Optimizing nitrogen use efficiency in European livestock systems: From feed to plant growth

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Contents

Abstract

Nitrogen (N) is a crucial nutrient for agriculture. Its unlimited provision by the Haber-Bosch process has led to high N surplus in agriculture, causing severe negative environmental externalities. The reduction of N losses in livestock systems and consequent enhancement of their nitrogen use efficiency (NUE) represents a key lever to substantially reduce the N surplus to meet European environmental policy targets. This review evaluates the potentials to improve the NUE of European cattle and pig livestock systems from feed to plant N use along the whole livestock system cascade. It discusses options to improve feed NUE and the current measures to reduce N losses at housing, manure storage and application as well as crop N recovery. The findings are synthesized at the overall livestock system level using exemplary case-study farms from selected European regions, to highlight potentials for livestock system NUE improvements in the context of the current and future European legislations. All case-study farms have already implemented measures for N loss mitigation, and their livestock system NUE reaches values between 55 % and 75 %. With the implementation of additional measures such as liquid manure acidification, low emission technologies and best field management, livestock system NUE can reach values between 75 % and 85 %. In order to achieve such high livestock system NUE and meet environmental goals, a scaled implementation of best available technologies is essential, and has to go hand in hand with a legislative harmonization and a targeted development of additional reduction measures and auxiliary tools for farmers.

1. Introduction

Nitrogen (N) is a key plant nutrient in agriculture. Its nearly unlimited provision by the Haber-Bosch process was the basis to increase the agricultural productivity dramatically and allowed a world population of more than 8 billion people in 2024 (Erisman et al., 2008; UNFPA, 2024; Zhang et al., 2015). Consequently, during the 20th century, global N input as fertilizer applications (manure and mineral N) and biological N_2-fixation by legumes increased 100-fold (Houlton et al., 2019). Europe is one of the most intensive agricultural regions worldwide with a high productivity per agricultural area (Godinot et al., 2016), corresponding with high N surpluses (N inputs to agriculture which are not recovered by crops or animal products) in several regions (Dalgaard et al., 2012; Godinot et al., 2016; Klages et al., 2020; Leip et al., 2011). The overall N surplus calculated for EU-27 countries using the OECD gross N budget approach was 40 kg N ha^{-1} in utilized agricultural area in 2014 (EUROSTAT, 2017), with regional variations between −12 and +195 kg N ha^{-1}. Nitrogen budgets based on both soil surface and farm-gate approaches reveal even higher surpluses, which are commonly but not necessarily related to higher stocking rates at farm-level or animal density on regional level (Godinot et al., 2015; Leip et al., 2011).

The N surplus (input – output) gives a quantitative approximation of N potentially lost from the system (Klages et al., 2020). N surplus data can be transformed to N use efficiency (NUE) of the system as ratio between output : input. Globally, NUE in manures and commercial fertilizers declined from about 60 % to 46 % from 1900 to 2000 (Houlton et al., 2019). Based on the farm-gate approach, NUE of the EU-27 in the beginning of the 21th century (2000–2003) was as low as 31 % on average (Leip et al., 2011). The variability was huge with a NUE of an average of 50 % in Romania and Bulgaria with extensive agriculture and crop production and 15 % in Slovenia and Ireland, characterized by a strong specialization towards livestock products. This is confirmed by the NUE values of selected agricultural products, which vary largely between crops (90 %) milk (39 %) and beef cattle (26 %) productions (Godinot et al., 2016).

High N surpluses and low NUE in livestock production are a major cause of severe negative environmental externalities and human health (De Vries, 2021). Excess N is a major source of air, soil and water pollution with subsequent climate impacts. Specifically, this leads to reduced biodiversity in rivers, lakes, wetlands and sea. Global industrial and anthropogenic symbiotic fixation of N surpassed the planetary boundary by a factor of 2.3 (De Vries et al., 2013; Richardson et al., 2023).

The European Union's Farm to Fork Strategy aims to reduce the N surplus in agriculture by at least 50 % and the use of N fertilizers by at least 20 % by 2030 in order to substantially reduce the N losses (EU 2019). A key lever to achieve this goal is a reduction of N losses in livestock systems and thus enhance their NUE (Leip et al., 2022).

This review assesses the potential to improve the *livestock system NUE* of European cattle and pig livestock systems (Fig. 1). *Livestock system NUE* is defined here as the combined NUE of (i) *feed NUE*, (ii) *manure cascade NUE* and (iii) *grazing NUE*. *Feed NUE* considers the N flows from feed intake to animal products (Section 3.1). The *manure cascade NUE* considers the N flows from excreted N at all stages of livestock husbandry including livestock housing (Section 3.2), storage (Section 3.4), application (Section 3.5)

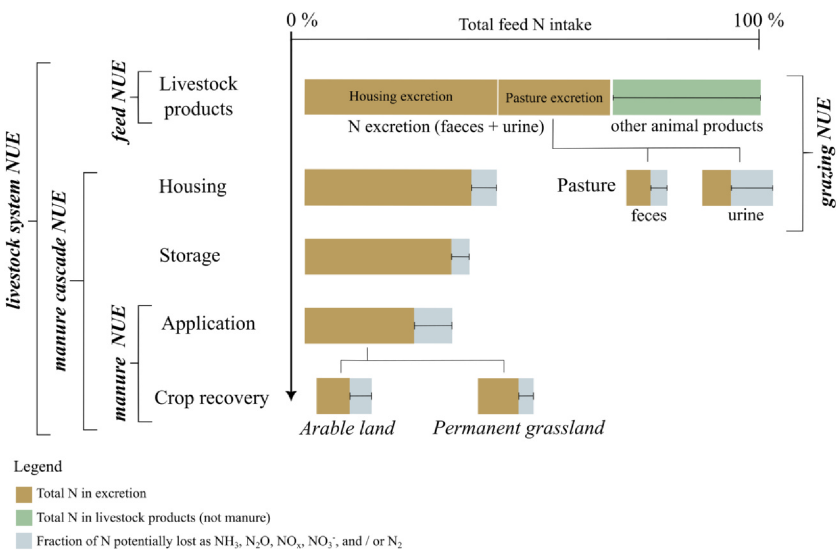

Fig. 1 Steps of the overall manure *livestock system NUE* assessment, as the combination of *feed NUE*, *manure cascade NUE* (housing, storage, application losses and crop N recovery) and g*razing NUE*. *Manure NUE* describes the N recovery of crops related to fertilized manure from storage. The N flow related to *other animal products* entails the sum of all animal products, except the N excreted. N from both liquid and solid manure can be lost as ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxides (NO_x), dinitrogen (N_2) or nitrate (NO_3^-). The bar lengths are for conceptual illustration only and are not based on specific measurements.

as well as fertilizer crop N recovery (Section 3.6). Manure processing (Section 3.3) can strongly influence the emissions of the subsequent stages of the *manure cascade*. The *grazing NUE* includes the plant N recovery from directly excreted urine and dung (Section 3.7; Fig. 1).

This review discusses the current measures to reduce N losses and to improve NUE at each stage of the cascade. It synthesizes results to show the current and potential optimization of the *livestock system NUE*, based on a set of exemplary European pig and dairy case-study farms and in the context of actual and future European legislations.

2. Aspects of legislation

Environmental impacts caused by excess availability and use of N, in particular in regions with high animal densities and surplus of organic fertilizers (De Vries, 2021), stem from losses in housing, manure storage and application to the fields, due to the limited uptake of N by the plants. This led to an increased regulation of N use in agriculture through policies to foster environment and resource protection (Wuepper et al., 2024). Up to date, laws, regulations, guidelines, monitoring and legal enforcement schemes exist on national and international scale (Bjerg et al., 2023). In general, legislation on animal manure handling and use is differentiated between point and diffuse sources for losses (Sommer and Knudsen, 2021). The reduction of losses from point sources contains for example building standards for housing systems, waste water management and minimum requirements for storage capacity. While there seems to be general agreement that there should be no direct discharge from manure storage to surface waters for instance, the legislation related to construction and infrastructure is more heterogenic in Europe (Bjerg et al., 2023). This is likely due to the relative scarcity of studies related to geographical distribution (Vitaliano et al., 2024), but also related to historically different development and implementation of new technologies and knowledge. Therefore, at European level such legislations are mostly facilitated by guidelines declaring mitigation potentials for different available technologies, such as the management of intensive pig husbandry (Santonja et al., 2017). The specific legislation is then left for individual countries and therein regions to be organized. At the international level, the UNECE Guide for NH_3 mitigation gives recommendations for mitigation of N losses (Bittman et al., 2014). However, certain aspects are taken up even in international legislation and are regulated in detail, as for instance the storage capacity over time obligated in nitrate vulnerable zones (NVZ), given in the nitrate directive (EU, 1991).

Diffuse N losses are more difficult to quantify and control. For livestock manure, they are related to the number of livestock and area used for the husbandry systems, all aspects affecting the N content of the manure and the achieved NUE in the fertilized cropping systems (Table 1). Here, legislation often uses thresholds or cap values for N fertilizer use or number of animals held per area (Laakso and Luostarinen, 2019). The EU nitrate directive for instance gives a maximum of 170 kg of N applied by farm manure including that deposited by grazing animals per hectare (EU, 1991). Other ways are an N fertilizer quota per farm as implemented in Denmark (Sommer and Knudsen, 2021), or the strict link to fertilizer planning as found in Germany (Löw et al., 2021) and many other European countries (Jordan-Meille et al., 2023). All of these approaches are based on estimated manure NUE values, which differ in assessment methodology and vary largely between countries (Table 1). The approaches range from assuming adoption of best technology and knowledge, thus achieving high manure NUE to force farmers to reduce their absolute N inputs (Sommer and Knudsen, 2021), to using average or realistic plant available N from manures, often oriented on traditional manure management, resulting in relatively low manure NUE (Table 1). Some countries additionally set limits for animals held per area, or manure used per area of land, as for instance in the water protection law of Switzerland (Table 1), where the limit of three animal units is supposed to restrict the fertilization of N to an absolute amount of 315 kg N per hectare.

Due to the complexity of underlaying processes to N losses, actual goals for low N emission legislation often aim at implementing and fostering sustainable production principles, good agricultural practice and best available technologies. Such can be applied as strict minimal standards in laws or as guidelines for orientation to technologies being subsidized for faster adoption by farmers (Gemtou et al., 2024; Groher et al., 2020). Additional guidelines comprise building standards, such as different housing systems, wastewater management and storage capacities and define different livestock feeding strategies, loss reduction technologies for manure spreading, and measures for efficient fertilizer use in time and space. The focus of country or region-specific regulations often follows major causes or consequences of N surplus, such as livestock density or ground water pollution (EU, 1991; Ragonese et al., 2024), respectively. Consequently,

Table 1 Aspects of nutrient and fertilizer management legislation of selected countries in Europe.

	Country examples					
	Denmark	**Germany**	**Switzerland**	**The Netherlands**	**France**	**Italy**
Field specific fertilization plan	Mandatory	Mandatory	Not mandatory but recommended	Since 2023 mandatory for all farms	Mandatory for all farms in NVZ	Mandatory for farms utilizing > 6000 kg N if not in NVZ; > 3000 kg N if in NVZ
Farm nutrient budget	Fertilizer accounting required from all farms	Currently not in place	Mandatory accounting on farm level (Suisse-Balance) using plant available N estimate (minus losses) and average crop N demand (not fertilizer plan)	Application standards for N (plant available N) and P, but not a nutrient budget	N budget < 50 kg N/ha in NVZ and for large animal farms (to ensure sufficient land and crop demand for manure application)	
Maximum amount of N from manure (kg ha^{-1})	170 (230 in cattle derogation farms), from 2025 no derogation	170	315 (equivalent to three livestock units). Cantonal adjustment possible	from 2026: 170	170 at farm level in NVZ	170 in NVZ, 340 outside NVZ

(continued)

Table 1 Aspects of nutrient and fertilizer management legislation of selected countries in Europe. (*cont'd*)

	Country examples					
	Denmark	**Germany**	**Switzerland**	**The Netherlands**	**France**	**Italy**
Maximum amount of total N from fertilizer	N norm (quota) for each crop	According to fertilizer plan, field specific	Maximum defined by farm level balance, not field specific	Legal applications standards, for soil and crop type. Range 0 kg N/ha for leguminous cover crops to 385 kg N/ha for grassland	According to fertilizer plan, field specific	Maximum defined by crop specific values
Number of manure categories used		9	11	20 types for which efficiency coefficients apply: 7 liquid manure and 13 solid manures		3
In legislation implied Manure N use Efficiency (% of total N) (Cattle / pig)	75 % / 80 %	60 % / 70 %	60 % / 60 %, minus additional farm specific losses	60 % for cattle slurry on farms without grazing; 45 % for farms with grazing. 60 % (peat, clay) and 80 % (sand) for pig slurries	Region specific MFE values, depending on manure type, crop type and period of application	Cattle: 24–62 %. Pig: 28–73 % depending on soil texture, application date, method and crop

Definition of Manure N use Efficiency	MFE	Minimum effectiveness in the year of application in % of total N content	Estimated farm level nitrogen recovery value	MFE	MFE	Not defined
Mandatory measures (program), examples	Mandatory cover crops when using manures	Immediate incorporation of manures	Cover of manure storage and use of trailing hose	N, P and manure application standards, close periods, catch crops after silage maize, crop rotation restrictions, buffer strips near ditches, etc.	Prohibited near water bodies and buildings and may require incorporation, depending on manure type, region, etc.	
Voluntary measures (program)	Measures aiming to improve the nutrient value and reduce ammonia emission of manure. Some are subsidized. (Cover crops, early seeding of winter cereal, etc.)	Agri-environment and climate-protection measures with subsidies	Agri-environmental support scheme with subsidies on cover crops, reduced N use and loss reduction measures for instance	Voluntary measures in the so-called DAW-program to increase nutrient use efficiency (some subsidized)	Subsidies in specific areas for "systemic" changes, that can include a decrease in fertilizer use	Agri-environmental support scheme with subsidies
Nitrate vulnerable zones or action program defined	Whole country	National action plan for whole country with additional measures in areas with $NO_3 > 50$ mg/l	Catchment based	Whole country	Regional according to NVZ and animal density	Regional

(continued)

Table 1 Aspects of nutrient and fertilizer management legislation of selected countries in Europe. (*cont'd*)

	Country examples					
	Denmark	**Germany**	**Switzerland**	**The Netherlands**	**France**	**Italy**
Ammonia emission abatement measures Housing/ storage/ application	Yes/yes/yes	Yes/yes/yes (partly mandatory, partly voluntary)	Yes/yes/yes (partly mandatory, partly voluntary)	Yes/yes/yes		National emission ceilings
Greenhouse gas abatement measures (Methane and nitrous oxide emissions)	Planned		Voluntary, on cantonal or even regional level, also labels	Not yet part of legislation, but part of a decision support tool for dairy farmers by which dairy industry stimulates farmers to decrease their CO_2 footprint		Supported by regional Rural Development Plan

NVZ refers to nitrate vulnerable Zones, MFE to mineral fertilizer equivalent.

good agricultural practices and best available technologies in NVZ aim to reduce nitrate (NO_3^-) leaching (Curk and Glavan, 2023; Frick et al., 2023a) or focus on livestock dense regions on mitigation of NH_3 emissions (Emmerling et al., 2020; Öttl et al., 2023).

To quantify N surplus, nutrient budgets are important tools complementing legislation. On international, national and also at regional levels, farm or field level budgets are used as indicators to control sustainability targets and agri-environmental impacts (Klages et al., 2020). Depending on data input and structure, a net and gross balance can be differentiated and many individual adjustments exist (Klages et al., 2020; Oenema et al., 2003). At operational level (farm and field), the main goal is an efficient use of resources, such as fertilizer and feed, and the inspection of legal enforcement. On regional and national levels the focus is on N monitoring as a foundation for political and legal decisions, often in combination with other agri-environmental targets.

Every country has different levels of obligation in their legislation, which are reflected by the mandatory or voluntary character of legislation and legal enforcement but also by the robustness and type of implementation of used numbers and information resources (Table 1). The level of documentation of use of good agricultural practice and best available technologies ranges from self-declaration with little options for reality check to digital accounting systems, including trade and transport of fertilizers. Furthermore, such aspects can be differently embedded into subsidy schemes and legislation. However, measures that are defined as obligation in legislation cannot be subsidized in EU countries. Seeing the large variability of legislation and integration of knowledge between countries (Bjerg et al., 2023) it seems obvious that implementation of knowledge in particular about NUE into decision support tools, models and consequently legislation remains a major challenge to support farmers and policy makers (Velthof et al., 2024).

3. Exploring the N losses and NUE from feed to long-term manure utilization

The feeding strategy largely affects the livestock system NUE by the feed-to-food N conversion efficiency (feed NUE), where a high efficiency leads to less N excretion and thus potentially reduced N losses. Decomposition of animal excrements (manure) leads to NH_3 losses, the dominant gaseous N loss from agriculture (Häussermann et al., 2021) besides nitrous oxide (N_2O), a potent greenhouse gas (Aryal et al., 2022; Ravishankara et al., 2009), dinitrogen (N_2), which is harmless to the environment, but a loss from a NUE perspective (Selbie et al., 2015), and NO_3^- leaching, causing contamination of surface and drinking water or eutrophication of freshwater and coastal marine ecosystems (Sebilo et al., 2013). Details of the loss of each N species at each stage of the livestock system (Fig. 1) are described in the following sub-sections.

3.1 Feeding strategies in cattle and pig management

Nitrogen ingested by farm animals is lost either as indigestible protein N through feces in complex structures, which are not easily mineralized, or as metabolized N compounds such as urea in urine. Feed NUE of a productive farm animal corresponds to the ratio of human edible animal products N over feed N ingested. This can be assessed either directly via the measured N content in the animal product, particularly for dairy cows and laying hens, or indirectly via the measurement of N contained in the excreta, where the N amounts excreted is at least 65 % of N intake per day (Groenestein et al., 2019; Xiccato et al., 2005).

3.1.1 Dairy cows

Feed NUE in dairy cows is usually assessed via the comparison of the daily milk protein yields over feed N intake, or by using models, which estimate the cow's lifetime feed NUE. The latter includes N turnover during heifer growth and the meat protein from the slaughtered cow. An effective indicator for short-term alterations in dairy cow feed NUE is milk urea N (Nousiainen et al., 2004), which mostly originates from ruminal NH_3 (Beltran et al., 2021; Kidane et al., 2018). This parameter is regularly analyzed by breeding associations.

Average feed NUE (calculated as milk N over feed N) in dairy cows is approximately 26 % (with a range between 18–35 %; Foskolos and Moorby, 2018; Groenestein et al., 2019). Consequently, on average 74 % of dietary N, amounting to 200–400 g N d^{-1}, are excreted (Foskolos and Moorby, 2018; Pozo-Leyva et al., 2021), particularly as urine N (Kälber et al., 2012; Kapp-Bitter et al., 2023).

The main factor influencing feed NUE in cattle is the dietary balance between feed crude protein (CP) and carbohydrates, as they influence the microbial protein synthesis in the rumen, and consequently also potential NH_3 production, which is ultimately excreted as urine (Van Duinkerken et al., 2011). Reducing dietary CP levels could increase the feed NUE by more than 5 % (Table 2), without severe milk yield depression in both intensive (Chowdhury et al., 2023) and extensive dairy systems (Kidane et al., 2018). A higher feed NUE also means a shift in the share of N excreted with feces and urine towards stable organic N compounds in feces, decreasing the risk of subsequent N losses during manure handling (Broderick et al., 2008; Huhtanen et al., 2008). An improved dietary balance could also be reached using feed supplements (Table 2). Several plant-based supplements (Table 2) have been tested for their potential to counteract ruminal protein excess and reduce urinary N losses, such as: tannin-rich herbs like burnet, chicory or buckwheat (Cheng et al., 2017; Kälber et al., 2012; Kapp-Bitter et al., 2023), tannin-rich extracts (Herremans et al., 2020; Kapp-Bitter et al., 2020), isolated cinnamaldehyde, essential oils (Cantet et al., 2023). These studies show some promising results, but mostly with limited size and at dosages and supplementation schemes which are often not applicable in broad practice. A latent influence of herb phytochemicals from multispecies swards towards better NUE can still be hypothesized based on the experimental evidence.

Challenges in dietary balance also exist for grazing cows, because well managed pasture swards often contain excess dietary protein. Grazing without supplementation of extra carbohydrates may lead to feed NUE as low as 18–20 % (Akert et al., 2020; Leiber et al., 2006). Corn silage or cereals used as supplements showed an increase in feed NUE between 3–7 % (Akert et al., 2020; Almeida et al., 2020; Beltran et al., 2021). Feed NUE could also be improved via changes in the sward composition and e.g. the integration of plantain (*Plantago lanceolata* L.), for its effect on animal metabolism and urine N excretion (Gregorini et al., 2016; Gregorini et al., 2018). The presence of plantain in a grazed sward reduced total N (TN) excretions and N concentration in the urine of dairy cows (Cheng et al., 2017; Totty et al., 2013). However, the positive effect of plantain shows high seasonal variations, ranging from 15 % to 50 % reduction for summer or winter urine applications, respectively (Welten et al., 2019). Timing of supplementation, related to the ruminal degradability of the supplemented carbohydrates, plays an important role for success (Dickhoefer et al., 2022). Moreover, season is an essential factor for feed NUE in grazing systems with better values in spring (Beltran et al., 2021) and particular poor levels in autumn, which are caused by particular high N excess in the swards (Akert et al., 2020). Improving feed NUE in grazing systems does not only mean reduction of soluble N release but also an efficient use of the pasture-born protein for milk or beef in order to

Table 2 Feeding strategies, urine and feces N indicators and potential to increase feed NUE

System	Factor	Effect	NUE (%)	urine N g/d	feces N g/d	References
Dairy cows	Carbohydrate supplementation when grazing	average	25	190	130	1,2,3,4
		NUE shift potential	5.5	−50	0	
	CP reduction to 150 or even 135 g/kg DM in barn	average	27	150	180	5,6
		NUE shift potential	7	−80	0	
	Plant phytochemical supplements	average	26	160	130	7,8,9
		NUE shift potential	2	−45	15	
Beef cattle/heifers	Carbohydrate rich feeds (fiber, starch, oils)	average	26			10
		NUE shift potential	10			
	Lower dietary CP	average	32.5	35	44	11
		NUE shift potential	3	−18	0	
	Plant phytochemical supplements	average	33	43	52	11,12,13
		NUE shift potential	7	−11	0	
	Season (autumn is the problem)Shift means if other season than autumn	average	22	101	94.4	1
		NUE shift potential	8	−20		

Pigs	use feedmill mixtures instead of own (Switzerland)	average	31			14
		NUE shift potential	2			
	Optimize dietary CP/per unit energy (non-linear, optimum in the middle)	average	33	23	6.5	14,15,16
		NUE shift potential	4	−18	−1.1	

[1]Beltran (2021); [2]Akert, 2020; [3]Dickhoefer (2022); [4]Almeida et al. (2020); [5]Chowdhury (2023); [6]Kidane et al. (2018); [7]Herremans (2020); [8]Kapp-Bitter (2023); [9]Kälber (2012); [10]Angelidis (2021); [11]Zhou (2019); [12]Cheng (2017); [13]Fonseca (2024); [14]Sollberger (2013); [15]Acciaioli (2011); [16]Hlatini (2020).

serve human edible protein from non-arable resources (Leiber et al., 2017) at optimal conversion ratios. The specific trade-offs between use of arable land for the supplements and added value by optimizing pastureland utilization need to be calculated in each case. Practically applicable models for such calculations are urgently needed.

3.1.2 Beef cattle

For the feed NUE assessment in beef cattle, equations using total retained feed N, i.e. also including tissues which are not human edible (Angelidis et al., 2021; Beltran et al., 2021) are applied. Additionally, urinary and fecal N excretions (Angelidis et al., 2021), as well as blood urea or NH_3 can be used as indicators. However, the latter can be assessed only in experimental settings. The assessment of feed NUE in beef cattle takes into account the ratio of food or total body protein yields of the slaughtered animals to estimate total feed N intakes. Similar to dairy cows, the physiological basis of N digestion and metabolism can be used to improve feed NUE achieved by lowering urine N excretion, reducing the urine N to fecal N ratio and optimizing grassland protein conversion.

Beef production systems can be clustered into feedlot fattening, grassland-based fattening and suckler cow systems. In feedlot and grassland-based fattening, average feed NUE is 26 % (Table 2) with a range between 10–50 % (Angelidis et al., 2021; Xiccato et al., 2005). Feedlot systems show a higher feed NUE but do not account for N input and losses during feed production (Beltran et al., 2021). When considering human edible food used as feed for cattle, grazing systems show higher feed NUE (Molossi et al., 2020). Suckler cow systems on the other hand show lowest feed NUE values (6 % on mountain pastures and 9 % in lowland grasslands; Estermann et al., 2001). Detailed assessment is complex due to the presence of two trophic levels (lactating cow and growing calf).

Average daily N excretion per beef animal ranges between 100–200 g N with higher shares from urine than from feces (Bougouin et al., 2022; Souza et al., 2024).

Similar to dairy systems, sufficient supply of carbohydrates via fiber, oils, starch and molasses, and low or reduced dietary CP levels increase feed NUE (Angelidis et al., 2021) and shift N excretion from urine to feces (Zhou et al., 2019). Supplementing by-products from the food industry rich in condensed tannins improved feed NUE of fattened Nellore bulls (Fonseca et al., 2024). In the study of Zhou et al. (2019), tannin rich supplements did not affect feed NUE, but decreased absolute and relative urinary N excretion. The experimental within-system potential to increase

NUE with feeding interventions ranges from 3 % with reduced dietary CP (Zhou et al., 2019) up to 10 % with added tannins (Fonseca et al., 2024, Table 2). Between-system differences in feed NUE can be large (Angelidis et al., 2021; Beltran et al., 2021), however, if systems like feedlot versus grazing are compared, more holistic models are needed in order to embed the animal level feed NUE perspective into land-use efficiency, feed-food competition and other relevant sustainability variables (Molossi et al., 2020; Schader et al., 2015).

3.1.3 Fattening pigs

Average feed NUE of commercial pig fattening is around 33 % (Groenestein et al., 2019; Sollberger et al., 2013), with a global range between 10–44 % (Gerber et al., 2014) and a European-North American range of 30–43 % (Shurson and Kerr, 2023, Table 2).

Similar to cattle, the reduction of CP and the ratio of CP per unit of metabolizable energy are the most crucial factors to increase the overall feed NUE (Acciaioli et al., 2011; Hlatini et al., 2020; Shurson and Kerr, 2023; Sollberger et al., 2013). Optimal CP ratios of approximately 150 g CP kg^{-1} feed (Hlatini et al., 2020; Sollberger et al., 2013) generally increase feed NUE (Table 2). If part of the carbohydrates are delivered in the form of dietary fibers, this is not only a benefit for animal health and welfare (Holinger et al., 2018), but may result in a considerable shift of N excretion from urine to feces (Shurson and Kerr, 2023).

Amino acid supply is crucial for NUE in non-ruminants. Therefore, including synthetic or isolated amino acids in diet of pigs is the most developed and propagated pathway to increase feed NUE and reduce dietary CP (Shurson and Kerr, 2023). As isolated amino acids are prohibited in organic systems, these farms rely even more than conventional pig producers on feed components with a particularly high protein quality (Quander-Stoll et al., 2021), in particular with high concentrations of lysine, methionine and threonine, to reach satisfactory feed NUE. In addition to amino acid supplementation, dietary phytase could improve protein digestibility and utilization rates by 2.8 % (range 0.9–4.1 %, Shurson and Kerr, 2023).

Phase feeding is an important approach to reduce N excretion rates (Pierer et al., 2016) and consequently enhance feed NUE. Further factors are genetics and sex (Shurson and Kerr, 2023), with advantage for boars, which are however still seen as a challenge regarding sensory shortcomings. Based on the cited evidence, a within-system improvement of feed NUE up to 5 % appears feasible through dietary improvements.

3.2 Livestock housing

At the housing level, N is mainly lost via NH_3 volatilization from emitting surfaces. This includes floors inside the barn, adjacent outside yards soiled with animal excretions and liquid manure pits situated beneath perforated barn floors, which store some or all animal excretions. The air exchange in animal housings is ensured with mechanical or natural ventilation, and influences the amount of N lost via NH_3 volatilization. Hence, the housing systems should be classified according to their ventilation system, manure management and storage strategy (Guo et al., 2022; Philippe et al., 2011; Sanchis et al., 2019).

Recent results from a large measurement campaign of naturally ventilated dairy housing systems in Germany indicate, that no significant difference in NH_3 emissions between barns with inside slurry storage compared to external storage exist (KTBL, 2024). The same applies to perforated or solid floor which do not differ in NH_3 emissions (Poteko et al., 2019).

For dairy cows, different yearly emission values are available, which rely either on multi-farm measurements (and meta-analyses of multiple measurement studies) or values based on expert judgment. The measured values of loose housing with cubicles, solid or slatted floor and slurry production, which are common systems in Europe, vary between 9.8 and 13.8 kg NH_3 cow^{-1} y^{-1} (Kai et al., 2017; KTBL, 2024; Poteko et al., 2019; Schep et al., 2022; Schrade et al., 2012). Guideline values from e.g. Austria, Germany and the Netherlands are typically somewhat higher and range between 13.0 to 15.8 kg NH_3 cow^{-1} y^{-1} (RAV, 2024; Normenausschuss, 2011; Öttl et al., 2023). For fattening pigs, the yearly NH_3 emissions values range from approximately 3.0 to 3.6 kg NH_3 animal $place^{-1}$ y^{-1}.

Numerous mitigation measures to reduce gaseous N losses in either the barn or the yard are available and reported in the literature. These measures directly influence the formation process of NH_3, and can be divided into: (i) Reduction of the active emission surface: A number of structural-technical solutions for the reduction of emitting surfaces are commercially available and reported, for both dairy and pig housing systems. In addition, an increased cleaning efficiency, i.e. frequent scraping combined with water cleaning can further reduce the volatilization of NH_3 from the emitting surfaces. (ii) Rapid segregation of urine and feces: Segregation is mostly achieved by construction measures, with an adapted surface design either directly on the floor or below slatted floors in the slurry channels.

(iii) Alteration of the physical-chemical conditions: chemical or physical alteration of manure can be achieved by either decreasing pH-level, urease activity, temperature and/or turbulence over the emitting surface. (iv) Air cleaning: NH_3 emissions can be efficiently reduced (up to 90 %) from mechanically ventilated pig and poultry houses using exhaust air scrubbers.

3.2.1 Dairy cows

i) Reduction of the active emission surface: Elevated feed stalls with divider bars in dairy barns prevent cows from moving along the feeding fence, keeping the area clean and reducing the emitting surface. This design reduces NH_3 emissions by 8–19 % and can be easily implemented into newly constructed barns (Table 3; Zähner et al., 2019). Floor systems with sloped slats and ventilation flaps enable a quick urine drainage and minimize the air exchange between slurry pit barn air, and reduce the emissions between 10 % (Janke, 2024) to 50 % (Meijer, 2018). Additional emission reductions of regularly scraped surfaces is the practice of water cleaning after or while scraping on dairy barns with slatted floors, where reductions between 14 % (Ogink and Kroodsma, 1996), 30 % (Kroodsma et al., 1993) or up to 37–40 % (Van Dooren et al., 2022) can be achieved (Table 3). Applied on a plane floor after scraping, an emission reduction of 15 % could be shown (Braam et al., 1997b).

Floors with longitudinal and cross-wise grooves enhance the surface-to-volume ratio and facilitate quick drainage. Early studies in the 2000s showed up to 46 % mitigation with and 35 % without perforated grooves (Swierstra et al., 2001). A recent study found a 35 % reduction with rubber-based flooring (Winkel et al., 2020). However, similar systems in commercial barns showed no significant emission reduction due to site-specific factors (Janke, 2024). Efficiency depends on operational conditions, with potential mitigation ranging from 35 % to 46 % under controlled conditions (Table 3). In essence, results may vary or emissions could even increase without close supervision (Van Bruggen et al., 2024).

ii) Rapid segregation of urine and feces: Sloped floors with adapted scrapers and urine gutter can reduce NH_3 emissions between 20 % (Braam et al., 1997a; Zähner and Schrade, 2020) and 50 % (Braam et al., 1997b; Swierstra et al., 1995). The range depends amongst others from the slope inclination and the positioning of the urine gutter in the housing system (Braam et al., 1997b; Swierstra et al., 1995).

Table 3 Compilation of mitigation measures for housing systems of dairy cows and fattening pigs.

Measure of mitigation potential	NH_3 reduction potential in %		
	Nr. of studies	Median	Individual values
Cattle			
Reduction of the active emission surface			
Elevated feed stalls	1[1]	13.5	
Sloped slats and ventilation flaps	2[2,3]	30	10[3], 50[2]
Frequent scraping combined with water cleaning	4[4,5,6,7]	22.5	14[4], 15[7], 30[5], 38.5[6]
Floors with grooves	3[8,9,10]	35	0[10], 35[9], 46[8]
Rapid segregation of urine and feces			
Sloped floors with adapted scraper and urine gutter	4[11,12,13,14]	35.5	20[14], 21[13], 50[11,12]
Chemical/physical alteration			
Slurry acidification	2[15,16]	36.5	30[15], 43[16]
Urease inhibitor	2[17,18]	39.3	20.5[18], 58[17]
Barn cooling	2[19,20]	positive effect	
Pigs			
Reduction of the active emission surface			

Sloped sidewalls of slurry channels	1[21]	32	
Rapid segregation of urine and feces			
Sloped pit floor with urine gutter and V-shaped Scraper	4[22,23,24,25]	45	40[22], 40[23], 50[25], 54[24]
Chemical/physical alteration			
Slurry acidification	3[26,27,28]	60	39[28], 60[27], 70[26]
Urease inhibitor	2[29,30]	24.5	20[29], 29[30]
Slurry cooling	3[31,32,33]	47	27[31], 47[33], 62[32]
Air cleaning			
Air scrubbers	2[34,35]	80	

[1]Zähner et al., 2019; [2] Meijer et al., 2018; [3] Janke et al., 2024; [4] Ogink et al., 1996; [5] Kroodsma et al., 1993; [6] Van Dooren et al., 2022; [7] Braam et al., 1997b; [8] Swierstra et al., 2001; [9] Winkel et al., 2020; [10] Janke et al., 2024; [11] Swierstra et al., 1995; [12] Braam et al., 1997b; [13] Braam et al., 1997a; [14] Zähner et al., 2020; [15] Kasper et al., 2022; [16] Andersen et al., 2013; [17] Bobrowski et al., 2021b; [18] Bobrowski et al., 2021a; [19] Poteko et al., 2019; [20] Qu et al., 2021; [21] Hagenkamp-Korth, 2023; [22] Loussouarn et al., 2014; [23] Lagadec et al., 2019; [24] Landrain et al., 2009; [25] Hagenkamp-Korth, 2023; [26] Kai et al., 2008; [27] Petersen et al., 2016; [28] Overmeyer et al., 2023; [29] Schulte et al., 2022; [30] Calvet et al., 2022; [31] Andersson et al., 1998; [32] Myczko et al., 2007; [33] Hagenkamp-Korth, 2023; [34] Melse et al., 2005; [35] Van Der Heyden et al., 2015.
All measures act on liquid manure.

iii) Alteration of the physical-chemical properties: The few studies available on the effects of acidification in dairy cow housing systems suggest an average mitigation potential between 30 % (Kasper et al., 2022) and 43 % (Andersen, 2013) in naturally ventilated dairy barns with slatted floors. With the application of urease inhibitors in the barn, a reduction of 10 % (Bobrowski et al., 2021a) and up to 58 % (Bobrowski et al., 2021b) can be achieved. Lower reductions were measured in naturally ventilated barns with solid floors. At the same time, substantial emission reductions were measured in a mechanically ventilated experimental dairy barn with slatted floor (Table 3). Several meta-studies have shown a reduction of emissions with reduced barn air temperature (Poteko et al., 2019; Qu et al., 2021). Hence, insulation of roofs, additional air conditioning, and shadowing exercise yards will most likely lead to lower NH_3 emissions, but no data exists for a quantification of the expected effect.

3.2.2 Fattening pigs

i) Reduction of the active emission surface:

With sloped, diagonal sidewalls of slurry channels, the contact area of the slurry surface with the surrounding air is reduced. Compared to a slurry channel with vertical sidewalls, the sloped walls provide significantly less emitting surface at the same filling level (Ndegwa et al., 2008; Philippe et al., 2011). The German administrative directive for air (TA-Luft) lists this measure as best available technology with a mitigation potential of 50 %, while latest results from case-control measurements showing a mitigation of 32 % for sloped sidewalls of slurry channels (Table 3; Hagenkamp-Korth, 2023).

ii) Rapid segregation of urine and feces: A partly slatted floor, where the floor of the pit has a slope of approx. 10 % and is frequently cleaned with a V-shaped scraper and a central urine gutter can reduce emissions between 40 % (Lagadec et al., 2019; Loussouarn et al., 2014) and 54 % (Landrain et al., 2009) as shown for mechanically ventilated pig barns (Table 3). Similar mitigation potentials were measured in naturally ventilated pig barns with exercise yards (Hagenkamp-Korth, 2023).

iii) Alteration of the physical-chemical properties: Studies on slurry acidification in pig barns report mitigation effects of 50–70 % (Kai et al., 2008; Petersen et al., 2016) in commercial systems with partly in-barn slurry storage, and 39 % for retrofitting systems with full in-barn storage

(Overmeyer et al., 2023). Many best available technology lists suggest slurry acidification with expected mitigation potentials ranging from 60 % (France), 64 % (Denmark), up to 65 % (Germany).

Several studies investigated urease inhibitors in pig fattening houses with fully-slatted floor and under-floor slurry pit. The application takes place either on the surface of the slats (Hagenkamp-Korth, 2023; Schulte et al., 2022), or under the slats directly on the slurry (Calvet et al., 2022). For the slat-surface application, a relative emission reduction of 19–21 %, while for the application on the slurry, a mitigation of 29 % was found (Calvet et al., 2022; Hagenkamp-Korth, 2023; Schulte et al., 2022).

The cooling of slurry surface in manure channels was investigated under controlled conditions in the lab and under real circumstances with different designs, with mitigation potentials between 47 % (Andersson, 1998) and 67 % (Myczko et al., 2007). A more recent measurement campaign in Germany investigated the mitigation efficiency of cooling fins (Hagenkamp-Korth, 2023). With a reduced surface temperature ≤15 °C, a mitigation potential for NH_3 emissions of 47 % was measured. Overall, the cooling of slurry can mitigate NH_3 emissions in the range of 47–67 %.

iv) Air cleaning: Exhaust air scrubbing can be achieved using biotrickling filters, where NH_3 is converted to NO_3^- by nitrifying organisms, or with acid scrubbers, where water is acidified to a pH < 4. Exhaust air scrubbers achieve an NH_3 removal between approximately 50−90 %. Removal rates superior to 80 % are possible if the air scrubbers are properly operated. However, this can only be achieved if NH_3 emissions are tackled with an automated process control to adjust the pH and the N content of the process water (Melse and Ogink, 2005; Van Der Heyden et al., 2015). The trapped N can be used as fertilizer and contribute to improving the fertilizer NUE.

3.3 Manure processing

The processing of manure contributes indirectly to either a reduction or an increase in N losses as it changes individual manure properties, such as pH, ammonium (NH_4^+) or dry matter (DM) content (Fangueiro et al., 2015a; Möller and Müller, 2012).

3.3.1 Liquid manure

Solid-liquid separation is a common and relatively simple processing technique to mechanically separate liquid manure into a solid fraction (SF) and a liquid fraction (LF) (Fangueiro et al., 2012b; Tampio et al., 2016), where mostly a screw press separator or a decanter centrifuge is employed. Solid-liquid separation can pursue different objectives, e.g. the reduction of the fiber content to reduce the viscosity of the LF, increase the manure NUE by reducing the C:N ratio in the LF or a more targeted application of nutrients to crops (Foged et al., 2012). The NH_4^+-N content does not substantially differ between the LF and the raw liquid manure, whilst the DM content strongly declines in the LF due to solid liquid separation (Hjorth et al., 2011; Møller et al., 2000). Of the initial total fresh matter, 75–80 % ends up in the LF and 20–25 % in the SF (Fangueiro et al., 2012b). While the LF has a high concentration of mineral N, potassium and low organic matter contents, most phosphorus and organic C are transferred into the SF (Foged et al., 2012; Hjorth et al., 2008; Möller et al., 2010; Tampio et al., 2016).

The solid-liquid separation of liquid manure is mostly a prerequisite for further processing such as NH_3 stripping or ultrafiltration (described later).

Anaerobic digestion converts the easily degradable carbon in manure into carbon dioxide and methane, with a carbon degradation potential between 30 % and > 90 % (Möller and Müller, 2012). Anaerobic digestion changes the manure properties and leads to an increase in total ammoniacal N (TAN):TN ratio as well as pH and a decrease of DM, volatile solids and viscosity (Möller, 2015; Möller and Müller, 2012).

Nitrification of the NH_4^+-N in the LF or in segregated urine can be achieved in specific reactors where under aerobic conditions, microorganisms convert NH_4^+ or dissolved NH_3 to nitrite and then to NO_3^- (Akizuki et al., 2021; Pelayo Lind et al., 2021; Takemura et al., 2017). The process consumes oxygen and releases protons, and thereby acidifies the substrate (Fumasoli et al., 2016; Henze and Comeau, 2008; Rodríguez-Gómez et al., 2021). Other than acidification with an external acid, lowering pH by nitrification will not affect the liming effect of slurry, because the processes are merely shifted from the soil to the reactor.

Plasma activation is a new method of acidification that employs electrically powered air plasma to generate nitric oxide (NO), which is readily oxidized to nitrogen dioxide (NO_2). Nitrogen dioxide is dissolved in water to create nitric acid (HNO_3) and NO_3^- in the aqueous phase, which is then used to treat liquid manure or urine, reducing the pH. It also increases TN content of the liquid manure (Graves et al., 2019).

Ammonia stripping or vacuum evaporation aims to reduce NH_4^+-N contents and/or the water content of the LF, while volatilized NH_4^+-N is collected in a gas scrubber. In an inventory of two systems Fechter et al. (2023)

found that 73–87 % of TN and 60–93 % of NH_4^+-N are in the LF. A subsequent vacuum evaporation of the LF removed 34 % of the initial water and left a DM-rich concentrate. At the same time, an ammonium sulfate solution containing 21 % of initial TN and 34 % of initial NH_4^+-N is produced.

Full-scale cascaded membrane filtration systems comprising prior separation with decanter centrifuges, microfiltration and reverse osmosis via membranes are more difficult to apply (Van Puffelen et al., 2022). Such systems are able to process liquid manures resulting in one or two solid fractions and, a liquid microfiltration concentrate that can be taken to different purification levels, up to water purified through reverse osmosis (Van Puffelen et al., 2022). The goals are to separate NH_4^+-N and P and to reduce the amounts of water in the system. Approximately 35 % of TN and 27 % of the NH_4^+-N ended up in the SF, another 35 % of TN and 39 % of the initial NH_4^+-N in the microfiltration concentrate and 25 % of the TN and 34 % of the initial NH_4^+-N in the reverse osmosis concentrate.

3.3.2 Solid manure

Similar to liquid manure, solid manure can be anaerobically digested (Section 3.3.1). Composting is an aerobic microbial process where organic matter degrades, resulting in a more stable product (Jensen, 2002; Tambone et al., 2015). Consequently it is often used to treat solid manure and the SF of digestates after solid-liquid separation. Organically bound nutrients are partially mineralized, but conversely, inorganic nutrients are also partially bound organically (e.g. N; Bernal et al., 2009). The smaller the C:N ratio and the higher the NH_4-N content of the solid manure, the higher the potential of gaseous N losses. This applies to digestates, pig and poultry manure (Hao et al., 2004; Hüther et al., 1997; Kirchmann, 1985; Mahimairaja et al., 1994; Petersen and Sørensen, 2008b), and cattle manure (Petersen et al., 1998).

Drying and pelletizing of solid manures aims to produce a dry, stable, hygienic, homogeneous and transportable product. NH_4^+-N is largely removed by drying (Sistani et al., 2001) and released as NH_3 if not captured by an air scrubber. Reducing the pH (e.g. by adding sulfuric acid) before drying can significantly reduce N losses (Pantelopoulos et al., 2016; Sommer et al., 2015).

All processing methods that keep N in a closed environment while reducing the amounts of fresh matter or decreasing the viscosity of the final product may lead to reduced emissions during storage or after field application. Conversion of manure N into a mineral N fertilizer via air scrubbing will result in a mineral fertilizer with a high NUE. Major concerns arise where a SF is generated containing relevant amounts of TAN, very prone to gaseous N losses. A potential mitigation measure is to dry the solids combined with TAN recovery in an air scrubber. However, new processing technologies like NH_3 stripping and/or vacuum evaporation, plasma activation or full-scale cascaded membrane filtration are energy demanding technologies, which are not evaluated for the full life cycle chain and costs (Angouria-Tsorochidou et al., 2022). There are no or only scarce data for overall manure NUE for all products generated in the processing chain.

3.4 Manure storage

Liquid and solid manures are either directly applied to the fields or stored for a certain period before application. Liquid manure is anoxic and denitrification losses may only occur in sites where oxygen interacts with liquid manure, such as in crusts formed in the tanks (Amon et al., 2006a; Amon et al., 2006b). In contrast, oxygen can diffuse into solid manures (Amon et al., 2006b) and cause substantial losses of N_2, NO_x, and/or N_2O.

For both liquid and solid manures, NH_3 emission play a significant role for N losses (Amon et al., 2006b; Webb et al., 2012). The review by Kupper et al. (2020) showed NH_3 baseline emission factors of uncovered storage of 0.08 g NH_3 m^{-2} h^{-1} and of 16 % of TAN for cattle liquid manure and of 0.24 g NH_3 m^{-2} h^{-1} and of 15 % of TAN for pig liquid manure. Kupper et al. (2021) have shown that emissions per unit of TAN depend on the annual TAN flow into the storage, its area, the annual average emission on an area basis and the liquid manure residence time in the store. These parameters exhibit high variation between farm production systems and are not reflected by the emission factors per unit of TAN. For example, Kupper et al. (2021) determined an emission factor of 3.3 % of TAN with an average liquid manure residence time of 51 days due to frequent liquid manure application after the harvest of fodder and to arable crops instead of mineral fertilizer. The baseline emission factors for N_2O are 0.002 g N_2O m^{-2} h^{-1} or 0.13 % of N for cattle liquid manure and 0.002 g N_2O m^{-2} h^{-1} or 0.10 % of N for pig liquid manure (Kupper et al., 2020).

NH_3-N emissions from cattle solid manure have been reported in the range between approximately 3 % and 15 % of TN (Hou et al., 2015; Lemes et al., 2023; Webb et al., 2012). For solid pig manure, the NH_3-N emissions lie between 20 % and 30 % of N (Petersen et al., 1998; Webb et al., 2012). This correspond to approximatively 45 % of TAN (cattle) and

80 % of TAN (pig), under the assumption that of the TN, 20 % and 25 % are TAN, for solid manure from cattle and pigs respectively. N_2O-N emissions are < 1 % of N but data remain sparse (Webb et al., 2012). N losses due to NO_3^- leaching from manure heaps are less than ca. 5 % of TN (Doody et al., 2013). N losses as N_2 and NO_x from solid manure are also expected, but data are lacking.

A limited number of studies have investigated N losses from SF. For example, TN losses of the SF of untreated and anaerobically digested pig manure can vary between 10 % and 55 %, as determined by Petersen and Sørensen (2008b) using a mass balance, and Hansen et al. (2006) found N_2O emissions of 4.8 % of N for SF from pig liquid manure. Data on N_2O losses are however even more limited and uncertain.

3.4.1 Liquid manure

3.4.1.1 Technical measures to reduce emissions

Covering liquid manure reduces NH_3 volatilization by decreasing the turbulence at the emitting surface. Storage coverings include impermeable structural covers (e.g., lid, tent covering), impermeable synthetic floating covers, permeable synthetic floating covers (e.g., plastic fabrics, expanded clay) and permeable natural floating covers such as chopped straw. Slurry bags are another option (Kupper et al., 2020; Sommer et al., 2022). The average reduction in NH_3 emission using covers ranges between about 50 to 90 % (Kupper et al., 2020). The formation of a natural crust on the liquid manure tank surface also reduces the NH_3 emissions (Amon et al., 2006a), but its formation and stability can be controlled to a limited extent only (Kupper et al., 2020; Wood et al., 2012) and thus is mostly not considered as a targeted NH_3 emission abatement technique.

The difference in NH_3 emissions of the pig separated LF are small compared to the untreated liquid manure, while storing the LF of cattle liquid increases the NH_3 emissions by 23 % but decreases N_2O emissions by 43 % (Kupper et al., 2020). Anaerobically digested liquid manure tends to have higher NH_3 emissions compared to raw liquid manure (Kupper et al., 2020). The acidification of liquid manure to a pH of 5.5 to 6.0 reduces the NH_3 emissions by 71–77 % compared to untreated liquid manure. Data on the impact of anaerobic digestion and acidification on N_2O emissions are sparse and thus still uncertain (Kupper et al., 2020). The effect of nitrification and plasma treatment on NH_3 emissions from storage have not yet been investigated but due to the lower pH of the liquid manure after the treatments, it seems likely that the emissions of NH_3-N are reduced. It can be assumed to potentially have a strong effect on N_2O-N emissions (Oenema and Velthof, 1993; Stevens et al., 1995) depending on the targeted pH (Oenema and Velthof, 1993), but experimental data are lacking.

3.4.1.2 Liquid manure management options to reduce emissions

Disturbance of the liquid manure surface due to stirring induces increased emission levels for NH_3 (Kupper et al., 2020). This is mainly due to breaking the barrier to the gas transfer provided by the natural crust and the time after stirring required to build up a new crust (Kupper et al., 2021). Therefore, reducing stirring of liquid manure stores substantially decreases NH_3 emissions. Dilution of cattle liquid manure has been shown to reduce NH_3 and N_2O emissions by approximately 50 % (Kupper et al., 2020), by i.e. facilitating slurry infiltration into the soil and fast NH_4^+ binding to soil surfaces.

3.4.2 Solid manure

3.4.2.1 Technical measures to reduce emissions

Covering solid manure stores with impermeable plastic sheets reduces the NH_3 emissions by 50 % or more (Chadwick, 2005; Lemes et al., 2023; Pardo et al., 2015) by lowering the turbulence at the emitting surfaces, where covering the entire heap reduces emissions more than partial covering. Storing solid manure in a container with a cover reduces emissions to a similar extent (Lemes et al., 2023). N_2O emissions were either unaffected or reduced by covering due to less contact with oxygen from the atmosphere (Chadwick, 2005; Hansen et al., 2006; Pardo et al., 2015). Emissions of NH_3 decrease with increasing density of manure heaps (Webb et al., 2012), i.e., compacting manure heaps contributes to emission reduction as well.

Hou et al. (2015) found a NH_3 emission factor for composting of 39 % N (range 17.3–45.3 % of TN). Thomsen (2000) found higher losses of NH_3 for composting of cattle manure (46 % of TN) compared to anaerobic storage (18 % of TN). This is in line with the review of Pardo et al. (2015), reporting substantially higher emissions of NH_3 for composting than for anaerobic storage. N_2O emissions do not differ in emission between composting and anaerobic storage (Hou et al., 2015; Pardo et al., 2015).

3.4.2.2 Management options to reduce emissions

Frequent removal and field spreading of solid manure stores or direct application of solid manure extracted from the housings can be an option to reduce emissions from storage. However, due to conversion of TAN to organic bound N during storage as found by Kirchmann and Witter (1989), less TAN is available that can be volatilized as NH_3 during field application if solid manure is stored. Therefore, the potential to mitigate NH_3 emissions due to direct application of solid manure is uncertain but likely to be limited due to N immobilization during storage although the extend thereof is not known. Furthermore, it will depend on the time slot between spreading and soil incorporation (Webb et al., 2004; Webb et al., 2005; Webb et al., 2012).

3.5 Manure application

The viscosity of the liquid manure influences its infiltration into the soil whereas a high infiltration rate reduces emissions. The DM content is mostly used as a proxy for viscosity, where an increasing DM stands for a higher viscosity.

Baseline emissions for the application of undiluted liquid manure from cattle and pigs by broadcasting are approximately 30–50 % of TAN (ALFAM2 model, version 3.9; Hafner et al. 2024). For solid manure, baseline emissions for spreading without incorporation can be estimated at ca. 60 % to 80 % of TAN based on Webb et al. (2012). However, these data are uncertain since they are mostly based on wind tunnel measurements, which are likely to overestimate emissions (Hafner et al., 2024; Pedersen et al., 2024; Sommer and Misselbrook, 2016). Van Der Weerden et al. (2021) reported N_2O emission factors for liquid cattle and pig manure ranging from ca. 0.003 to 0.01 kg N per kg of N applied. Skiba et al. (2021) suggest an emission factor for NO of 0.006 kg N per kg of N applied, but considered it highly uncertain. The emission factors for N_2O and NO apply to both liquid and solid manure.

3.5.1 Liquid manure

3.5.1.1 Technical measures to reduce emissions

With low emission application techniques for liquid manure the size of the emitting surface is reduced by placing the liquid manure in bands close to the soil surface or soil injection while avoiding soiling of plant foliage. Concomitantly, the liquid manure bands are more shaded by plants, which may reduce the temperature and the turbulence at the emitting surface. Webb et al. (2010) reported abatement efficiencies of NH_3 emissions for the trailing hose of 35 % on average (range: 0–75 %), for the trailing shoe of 65 % (range: 38–74 %) and for open-slot injection of 75 % (range: 23–99 %). These values, were confirmed largely by later studies (Hafner et al., 2024; Häni et al., 2016; Pedersen et al., 2020). Pedersen et al. (2020)

showed lower NH_3 emissions by 19 % for trailing shoes application compared to a trailing hose, and highlighted that it is crucial to apply liquid manure with a trailing hose at the ground surface, as applications at 20 cm above the canopy increased NH_3 emissions by 40 %.

Emissions for N_2O and NO_x are generally lower by a factor of 10 to 1000 as compared to NH_3 emissions and can be omitted for considerations regarding N efficiency. This is in line with studies on N budgets where N_2O and NO were considered as irrelevant for N losses (Ammann et al., 2009; Jones et al., 2017). Despite these low losses, there is large consensus that NH_3 abatement techniques, which significantly increase the risk of greater GHGs emissions, namely N_2O, should be avoided. The use of methods which reduce NH_3 emissions likely lead to an increase of the amount of manure-N entering the soil and thus enhance direct emissions of N_2O as has been shown by several studies (e.g. Maris et al., 2021; Webb et al., 2014). However, indirect emissions of N_2O, arising following the deposition of NH_3 to land, will be reduced (Webb et al., 2010) if techniques for emission reduction from manure application are employed. Also, the use of mineral N-fertilizers which are a source of N_2O emissions can be reduced (Velthof and Mosquera, 2011). It is unclear to which extent higher direct and indirect emissions compensate each other because several factors can be relevant: (i) uncertainties of emission factors used for direct and indirect N_2O emissions, whereas the latter are likely to be highly uncertain; (ii) the extent to which higher NH_3 losses due to not using low emissions techniques would be compensated by farmers with higher application rates of mineral N-fertilizers. It has also to be noted that mineral N-fertilizers could have lower emission factors for N_2O than manure (Petersen et al., 2023). In contrast, Velthof and Mosquera (2011) found that N_2O emission from a nitrate fertilizer applied to grassland is higher than from cattle slurry. Additionaly, mineral N-fertilizers applied in combination with manures have been shown to exhibit higher N_2O emissions than sole application of manure (Charles et al., 2017).

A study based on literature data and model calculations including both the LF and the SF found a decline in NH_3 losses of up to 12 % depending on the application technique and the liquid manure type (Pedersen et al., 2022b). Emission data from field application of urine segregated in the housing compared to non-segregated liquid manure are scarce. Katz (1996) compared liquid cattle manure consisting of urine and feces with cattle urine from a traditional tied housing system, where urine was drained and stored separately from solid manure, and showed that the emissions were similar for both manure types. Whitehead and Raistrick (1993) found NH_3 losses of cattle urine at laboratory scale of 26 % of urinary N applied and Lockyer and Whitehead (1990) from field application using wind tunnels between 4 to 27 % of urinary N applied.

During anaerobic digestion, the TAN content and pH of the liquid manure increases while DM and viscosity declines, inducing opposing effects on NH_3 emissions (Möller, 2015). This seems to be confirmed by Pedersen and Hafner (2023) who did not find a consistent difference in emissions from digested and undigested liquid manure. The available literature data predominantly refers to liquid manure without co-substrates. Currently, additional feedstocks are used as co-substrates with livestock slurries in agricultural biogas plants. This may lead to higher DM contents which can increase NH_3 emissions (Efosa et al., 2023). Yet, insufficient emission data are available for the resulting liquid digestates, and an assessment of their emissions remains difficult (Pedersen and Hafner, 2023).

In their review, Fangueiro et al. (2015a) and Fangueiro et al. (2015b) found emission reduction rates between 15 % and 80 %, based on the utilization of different acids for cattle and pig liquid manure. This was confirmed by the ALFAM2 model, which is an empirical model based on a series of experimental datasets of acidified liquid manures (Hafner et al., 2019). It predicts a reduction in emissions by ca. 35 % if liquid manure is acidified to a pH of 6.4 as compared to untreated liquid manure with a pH of 7.5. More recent data have confirmed these earlier findings (Pedersen et al., 2022a; Pedersen and Nyord, 2023; Seidel et al., 2017). In an experiment where sulfuric acid was added to digestate to cover the crop sulfur requirement, the pH was lowered in the digestate by approx. 0.3 units and the NH_3 emissions were reduced by 13 % on average as compared to the non-acidified digestate (Pedersen and Nyord, 2023).

Rollett et al. (2023) have found an NH_3 emission reduction of almost 80 % for a plasma treated digestate compared to the untreated digestate. However, when combined with inorganic fertilizers, N_2O emissions increased substantially (Lloyd et al., 2024).

Besides acids, Mccrory and Hobbs (2001) found no effect of additives (e.g. biochar) on NH_3 emissions reduction (Zhang et al., 2023). This was confirmed by more recent studies where additives were specifically investigated in the context of liquid manure application (Owusu-Twum et al., 2017; Sun et al., 2014). Emission data on urease inhibitors added to liquid manure remain sparse and inconclusive (Lasisi et al., 2020; Li et al., 2014). Therefore, additives are not further discussed in this context.

There are treatments which can increase N_2O emissions. In a field experiment lasting over 83 days, cumulative N_2O fluxes were found to be almost five times greater from plasma treated pig slurry compared to the untreated slurry (Lloyd et al., 2023). In an incubation study, the N_2O emission from the concentrated LF (mineral concentrate) of pig slurry was higher than that from untreated slurry (Velthof et al., 2020). The discussion if slurry acidification increases N_2O emissions is still ongoing. In a field experiment over two years using cattle slurry applied to grassland, N_2O emissions from slurry acidified to pH 6.5 and 6.0 using H_2SO_4 tended to be higher than from non-acidified slurry, although the differences were not statistically significant (Seidel et al., 2017). In an incubation experiment, LF of acidified pig slurry exhibited statistically significant higher N_2O emissions than untreated slurry in soil near saturation but vice-versa in soil at field capacity, while the solid fraction was less affected by acidification regarding N_2O emissions (Goméz-Muñoz et al., 2016). Results from other studies suggest that slurry acidification using sulfuric acid does not influence direct N_2O emissions (Fangueiro et al., 2015b; Fangueiro et al., 2018; Malique et al., 2021; Nyameasem et al., 2023).

3.5.1.2 Management options to reduce emissions

Incorporation of liquid manure disrupts air exchange between liquid manure and the atmosphere thereby reducing NH_3 emissions. Webb et al. (2010) reported an emission abatement of at least 90 % due to immediate incorporation of liquid manure by plowing. Any delays in incorporation drastically reduces the potential mitigation effect (Hafner et al., 2018). The ALFAM2 model suggests an emission reduction by approximately 80 % due to deep incorporation (mouldboard plow) and by ca. 60 % with shallow incorporation (harrow) within one hour after liquid manure application. Even after 96 h, an emission reduction of ca. 15 to 25 % can be achieved.

Dilution of liquid manure with water reduces the DM content and thus enhances soil infiltration, and reduces the TAN concentration. According to the ALFAM2 model (version 3.9), a dilution of 1:1 (liquid manure: water) reduced the NH_3 emissions from cattle and pig liquid manure by approximately 30 % and 40 % applied with a trailing hose and broadcasting, respectively (Hafner et al., 2019). (Mkhabela et al. (2009) observed an emission reduction of pig liquid manure by 41 %, 31 % and 25 % for a dilution of 1:1, 1:0.5 and 1:0.25, respectively. Stevens et al. (1992) reported that a 1:1 dilution of cattle liquid manure reduced NH_3 emissions by 50 % compared to undiluted liquid manure.

In the early morning hours and in the evening, temperature and wind speed are often lower. In addition, dew on the leaves of the canopy can absorb NH_3. These effects might explain the lower emissions (ca. 25 to 35 %) when liquid manure was applied in the evening or in the morning as compared to noon (Häni et al., 2016).

Several experiments demonstrated that emissions after liquid manure application are temperature dependent. In 16 experiments, emissions in the summer months were higher by ca. 40 % and 65 % than in fall or spring for broadcast and trailing hose application, respectively (Häni et al., 2016; Huijsmans et al., 2018). Hafner et al. (2019) reported an increase in emissions by 50–120 % with an increase in temperature from 13 to 35 °C based on a scenario from the ALFAM2 model. A model from Pedersen et al. (2021b) based on data from 19 experiments using dynamic chambers revealed a positive response of the cumulative NH_3 emission on the temperature up to approximately 14 °C. Above that, the temperature effect was found to be insignificant.

For an increase of wind speed from 2.7 m s^{-1} to 10 m s^{-1}, Hafner et al. (2019) reported an increase in emissions by 13 % and approximately 30 % for broadcasting and low emission application techniques, respectively.

Rainfall can substantially reduce the emission as shown by field experiments (Sanz-Cobena et al., 2019) and filed observations (Hafner et al., 2019). At a precipitation rate of 1 mm h^{-1}, the ALFAM2 model (version 3.9, Hafner et al., 2024) predicts an emission reduction for undiluted liquid manure between 50 % and 60 % with higher numbers for cattle liquid manure and band spreading than for pig liquid manure and broadcasting. In an experiment with simulated rainfall of 6 mm m^{-2} after pig liquid manure application NH_3 losses were reduced by 45 % (Mkhabela et al., 2009). It can be assumed that the shorter the time span between liquid manure application and rainfall, the higher the emission reduction.

A higher canopy height can provide additional shading and a reduction of the turbulence of the emitting surface if liquid manure is band spread (Maguire et al., 2011). Häni et al. (2016) found an emission being lower by almost 50 % if liquid manure was applied onto a ley with a canopy height of 18 cm as compared to 10 cm.

3.5.2 Solid manure

3.5.2.1 Technical measures to reduce emissions

Anaerobic digestion of solid manures could be an option to reduce NH_3 emissions from solid manures. In the digester, solid substrates are converted to liquids, which can be applied using low emission techniques. In contrast, the only technique to reduce emissions from solid manure application is incorporation, which is not applicable e.g. in established crops during the growing season, for grassland and no-till systems. However, insufficient emission data are available for liquid digestates that include a large proportion of solid manure (Pedersen and Hafner, 2023). As a result, the impact of solid manure liquefaction due to anaerobic digestion on emissions after field application remains unclear.

3.5.2.2 Management options to reduce emissions

The review of Webb et al. (2013) showed an emission reduction between 80–90 % for solid manure incorporated within less than four h after plowing depending on the device used (disc, harrow, chisel plow), respectively. Incorporation after more than 24 h, still resulted in 30–50 % emission reduction.

Mean annual N_2O emissions were found to be higher from autumn than spring applications for several manure types applied to wheat (Bell et al., 2016). No effect due to incorporation of cattle, pig farmyard manure and other solid manure types by disc or tine on N_2O emissions after 60 days was observed but incorporation by plow increased direct emissions of N_2O compared with surface application. Direct emissions of N_2O were substantially higher at the site with a sandy soil than at the site with the heavy clay soil (Webb et al., 2014). Webb et al. (2016) stated that immediate incorporation by plow increased direct N_2O emissions from the sandy soil and that all methods of immediate incorporation in autumn increased modeled estimates of NO_3^- leaching. However, these increases were only a small proportion of the NH_3-N conserved, which was probably either recovered by the crop or retained by soil particles. Therefore, concerns regarding ‘pollution swapping’ should not be a barrier to the immediate incorporation of solid livestock manures to reduce emissions of NH_3.

The emissions from solid manure last over a larger time period, which has different implications regarding the influence of meteorological conditions on the emission level compared to liquid manure. The timing over the day is likely to be less important for solid manure than for liquid manure, but emission data from solid manure addressing the timing of application are scarce. It can be expected that the emissions are higher with spreading in the warm seasons, as Bell et al. (2016) found NH_3 emissions to be greater from spring than autumn applications although the differences were not statistically significant.

Shah et al. (2012a,b) investigated the influence of rainfall amount on NH_3 emissions and found an increase of the apparent N recovery (ANR) by ca. 20 % and 30 % with 5 and 10 mm of irrigation after application of solid cattle manure. Additional data addressing the influence of other meteorological parameters on NH_3 emission are sparse.

3.6 Manure NUE

In this section we summarize field studies on manure NUE in the application year (*short-term*) and after repeated application during several years (*long-term*) in cropland and grassland systems. We discuss major drivers of manure NUE from application to crop N recovery (Fig. 1), and the potential to improve manure NUE in croplands and grasslands. Losses by NO_3^- leaching are major driver determining *long-term* NUE, and are discussed in this section while NH_3 losses determining primary *short-term* NUE were already presented in Section 3.5.

3.6.1 System boundaries

The studies reported here refer to manure NUE measured in European field experiments from application (after storage) to crop N uptake for solid or liquid, cattle and pig manures.

To quantify NUE we used field studies carried out in Europe using manure alone (not combined with other organic or mineral fertilizers). We excluded laboratory incubation experiments, pot experiments, and field experiments with excessive manure application rates or with manure processing other than selected. Only for ^{15}N experiments we included field studies outside Europe. Table 4 shows the classification of the criteria for studies selection to summarize manure NUE from cropland and grassland (Section 3.5.6).

3.6.2 Definitions of manure NUE

The fertilizer NUE (NUE_{fert}) defines the amount of additional N recovered in the crop from fertilizer, which can be manure or a mineral N fertilizer (Congreves et al., 2021; Jensen, 2013). It can be calculated with the *difference approach* from the increase in N uptake between a fertilizer treatment ($N_{crop\ fert}$) and a zero N fertilization control treatment ($N_{crop\ control}$) divided by the amount of N applied (N_{fert}) or by the difference in N uptake of two different fertilization treatments ($N_{crop\ fert1} - N_{crop\ fert2}$) divided by the difference of fertilizers N applied ($Nfert_1 - Nfert_2$).

Table 4 Selection criteria for European studies considered.

Livestock category[a]	**Manure type**	**Application technology**	**Processing**[b]
Cattle (43)	Solid (26) (cattle 21, pig 5)	Broadcast (15; liquid, solid)	Without (45)
Pig (22)	Liquid 49 (cattle 28, pig 21)	Band application (7; liquid)	Separation (2)
		Injection (28; liquid)	Digestion (11)
			Acidification (2)
			Composting (3)

Criteria for exclusion of studies or treatments within studies were: Combined mineral/manure treatments; very high or very low amounts of N fertilization which cannot cover or exceed crop demand largely.
Numbers in brackets indicate number of studies with treatments meeting selection criteria.
[a]Studies with ^{15}N labeling, which focus on processes, were included also from North America.
[b]Liquid solutions from manure processing with ultrafiltration/reverse osmosis are discussed separately.

NUE is expressed as a percentage of fertilizer N applied:

$$NUEfert\,(\%) = \frac{N_{crop\ fert} - N_{crop\ \ control}}{N_{fert}} \times 100$$

or

$$NUEfert\,(\%) = \frac{N_{crop\ fert\ 1} - N_{crop\ \ fert\ 2}}{N_{fert\ 1} - N_{fert\ 2}} \times 100$$

The *difference approach* builds on the assumption that the additional N uptake by the fertilized treatment is only derived from the applied additional N fertilizer, implying no interaction with soil derived N and type or amount of applied fertilizer and that no other element than N would limit plant growth (Jensen, 2013).

In most experiments, for practical reasons, only N uptake in the aboveground plant parts is determined. The NUE calculated in this way is termed the *apparent fertilizer NUE.*

In addition, using *^{15}N tracers* allow a source specific evaluation of NUE e.g. for TN, if organic and mineral N forms are labeled homogenously or for a specific source like mineral or organic N forms if only one of the sources are labeled.

The NUE of the *^{15}N tracer approach* can then be calculated as

$$NUEfert\,(\%) = \frac{N_{cropfert} \times \%N_{dff}}{N_{fert}} \times 100$$

whereas N derived from fertilizer (N_{dff}) is calculated as

$$Ndff\,(\%) = \frac{N^{15}_{fraction}\,Crop_{fert}}{N^{15}_{fraction}\,Fertilizer} \times 100$$

In *long-term* experiments lasting several decades, the fertilizer NUE can also be calculated with a *balance approach*. The fertilizer NUE in the *balance approach* is defined as harvested N, which is removed from the field ($N_{crop\ fert}$) divided by N fertilizer inputs over several years.

It can be calculated as

$$NUEfert\,(\%) = \frac{N_{crop\ fert}}{N_{fert}} \times 100$$

In the case of additional non-fertilizer N inputs (N_2-fixation, atmospheric deposition, seeds, and soil N stock change), the calculation with the *balance approach* can be normalized for fertilizer NUE according to Oberson et al. (2024):

$$NUEfert\,(\%) = \frac{N_{crop\ fert} - N_{seeds} - N_{fixation} - N_{deposition} + \Delta N_{soil}}{N_{fert}} \times 100$$

Over the *long-term* the *difference approach*, the *^{15}N tracer approach* and the *balance approach* tend to converge to similar NUEs (Quan et al., 2021).

The *mineral fertilizer equivalent* (MFE) of manure enables to quantify the amount of mineral N fertilizer that can be substituted by manure N:

$$MFE\,(\%) = \frac{NUE_{manure}}{NUE_{\min fert}} \times 100$$

Short-term manure NUE is defined as the NUE in the application year by the main crop, which is typically one crop per season in European temperate climates. *Short-term* manure NUE is calculated in most studies with the *difference approach* and given as *apparent NUE* or with the *^{15}N tracer approach*. Results from both methods are shown separately in this section.

The *long-term* manure NUE includes the legacy effects of residual N in the soil that has not been used in the application year and is reported from experiments with repeated manure application with a duration longer than one decade. It is calculated either with the *difference approach* or the *balance approach*.

3.6.3 Factors affecting manure NUE

3.6.3.1 Short-term manure NUE

The *short-term* manure NUE is influenced by manure composition, the dose and timing of application in relation to the crop requirement, soil properties, and the environmental conditions during and after application. The most important predictors of the *short-term* availability of manure-N are the mineral N content (as NH_4^+-N or NO_3^--N), the C:N ratio and the composition of the organic substances (Jensen, 2013). Because NH_4^+-N is the main determinant of short term NUE, the proportion and the fate of it after application are highly relevant. The concentration of NH_4^+-N and its proportion of total manure N varies widely, depending on manure type (Jensen, 2013; Webb et al., 2013), which in turn depends on type of livestock, feeding, husbandry and production system. While feeding mostly affects the urine N to feces N ratio (Pagliari et al., 2020; Sørensen and Fernández, 2003; Sørensen et al., 2003), the husbandry system determines whether manure is collected as liquid or as solid manure, of which the latter has a higher DM content and a lower share of NH_4^+-N to TN (roughly about 20 % of TN in solid manure compared to 40–80 % in most slurries, Webb et al., 2013). Adding litter straw in barns influences the N availability. Ammonia losses and other gases, especially N_2, during and after application will reduce the *short-term* manure NUE. Thus, every measure that reduces NH_3 volatilization potentially increases *short-term* NUE (Section 3.5).

After application to soil, NH_4^+ may be directly taken up by crops, get adsorbed by soil colloids, be microbially immobilized or be converted into NO_3^-. Nitrate may be taken up by plant roots, but is also prone to losses via leaching or denitrification. Hence, the synchrony between the N demand of the plants and the supply of plant available manure N (as NH_4^+ and NO_3^-) in the soil solution, both in terms of timing and quantity, is highly relevant for manure NUE. Manure NUE is therefore expected to be higher for crops with a longer growing period (e.g., maize, temporary and permanent grassland, winter cereals) than with a shorter growing period (e.g., spring cereals), because of a longer lasting plant cover with an active root system and the technical possibility to split manure N applications. Whilst this has to the best of our knowledge not yet been proven by comparing the NUE of ^{15}N labeled manure, supporting evidence comes from studies with undersown cover crops. For instance a ryegrass cover crop undersown to spring barley recovered an additional 1–3 % of ^{15}N labeled cattle and pig liquid manure (Sørensen, 2004; Sørensen and Thomsen, 2005).

Manure also contains organic N, and via the addition of organic C stimulates microbial growth, resulting in an immobilization-mineralization turnover with both mineralization of organic N and immobilization of mineral N occurring simultaneously. The net mineralization of organic manure N is related to the C:N_{org} ratio. Generally, at C:N_{org} ratios > 15, mineralization process is dominant and at ratios < 15 immobilization , (Bhogal et al., 2016; Webb et al., 2013). However, even within a given C:N_{org} ratio, the mineralization vs. immobilization rate will vary as a function of the molecular structure of the organic compounds (Bosshard et al., 2011; Levavasseur et al., 2022). The net mineralization of organic N forms contained in manure is usually low, e.g. −1 %, 3 %, 6 % and 20 % of organic N contained in cattle liquid, cattle solid, pig solid and pig liquid manure, respectively (Levavasseur et al., 2022). Importantly, also soil properties, such as the mineral N content in the soil, soil texture, moisture and temperature will affect the mineralization rate of organic manure N (Fangueiro et al., 2012a; Griffin et al., 2002).

Using ^{15}N labeled cattle liquid manure with a NH_4^+-N:TN proportion of 62 %, Frick et al. (2023b) showed that in the year of application 19 to 22 % of liquid manure N was recovered in the aboveground biomass of a maize crop or a temporary grassland under Swiss arable conditions. Liquid manure N was rapidly (within days to weeks) incorporated into the microbial N pool and at the end of the vegetation period, more than half of the applied manure N was still stored in the topsoil, mostly in soil organic N. In the next spring, 75–85 % of the residual N from liquid manure was found in the soil organic N pool, while about 10–20 % were in the microbial N and less than 2 % in the mineral N pool. For more details on the factors controlling the *short-term* manure NUE the reader might refer to the comprehensive reviews by Webb et al. (2013), Jensen (2013), and Möller and Schultheiß (2014). In addition, Chalk et al. (2020) provide an overview on ^{15}N tracing studies for elucidating the *short-term* fate of manure N in the soil-plant-system.

3.6.3.2 Long term manure NUE

In addition to the factors affecting *short-term* NUE, *long-term* NUE is determined by N losses and the dynamics of N in soil organic matter (SOM). The residual fertilizer value of manure N stabilized in the soil organic N pool in the years following application is low and diminishes with time, e.g., <5 % recovery of originally applied manure in the first succeeding crop, <2.5 % in the second succeeding crop (Frick et al., 2022). Manure composition affects both the *short-term* availability of N and its *long-term* residual effect, with higher residual effects typically occurring with greater N_{org}:TN ratios (Webb et al., 2013). Even though residual fertilizer N could be traced in the soil and the crops 25 years after application (Sørensen et al., 2023), mineralization and crop uptake rates are low (Smith and Chalk, 2018). Hence, the residual effect of N_{org} is usually accounted only for a couple of years when determining *long-term* NUE of a single application. Nonetheless, with repeated manure applications, these residual effects sum up (Fig. 2), explaining substantially higher NUE of manures in the *long-term* as compared to *short-term* NUE (Berntsen et al., 2007).

Stable agricultural rotation systems with repeated manure application tend to reach SOM equilibrium. When the soil organic N stock remains constant over the years, losses due to N leaching and denitrification largely determine *long-term* NUE. At equilibrium, the N mass balance of N total inputs, i.e. by fertilizers + fixation + deposition − N removal by harvest − gaseous N losses is in the range of the average N leaching loss. N leaching losses include NO_3^-, NH_4^+ and dissolved organic N species. Nitrate is the predominant N species in the soil solution in most European agricultural systems under aerobic soil conditions and contributes > 90 % to N leaching.

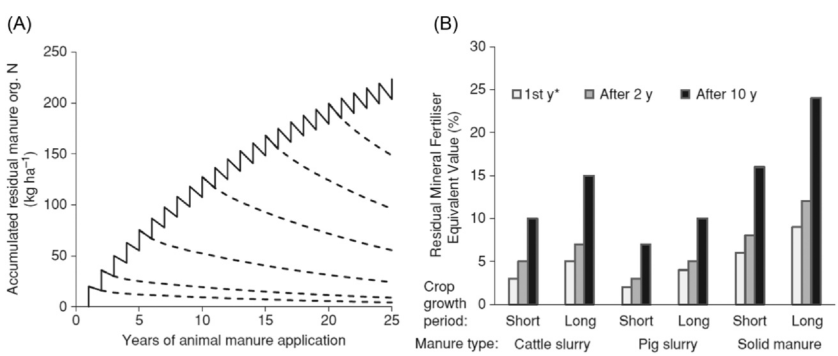

Fig. 2 Modeled (A) residual manure N_{org} accumulated (full line) in soil over 30 years with annual applications of 100 kg total-N ha^{-1} in pig liquid manure (25 % organic N, dashed line indicating decay of organic N after first, second, fifth, 10th, 15th and 20th application) and (B) the corresponding residual MFE value of 1, 2 or 10 years of repeated applications of different types of animal manure, compared with soil only receiving mineral fertilizers. MFE given in per cent of annual manure total-N application for crops with a short (e.g. spring cereals, 50 % of mineralized N available for crop) or long growing period (e.g. maize, beet, 75 % of mineralized N available for crop). Solid manure: cattle FYM and deep litter. Effect in the year after application, the first year with residual N effect. *©University of Copenhagen; data based on principles in Sørensen et al. (2002). Graph from Jensen et al. (2013), with permission from L. S. Jensen.*

Under anaerobic conditions, the main N loss paths switch to denitrification losses. A study using ^{15}N-labeled liquid cattle manure showed that less than 5 % of cumulated amounts of NO_3^- leaching over a sequence of three crops originated from direct leaching of the labeled fertilizers (Frick et al., 2022). In systems with regular manure application, 1 to > 2 % of total soil organic N is estimated to be mineralized to NO_3 annually.

In most European regions, any available NO_3 in autumn, winter and early spring is prone to leaching losses. *Long-term* N surplus is a dominant factor of N leaching. However, in the *short-term*, N surplus is not a good predictor for NO_3^- leaching (De Notaris et al., 2018; Nouri et al., 2022; Pedersen et al., 2021a), and autumn vegetation is important to reduce N leaching (Thorup-Kristensen et al., 2003). Application of available N in manures and fertilizers above the economic optimum for crop growth increases the risk for NO_3^- leaching significantly (Goulding et al., 2000). It is often challenging to predict the availability of N in manures precisely and therefore there is a higher risk for overfertilization and thereby increased NO_3^- leaching by use of manures (Sørensen and Jensen, 2013).

North-European studies have shown that NO_3^- leaching increases proportionally with the total N input after manure and mineral fertilizer application in spring (Pedersen et al., 2021a; Schröder et al., 2010; Thomsen et al., 1997). These studies indicate that the proportion of applied N that is lost by leaching in the first year is nearly the same for mineral N and organic N inputs. As TN application is normally higher with manures than with mineral fertilizers this also implies that NO_3^- leaching will be higher by manure application. Thomsen et al. (1997) found leaching losses from ^{15}N-labeled manures and mineral N at 3–5 % of the TN input in the first winter season after application. Pedersen et al. (2021a) observed a leaching rate from both mineral N and solid manure of 8 % of the TN input in the first winter season after a spring barley crop, but the loss rate could be reduced to 1–2 % by growing a cover crop after the barley. In the second year after application, Pedersen et al. (2021a) found leaching rates between 4–7 % of the original N input. On top of the leaching loss from the manure N input there is a leaching loss from soil N, that is variable depending on climate, soil conditions and the mineralization of N from SOM.

The residual organic N from manure application can be leached after mineralization and nitrification and is more prone to NO_3^- leaching than N applied in spring because part of the mineralization takes place in periods with poor plant growth and surplus precipitation. The proportion of mineralized N leached is very dependent on soil vegetation, e.g. Sørensen et al. (2023) in an area with high precipitation found leaching losses of mineralized N at 33–40 % after spring barley, 14–16 % after cover crops, and only 2–4 % on permanent grass. Losses are expected to be lower in drier areas.

Manures should only be applied when crops can utilize the available N. However, application of the manure at the optimal time is not always practicable. When applying solid manures in autumn on bare soil or before sowing winter wheat, extra NO_3^- leaching equivalent to the NH_4^+ content has been observed (Sørensen and Rubæk, 2012; Thomsen, 2005). In some cases, the extra leaching loss from solid manure in autumn was equivalent to 30 % of manure TN (Sørensen and Rubæk, 2012). These losses are reduced under conditions with less precipitation (Frick et al., 2022; Karimi and Akinremi, 2018).

Compared to NO_3^- leaching, N losses due to denitrification (N_2) can also be substantial. Based on the MITERRA model, Velthof et al. (2009) estimated on average 44 kg N ha agricultural land^{-1} y^{-1} as N_2 from denitrification and 16 kg N ha^{-1} y^{-1} from leaching, whereas losses due to NH_3 volatilization were 17 kg N ha^{-1} y^{-1}. Jones et al. (2017) established an N budget for a grassland site over time based on measurements and model-based data. They reported losses from leaching of N of 53 kg N ha^{-1} y^{-1} and N_2 emissions due to denitrification of 29 kg N ha^{-1} y^{-1}, which combined were substantially greater than those from NH_3 volatilization with 39 kg N ha^{-1} y^{-1}. Based on model calculations of Hutchings et al., 2023 for Switzerland, a loss due to diffuse N inputs into water bodies of 35 kg N ha^{-1} y^{-1} for arable land and 15 kg N ha^{-1} y^{-1} permanent grassland, respectively, can be estimated. This is comparable to NH_3 losses from grassland of three to six slurry application events using a trailing hose estimated for Swiss conditions based on the ALFAM2 Model (version 3.9; Hafner et al., 2024).

In contrast to NH_3, mitigation options to reduce N losses following denitrification related to manure application are limited to crop management, i.e. optimizing N application rate to match the N demand of the crops. Application of nitrification inhibitors has also been shown to significantly reduce soil denitrification rate and N_2 emissions (Pan et al., 2022). The same principles may apply for leaching of NO_3^-. When SOM is not at equilibrium (stock change occurs), the N stabilized following SOM accumulation or release due to SOM loss results in an apparent decrease or increase of NUE, respectively. Hence, factors affecting SOM dynamics such as the land-use trajectory (Guillaume et al., 2021) and management practices (Chenu et al., 2019), have also an impact on *long-term* NUE, along with sites' pedoclimatic characteristics. Zavattaro et al. (2017) found a higher manure NUE in light-textured soils and warm climates than in medium-textured soils and cool climates. As SOM is rarely at equilibrium, it is important to consider changes in SOM (ΔN_{soil}) when estimating the *long-term* fertilizer NUE.

3.6.4 Differentiating factors for manure NUE between cropland and grassland

When assessing manure NUE, it is important to differentiate between cropland and grassland, and more specifically *short-term* grassland leys and *long-term* permanent grasslands.

In relation to the *short-term* manure NUE, grass has a long growing season and year-round ground cover which results in a lower risk of N losses outside the main growing season. Also, the timing and release of N is less critical in grasslands in comparison to croplands. Finally, on grasslands, direct incorporation of manure through plowing to minimize NH_3 losses is not possible, and liquid manures cannot be so deeply injected as by injection before sowing a new crop. Deep injection of slurry in arable soils has a much lower NH_3 emission (2 % of TAN) than sod-injection in grassland (17 % of TAN). Thus, the potential to reduce NH_3 emission is higher for arable land than for grassland (Van Der Zee et al., 2024). As a result, the *short-term* NUE on grassland is highly dependent on the application method and meteorological conditions during and after application. *Short-term* NUE is more affected by manure composition, application methods and conditions than by inherent differences between crop- and grassland.

The difference between cropland and specifically permanent grasslands is more pronounced in *long-term* NUE. Permanent grasslands are not plowed for years, allowing a build-up of SOM – including residual manure N – in the soil up to saturation. Even though the annual release of residual organic N from manure is relatively small, the cumulative effect of repeated manure application over multiple years can have a large impact on the *long-term* manure NUE (Fig. 3; Cela et al., 2011; Schröder et al., 2007; Sørensen, 2004). Due to a continuously active root system in permanent grasslands, N losses by NO_3^- leaching and denitrification from mineralized residual manure N are lower than in rotational systems (Sørensen et al., 2023). On the other hand, it is hard to reduce NH_3 losses after manure

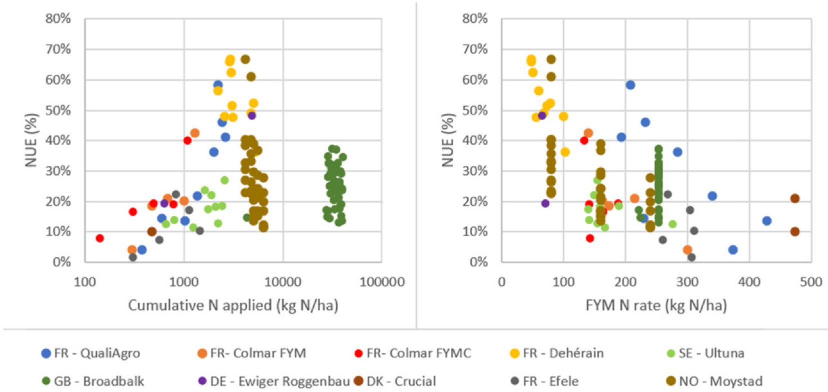

Fig. 3 Nitrogen use efficiency of cattle solid manure in ten *long-term* field experiments, depending on (A) the cumulative N applied and (B) the application rate. Manure NUE was calculated with the difference approach.

application in grassland systems, as the manure cannot be incorporated or deeply injected without disturbing the grass. Therefore, the *long-term* NUE of manure applied to permanent grassland compared to cropland depends very much on the local pedoclimatic conditions.

3.6.5 Summary of manure NUE from cropland and grassland experiments

3.6.5.1 Cropland manure NUE

The *short-term* manure NUE values found in the literature (Table 5) show substantial variation within manure types and cropping systems, among manure types, and among measurement methods. Reasons for this large variation include differences in manure composition, climates, soils, crops, application methods, and different NUE measurements (from one month to one year).

Liquid cattle manure used on cropland has a median NUE of 22 % and 32 %, estimated with the ^{15}N tracer and the difference approach, respectively; the latter value is backed up by a rather large data set (58 observations from 10 studies). For both approaches the variation within this manure type is large. The median MFE is 54 %.

For solid cattle manure we did not find measurements with the tracer approach, and too few data (two studies only) for the difference approach (median manure NUE 21 %, not reported in the table).

A large dataset is available for liquid pig manure (52 observations from 11 studies), showing a median NUE of 48 % and a MFE of 75 %. This

Table 5 Short-term NUE of cattle and pig liquid and solid manures measured in experiments across Europe.

Manure type and cropping system		NUE manure quality drivers (average)			NUE indicators			
		Dry matter ($g\ kg^{-1}$ FW)	TAN[a]: TN (%)	C:N	Apparent NUE[b] (%)	^{15}N recovery (%)	MFE (%)	Mineral fertilizer NUE (%)
Cropland								
Liquid	Cattle	71	57	8.7	32 (10, 43) [10,58]	22 (17, 43) [4,7] with ^{15}Ntot (insufficient $^{15}NH_4^+$ data)	54 (26, 68) [11,68]	64 (43, 75) [10,58]
Solid	Cattle	226	11	16.3	Insufficient data	n.a.	Insufficient data	Insufficient data
Liquid	Pig	50	75	3.5	48 (19, 64) [11,52]	No data for ^{15}Ntot insufficient $^{15}NH_4^+$ data	75 (41,96) [11,61]	65 (51, 94) [11,52]
Solid	Pig	291	32	8.7	16 (7, 27) [3,31]	insufficient ^{15}Ntot data No data for $^{15}NH_4^+$	30 (12, 54) [3,31]	56 (50, 79) [3,31]
Grassland								
Liquid	Cattle	83	53	7.8	29 (14, 42) [5,47]	n.a.	36 (15, 57) [5,47]	82 (70, 109) [5,47]

(continued)

Table 5 Short-term NUE of cattle and pig liquid and solid manures measured in experiments across Europe. (*cont'd*)

		NUE manure quality drivers (average)			NUE indicators			
Manure type and cropping system		**Dry matter (g kg^{-1} FW)**	**TAN[a]: TN (%)**	**C:N**	**Apparent NUE[b] (%)**	**^{15}N recovery (%)**	**MFE (%)**	**Mineral fertilizer NUE (%)**
Solid	Cattle	n.a.	12	14.0	Insufficient data	n.a.	Insufficient data	Insufficient data
Liquid	Pig	69	65	6.0	49 (35, 63) [4,30]	n.a.	81 (72, 102) [4,30]	63 (43, 70) [4,30]

Short-term NUE studies cropland= >application year.
[a]TAN=Total ammoniacal nitrogen (TAN=NH_4^+ + NH_3).
[b]Difference method. n.a.: no papers were found to fill the cell in the table.
NUE indicators presented as median (10th, 90th percentile) and underneath are [*number of studies, number of observations*].
Insufficient data: the papers related to the cell in the table were only 1 or 2.
Following studies were reviewed: Antoun et al., 1985; Baral et al., 2017; Cavalli et al., 2014; Cavalli et al., 2016; Chantigny et al., 2007; Chen et al., 2022; Chirinda et al., 2010; Fontaine et al., 2020; Frick et al., 2023b; Frick et al., 2022; Gómez-Muñoz et al., 2017; Gutser and Dosch, 1996; Hernández et al., 2013; Herrmann et al., 2013; Houot and Chaussod, 1995; Jackson and Smith, 1997; Jensen et al., 2000; Johnston, 1969; Kai et al., 2008; Kismányoky and Tóth, 2010; Klop et al., 2012; Lalor et al., 2011; Levavasseur et al., 2021; Lipavský et al., 2008; Martínez et al., 2017; Morvan et al., 2020; Muñoz et al., 2003; Nannen et al., 2011; Nedvěd et al., 2008; Oberson et al., 2024; Obriot, 2016; Olesen et al., 2009; Pantelopoulos et al., 2017; Pedersen et al., 2021a; Petersen, 2005; Petersen, 2006; Petersen and Sørensen, 2008a; Powell et al., 2005; Reijs et al., 2007; Riley, 2007; Rühlmann, 2003; Schils and Kok, 2003; Schröder et al., 2013; Schröder et al., 2005; Schröder et al., 2007; Sigurnjak et al., 2017; Sørensen, 2004; Sørensen and Amato, 2002; Sørensen and Eriksen, 2009; Sørensen and Fernández, 2003; Sørensen and Jensen, 1998; Sørensen and Rubæk, 2012; Sørensen and Thomsen, 2005; Sørensen et al., 2003; Spiegel et al., 2010; Thomsen, 2005; Trochard et al., 2012; Velthof and Rietra, 2019; Vullioud et al., 2006.

shows that N in pig liquid manures is more available in the application year compared to cattle liquid manure. It is also worth noting that the 90th percentile of the MFE for pig liquid manure is 96 %, indicating that in 10 % of cases an almost complete substitution of manure-N with mineral fertilizer-N can be expected. The values of MFE higher than 96 % come from pig liquid manures with a TAN:TN ratio of 82 % on average which explains their high *short-term* effect. However, in other cases the NUE of mineral fertilizers was not particularly high even for pig liquid manure (43 % on average). Finally, for solid pig manure the median NUE calculated with the difference approach is rather low (16 %, MFE=30 %, only three studies).

Regarding the *long-term* effects, only few published data were found and a quantitative analysis was not possible. However, we used available raw data from ten *long-term* field experiments to compute the NUE value of solid cattle manure with the difference approach. Using only the most recent year available for each *long-term* experiment, a median NUE value of 35 % was calculated, only slightly higher than the previously reported median *short-term* value (21 %). However, the 90th percentile of the data set revealed a *long-term* manure NUE of 56 % and a MFE of 82 %. A significant increase in NUE with cumulative N input was observed (Fig. 3A), illustrating the effect of repeated applications in the long term on SOM increase and associated soil N supply. A significant decrease of NUE with increasing application rate was also observed (Fig. 3B), which could be due to risk of high N losses by high application rates, but it could also be due to higher application rates used for manure types with an expected low N availability. Both factors explained 32 % of the variance of manure NUE in these *long-term* experiments.

A similar increase of manure NUE over time, after repeated applications, was observed for liquid cattle manure by Nevens and Reheul (2005). After 19 years of continuous application of dairy cattle liquid manure together with inorganic fertilizer, they observed that NUE increased from an average of 16 % in the first three years to an average of 61 % in the last three years.

In their review, Zavattaro et al. (2017) indicated that the MFE for cattle solid and liquid manure, used at approximately the same rate as mineral fertilizers, is on average 72 % and 57 %, respectively. One reason why these values are higher than those reported in Table 5 is because they also include *long-term* effects.

No studies were found reporting *long-term* NUE for solid pig manure.

3.6.5.2 Grassland manure NUE

Short-term NUE of liquid cattle manure on grassland is similar to that of croplands (29 % with the difference approach), while a higher NUE of mineral fertilizers on grasslands generates a smaller MFE (36 %) than in cropland. As for cropland, also for grassland the data about solid cattle manure are too few (one study only) to allow a summary to be presented.

Data estimated with the difference approach for liquid pig manure on grassland are similar to that on cropland, with a median NUE of 49 % and a median MFE of 81 %; data abundance is smaller (30 observations from 4 studies). This indicates that like cropland, grassland N from pig liquid manure is more available for crops in the year after application compared to liquid cattle manure. No data are available for solid pig manure on grassland.

We found no studies reporting *long-term* NUE for liquid pig manure on grassland and only two studies for solid pig manure, which do not allow a summary to be presented.

3.6.6 Potentials to improve manure NUE

This section discusses the potential manure NUE under best available practice in cropland and grassland.

Measures at barn, yard and storage level as well as processing of manure lead to (i) more available TN from excretion for field application and (ii) higher quality of liquid manure with a higher TAN/TN ratio (see Section 4). Thus, pollution swapping at the field scale must be avoided by applying modern application technologies and best crop management. Fig. 4 (Section 3) illustrates the main measures that can be applied at the respective stage of the manure cascade and the effects on manure quality and cropland/grassland NUE.

Studies which integrate all N loss mitigating measures along the manure cascade and evaluating the NUE of crop productivity at the end of the cascade are not available. Therefore, for *short-term* NUE, we report the 75 % to 95 % percentile of observations in considered studies assuming that those studies show the achievable potential under best practice. We give a range of values, because climate and site factors influence the potential NUE and highest NUEs cannot be achieved on all sites (Zavattaro et al., 2017). The potential *long-term* NUE is here defined as regular (yearly) manure application over a period of two decades or longer. For *long-term* NUE only a small number of studies are available. Therefore, we discussed the potential *long-term* NUE based on single studies and model outputs (Jarosch et al., 2018; Sørensen et al., 2017).

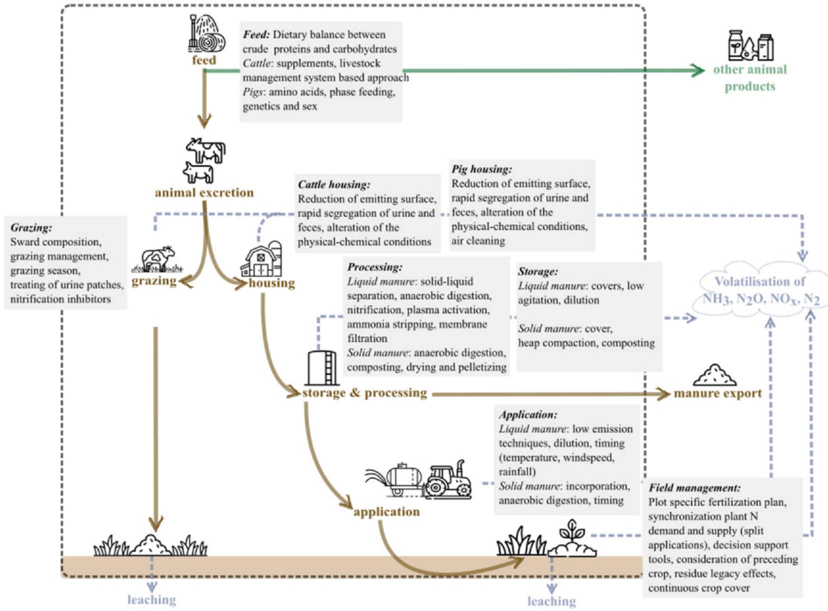

Fig. 4 Overview of the discussed measures reducing N losses and increasing the NUE at the different steps of the livestock system.

3.6.6.1 Liquid manure cropland

Short-term NUE shows a potential of 40–50 % and corresponding MFE of 60–70 % of applied TN for cattle liquid manures as well as an NUE of 60–75 % and an MFE of 75–95 % for pig liquid manures (Table 6). Sørensen and Eriksen (2009) found MFE of 100–103 % for pig liquid manure and 61–68 % for cattle liquid manure after acidification and injection in Denmark. Similar high values were reported by Kai et al. (2008) who found MFE of 80–100 % with acidified pig liquid manure, but surface application. Mineral concentrates from further processing like ultrafiltration followed by reverse osmosis does not significantly improve the MFE compared to other measures. Schröder et al. (2014) found MFE of 72 % to 84 % and Bosshard et al. (2010) found a MFE of 86 %, which were not different from the previously digested pig liquid manure. Velthof et al. (2020) found MFEs of 72–84 % for maize and 54–81 % for grasslands. The ultrafiltration retentate, one product of the process chain, had a NUE of 43 % and a MFE of 68 % in the application year (Bosshard et al. 2010). However, data on the overall manure NUE of mineral concentrates and the corresponding separated solid fractions and retentates are not available.

Table 6 Potential nitrogen use efficiency (NUE) and nitrogen fertilizer equivalent (MFE) of solid and liquid cattle and pig manure if best available practice (see Fig. 2) is applied. Ranges for NUE and MFE in the application year are derived from 75 % to 95 % percentiles of evaluated studies listed under Table 5. Long-term NUE and MFE refer to regular application of more than two decades and take into account legacy effects of residual manure N remaining in soil from previous applications. Ranges are derived from single long-term studies and modeling approaches (n.a. no data available)

Category		Short-term (application year)			Long-term		
		n^1 / n^2	NUE	MFE	NUE	MFE	Studies
Cropland			(%)	(%)	(%)	(%)	
Liquid	Cattle	58/10	40–50	60–70	60–80	75–90	4, 5, 7, 8, 9
	Pig	52/11	60–75	75–95[a]	65–85	80–95	1, 2, 3, 5
Solid	Cattle	4/3	20–35[b]	35–60[b]	55–70	70–85	4, 5, 6, 8, 9
	Pig	31/4	25–35	40–70	55–70	70–85	5
Grassland							
Liquid	Cattle	71/6	40–50	50–70	n.a.	n.a.	
	Pig	1/1	40–65[c]	70–90[c]	n.a.	n.a.	
Solid	Cattle	1/1	20[d]	30[d]	n.a.	n.a.	
	Pig	0/0	n.a.	n.a.	n.a.	n.a.	

n^1 number of observations.
n^2 number of studies.
[1]Hernández et al. (2013); [2]Martínez et al. (2017); [3]Morvan et al. (2020); [4]Oberson et al. (2024); [5]Petersen and Sørensen (2008a); [6]Rühlmann (2003); [7]Schröder et al., 2005; [8]Spiegel et al. (2010); [9]Zavattaro et al. (2017).
[a]Sørensen and Eriksen (2009) found MFE 100–103 %.
[b]Based on three studies (Cavalli et al., 2014; Petersen and Sørensen, 2008a; Schröder et al., 2013).
[c]Based on three studies (Delin et al., 2012; Van Middelkoop and Holshof, 2017; Velthof and Rietra, 2019).
[d]Based on only one study (Schröder et al., 2007).

Across all liquid manure types, both animal categories studied revealed a strong increase of *short-term* NUE due to band application (NUE median 32 %; n = 7) or injection (NUE median 39 %; n = 79) compared to broadcast application (median 25 %; n = 134).

In a comprehensive review of studies including *long-term NUE* of cattle liquid manure Zavattaro et al. (2017) analyzed 12 *long-term* experiments all over Europe. The top third of observations revealed MFE from 72–81 % for cattle liquid manure. High NUE between 73 % and 92 % (MFE: 85–95 %) were found in a recent study by Oberson et al. (2024) for combined fertilization of cattle solid and liquid manure on a fertile soil in Switzerland over 35 years. Spiegel et al. (2010) found cattle liquid manure NUE of 56 % (MFE = 73 %) after 21 years regular liquid manure application in Austria.

For pig liquid manure *long-term NUE* data are scarce. The potential N legacy effect for pig liquid manure is lower than for cattle liquid manure due to higher TAN/TN ratios and lower absolute N_{org} inputs. Hence, we assume *long-term* NUE and MFE to be only slightly higher than *short-term* observations. *Long-term* NUE of 64 % to 77 % (MFE: 77–95 %) were reported (Hernández et al., 2013; Martínez et al., 2017; Morvan et al., 2020).

Based on the Danish report on "Fertilizer value of nitrogen in animal manures" (Petersen and Sørensen, 2008a) and an empirical modeling approach (Jarosch et al., 2018; Sørensen et al., 2017), the potential NUE for cattle liquid manure is between 60–80 % (potential MFE: 75–90 %) in the *long-term*. For pig liquid manure the potential NUE is between 65–85 % (potential MFE: 80–95 %) in the *long-term* (Table 6).

3.6.6.2 Solid manure cropland

Solid pig manure shows a potential *short-term NUE* between 25 % and 35 % (MFE: 40–55 %). However, for cattle solid manure data are surprisingly scarce. The potential NUE of 20 % (MFE= 30 %) is based on only two studies (Cavalli et al., 2016; Schröder et al., 2013).

For solid cattle manure the *long-term NUE* is based on the same studies that had been taken into account for *short-term* NUE (Table 6). Potential NUE ranges from 55–70 % and MFE from 70–85 %. For pig solid manure NUE we found no *long-term* studies. Potential NUE of 55–70 % and MFE of 70–85 % are based on Petersen and Sørensen (2008a) and modeling approaches (Table 6).

3.6.6.3 Grassland

Grassland studies reveal a potential *short-term NUE* of 40–50 % and corresponding MFE of 50–70 % for cattle liquid manure. For pig liquid manures we found only one field study for mineral concentrates processed from pig liquid manure applied on grassland in the Netherlands which showed mean NUE of 51 % (35–65 %) but high mean MFE of 83 % (63–93 %) (Van Middelkoop and Holshof, 2017). Also here, there is no information about the overall manure NUE if separated solid products are taken into account.

Data for solid manures are also scarce. We found only one study for cattle solid manure (Schröder et al., 2007) which shows an NUE of 20 % and a MFE of 30 % in the application year (Table 6).

As an approximation for *long-term* NUE only two studies for liquid cattle manure are available revealing an NUE of 40–50 % and MFE of 60–75 % (Schils and Kok, 2003; Schröder et al., 2007). However, these studies cover a period of only 2–4 years, which does not meet our definition of *long-term* NUE. Thus, for longer periods higher values could be expected. We found no reliable data for *long-term* application of solid manures on grassland.

3.7 Pastures – direct excretion during grazing

3.7.1 N losses from direct N excretion

Grazed grasslands receive N inputs from grazing animal excreta compared to cut grasslands. The uneven distribution of this N source results in patchy hotspots of high N concentration (Mcculley et al., 2005), from which N is lost particularly through leaching and denitrification (Cameron et al., 2013). Nitrogen deposited as urine is more prone to losses than manure spreading because it is concentrated in patches, a defined small zone where two factors act simultaneously. The first is the high N concentration in urine contains urea-N and also NH_4^+-N that are both quickly converted to NO_3^--N. The second is a concentrated liquid flux that transports soluble N deep in the soil profile, often below the depth where most of roots are. The area of effects of urinary N can be moderately larger than the original patch area due to lateral diffusion and due to root uptake by neighboring plants. However, this effect is limited (Cichota et al., 2018) and consequently, N leaching is concentrated below urine patches. Conversely, the release of inorganic N from dung is a relatively slow process, and dung contains less N compared to urine. Therefore, leaching and other losses under dung patches are considered less critical than that of urine (Cai and Akiyama, 2016; Lantinga et al., 1987).

Literature syntheses of relative N loss pathways from urine (and dung) deposited on pasture were reported by Selbie et al. (2015) and Cai and Akiyama (2016). They combined results of various studies that measured N loss pathways and/or N recovery in harvested biomass during one month to more than one year. Their values (Table 7) were used to roughly

Table 7 Average relative N losses from cattle excreta deposited on grazed pastures via different pathways (after Selbie et al., 2015 and Cai and Akiyama, 2016).

Literature source		**N losses**				**N recovery**
Unit (%)		**NH_3**	**NO**	**N_2O**	**NO_3^-**	**biomass**
Selbie et al. (2015)	Urine	13		2	20	41
Cai and Akiyama (2016)	Urine	12.4	0.1	0.8	24.1	31.4
	Dung	1.7	0.0	0.3	16.4	10.4
	Excreta total	9	0.1	1	20	27

Values represent fractions of applied urine or dung N, respectively. The N loss and recovery fractions for total excreta was estimated for typical shares of excreta N in urine (2/3) and dung (1/3) for dairy cows (Voglmeier et al., 2019).

estimate N loss fractions for total excrements assuming typical shares of excreted N in urine and dung, as given in Voglmeier et al. (2019). The results indicate largest average N losses occur via NO_3^- leaching, followed by NH_3 volatilization. According to Cai and Akiyama (2016), leaching losses of NH_4^+ (3 %) and dissolved organic N (4 %) can also play a role. The gaseous losses as N_2O and NO are generally small (less than 1 % of TN, Chadwick et al., 2018) and thus are of negligible relevance for pasture NUE. Yet, it needs to be noted that in recent years, several studies reported high N_2 emissions from pasture urine patches of 26 % (Selbie et al., 2015), 44 % (Burchill et al., 2016), 14–32 % (Ding et al., 2022). The representativeness of these high emissions is still unclear and they were not considered in the synthesis study results in Table 7.

With the estimated average total excreta N recovery in biomass (= *short term* NUE) of 27 % (Table 7), a considerable gap in the N budget of pasture excreta remains. For urine, Selbie et al. (2015) attributed the difference to immobilization, i.e. the storage of N in organic form in the soil. Smit et al. (2021) reported that field-N-balances of grazed grass-clover-leys were not an appropriate indicator to predict N-leaching losses, because soil and plant residues can accumulate part of N and prevent its loss. The stored organic N can be re-mobilized in the medium term and thus increase the total NUE of the pasture excreta.

3.7.2 Selected measures mitigating N losses from direct N excretion

Nitrogen losses from grasslands are influenced by pedoclimatic conditions, pasture types, and grazing management practices. Effective management tailored to these factors mitigates N losses and environmental impacts.

Regarding pedoclimatic conditions, sandy soils are prone to NO_3^- leaching to a greater extent, while clayey soils enhance denitrification and N_2O emissions (Cameron et al., 2013).

Different pasture types vary in N dynamics and efficiencies. For instance, monoculture ryegrass pastures lead to higher losses compared to mixed swards or those with N-fixing legumes such as clover (Grange et al., 2021). Ryegrass pastures rely heavily on nitrogen fertilization, leading to substantial inputs but also losses through leaching and volatilization (Di and Cameron, 2002). Ryegrass/clover pastures benefit from clover's nitrogen fixation, reducing synthetic fertilizer needs and enhancing soil nitrogen availability (Grange et al., 2021). Multi-species swards optimize nitrogen capture and utilization through diverse root structures and nutrient cycling, resulting in reduced losses and improved soil health (Sanderson et al., 2004). These systems enhance resilience to environmental stresses, stabilizing nitrogen budgets. Thus, while intensive management is required for ryegrass pastures, ryegrass/clover and multi-species swards offer sustainable alternatives with better efficiency and lower environmental impacts.

Several studies have compared the effect of the sward composition in particular on N leaching. Despite legumes can add extra N to the soil system via symbiotic fixation, their presence in the sward can significantly reduce N leaching, probably because of their deep root system. Saarijärvi et al. (2007) reported N leaching losses of 17 and 9 kg N ha^{-1} from grass and grass-clover treatments, respectively. The different species have different optimal growing conditions, therefore N uptake may vary across seasons. Conversely, considerable leaching losses of inorganic N from lucerne (>200 kg N ha^{-1}) were reported by Welten et al. (2019) due to the relatively low plant growth rates during winter (<15 kg DM ha^{-1} day^{-1}) that led to low total recovery of urine-N by lucerne plants (<20 % of the applied urine-^{15}N). Talbot et al. (2021) reported positive effects of replacing perennial ryegrass with white clover with Italian ryegrass and/or plantain, in a shallow stony soil.

In some cases, the sward composition influences N losses also because it has an effect on animal metabolism. Plantain (*Plantago lanceolata*), in particular, can reduce total N excretions and N concentration in the urine of dairy cows with consequent reductions in N leached from pastures. The positive effect of plantain shows high seasonal variations, being more efficient in winter (50 % reduction of N leaching losses compared to ryegrass) than in summer (−15 %) with an overall reduction of 39 % in the total amount of inorganic-N leached across the three seasons (53 vs. 87 kg N ha^{-1} leached relative to ryegrass, Welten et al., 2019).

Among grazing management practices, N fertilizer intensity and grazing management influence all kind of N losses; NO_3^- leaching and N_2O emissions were found higher in intensive dairy farming (De Klein and Ledgard, 2005). In particular, reducing the stocking rate can have an impact on N leaching losses. However, this effect can also be counter-intuitive. Roche et al. (2016) reported that the concentration of NO_3^--N in drainage water and the quantity of NO_3^--N leached ha^{-1} y^{-1} declined linearly with increasing stocking rate. This unexpected result was possibly due to the fact that higher stocking rates were associated with shorter milking periods per cow, resulting in a reduction in urine N excretion per cow during autumn, the climatically sensitive period for N leaching. The reduction in urine N excretion per cow led to an increase in urinary N spread, greater plant recovery of N, and reduced losses from urine patches.

The N load in urine patches has strong effects on N leaching and varies largely with variations in volume and N concentration of urination events, but literature data on urine production and its concentration are not numerous (Haynes and Williams, 1993; Selbie et al., 2015). Marsden et al. (2021) using accelerometers found that sheep produced 159 $\pm$ 1 mL per urine event (ranging between 17 and 745 mL), with a daily volume of 2.15 $\pm$ 0.04 L. Some indicators of animal performance are correlated with N leaching losses, for instance Beck et al. (2023) found significant correlations with milk urea nitrogen concentration, DM intake, CP content, and body weight. The overall feeding strategy has an influence on the potential N excretion, and was discussed in Section 3.1.1. In addition, Marshall et al. (2020) reported that breeding and selecting for dairy cows with low concentrations of urea in milk can reduce urinary urea N deposition onto pasture and consequently the negative environmental impact of pastoral dairy production systems in temperate grasslands. In contrast, Ariyarathne et al. (2021) stated that the reduction of excretion using genetically-improved cows may be negligible at the whole farm level, due to the influence of stocking rate on the per hectare N excretion.

Modifying the grazing season can have an effect on N leaching losses. Talbot et al. (2021) reported that N leaching losses were high when urine was deposited during warmer months (late summer/early autumn) compared to cooler autumn months (>198 kg NO_3^--N ha^{-1}). Corré et al. (2014) reported that the climatic conditions during the grazing day had a significant effect on N leaching, but this effect was not consistent over the years.

Grazing during periods of lower emissions reduces nitrogen losses from animal excreta (Krol et al., 2016; Talbot et al., 2021). Restricting grazing during high-emission times minimizes nitrogen losses (De Klein and Ledgard, 2005). Changes are also influenced by the season. In autumn, urine-N concentrations of lactating cows were higher in the evening ($5.8\,g\,N\,L^{-1}$) than they were in the afternoon or early morning (mean of $4.2\,g\,N\,L^{-1}$). In late spring–summer, the concentrations were higher ($P < 0.001$) in late morning ($8.0\,g\,N\,L^{-1}$) than they were in the early morning, afternoon or evening (mean of $6.3\,g\,N\,L^{-1}$) (Cosgrove et al., 2017).

Synthetic nitrification inhibitors can be applied to pastures as a separate agronomic application to reduce N_2O and NO_3 leaching from urine patches (Selbie et al., 2014), or specifically be applied to individual urine patches if soil electrical conductivity is an effective indicator (Jolly et al., 2021), added in a bolus so the nitrification inhibitor is targeted with the urination event (Ledgard et al., 2008). Laboratory studies show, that nitrification inhibitors slow NH_4^+ conversion to nitrate, while biological inhibitors further reduce leaching and N_2O emissions (Zaman et al., 2009).

Integrated strategies enhance N use efficiency and sustainability in grassland management. Effective management practices are crucial for optimizing N use efficiency and minimizing negative impacts in both systems.

4. Overall manure livestock system NUE

4.1 Baseline information for the modeling study

In the previous sections, the potential feed NUE (Section 3.1), various measures to reduce N emissions along the manure cascade (Sections 3.2–3.5), the manure NUE (Section 3.6) and the grazing NUE (Section 3.7) were introduced and discussed. In this section, data from six European case-study farms (Fig. 5) were used to quantify the influence of selected measures on the Manure cascade NUE and overall Livestock system NUE (Fig. 1). To this end, four scenarios were modeled: the *current management* (CM) depicts the current livestock and manure management given from the farm dataset, the *past management* (PM) scenario represents manure management options that were applied in the past, but might still be practiced in European farms, and the *optimized management 1* (OM1) and *optimized management 2* (OM2) scenarios, describe a set of measures, that could potentially increase the overall Livestock system NUE (Table 8).

The modeling was implemented on six *good agricultural practice* example farms located in selected regions of Denmark, France, Germany, Italy, Switzerland and The Netherlands (Fig. 5). The case-study specific baseline

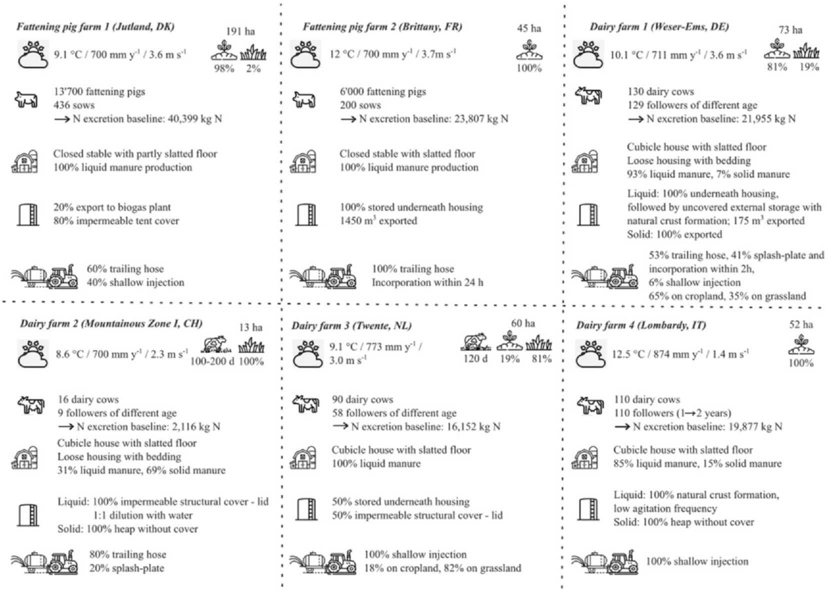

Fig. 5 Description of the six selected case-study farms, with information about yearly average climate parameters, land management, livestock and manure management. The farm data were provided by the co-authors. For the modeling approach, the exported manure was managed the same as on the case-study farm (e.g. the 1450 m^3 liquid manure exported from pig farm 2 are applied with trailing hose and then incorporated within 24 h). To simplify the scenario implementation of pig farm 1, 100 % cropland instead of 98 % was assumed.

farm information on number of livestock and its management, and the share of cropland and permanent grassland was kept constant for all scenarios, while the manure management was changed for the scenarios PM, OM1 and OM2 (Table 8). The manure management described in Fig. 5 is depicted in the CM scenario.

The N losses and TN flows from excretion to both liquid and solid manure including the application were quantified using the *Swiss Ammonia and Greenhouse gas Emission model* (SAGE, Winter et al., 2024), which is mainly based on the AGRAMMON model (Kupper et al., 2022), the ALFAM2 database and model (Hafner et al., 2018; Hafner et al., 2019) and the Swiss greenhouse gas emission inventory (FOEN, 2024). The SAGE model corrects baseline NH_3 emissions at the housing and storage level using specific correction factors for each measure. In this case, the specific N emission and correction factors were either retrieved from literature mentioned in Sections 3.2–3.5, or from Kupper (2022), and implemented

Table 8 Description of management measures that were applied to the case-study farms to depict the scenarios *past management* (PM), *optimized management 1* (OM1) and *optimized management 2* (OM2).

Step of the cascade	Livestock management system	Past management	Optimized management 1	Optimized management 2
Feeding:	Both livestock management systems	Same as current management	Reduction of N excretion by 5 % via an improved utilization of dietary crude proteins per unit metabolizable energy	
Housing:	Dairy cows	Same as current management	sloped floors with adapted scraper and urine gutter Elevated feeding lots	
	Fattening pigs:	Same as current management	Air-scrubbers: the N saved via air-scrubbers is added to the liquid manure prior to application	
Processing and storage:	Both livestock management systems	**Liquid manure:** Natural crust formation in external tank with no cover or, storage underneath livestock housing **Solid manure:** No cover of heap	**Liquid manure:** Cropland: *Solid-liquid separation* Acidification of liquid phase to pH 6 External tank with cover or, storage underneath livestock housing Grassland: acidification of liquid phase to pH 6 External tank with cover **Solid manure:** Cover of heap and of solid phase	**Liquid manure:** Cropland: Acidification of liquid phase to pH 6 External tank with cover or, storage underneath livestock housing Grassland: acidification of liquid phase to pH 6 External tank with cover **Solid manure:** Cover of heap

Application:	**Liquid manure** Application with broadcast	**Liquid manure** Timing of application (temperature, rainfall) Cropland: application with open-slot injection Grassland: application with trailing shoe **Solid manure** Cropland: Direct incorporation within 1 h

At the steps feeding and housing, the same measures for both OM1 and OM2 were implemented. Differences between the three scenarios lay particularly at the processing and storage and application step, where in OM1 the liquid manure undergoes a solid-liquid separation step prior to acidification, if applied to cropland. For case-studies with 100 % grassland, no differentiation in methods is made between OM1 and OM2. Information regarding the measures implemented for the *current management (CM) scenario are described in* Fig. 5 for each case-study separately.

on the case-study farms. For the quantification of NH_3 emissions at application using ALFAM2, a baseline DM of 4.75–5.00 % (pigs) or 7.5 (cattle) and a pH of 7 for liquid manure were assumed. For the dairy farm 2 located in Switzerland, the liquid manure was diluted with water and had lower DM content. Direct and indirect emissions of N_2O, NO_x, N_2 were calculated as fractions of TN at each step of the cascade.

In order to calculate the livestock system NUE, the output of the SAGE model was linked with the feed NUE and manure NUE. The feed NUE in the *current management* (CM) and *past management* (PM) scenarios corresponds to 27 % (dairy cows) and 44 % (pigs) (Section 3.1.1), i.e. of the TN intake, 73 % (dairy cows) and 66 % (pigs) end up in the excretion. For the OM1 and OM2 scenarios it was assumed that each livestock category excretes 5 % less N than in the CM, based on an improved utilization of dietary CP per unit metabolizable energy (Section 3.1.1). The intake N amounts were kept constant for all four scenarios. The manure TN available after storage was multiplied by the manure NUE factors described in Table 6 to calculate the plant N recovery. The range of the potential *long-term* manure NUE factors given in Table 6 is linked with the application technique used for liquid manure application, specifically broadcast and injection, which were mostly used in the studies considered in Table 6. Exemplified for the cattle liquid manure case, this means that the highest potential *long-term* manure NUE (*long-term* cattle liquid manure in cropland=80 %, Table 6) is achieved with injection, and the lowest potential *long-term* manure NUE (*long-term* cattle liquid manure in cropland=60 %, Table 6) is achieved with trailing hose. The *long-term* manure NUE for broadcast was assumed to be 63 % of the *long-term* manure NUE with injection (80 % × 0.63 = 50.4 %), as presented in Section 3.6.2. Additionally, it was assumed that the *long-term* manure NUE potential for grassland equals to the *long-term* manure NUE potential for cropland (Table 6), for both liquid and solid manure. For the two case-studies with grazing (Fig. 5), the grazing NUE factor of 27 % (Section 3.7) was used.

Manure cascade NUE and livestock system NUE are illustrated in Fig. 1 and are defined as:

$$Manure\ cascade\ NUE\,(\%TN) = \frac{\sum N_{crop\,(croplan,\ grassland)}}{N_{excretion\ housing}} \times 100$$

$$Livestock\ system\ NUE\,(\%TN) = \frac{\sum N_{crop\,(croplan,\ grassland,\ grazing)} + N_{animal\ \ products}}{N_{intake}} \times 100$$

Where $\sum N_{crop\ (cropland,\ grassland,\ grazing)}$ corresponds to the sum of N recovered in plants under different field management systems, $N_{animal\ products}$ is the N that is allocated in all animal products except manure, $N_{excretion\ housing}$ is the amount of N that is excreted at housing, i.e. the N amount that will follow the manure cascade (Fig. 1), and N_{intake} corresponds to the amount of N ingested by the animal.

4.2 Effects of measures to increase the overall livestock NUE

All the case-study farms already implement measures to reduce N emissions along the manure cascade, particularly at the storage and application level, resulting in relatively high livestock system NUE of 54–76 % in the CM scenario (Table 9 and Fig. 6). Both pig case-study farms could reach the highest overall livestock system NUE with OM2, by implementing air-scrubbers in the barn, cover the liquid manure during storage, and add a strong acid during the application to cropland via injection (87 % and 86 % Livestock system NUE, Table 9). Dairy farms 3 and 4 have already implemented most measures in the CM and can thus increase their Livestock system NUE by only 5–6 % with the measures proposed in OM2 (Table 9). Differently, dairy farm 2 reached only 54 % livestock system NUE under CM, which is very close to the PM scenario (52 %, Table 9 and Fig. 6). Consequently, it has the highest optimization potential (21 % increase in both OM scenarios).

The livestock system NUE of CM is higher than that found in the other studies (Leip et al., 2011; Quemada et al., 2020). This can be partially ascribed to the methodological approach used in the SAGE modeling, with a different definition of livestock system NUE and the utilization of updated emission factors and the ALFAM2 model, which reveal lower emission factors than previously used. The manure management measures implemented in the CM scenario emphasizes the regional peculiarities in which the farms are situated. Dairy farm 2 represents a full-grassland based Swiss dairy farm located in a mountainous region with around 30 % solid manure production, where typical management consists in the partial application of liquid manure via broadcast (slopes > 18 % inclination), grazing and mandatory covering of the external liquid manure storage (Table 1). Thus, in this case the topography limits the implementation of low emission application techniques, and the OM1 and OM2 scenario might therefore not be fully implementable. The EU countries are legally obliged to implement best available technologies to avoid exceeding the N emission limits (Bjerg et al., 2023 and Table 1). Denmark and The

Table 9 Manure cascade NUE (% TN excreted in housing) and Livestock system NUE (% TN intake) of the six case-studies for the four *scenarios past management* (PM), *current management* (CM), *optimized management 1* (OM1) and *optimized management 2* (OM2).

Management scenario		PM	CM	OM1	OM2	Delta CM and OM2
		Manure cascade NUE (% TN excreted in housing)				
Case-study farms	Pig 1	46	68	75	81	13
	Pig 2	46	57	73	80	23
	Dairy 1	42	52	64	71	19
	Dairy 2	39	40	66	66	26
	Dairy 3	38	62	60	67	5
	Dairy 4	42	66	64	69	3
		Livestock system NUE (% TN intake)				
	Pig 1	60	76	82	87	11
	Pig 2	60	68	81	86	17
	Dairy 1	54	62	74	79	17
	Dairy 2	52	54	75	75	21
	Dairy 3	53	72	72	77	6
	Dairy 4	54	73	74	78	5

The delta is calculated by subtracting the OM2 scenario from the CM scenario, to obtain the highest potential Manure cascade NUE or Livestock system NUE increase.

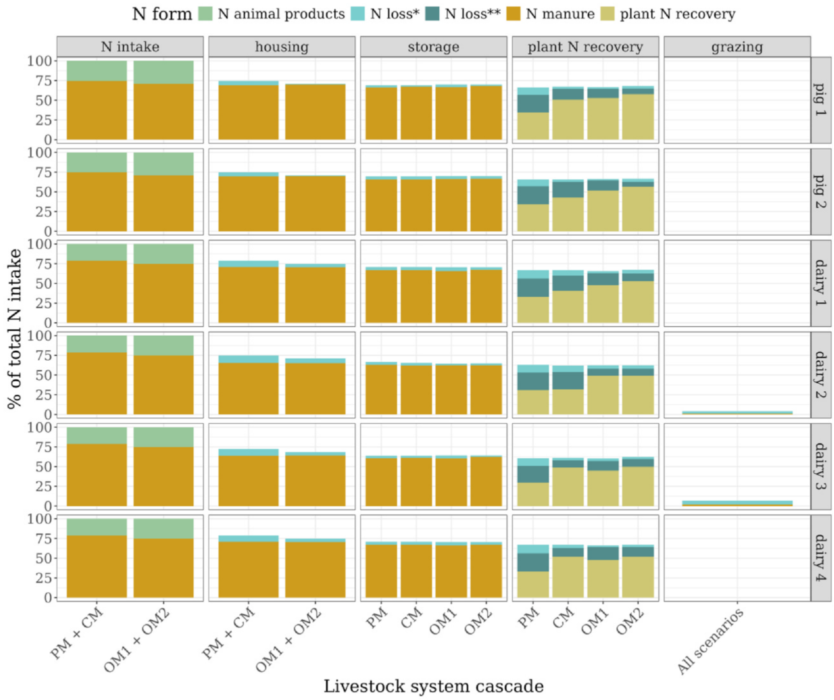

Fig. 6 Relative changes of total N intake (%) within the livestock systems for the four scenarios: *past management* (PM), *current management* (CM), *optimized management 1* (OM1) and *optimized management 2* (OM2). PM + CM and OM1 + OM2 are grouped at the *N intake* and *Housing* steps as the scenarios do not differ at these stages (e.g. both PM and CM implement the same measures at housing, measures described in Fig. 5 and Table 8). Similarly, the grazing management was kept constant for all scenarios. Both *N loss** and *N loss*** represent all N forms lost along the cascade. *N loss*** is represented with a different color to differentiate between the N losses during manure application and the *long-term* field N losses.

Netherlands go beyond these requirements, France and Germany follow the requirements, while in the region of Lombardy (Italy), an action plan to further improve the air quality has been signed in 2017 (Bjerg et al., 2023). Thus, the current management of the case-study farms located in these countries reflect the efforts made to fulfill this legal framework. Further measures as suggested in the OM scenarios could be or are already implemented in these regions and therefore represent realistic potentials for additional livestock system NUE improvements. The case-studies represent examples of good practice, given in the regional and legislative framework. It remains unclear, how many farms have already reached the CM or the

OM livestock system NUE. Nevertheless, the results highlight that a shift from PM to CM is absolutely necessary and from CM to OM already possible and should be promoted by national governments.

The highest proportion of N losses along the manure cascade for liquid manure occur with manure application and in subsequent years (plant N recovery, Fig. 6), defined as the manure NUE (Fig. 1), while for solid manures and separated solids the highest proportion of N losses usually occur in the housing and during manure storage. Manure quality (Jensen, 2013), application technique (Webb et al., 2010), canopy type and development (Häni et al., 2016), timing, based on climatic conditions (Section 3.5), and soil properties (Fangueiro et al., 2012a; Griffin et al., 2002), influence the *short-term*, and the *long-term* manure NUE (Section 3.6). Some of these factors can be controlled and thus considered for fertilizing planning, while others remain difficult to be implemented, when farmers are forced to weigh their actions depending on the current weather conditions. The manure type and field management specific *long-term* manure NUE values explored in Section 3.6 provide a baseline to further develop currently available decision support tools (Nicholson et al., 2020), to achieve an efficient N utilization in the *long-term*.

5. Final discussion and conclusions

The review highlights the potential to improve the N use efficiency of dominant European cattle and pig livestock systems. By following good agricultural practice, livestock system NUE can be further improved – from currently 55–75 %, to 75–85 % – if best available technology is applied. Such high efficiencies are probably the maximum of what can be achieved in agricultural systems due to the volatile and mobile nature of N species and require optimization measures at all stages of livestock systems from feeding to the manure cascade to fertilization optimization (Fig. 2). A number of farms in Europe still work below good agricultural practice standards with low livestock system NUEs of 50–60 % (PM scenario, Table 9). Their potential to improve livestock system NUE is large for all farm types and animal categories (Table 9).

The modeling approach is based on single emission factors at each stage of the livestock system. However, the review highlights that for each measure, there usually is a certain range in the measured emissions. Thus, experimental validation of modeling results on operational farms over the whole system cascade is still lacking, but would be necessary to consolidate the findings of the modeled case-study farms.

The N loss mitigation potential is largest for farms with closed housing system and subsequent manure processing, mainly for pig farms (Table 9). On the contrary, improvement of livestock system NUE at farms with a high grassland proportion and grazing is limited, due to the very low grazing NUE of 27 % (Table 7). This should not be underestimated in the European context, because permanent grasslands cover 34 % of the European Union's agricultural land (Schils et al., 2022) and grazing plays a central role for dairy production with land shares of > 70 % in many countries (Finland, France, Ireland, Norway, Sweden, Switzerland, The Netherlands, United Kingdom) and > 20 % in most European countries (Van Den Pol-Van Dasselaar et al., 2020). Additionally, large uncertainties exist on *long-term* manure NUE for solid and liquid manure particularly on grassland, since experiments lasting more than two decades are rare (Tables 5 and 6).

Future research should focus on tradeoffs associated with mitigation measures. Liquid manure acidification is one of the most efficient technologies (Section 3.3). However, it requires considerable amounts of concentrated acids, in most cases sulfuric acid, which surpasses crop demand by a factor of 5–15 (Loide et al., 2020). This may lead to leaching of sulfate to ground and surface waters, eventually creating problems for water quality (EU, 2000; EU, 2006), and cause accelerated acidification of soils (Zireeni et al., 2023). However, long term observations of soil acidification under field conditions are needed.

One of the drawbacks of the solid-liquid separation technology is the high content and potential losses of NH_4^+-N from the solid fraction during storage and application. This is reflected in lower manure cascade NUE in OM1 compared to OM2 (Table 9). Even though options for additional processing are available, future research should focus on further improving the N utilization of solid manure. In the case of the liquid fraction, new processing technologies like NH_3 stripping and/or vacuum evaporation, plasma activation or full-scale cascade membrane filtration are energy demanding technologies and are not evaluated for the full life cycle chain (Angouria-Tsorochidou et al., 2022). Additionally, data on overall manure NUE for all products generated in the production chain of these technologies are scarce.

Considering absolute quantities, measures after storage at the end of the manure cascade have the highest potential to improve NUE (Fig. 6, Table 8). Therefore, implementing N loss mitigation measures at earlier stages of the manure cascade like housing or storage can be subjected to pollution swapping, if no measures are implemented at the end after storage.

New crop management approaches to improve fertilizer NUEs after application and thus *long-term* NUE are an upcoming challenge (Velthof et al., 2024). This is true for manure as well as for mineral fertilizers, which reveal also low mineral fertilizer NUEs between 60–80 % (Table 5). In order to improve the *long-term* manure NUE assessment, site specific modeling approaches combined with earth observation technologies (drones, satellites) and consideration of the N inputs via N deposition, symbiotic N_2 fixation, crop residues and the N mineralization potential of the soil in combination with organic and mineral fertilizer N inputs have to be considered. This is especially important because more frequent extreme drought and wet periods in the future (Brás et al., 2021) will have a negative impact on yields and thus on NUE (Nóia Júnior et al., 2023). Climate change will pose immense challenges to the farmers and in this setting, a NUE increase can only be achieved in cropping and grassland systems, which are resilient against the changing environment.

Legislation on EU level have set only a number of shared obligations like a maximum N application with manure of 170 kg N ha^{-1}. The large differences in legislation points towards the need for harmonization and standardization to allow better integration with monitoring as it is envisaged and partly facilitated by the EUs nitrate directive. Harmonizing standards and monitoring principles shared targets and even legislation and enforcement strategies might lead to a broadly higher NUE. This may even help to close the gap between NUE of actual implemented measures and environmental effects, which is often unclear for farmers, policy makers and legislation authorities. Examples for good and best available techniques and practices are available on country (Bjerg et al., 2023) and farm level (this study), showing that quantitative monitoring combined with research can make the difference and should be strengthened to foster evidence-based policies and quantitative targets that can be implemented in legislation to increase NUE in general. This is supported by a consequent digitalization strategy, allowing aggregation of information from field to farm and finally to national and continental level.

As a first step, a broad implementation of best available technologies on the majority and particularly large farms is required, but additional mitigation measures complementing currently available technologies have to be developed to meet all environmental goals. Overall, livestock densities related to carrying capacity of a specific site combined with best available technologies is the most sustainable way to avoid harmful environmental impacts (De Vries et al., 2023).

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