Joined-up Nitrogen guidance for air, water and climate co-benefits

Theme 3: Field application of organic and inorganic fertilizers

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1. Types and quantities of materials being applied

Nitrogen is applied directly to agricultural land as a crop nutrient in the form of manufactured inorganic fertilizers, as livestock manures or as other organic amendments deriving from waste or by-products (e.g. sewage sludge, digestate from anaerobic digestion, composts). Managed land will also receive nitrogen inputs more indirectly from recycling of crop residues, from dung and urine deposition by grazing livestock and through N fixation by legumes. Together, these direct and indirect inputs total approximately 25,000 Gg N per year for the EU28 (Fig. 1). In addition to this is a further 2,000 Gg N per year input from atmospheric deposition but management of that is considered outside the scope of this background document. The characteristics of these different sources of N and their management are important in determining the agronomic value to crop and forage production and potential environmentally damaging impacts.

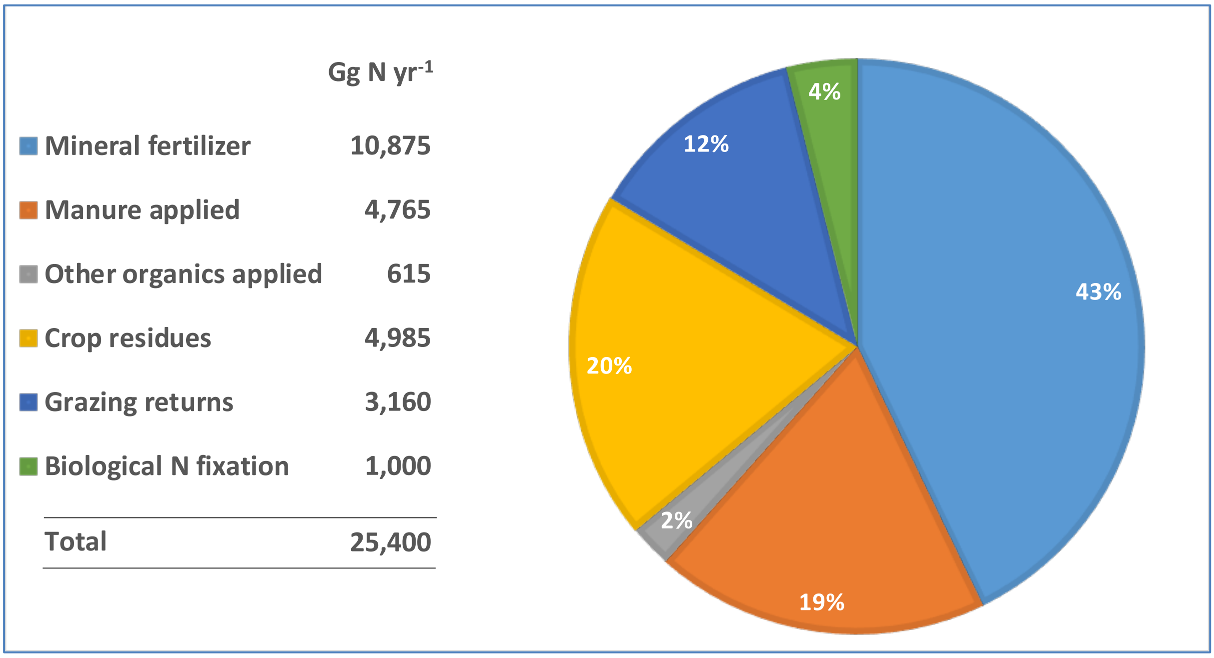

Figure 1. Estimate of N inputs to agricultural soils for EU28 (Gg N yr^{-1}) for 2014. Values derived from the 2016 GHG inventory submission to UNFCCC by the European Union (http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/9492.php) with the exception of Biological N fixation which was derived from Leip et al. (2011a) for the year 2002.

1.1. Inorganic mineral fertilizers

Manufactured inorganic mineral fertilizers represent the largest category of N inputs to agricultural land in the European Union (Fig. 1). There are a number of different formulations and blends of N-containing fertilizers used in Europe but these can be broadly considered to deliver nitrogen in the chemical form of ammonium, nitrate or urea. Ammonium and nitrate are directly available for plant uptake, although ammonium will also convert to nitrate in the soil through the microbial process of nitrification. These two forms of N will behave differently in the soil, with ammonium more susceptible to losses via ammonia volatilization while nitrate is more susceptible to losses via denitrification and leaching. Urea hydrolyses after application to form ammonium (and subsequently nitrate); the hydrolysis process is associated with an increase in pH which greatly increases the susceptibility to losses via ammonia volatilization. Straight N fertilizer products include ammonium nitrate (AN), calcium ammonium nitrate (CAN), urea and urea ammonium nitrate (UAN, a liquid formulation). Anhydrous ammonia is a liquid (gas under pressure) fertilizer that special equipment and safety measures for application. Combinations with other nutrients include ammonium sulphate, diammonium phosphate and potassium nitrate. Ammonium nitrate and CAN represent the major fertilizer forms used in Europe, with urea (either as urea or UAN) accounting for approximately 17% of total fertilizer N use in the EU28 in 2014 (based on background data supplied with the 2016 European Union GHG submission to the UNFCCC). Other nitrogen fertilizers including inhibitors and slow release formulations are discussed in Section 6.1.

1.2. Livestock manures

The major livestock types for which managed manure is applied to land are cattle (dairy and beef), pigs and poultry. Nitrogen will be present in organic and inorganic (ammonium and nitrate and, for poultry, uric acid and urea) forms. Manure characteristics depend on livestock diet and performance, housing and storage systems (including bedding use) and any subsequent processing prior to land application.

For cattle and pigs, manure type can be categorized as either slurry, consisting of mixed urine, faeces and water with very little bedding material and with a dry matter content typically in the range 1-10%, or as farm yard manure (FYM) consisting of urine and faeces mixed with large amounts of bedding material (typically straw) having higher dry matter content. Slurries will typically contain 40-80% of the N in the ammonium form with the remainder as organic N and none as nitrate. Farm yard manure typically contains a much lower proportion of the N in the ammonium form and may contain a small fraction in the nitrate form. Pig manure will typically have a higher total N and available (mineral) N content than cattle manure but this depends on water content.

For poultry, manure can generally be categorized as litter, deriving from systems where excreta are mixed with bedding (e.g. broiler houses) or as manure where excreta are collected, generally air-dried, without bedding material. Both have relatively high dry matter contents (>30%) and higher total N contents than cattle or pig manures. Between 30-50% of the total N may be in an inorganic form as uric acid or ammonium.

Manures will also vary regarding the content of other nutrients and application rates of all manures may be limited by the concentration of phosphorus (P) rather than N. The mineralization, availability and utilization of manure N is strongly influenced by C:N ratio.

1.3. Other organic N amendments

A range of other N-containing organic amendments are applied to agricultural land and while the total applied is currently small, this is likely to increase (and be encouraged) as the concept of the circular economy becomes more prevalent. These materials may be liquids (e.g. digestates) or solids (e.g. composts), deriving from human wastes, food processing, green wastes, etc., and for the purposes of this background document they will be implicitly included in discussions regarding management of livestock manures. Even though this recycling is important for the overall sustainability of society, the additional N added to agricultural systems are likely to be small compared to manure and fertiliser inputs. However, processing such organic amendments (e.g. anaerobic digestion) may increase the plant availability of N.

1.4. Crop residues

The quantity of N returned to agricultural soils through crop residues is of a similar magnitude to that applied as livestock manures (Fig. 1). These will include above and below ground residues, the N content of which will depend largely on crop type, yield and fertilizer management. The N will be almost entirely in an organic form, the rate of mineralization of which will depend on a number of crop, soil and environmental factors, and the potential for N losses will be mostly through nitrate leaching and denitrification rather than ammonia volatilization. In some cases (e.g. for high C residues) N from residues will be stored in soils as organic matter.

1.5. Grazing returns

Cattle and sheep can spend a substantial proportion of the year at pasture grazing depending on regional soil and climate characteristics and management systems. During grazing, dietary N not retained by the animal is deposited directly back to the pasture as dung and urine. Dung contains mostly organic N forms, which will subsequently mineralize at a rate dependant on soil and environmental factors, whereas N in urine is predominantly in an inorganic form and immediately susceptible to losses via ammonia volatilization, leaching and denitrification (Selbie et al., 2015).

1.6. N fixation

Cultivated legumes are grown on a relatively small proportion of the European Union agricultural area and their production in Europe has been declining over several decades despite an increased reliance by Europe on imported grain legumes (Luscher et al., 2014). This somewhat paradoxical situation contributes to global imbalances in protein production and consumption and the EU is currently considering options to increase home grown legumes to reduce the reliance on imported protein predominantly for livestock feed. Clover is an important constituent in many grasslands across Europe but the quantity of N provided by pasture is highly uncertain. During the growing season, N fixed by legumes will be mostly utilized by the crop (legume or companion crop) but when active growth slows or ceases then fixed N may be released to the soil through mineralization with potential subsequent losses through leaching and denitrification, in particular if the grassland is ploughed as part of a rotation system.

The FP7 project Legume Futures recently estimated that biological N fixation in Europe provided an input of 0.81 Mt N fixed in EU27 in 2009 (225 kt N from grain legumes and 586 kt N from grassland which was broadly similar to the mean estimate of 1.12 Mt from four European N budget models (de Vries et al., 2011) and the value of 1.1 Mt submitted to the United Nations Framework Convention on Climate Change (EEA, 2008). Most of the difference occurs because the N budget models allow for ~5 kg ha^{-1} of N fixation by free-living microbes in all non-legume arable land, in contrast to the Legume Futures focus on legumes

2. Current estimates of Nitrogen losses

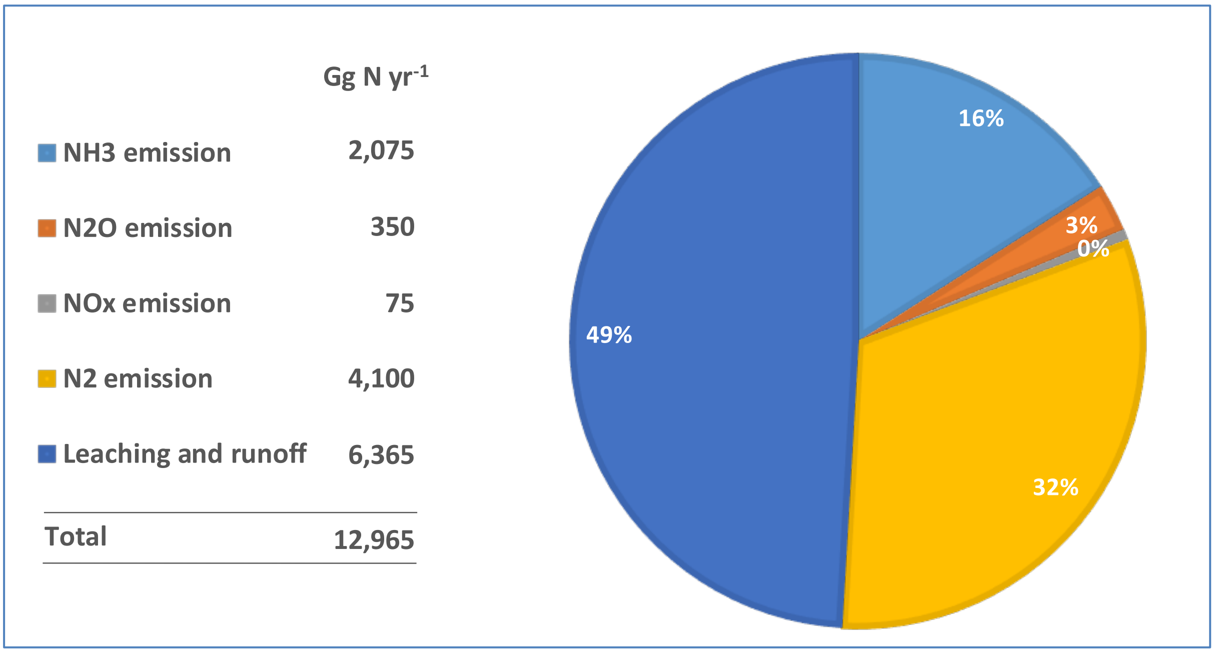

Figure 2. Estimates of N losses from agricultural soils in EU28 (Gg N yr^{-1}) for the year 2014. Values derived from the 2016 GHG inventory submission to UNFCCC by the European Union (http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/9492.php) with the exception of NO_x and N_2 emissions which were estimated as a ratio of reported N_2O emission based on values given by Leip et al. (2011a).

Estimates of N losses from agricultural soils for the EU28 are given in Figure 2, based on the 2016 European Union GHG submission to UNFCCC for ammonia, nitrous oxide and leaching and runoff losses and using the ratio of NO_x and N_2 to N_2O emissions for 2002 as reported by Leip et al. (2011a) to derive revised NO_x and N_2 emission estimates for 2014. These loss estimates are subject to large uncertainties, but imply that approaching 50% of N inputs to agricultural soils in the EU28 (including the estimate for atmospheric deposition) are subsequently lost to the environment through gaseous emissions, leaching and runoff. Of this, almost half is via leaching and run-off and another third as dinitrogen via denitrification. Dinitrogen is environmentally benign, but this represents a large loss of N which otherwise would have enabled agricultural N inputs to be reduced with subsequent savings in other parts of the system.

Emissions of ammonia, nitrous oxide and particularly NO_x account for smaller proportions of the total N loss from agricultural soils, but magnitude of loss doesn't necessarily equate with magnitude of impact. For nitrous oxide and ammonia, agricultural soils represent one of the most significant emission sources and therefore a key target area for interventions to meet national and international emission reduction targets.

3. Spatial distribution across Europe

Nitrogen inputs to agricultural soils vary considerably across Europe according to locations of livestock and crop areas and specific management practices, as driven by underlying factors including soils, climate and socioeconomics, as well as governance systems that regulate N inputs at farm scale. Mineral N fertilizer inputs tend to be higher across broad areas of NW Europe, while manure N inputs are more localised to areas with high livestock densities with particular 'hotspots' in the Netherlands and N Italy for example due to cost of transporting the manures (Fig. 3; NB data shown for 2005). Finely distributed N input estimates across space will have a greater level of uncertainty than national- or European-scale estimates and will vary across the region, particularly for manure N inputs where robust data on spatial variation in key factors influencing livestock N excretion (diet, management) and subsequent manure management practices may not be available.

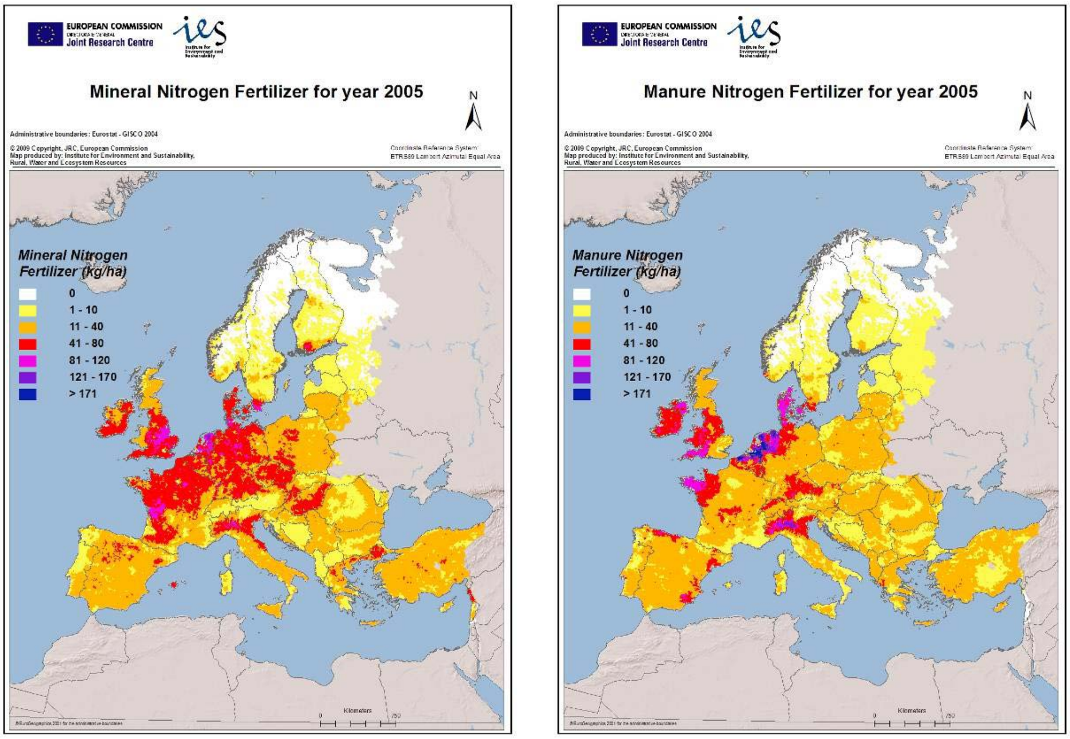

Figure 3. Estimates of spatial distribution of mineral fertilizer and manure N to agricultural land for 2005 (Bouraoui et al., 2009.).

Spatial distribution of the different N losses across Europe relies on modelling, which can be performed at different complexities. At the simplest, national N loss estimates can be spatially distributed using nationally averaged emission factors (e.g. per livestock head or per kg of fertilizer N) according to national survey data on livestock numbers, cropping and fertilizer use. More informatively, empirical or process-based models can be used which reflect the spatial (and temporal) distribution in underlying factors driving the loss processes (soils, climate) and therefore better reflect spatial distribution of losses, albeit with uncertainties associated with required parameters and data inputs and model accuracy and performance. Such an approach will result in different 'emission factors' for different regions (and different nationally-averaged emission factors). For example, Leip et al. (2011b) used the DNDC-Europe model to derive spatially distributed nitrous oxide emission factors across Europe and showed that while on average the default IPCC emission factor (1% of applied N being emitted as N_2O-N) was appropriate across Europe, spatial variability was large with national averaged emission factors ranging from 0.4 to 4.1%.

Similarly, there is currently significant uncertainty around the ammonia emission factor for urea fertilizer application to land, with the EMEP/EEA Air Pollutant Inventory Guidebook of 2009 giving an emission factor relating to the average spring temperature, but a subsequent revision in the 2013 Guidebook removing the temperature dependence and giving a default emission factor for urea of 24.3% of applied N being lost as ammonia-N. This implied a very large change in ammonia emission estimates for countries with lower spring temperatures (e.g. Germany) with potential consequences regarding compliance with agreed national emission ceiling targets. Further Guidebook revision is ongoing, with an expected return to a process-model approach able to reflect regional differences. However, European countries may apply different approaches in their national inventory models depending on the data they have available, from a Tier 1 approach which applies a default emission factor to all fertilizer N regardless of type, through Tier 2 which reflects fertilizer type in emission factor, to Tier 3 where a country can use its own measurement data or apply a more detailed model. This highlights the importance of having, and using a robust understanding of the N loss processes when compiling estimates of the different loss pathways and of using a consistent approach across all regions, despite different contexts and activity data.

Finally, the impacts of N losses from agricultural soils on the environment will also have a spatial dimension. A large proportion of ammonia emissions from N applied to agricultural soils will be redeposited locally, with potential impacts through eutrophication and acidification, but a proportion will also be subject to longer range transport and processes associated with aerosol formation with subsequent human health implications. Similarly, N losses through leaching and runoff will have a local, catchment and potentially regional effect on water quality depending on flow pathway and N transformation and reduction processes along this pathway. For these reactive N species therefore, a good understanding of source-receptor matrices is required including appropriate spatial and temporal distributions. In contrast, nitrous oxide has a global, rather than local impact as a greenhouse gas and dinitrogen is environmentally benign. For these gases an understanding of the spatial and temporal influences on their emissions is important, but such influences on dispersion and impacts need not be considered.

4. Management practices and influence on N losses

Nitrogen is the nutrient recovered in largest quantities from soil by agricultural crops, and the availability of nitrogen to crops has a major impact on yields. Management of the different N inputs to agricultural soils will influence the subsequent N cycling, N utilisation by crops and losses of N in different forms to the environment. Until now, focus has largely been on controlling individual N loss pathways – e.g. nitrate leaching (Nitrates Directive), ammonia (Gothenburg Protocol, NECD and Habitats Directive) and nitrous oxide (Kyoto protocol) and guidance given accordingly (e.g. TFRN Options for Ammonia Mitigation Guidance document). It is critical in trying to develop a more joined-up approach to N guidance to have a good understanding of how management practices and targeted mitigation measures might impact on the whole N cycle and not just one specific pathway. This section presents briefly the main management practices that will influence N utilization and losses from the different N sources. A summary of the impacts on the different N losses is given in Appendix 1. Additionally, the concept of precision agriculture for enhanced Nitrogen use efficiency is very relevant here and complements the management practices discussed. Enhancing N use efficiency is not only a question of proper fertilisation strategies, it also is also very much related maintaining a health crop, where good soil quality, good crop establishment and proper control of weeds, pests and diseases play major roles.

4.1. Inorganic mineral fertilizers

Use of fertilizer N commonly doubles crop yields, and the longer term economic benefits of N fertilizer use are even larger because fertilizer N serves to build soil fertility. Thus fertilizer N is vital to the profitability of crop production in all regions of the EU and N fertilizers are used by almost all farms other than those committed to 'organic' production.

Quantities of N required by crops (and used) are crudely related to their productivities. Thus productive crops need more N, whilst crops with short life-cycles or subject to drought need less N. Whilst plant breeding and improved agronomy have increased the efficiency of crop N use a little, most attempts to increase crop productivity are nevertheless also associated with increased N requirements (Sylvester-Bradley & Kindred, 2009). Best responses to fertiliser N are generally through application in spring, just prior to rapid crop growth, whilst soils are usually drying (evapotranspiration exceeding rainfall). Thus most gaseous losses occur soon after application but most leaching losses are delayed until after harvest; they arise from fertiliser N that has been immobilized (partly in crop residues) and re-mineralized. This legacy of immobilized N, especially from inefficient crops, tends to build soil fertility and reduce N requirements of succeeding crops (Sylvester-Bradley, 1996).

Guidance on N application rates and timings is often available at a national level (e.g. UK RB209) but, due to local factors and conditions including soil, weather, disease incidence etc, recommendations are generally imprecise with, at best, half of them differing by more than 50 kg/ha from the true optimum (Sylvester-Bradley et al., 2008). However, N losses relate better to absolute amounts applied than to imprecision, so they could be reduced more by improving fertiliser efficiencies or targeting for lower crop N contents (e.g. protein concentrations) than by than improving fertilizer recommendations.

Inhibitors can be incorporated into fertilizer products to reduce specific N loss pathways and improve efficiency (Abalos et al., 2014). Urease inhibitors used with urea fertilizer products are very effective at reducing ammonia emissions, and generally give an associated enhancement in crop yield, although the potential for nitrous oxide emissions (and nitrate leaching) may be marginally increased as more available N is retained in the soil. Nitrification inhibitors used with urea and ammonium based fertilizers can be very effective at reducing nitrous oxide emissions, although efficacy varies according to soil and weather conditions. Positive impacts on crop yields have been more difficult to show, and there is a potential increase in ammonia emission. The use of double inhibitors with urea-based fertilizers has been trialled to reduce all losses and improve N utilization with mixed results (e.g. Harty et al., 2016; Zaman et al., 2009).

4.2. Livestock manures

There has been considerable research and development of slurry application methods associated with lower ammonia emissions than surface broadcast application and these methods are well established, even if not well implemented across Europe. Impacts of these methods on other N pathways is perhaps less well established, with mixed evidence regarding increases in nitrous oxide emissions and benefits of ammonia emission reduction not always being apparent in higher crop yields or N uptake. A recent review showed that nitrous oxide emissions can range between 0.1-9.5% of the total N contained in the slurry, with this range being affected by slurry type, application method, soil conditions and climate (Chadwick et al., 2011). The use of trailing shoe and injection technology can dramatically reduce ammonia emissions and odour and thus reduces indirect nitrous oxide emissions. However, studies have shown that such techniques increase direct nitrous oxide emissions (Bourdin et al., 2014; Thorman et al., 2007). A recent comparison between splash plate and injection techniques in Ireland concluded that there was no significant difference in net greenhouse gas emissions from the two techniques (Bourdin et al., 2014). Slurry acidification as a means of reducing ammonia emissions is also very effective and in recent years has been demonstrated to be a practical option with significant implementation in Denmark. Effects on crop yields have generally shown to be positive, but longer term impacts on soil quality across the range of European soils and conditions still need further investigation.

Slurry dilution and application through fertigation in areas where irrigation is required is another option aimed at reducing ammonia emissions, through more rapid soil infiltration and, while the potential for nitrous oxide and nitrate leaching is to increase the risk of this is low if applied at agronomically sensible times and rates. Pre-processing, such as slurry separation, may also improve the ability to use the slurry nutrients more efficiently, but impacts on N flows will depend on the subsequent use of the liquid and solid fractions.

Rapid soil incorporation of manures by tillage significantly reduces ammonia emission, again with the potential to increase nitrous oxide emissions and nitrate leaching depending on timing and conditions. There is some discussion regarding practicalities of rapid incorporation, and what time period is considered ‘rapid’, and monitoring compliance of such a measure may present difficulties.

Nitrification inhibitors can be used to reduce direct nitrous oxide emissions and nitrate leaching associated with manure application to land, but have the potential to increase ammonia emissions and positive effects on yield or crop N uptake are small if seen at all.

Part of the N in manure is applied in organic form, which is not readily available for plant uptake, and which through mineralisation may enhance leaching losses outside of the main crop growing season. Anaerobic digestion of manures enhances the proportion of mineral (ammonium) part of the liquid manure, which enhances crop N uptake and reduces N leaching. However, anaerobic digestion also results in higher slurry pH which may increase ammonia volatilisation during storage and after application. The enhanced volatilisation of the digested slurry is to some extent mitigated by a more rapid infiltration into the soil due to changes in viscosity.

The focus for integrated guidance should therefore be on maximizing manure N (and nutrient) utilization through development of a nutrient management plan including fertilizer use depending on crop requirements, considering application rate, timing and method according to local soil and environmental conditions.

4.3. Legumes and crop residues

Leguminous crops input N to agricultural systems by biological N fixation, in which a symbiotic relationship is formed between the legume and N-fixing bacteria. Fixation (compared with mineral fertilizer use) is associated with reduced GHG emissions for two reasons; firstly, emissions from N manufacture are avoided, and secondly, the process of N fixation itself is associated with an emission factor of 0 (IPCC 2006), unlike inorganic N fertilisers. For grassland systems the challenge is to maintain an appropriate proportion of clover in the sward within a season and across multiple seasons and manage mineral N fertilization to achieve an optimal delivery of fixed N to the mixed sward. Crop residues are known to contribute to both nitrous oxide and ammonia losses, although this is related to residues quality, environmental conditions and method of incorporation. Recent research suggests that nitrous oxide emissions from crop residues may be lower than previously thought (Sylvester-Bradley et al., 2015). Management considerations should include the avoidance of high available soil mineral N contents when crop uptake is low.

Winter cover crops are used in some circumstances to minimize high soil available mineral N content over the high risk period for nitrate leaching but their success in increasing N use efficiency over the whole cropping cycle depends on effective management of the cover crop residue. Tillage options influencing N mineralisation will impact on potential N losses and uptake. In colder climates, freeze thaw cycles over the winter period can cause significant nutrient release and nitrous oxide emissions. In order to minimise N loss it is necessary to time tillage operations in order to optimise synchrony between N release and uptake by a subsequent crop N uptake.

4.4. Grazing returns

The management of grazing livestock and the impact of the N returns through dung and urine can be of significant importance for countries where there is a high reliance on grazing to feed ruminants (e.g. Ireland, UK). The key management tool available to influence soil N losses from grazing is to remove grazing animals prior to periods of high risk of N loss (via leaching and denitrification), i.e. having a shorter grazing period than consideration of soil condition and herbage availability alone would suggest. However, this has to be weighed against the implications for N flows occurring during the housing of the livestock, where ammonia emissions are likely to be greater. The use of appropriate forage species and fertilizer management to optimize the feed quality for the grazing animal will improve N utilization and reduce N excretion. Information on the influence of different grazing practices (set stocking, rotational paddock grazing, mob grazing) on herbage N utilization, N excretion and N losses is required to develop guidance on improved N use at the system level. The use of nitrification inhibitors to specifically reduce nitrous oxide emissions and nitrate leaching associated with urine patches represents another management tool for which cost-effective delivery mechanisms need to be developed and the issues of inhibitor retention in milk or meat need to be addressed.

4.5. Nitrogen use efficiency and precision management

The EU Nitrogen Expert Panel (2015) have introduced the concept of a target range for N use efficiency taking into account a minimum level of desired productivity, a desired maximum N surplus (per ha) and an acknowledgment that long-term mining of soil N reserves is unsustainable (Figure 4). The N output target must take into account both quantity and quality of product, whether that be food or feed, so that N is not wasted in a later part of the food production/consumption chain. This targeting of optimal N use efficiency might provide a starting framework for joined-up Nitrogen guidance for food, air, water and climate co-benefits. The concept of precision agriculture is very relevant to this, understanding the importance of all other factors being right (other macro-and micro-nutrient supply, water supply, soil 'health', management of pests and diseases) in order to achieve optimal N use efficiency.

Historically most fertiliser recommendation systems in Europe have taken little account of spatial variability in N cycling processes, despite the well established heterogeneity of N cycling processes within landscapes. Our rapidly developing ability to observe and analyse spatial and temporal heterogeneity of plant and soil condition, coupled with information and sensor technologies that are able to manage fertiliser, lime and tillage operations on a more spatially explicit basis are beginning to offer opportunities to develop precision N management. Optimising the use of such technologies will depend upon a further development in understanding of underlying soil processes, but offers the potential to deliver increased N use efficiency in agricultural systems {Diacono, 2013}. However, recent research shows that, so far, the imprecisions that apply at a field-scale, also apply at a sub-field scale (Kindred et al., 2015).

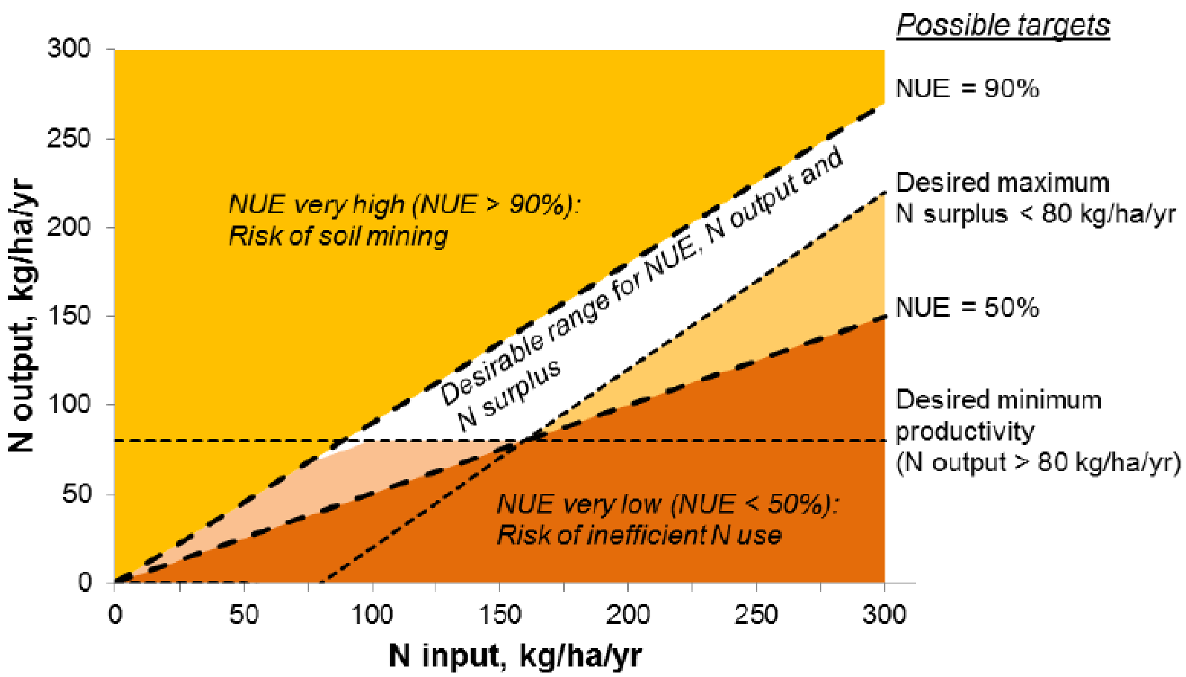

Figure 4. Conceptual framework of the N use efficiency (NUE) indicator (EU Nitrogen Expert Panel, 2015)

4.6 Water and N use efficiency

Water is a driver of the main environmental problems caused by excessive N inputs (mineral or organic fertilizers) in agroecosystems. Excessive water input, either by rain or irrigation, enhances nitrate contamination of water bodies or increases emissions of nitrous oxide. Therefore, sustainable intensification of agriculture should take into account management strategies towards increasing water and N use efficiency simultaneously.

In irrigated agriculture, water application is a management option that the farmers may use to enhance NUE and reduce losses (Quemada and Gabriel 2016). In this sense, new techniques such fertigation have great potential for increasing N use efficiency of fertilizers. Fertigation is a particular case of scheduled irrigation combined with nutrient application. In conventional fertilization the fertilizer is split in one, two or three applications. During a certain period, there is an excess of N in soil that could reduce NUE. With fertigation the fertilizers are dissolved and supplied with irrigation. The number of applications can be high and adapted to crop demand, therefore reducing N potential lost (Abalos et al 2014).

5. Existing guidance

- Options for Ammonia Abatement: Guidance from the UNECE Task Force on Reactive Nitrogen (http://www.clrtap-tfrn.org/content/options-ammonia-abatement-guidance-unece-task-force-reactive-nitrogen)
- HELCOM Baltic Sea Action Plan (http://helcom.fi/baltic-sea-action-plan) See p86-96 for agricultural measures
- EU Project report: 'Resource efficiency in Practice – Closing Mineral Cycles' (http://ec.europa.eu/environment/water/water-nitrates/pdf/Closing_mineral_cycles_final%20report.pdf) See p87 onwards. Also see project outputs - *Region-specific leaflets on best-practices* (http://ec.europa.eu/environment/water/water-nitrates/index_en.html)
- Mainstreaming climate change into rural development policy post 2013. European Commission 2014 (http://ecologic.eu/sites/files/publication/2015/mainstreaming_climatechange_rdps_post2013_final.pdf) See Table 3 for list of measures
- National fertilizer recommendations (e.g. UK RB209)
- National codes for good agricultural practice

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Table 1. Impact of management practices on Nitrogen losses from agricultural soils (for discussion)

Practice	Leaching/runoff	Ammonia volatilization	Nitrous oxide	Notes
Inorganic mineral fertilizers				
Appropriate rate and timing	↓	↓	↓	Timing (wet) that reduces NH3 my increase leaching or denitrification
Replace urea with AN	~	↓	~	Urea is cheaper and possibly safer. OK for some situations (injection)
Use urease inhibitor	~	↓	~	Can reduce synchrony between crop demand and availability of N. May increase leaching of urea.
Use nitrification inhibitor	↓	~↑	↓	Can reduce synchrony between crop demand and availability of N
Use slow release fertilizers	~↓	~↓	~↓	
Livestock manures				
Integrated N management plan	↓	↓	↓	
Apply slurries by band spreading/trailing shoe	~↑	↓	~↑	
Apply slurries by injection	~↑	↓	~↑	Shallow injection can create runoff channels
Slurry dilution for fertigation	~↑	↓	~↑	
Slurry acidification	~↑	↓	~↑	

Use nitrification inhibitors	↓	~↑	↓	
Rapid incorporation of manures after application	~↑	↓	~↑	
Anaerobic digestion	~↑	~↑	~↑	Depends on management of facility and subsequent digestate
Livestock grazing				
Shorter grazing season	↓	↑	↓	Needs to be assessed across the full system
Use nitrification inhibitors	↓	~↑	↓	
Tillage and cropping				
Use cover crops	↓	~	↓	
Use minimum tillage practices	~↑	~↑	↑	
Use legumes	↓	↓	↓	Potential for higher winter losses due to mineralization and freeze thaw
Crop residue management?	~↓	~↓	↓	
Choice of low protein varieties	↓	↓	↓	These have low N demands, but must be reconciled with end-users & markets.

Discussion questions

Questions to promote discussion to address the high level question of 'How to develop joined-up Nitrogen guidance for food, air, water and climate co-benefits'.

- Do we have good enough data on the different N inputs to agricultural soils – and at a fine enough resolution? What are the gaps and how can we fill them?
- Do we have good enough knowledge on the different N losses (including N2) from agricultural soils? What needs further developing and what data are required? (leaching/runoff to include DON)
- Can we change end-use specifications for crops, such that their N demands are reduced?
- Is it possible to have a robust and cheap methodology to predict N mineralized from an organic fertilizer under field conditions?
- Can we reconcile 'emission factor' approaches with different tiers of complexity for different countries with a more process-based approach reflecting regional differences?
- Are there exemplar practices that could be adopted/regulated more widely throughout Europe – with potential regional adjustments?
- Should fertigation technology be promoted for irrigated agriculture?
- Would an increased use of legumes help, and if so in what circumstances? Do current policies promote or discourage use?
- Should all mineral fertilizer products include inhibitors or other technologies that delay release?
- Would a major focus on guidance to improve NUE be effective in reducing losses in an integrated way?
- Would the enhancement of water and N use efficiency simultaneously provide advantages over optimization of water and N input separately?
- Should further zonal policy (e.g. NVZ) be considered to address the spatial nature of inputs/losses/impacts?
- Are certain forms of N loss more important than others and is this region/context specific?
- Can we develop an adaptive system to N use recommendations accounting for medium to long term weather forecasts?
- How can precision management approaches be used to reduce N loss and increase crop N recovery?