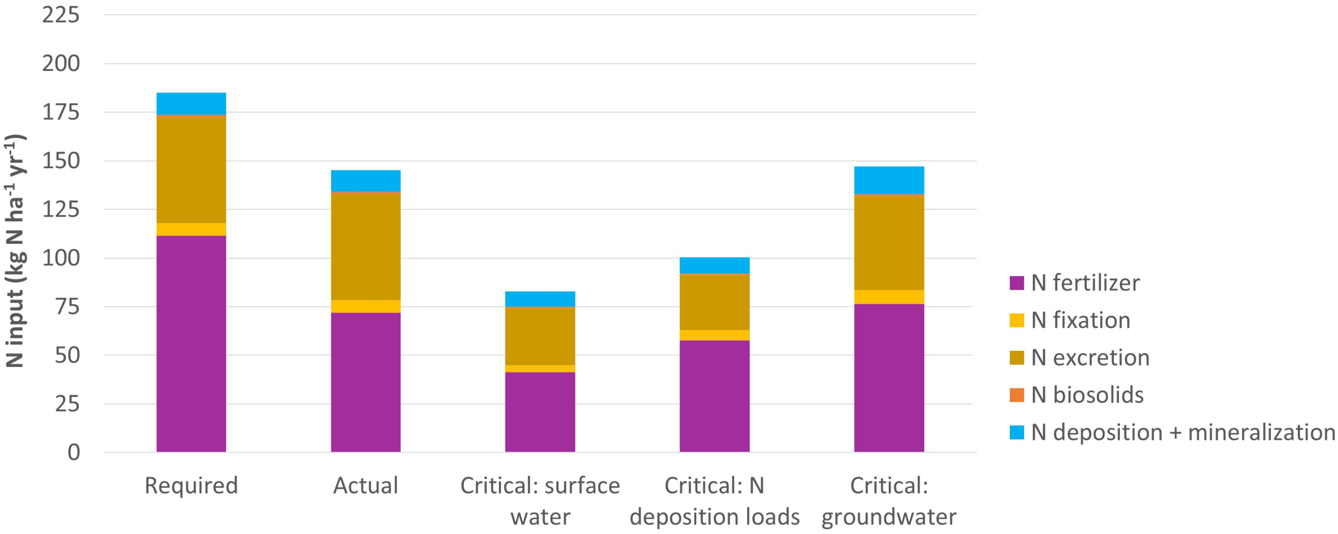

Figure 8: *Average required, actual and critical and nitrogen inputs for arable land, grassland (including fodder) and all agricultural land for the EU-27.*

Details of the annual required, actual (2010) and critical N budgets for all agricultural land in the EU-27, including the contribution of the various outputs to the critical N inputs for different criteria, are given in Table 1. The required, actual and critical N inputs are those presented in Figure 1. The table shows that the offtake, including the net removal of crops or grass from arable land or grassland, as a percentage of total N input is quite comparable at required, actual and critical N inputs, varying between 62-68%, increasing in the direction of required, actual and critical N budgets. In accordance, the N surplus as a percentage of total N input is also comparable, varying between 32-38%. Compared to the global average NUE, near to 45% (Bouwman *et al.*, 2013), the EU-27 values are relatively high.

The use of a critical NH_3 emission implies an N emission that is approximately 36% lower than the actual (year 2010) N emission (12.2 vs 19.2 kg N ha^{-1} yr^{-1}; see Table 1) and a critical N surplus that is approximately 35% lower (34.7 vs 53.0 kg N ha^{-1} yr^{-1}). The use of a critical N concentration in runoff of 2.5 mg N l^{-1} causes a European average critical runoff that is nearly 50% lower than the actual runoff (4.0 vs 7.8 kg N ha^{-1} yr^{-1}), while the related 'critical' denitrification is 57% lower than the actual denitrification (7.5 vs 17.3 kg N ha^{-1} yr^{-1}). This is possible because of spatial differences in the denitrification fraction, which is related to soil type, land use and precipitation. The critical N surplus is also about 50% lower than the actual surplus (26.4 vs 53.0 kg N ha^{-1} yr^{-1}; see Table 1). The European average critical N leaching rate, based on a NO_3-N concentration of 11.3 mg N l^{-1} is approximately 20% lower than the average actual N leaching (7.0 vs 8.7 kg N ha^{-1} yr^{-1}; see Table 1) but the N crop uptake is higher and the N surpluses are comparable, due to a higher N emission, illustrating that N inputs that are acceptable for groundwater will not only exceed the critical N runoff but also critical NH_3 emissions.

Table 1: *Average annual required, actual (2010) and critical N budgets for all agricultural land in the EU-27 for different criteria as calculated by INTEGRATOR.*

Source	**N budget EU-27 (kg N ha^{-1} yr^{-1})**				
	Required	Actual	At critical NH_3 emission	At critical N runoff to surface water	At critical N leaching to groundwater
Input to land					
Fertiliser +fixation	118.0	78.3	62.9	45.0	83.4
Excretion+ biosolids	55.7	55.7	29.0	30.0	49.4
N deposition	10.5	10.5	7.6	7.0	13.5
N mineralisation	0.8[1]	0.8[1]	0.8[1]	0.8[1]	0.8[1]
Total input	185.0	145.3	100.3	82.7	147.1
Output from land					
Crop offtake	116.6	92.3	65.6	56.3	97.2
Total surplus	68.4	53.0	34.7	26.4	49.9
- N emission (NH_3, N_2O, NO_x)	24.3	19.2	12.2	11.8	20.2
- Denitrification	22.6	17.3	11.3	7.5	15.1
- Runoff to surface water	10.2	7.8	11.2[2]	4.0	7.7
- Leaching to groundwater	11.3	8.7	-	3.2	7.0
Total output	**185.0**	**145.3**	**100.3**	**82.7**	**147.1**

[1] Net N mineralisation is only calculated for (drained) peat soils as we assumed no change in soil N pool (neither mineralisation nor accumulation) for mineral soils. The results, however, only refer to part of the peat soils, since approximately 2,500 crop-NCU combinations on peat soils were excluded because critical N inputs from fertiliser and excretion were negative because mineralisation alone leads to the exceedance of critical limits for runoff and/or leaching. This does not significantly affect results for critical N inputs, but does lower average mineralisation rates from 4.9 kg N ha^{-1} yr^{-1} to 0.8 kg N ha^{-1} yr^{-1}.

[2]. For the N budget at critical NH_3 emissions, runoff and leaching are not provided separately – the value presents the sum of N runoff + N leaching.

Today, in the EU, the risk of adverse environmental impacts of N inputs is thus highest for surface water quality, followed by biodiversity impacts due to air quality and then groundwater quality. This is also illustrated in Table 2 showing that the share of the agricultural area where actual N inputs exceed critical N inputs is highest for N runoff to surface water (76%), followed by NH_3 emissions to air (69%) and finally N leaching to groundwater (25%).

Note that N leaching to groundwater is still exceeded on about a quarter of the agricultural area, even though the average critical N input exceeds the actual N input. The percentage of the area where actual N inputs exceed critical N inputs in view of critical NH_3 emissions is higher for grassland and fodder than for arable land, reflecting the higher manure inputs to those lands. The reverse is true for percentage of the area where actual N inputs exceed critical N inputs in view of surface water and ground water quality, due to the higher N leaching and N runoff fractions from arable land as compared to grassland.

Table 2: *Percentage of area where actual N inputs exceed critical N inputs in relation to critical NH_3 emissions, critical N runoff to surface water and critical N leaching to groundwater.*

N flux	Arable	Fodder	Grass	Fodder + grass	Total
NH_3-N emissions to air	66%	76%	73%	74%	69%
N runoff to surface water	90%	67%	46%	54%	76%
NO_3-N leaching to groundwater	29%	21%	17%	18%	25%

3.2. Geographic variation within the EU in required, actual and critical nitrogen inputs.

3.2.1. Geographic variation in required and actual N inputs and their difference.

The geographic variation in actual N inputs, required N inputs to obtain target yields and the difference between required and actual N inputs is given in Figure 9. Results show high total N inputs in the Netherlands, Belgium and Luxembourg and the Po valley in Italy and to a lesser extent in Ireland and western UK, Denmark, Germany, Brittany in France (Figure 9 left), regions with well-known intensive livestock production. Required inputs are quite comparable with actual N inputs except for Portugal, parts of Spain, Poland and other scattered regions over Europe (Figure 9 middle), illustrating that there are clear yield gaps in those regions (see also Figure 9 right).

3.2.2. Geographic variation in actual and critical N inputs and their difference.

The geographic variation in actual N inputs, critical N inputs and the difference between actual and critical N inputs is presented in relation to the protection of terrestrial biodiversity from enhanced NH_3 emissions (Figure 10) and of surface water quality (Figure 11), which are mainly limiting the nitrogen inputs. Critical N inputs are only partly correlated with the actual N inputs (Compare Figure 10 and 11 middle and left). Critical N inputs are mainly determined by variations in critical N deposition levels (NH_3-N emissions), the fraction of the agricultural area in a region ((NH_3-N emissions and N runoff), precipitation surplus (N runoff and N leaching) and soil type,

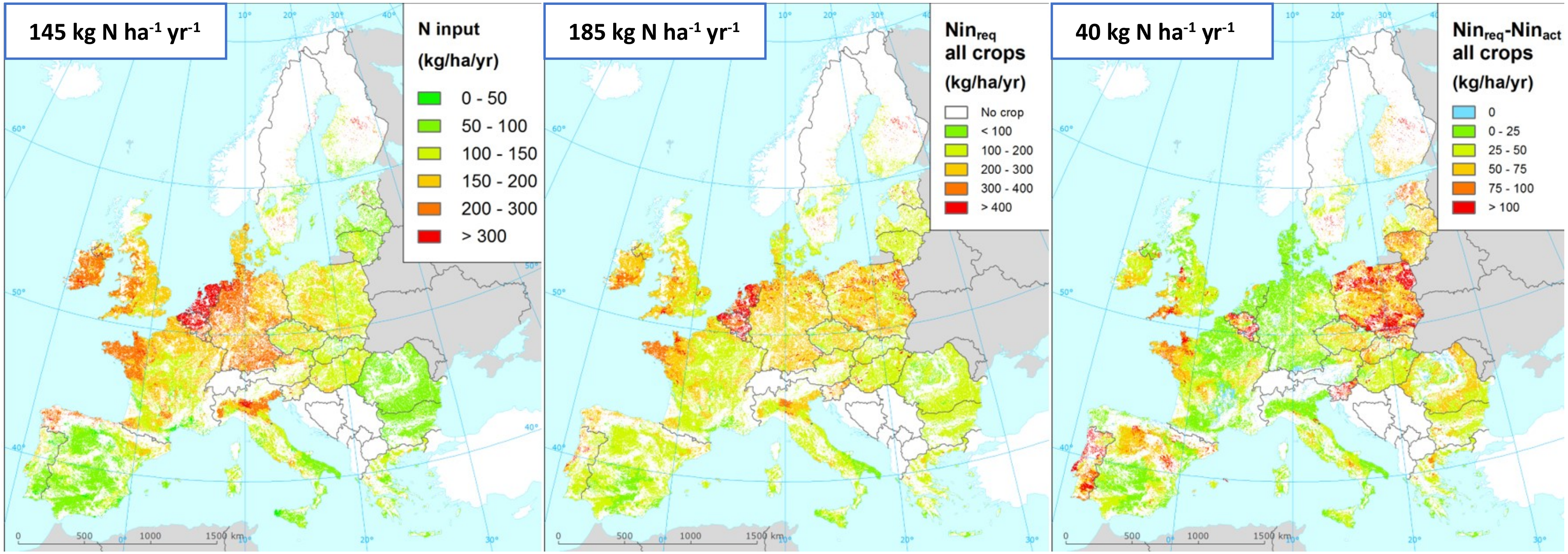

Figure 9: *Geographic variation in actual N inputs (left), required N inputs (middle) and the difference between required and actual N inputs (right).*

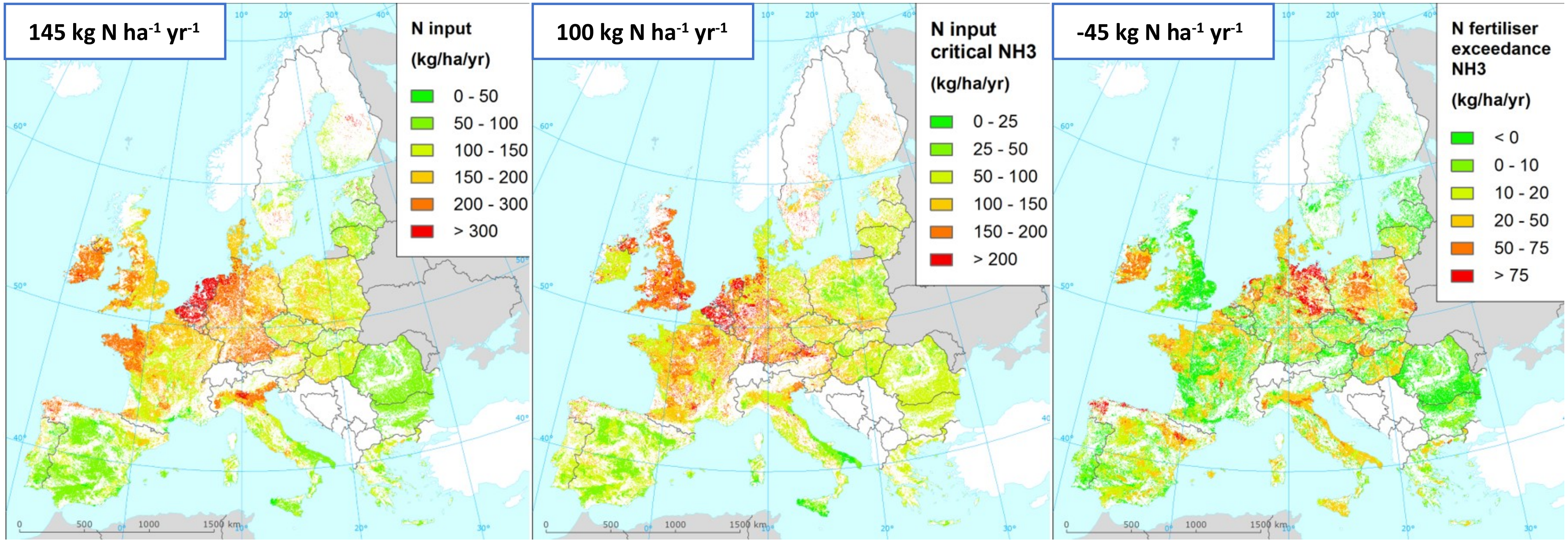

Figure 10: *Geographic variation in actual N inputs (left), critical N inputs in view of the protection of terrestrial biodiversity from enhanced NH_3 emission (middle) and the difference between actual and critical N inputs (right).*

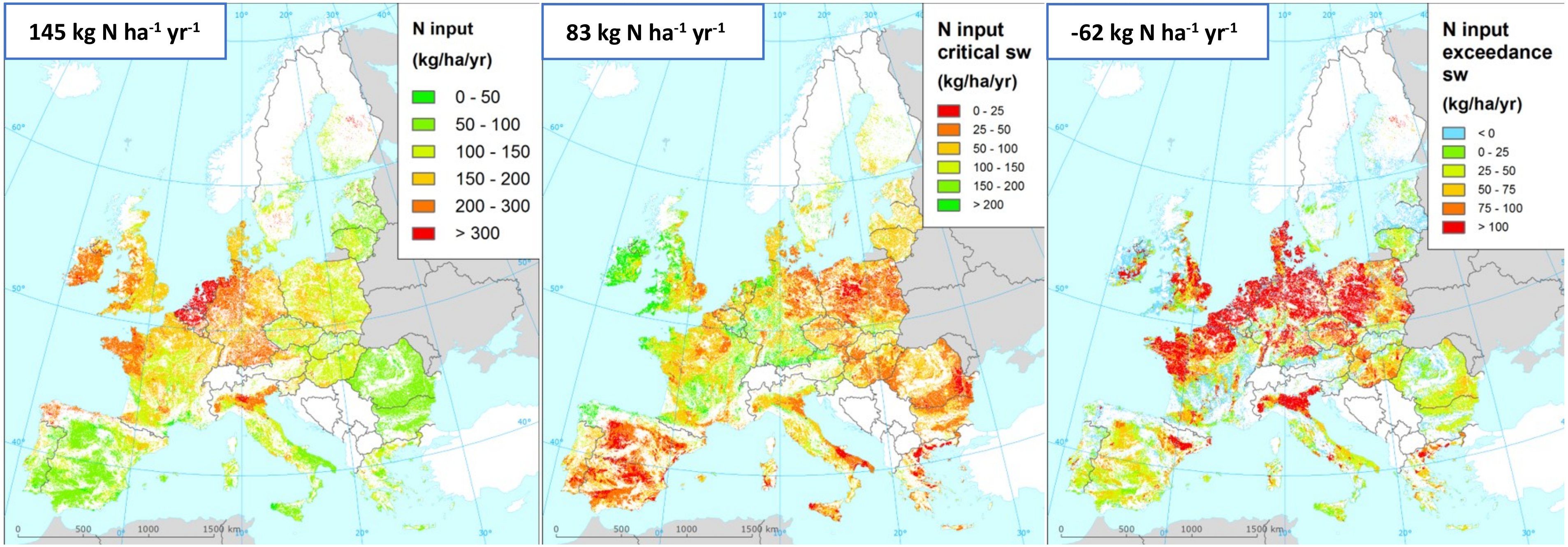

Figure 11: *Geographic variation in actual N inputs (left), critical N inputs in view of the protection of surface water quality (middle) and the difference between actual and critical N inputs (right).*

land use and slope (N runoff and N leaching). In general the largest exceedances occur in regions with the largest inputs.

With respect to NH_3-N emissions, exceedances are especially large in high density livestock regions with large N manure inputs, including Ireland and western UK (partly caused by intensive sheep grazing), the Netherlands, Belgium and Luxembourg, Northern Germany, Brittany in France and the Po valley in Italy (Figure 10 right).

With respect to N runoff to surface water, exceedances are especially large in Northern Europe, including the western parts of the UK, countries bordering the Baltic sea (Estonia, Latvia, Lithuania, Poland, Germany and Denmark), the Netherlands, Belgium, Luxembourg, North Eastern France and the Po valley in Italy (Figure 11 right). Despite the relative low N inputs in Poland, the exceedances of critical N inputs in relation to the protection of surface water quality are still high, which is due to low critical N inputs (Figure 11 middle) and the relative low N use efficiency in Poland (see also De Vries *et al.*, 2020).

3.3. Necessary ammonia emission fractions and nitrogen use efficiencies to reconcile food production and environment.

3.3.1. Results at the EU 27 level.

Figure 12 shows the share of agricultural land for different reduction targets for the NH_3 emission fraction for different NUE targets at actual or required N inputs. More specifically Figure 13 (left) shows the share where (i) no reduction in ammonia emission fractions are necessary (because actual or required N inputs are lower than critical N inputs in relation to air quality) and (ii) where reductions in ammonia emission fractions (NH_3 EF) are necessary to stay below critical NH_3 emissions with the necessary emission fractions being above 0.2; between 0.1 and 0.2, between 0.05 and 0.1, or below 0.05 (considered to be impossible).

At 67% of the land area, NH_3-N emission fractions for manure need to be reduced to a value below 0.2 to protect biodiversity at actual N inputs. However, at 25% of the land area, the necessary reductions in NH_3-N emission fractions are even below 0.05, this not being feasible. At required N inputs, to achieve target yields, these percentages increase to 83% and 43%, respectively. These values, however, refer to a situation in which the NH_3-N emissions from fertiliser remain the same. When the NH_3-N emissions from all fertilisers equal 2%, the land area at which NH_3-N emission fractions for manure need to be reduced to a value below 0.2 are 50% and 68% at actual yields and target yields, respectively. The land area in which the necessary reductions in NH_3-N emission fractions are not feasible (below 0.05) then reduce to c.10% and 20%, respectively.

The necessary NUEs to attain the actual crop yield at critical N runoff to surface water cannot be achieved on 17% of all agricultural land and on 25% at target crop yield (see Figure 12 right). This is based on the assumption that an NUE of 0.9 is the maximum plausible NUE that farmers can achieve. On

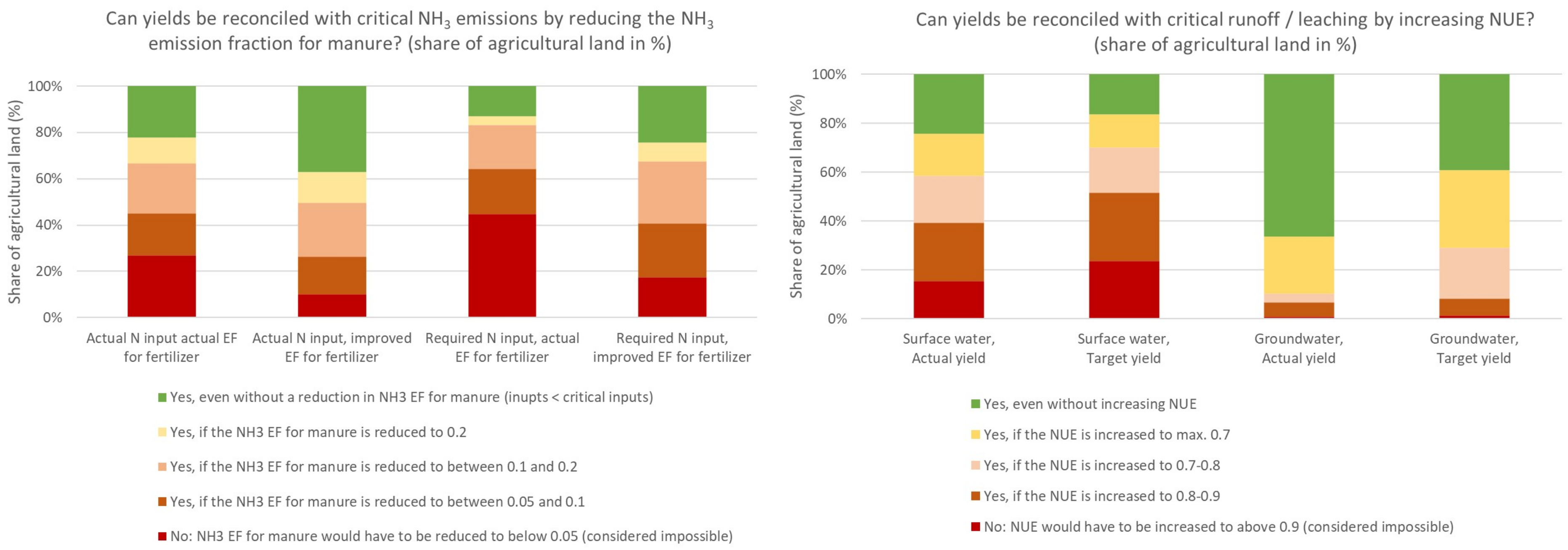

Figure 12: *Share of agricultural land for different reduction targets for the NH_3 emission fraction for manure at actual or required N inputs (actual yields and target yields) and assuming actual or improved EF for fertiliser (left) and for different NUE targets, for actual yields and target yields and for critical N runoff to surface water and critical N leaching to groundwater (right).*

the remaining land, the average NUE has to increase from 0.61 to 0.72 for actual N inputs (actual yields) and from 61% to 74% for required N inputs (target yields). This increase in NUE implies an automatic reduction in N inputs from organic and mineral sources of between 15-20%.

For groundwater, on the other hand, it is possible to stay below critical N leaching by increasing the NUE to a maximum of 90% on 98% of all agricultural land (both at actual and required N inputs). For actual N inputs, the average NUE has to increase from 62% to 64% and the necessary NUE is lower than 70% on 90% of agricultural land. For required N inputs, the average NUE has to increase from 62% to 67%, and the necessary NUE is lower than 70% on c.70% of the land (see Figure 12 right)

3.3.2. Geographic variation within the EU.

Maps of the geographic variation in actual NH_3 emission fractions, necessary NH_3 emission fractions to protect biodiversity and the exceedance between required and actual NH_3 emission fractions at actual (year 2010) N inputs show that a significant reduction in NH_3-N emission fractions is needed in the high density livestock regions where large exceedances in NH_3-N emissions occur, including Ireland and western UK (partly caused by intensive sheep grazing), the Netherlands, Belgium and Luxembourg, Brittany in France and the Po Valley in Italy (Figure 13). Exceedances, however, also occur in (large) parts of Germany, Poland, Spain, Greece and Czech Republic, mostly due to (very) low critical NH_3 emissions (Compare Figure 10 right and Figure 13 right). Actual and necessary NH_3-N emission fractions averaged for all agricultural land, excluding the NCUs where necessary NH_3-N emission fractions are less than 0.05, are 0.25 and 0.16, respectively, implying a necessary average reduction of 64%.

Maps of the geographic variation in actual NUE, necessary NUE to protect surface water quality and the exceedance between necessary and actual NUE at actual N inputs show that a significant increase in NUE is necessary in the area where large exceedances in critical N runoff to surface waters occur (Compare Figure 10 and Figure 14 right). This includes Eastern UK, north Germany, Denmark, the Netherlands, Belgium, Luxembourg, North Western France, the Po valley in Italy, Poland, and parts of Spain and Italy, including with high density livestock regions. Actual and necessary nitrogen use efficiencies for all agricultural land, averaged over the EU-27 are 61% and 76%, respectively, implying a necessary average increase of 18%.

4. DISCUSSION AND CONCLUSIONS.

4.1. Limitations of the approach.

4.1.1. Critical limits used.

Critical ammonia emission rates: The critical ammonia emission rates based on average critical N loads (deposition levels) at NUTS3 level. In a previous global study, De Vries *et al.* (2013) used uniform critical atmospheric NH_3

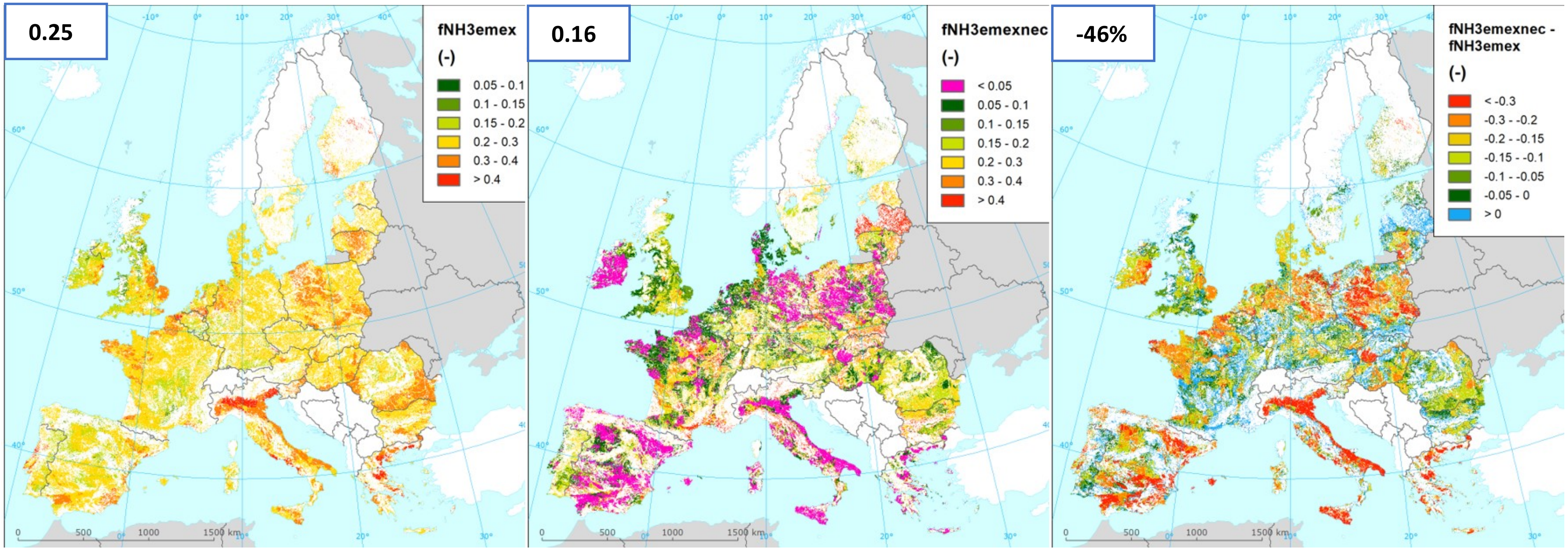

Figure 13: *Geographic variation in actual NH_3 emission fractions, required NH_3 emission fractions to protect biodiversity and the exceedance between required and actual NH_3 emission fractions at actual N inputs. Values in the maps indicate the average value for the EU-27.*

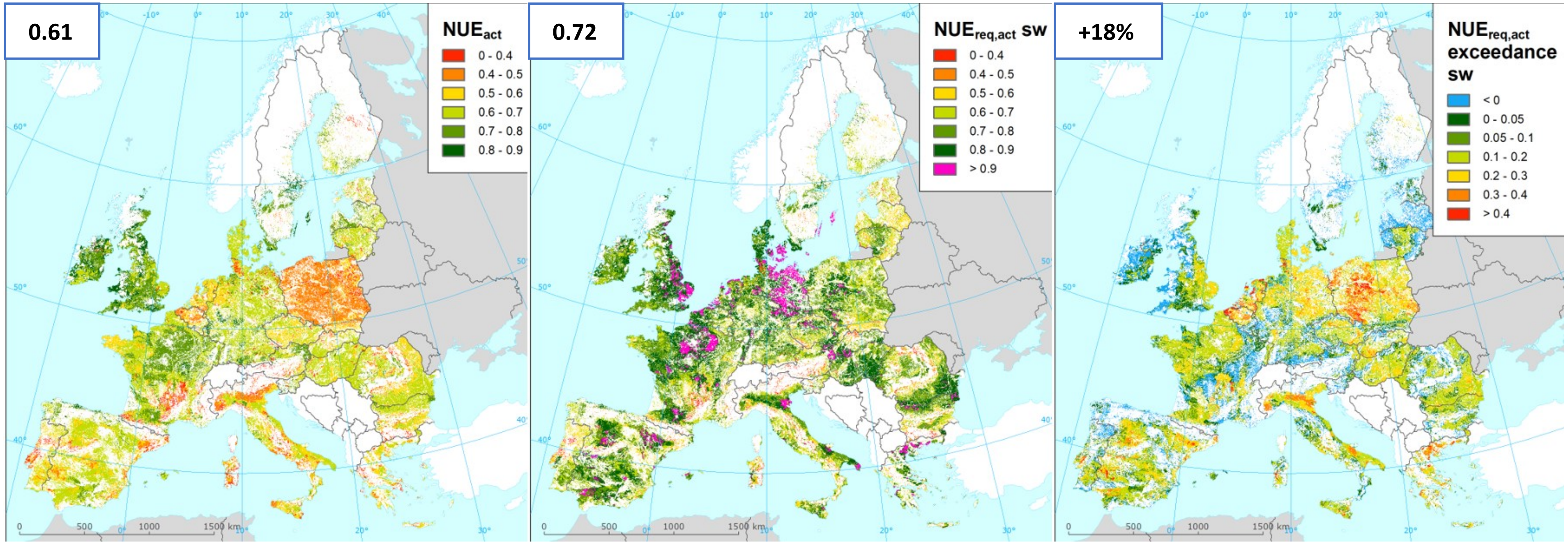

Figure 14: *Geographic variation in actual NUE, required NUE to protect surface water quality and the exceedance between required and actual NUE at actual N inputs. Values in the maps indicate the average value for the EU-27.*

concentrations to assess N impacts on biodiversity. However, this approach requires the use of an atmospheric dispersion model, and does not account for the diversity in local circumstances affecting the critical load of nitrogen on terrestrial ecosystems. Consequently, we based the critical NH_3 emissions on spatially explicit differences in critical N loads at the NUTS3 level, aggregated (averaged) from more detailed information at higher spatial resolution than NUTS3, as presented in the methods section. Although a NUTS region generally encompasses several hundreds of square kilometres, it should be realised that the transport distance of ammonia from a source can be several hundreds of kilometres. It has been estimated that approximately 30% of the emitted ammonia is deposited within a distance of 20 kilometre and this is still only c.80% within a distance of 250 kilometre[3]. There is thus a less direct connection between emission and deposition as assumed in this study.

Critical N concentrations in runoff and leachate: The approach that was taken to assess critical N inputs to agriculture was that the maximum critical N concentration in runoff from agriculture should not exceed 11.3 mg N l^{-1} (being the drinking water limit) and it should not exceed a critical N concentration in surface water of 2.5 mg N l^{-1}. Use of a limit value for runoff water from agriculture is, however, only a surrogate in terms of the surface water quality. Higher values can be acceptable due denitrification or N retention in surface water, while lower values may be needed because of a mixing of runoff water with point loads of N into surface water. A higher limit value in runoff water from agriculture could thus be acceptable due to denitrification in shallow groundwater, in riparian zones and in surface water, whereas a lower limit value for runoff water could be needed because of mixing of runoff water with point loads of N into surface water. This effect was assumed to compensate. The comparison with surface water quality measurements, however, indicated that our calculated N concentrations in runoff are generally higher than the concentrations in surface water.

N_2O emissions: N inputs also cause N_2O emissions, but since there are no clear limits for N_2O emissions, apart from a required reduction target, this aspect was not included in the assessment. Another argument for not including N_2O emissions is the fact that NH_3 emissions due to agricultural N inputs cause an enhanced CO_2 sequestration in response to elevated NH_3 deposition. This largely compensates for the global warming potential caused by N_2O emissions, such that the overall effect of N use in agriculture on greenhouse gas emissions Europe is near neutral (De Vries *et al.*, 2011a).

4.1.2. Low precipitation areas, chalk – karst regions and steep slopes.

The use of surface water quality criteria can lead to very low critical N surpluses in areas with a low precipitation surplus. This in turn will lead to a very low N uptake, especially in areas with a low NUE. In those regions, surface water quality can be affected at very low N inputs, as illustrated for e.g. Greece and Portugal. To avoid extremely low critical N inputs, we applied

[3] https://www.rivm.nl/stikstof

an arbitrary minimum value for the precipitation surplus of 25 mm/yr in the semi-arid regions in the Southern and Eastern part of the EU. This value remains arbitrary.

Furthermore, there are regions where there is no connection between precipitation surplus and recharge of aquifers or surface waters, such as chalk – karst regions in UK, France and central Europe.

One might also argue that the approach is less meaningful for regions where surface runoff and erosion are the main loss route for nutrients to surface water bodies, since these fluxes are more closely related to N application than to N surplus. Surface runoff and erosion are in general the main loss routes for P to surface water bodies, but for N this is generally not true unless the slopes are extremely steep, which is unlikely for arable land. Consequently, we have not excluded any land with the idea that direct surface runoff dominates the loss route to surface water.

4.2. Plausibility of the approach.

4.2.1. Plausibility of the calculated ammonia emissions.

A comparison of calculated country-level ammonia emissions compared to officially reported emission data for the year 2010 obtained from the EMEP database) shows reasonable to good comparisons for total agriculture and also for fertiliser application and manure application/grazing separately, while the NH_3 emissions from housing and manure storage show a consistent overestimation (Figure 15).

At the EU-27 level, emissions from total agriculture are slightly under-estimated (INTEGRATOR = 2,493 Gg N yr^{-1}, EMEP = 2,872 Gg N yr^{-1} -> INTEGRATOR 11% lower), emissions from fertiliser application are slightly over-estimated (INTEGRATOR = 530 Gg N yr^{-1}, EMEP = 459 Gg N yr^{-1} -> INTEGRATOR 16% higher) and emissions from manure application and grazing are nearly similar (INTEGRATOR = 868 Gg N yr^{-1}, EMEP = 878 Gg N yr^{-1} -> INTEGRATOR 1% lower). Manure storage and housing is however clearly lower at the EU-27 level (INTEGRATOR estimates 1,094 Gg N yr^{-1}, vs. 1,468 Gg N yr^{-1} for EMEP -> INTEGRATOR 25% lower). The lower total NH_3-N emissions might have caused an underestimation of the exceedance of critical NH_3-N emissions but the uncertainty in those critical emissions is very large and it is not clear whether this is the case in reality.

4.2.2. Plausibility of the calculated nitrate concentrations in groundwater.

A comparison of calculated NO_3 concentrations in leachate to ground water for the year 2010 with measured NO_3 concentrations in groundwater in the period 2008-2011 at country level (EC 2013), is given in Figure 16. The results show that the calculated ground water concentrations are comparable to measured concentrations at the EU-27 level, but concentrations are strongly overestimated in the Netherlands, Belgium and Poland and (slightly) underestimated in the UK, Germany, Romania, Slovakia, Slovenia and Spain. This may be due to the fact that we calculated leaching from the rootzone to ground water and not the ground water concentration. In addition, the location of the sampling sites is not given and the comparison thus refers to the percentage of sampling sites for the measurement as compared to the percentage of agricultural areas.

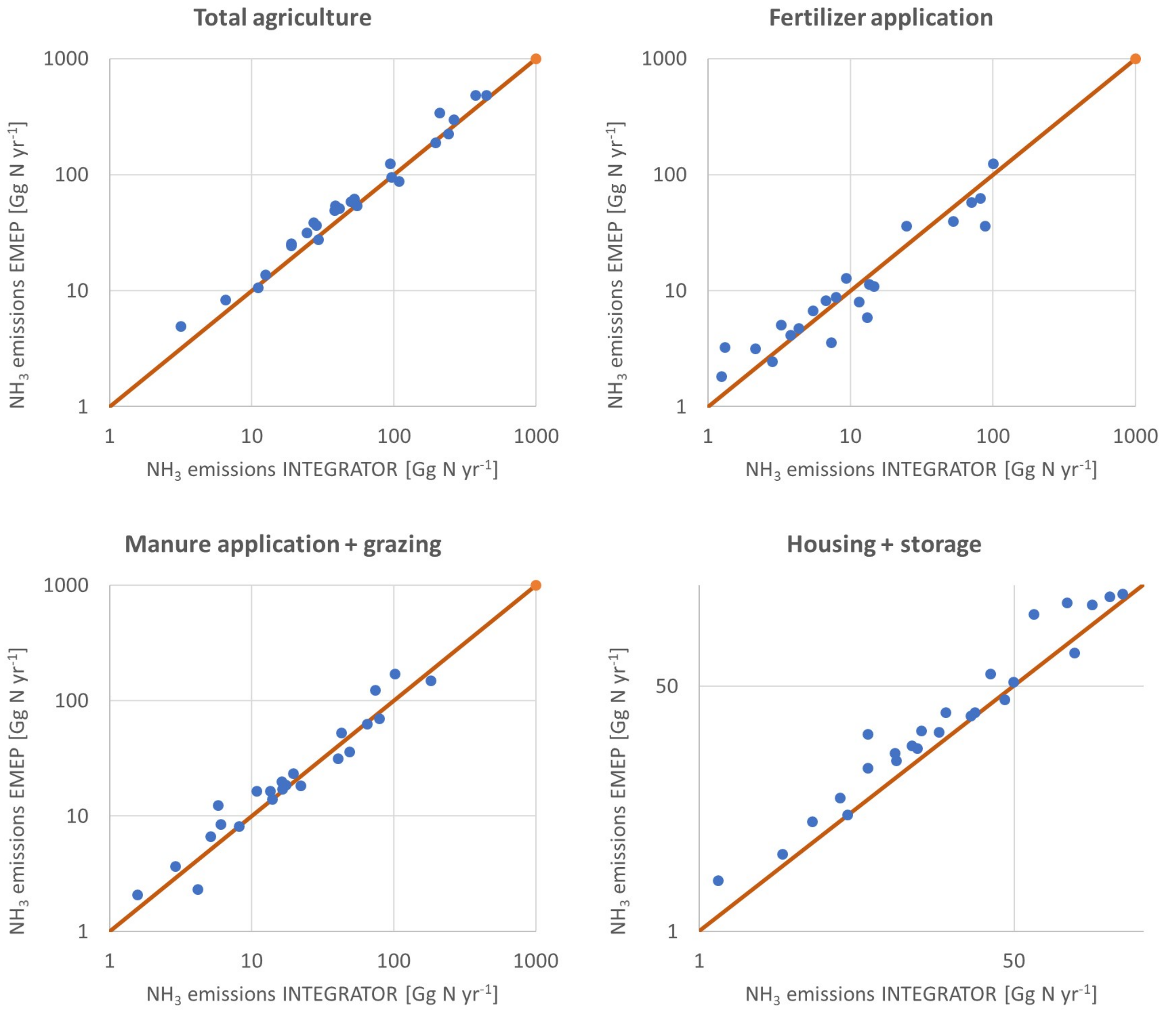

Figure 15: *Comparison of NH_3 emission estimates from EMEP (officially reported emission data level 2 for the year 2010 obtained from EMEP database) with NH_3 emissions from INTEGRATOR at country-level for (i) total agriculture (top left), fertiliser application (top right), manure application + grazing (bottom left) and housing and manure storage (bottom right).*

Measurements are also available of N concentrations in surface water, but these concentrations are not equal to N concentrations in runoff to surface water, calculated in our approach. It accounts for impacts of N removal in surface waters by denitrification and sedimentation, while it may also be

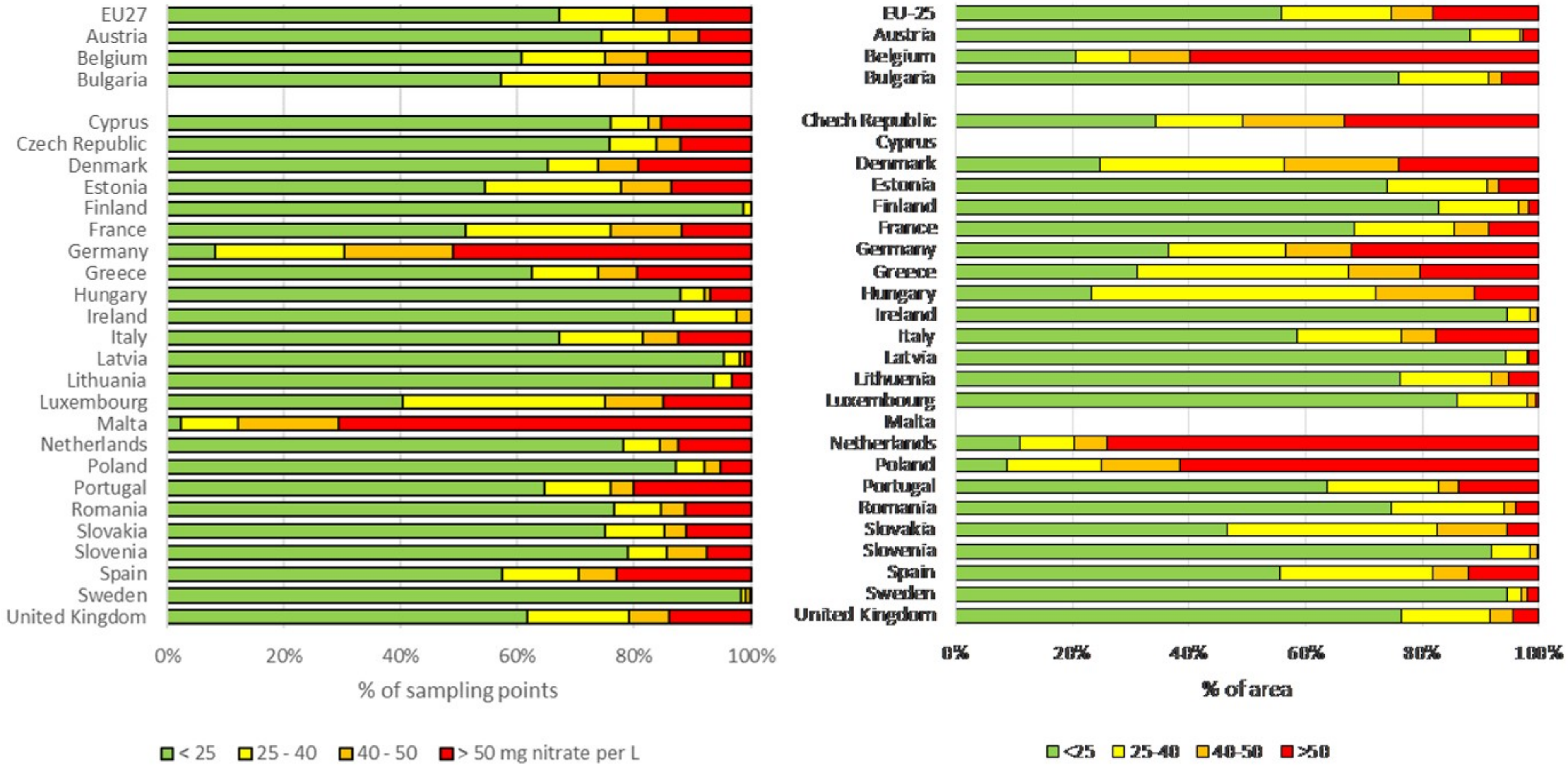

Figure 16: *Measured (left) and calculated (right) percentage of sampling sites (measurements) and areas (calculations) in a given NO_3 concentration range at country level in* EU-27 *countries.*

affected by waste inputs. The comparison showed that the calculated N concentrations in runoff to surface water are generally higher than the measured N concentrations in surface water, which can be due to the effect of denitrification and sedimentation. This means that our calculations may be an overestimate of the necessary reductions to protect surface water quality.

4.3. Conclusions.

4.3.1. Required, actual and critical nitrogen inputs.

At the EU-27 level, required N inputs are on average c.25-30% higher than actual inputs, with the difference being higher for arable land than for grassland and fodder. The average critical N inputs in relation to nutrient enrichment of terrestrial and aquatic ecosystems, using either a critical N deposition or a critical N concentration in runoff to surface water as criterion, are approximately 30% and 40% lower, respectively, than the actual (year 2010) N inputs. On average, the critical and actual N inputs are nearly equal when using a critical NO_3 concentration in groundwater of 50 mg NO_3 l^{-1} as the criterion. Comparison with measurements shows that the required reduction in relation to surface water quality is likely to be overestimated. In calculating the N runoff to surface water, we do not account for N removal in surface waters by denitrification and sedimentation and this effect seems larger than the neglect of N input from wastewater. An overall reduction in N inputs of 30% to protect air and water quality thus seems a reasonable average estimate.

As shown in the maps, the differences between required, actual and critical nitrogen inputs vary considerably between the different regions in Europe, implying that needed reductions (if even needed) should be implemented at a regional level. The exceedances in critical N inputs in relation to biodiversity losses in terrestrial ecosystems by NH_3-N emissions are mainly determined by variations in the N manure input and the critical NH_3-N emission rate, being mainly determined by the critical N load on terrestrial ecosystems and the fractions of agricultural land. Exceedances are especially large in high density livestock regions with large N manure inputs including Ireland and western UK, the Netherlands, Belgium and Luxembourg, Brittany in France and the Po valley in Italy.

The exceedances in critical N inputs in relation to eutrophication of aquatic ecosystems are mainly determined by variations in the total fertiliser and N manure input and the precipitation surplus and leaching (denitrification) fraction mainly determining the critical N inputs. Exceedances are especially large in Northern Europe, including the western parts of the UK, countries bordering the Baltic sea (Estonia, Latvia, Lithuania, Poland, Germany and Denmark), the Netherlands, Belgium, Luxembourg, North Eastern France and the Po valley in Italy, including high density livestock regions. Exceedances are, however, also high in regions with relative low actual N inputs, due the low N use efficiencies and/or low critical N inputs.

4.3.2. Ammonia emission fractions and nitrogen use efficiencies.

Actual and necessary ammonia emission fractions for all agricultural land, averaged over the EU-27 are 0.25 and 0.16, respectively, implying a necessary average reduction of 64%. However, we estimated that in c.25% of the area, the necessary reductions in NH_3-N emission fractions are not feasible at the actual yield level, when the NH_3-N emission fractions for fertiliser stay similar, but this reduces to c.10% when the NH_3-N emission fractions for fertiliser are all reduced to 2%. At target yields, the areas are c.40% and 20%, respectively.

The actual nitrogen use efficiency for all agricultural land, averaged over the EU-27 is 61%. This value has to increase by 25%, up to an average NUE of 72%, to attain the actual crop yield at acceptable N runoff to surface water. However, we estimated that in c.15% of all agricultural land the necessary NUE to achieve the surface water criterion is not feasible, as it exceeds a maximum plausible NUE of 90%. At target yields, this area is 25%. Considering the attainable NUE increases, a reduction in N inputs of 15-20% can be attained

In summary, at actual NUEs, it is expected that an overall reduction in N inputs of 30% at the EU level is necessary to protect air and water quality. However, with increased NUEs, the critical N input increases and the necessary reduction in N input in relation to environmental protection becomes lower. Overall, a reduction nearer to 15-30% seems reasonable, depending on the use of more efficient fertiliser application techniques.

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