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Evaluating the performance of biobased, recovered nitrogen fertilizers in European cropping systems using modelling

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ABSTRACT

Biobased fertilizers (BBFs) are gaining attention for their potential to advance a circular economy. This study used the Daisy model to evaluate the performance of three BBFs—ammonium sulphate (AS), digestate (DIG), and liquid fraction of digestate (LFDIG)—compared to baseline fertilization (mineral and manure) across ten European cropping systems. BBFs replaced baseline fertilization under three scenarios: (i) full replacement with equivalent total N input, (ii) full replacement with higher total N input due to BBFs' lower (<100%) fertilizer replacement values (FRVs, relative substitution rate of BBF to synthetic fertilizer), and (iii) partial replacement (only manure-N) with equivalent total N input. Results indicated that under both partial and full replacement scenarios with equivalent total N input, AS, DIG, and LFDIG had minimal impacts (<5%) on crop N yield, nitrogen use efficiency (NUE), and total N losses (gaseous and leaching) compared to the baseline. The soil organic N (SON) stocks either decreased or changed more slowly with AS and LFDIG. In scenario ii (higher total N input), BBFs led to increased N yield (2–18%) and N losses (avg. 76%), and decreased NUE (2–25%) relative to the baseline. DIG was the most effective at improving SON stocks (average increase 4.9 kg N $ha^{-1}y^{-1}$) and reducing N losses, followed by LFDIG and AS. The impact on N leaching varied, with higher leaching observed in annual cereal-based compared to semi-perennial grass-based systems. Implications are that BBFs should be applied assuming a high FRV (~100%), ensuring equivalent total N input when replacing baseline fertilization to prevent increasing N losses.

1. Introduction

Growing food demand has led to an increased reliance on synthetic fertilizers as a source of nitrogen (i.e., Haber – Bosch N; hereafter HB-N). Although the use of HB-N has played a key role in increasing crop yields (Stewart et al., 2005) and overall food production, which has doubled the amount of people that the world can sustain (Erisman et al., 2008), it has also disrupted the natural N cycle, creating an imbalance in nutrient budgets across geographical scales and ultimately increasing N emissions to the environment (Sutton et al., 2011), causing eutrophication, soil acidification, increased emission of greenhouse gases (nitrous oxide), water quality issues, and ecosystem degradation. Hence, reducing reliance on HB-N and finding alternative and more circular ways to fertilize crops, such as composts, products from anaerobic digestion, manures, slurries, cover crops, green manures, and materials produced from side streams sourced through nutrient recovery technologies, has been the frontier of agricultural research in recent years.

Animal manures, slurries and other farm residues can be used as an alternative to the HB-N. However, emissions and nutrient losses from manures and crop residues are a great challenge. The nutrient losses from nitrogen-rich plant residues could be substantial if mineralization is not synchronized with the nutrient uptake by the following crop (Kandel et al., 2020). Likewise, the use of raw manures and slurries often results in significant losses of nutrients through gaseous (NH_3, N_2O) and leaching (NO_3^-) pathways, which apart from being a direct loss of resources, causes severe environmental impacts (Aso et al., 2022). The wide-scale use of manures is also constrained by the N application legal limit (170 kg N ha^{-1} y^{-1}) set under the EU Nitrates Directive (ND) that applies to all nitrate-vulnerable zones (European Comission, 2013). Thus, due to environmental, legal and financial concerns regarding the use of HB-N, raw manures and crop residues, there is a strong need to develop and test safe, efficient, and environmentally benign alternatives.

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Anaerobic digestion (AD) has the advantages that it produces renewable fuel as well as provides a safe way to process farm-based organic waste streams (e.g. manure, slurry, straw), essentially favouring recycling and promoting a more circular economy (Chojnacka et al., 2020). The residual product of AD is called digestate (DIG), which can be used as a soil amendment and fertilizer (Möller and Müller, 2012). The DIG can be processed further through mechanical, chemical and biological treatments, which significantly change its composition, often to reduce the volume and increase the fertilizer value. The digestate-derived products include, but are not limited to, the liquid fraction of digestate (LFDIG; rich in N) and recovered ammonium sulphate (AS). The AS can be recovered from DIG through a series of processes involving ammonia stripping and scrubbing, and is considered similar to synthetic fertilizers as it is comprised of almost only mineral N.

The widespread use of products derived from manure, may still be limited by the EU Nitrates Directive's ceiling for nitrogen application. Since manures serve as feedstocks in anaerobic digestion, AS, DIG, and LFDIG may still be considered as manure-derived products and may only be applied in amounts not exceeding 170 kg N ha^{-1}. However, the Joint Research Centre of the EU proposed criteria that would permit the use of fertilizing products derived from manures through processing to exceed the aforementioned limit (Huygens et al., 2020). These products are suggested classified as REcovered Nitrogen from manURE (RENURE). The proposed criteria establish when manure-derived products meet the standard to be treated as "mineral fertilizers" and are exempted from the nitrogen application ceiling applicable to manures. It is possible that AD-derived products, particularly, AS, and LFDIG may eventually be classified as RENURE products, and therefore, exempt from the legal limit for manure-derived products (Reuland et al., 2021), which would eventually pave the way for wide-scale application of these products.

Nonetheless, the knowledge about the agronomic and environmental performance of AD-derived, bio-based fertilizers (BBFs) is still developing. Although a great deal of work has been done on the characterization (Reuland et al., 2021; Kovačić et al., 2022) and performance of BBFs (Möller et al., 2008; Möller and Müller, 2012), the large-scale application and testing of these products under diverse agro-environmental conditions is still lacking. Carrying out such an analysis through experimentation is not realistically feasible due to the involvement of a large number of agro-environmental factors. However, process-based agroecosystem models that not only simulate crop production but also track the cycling of C, N and water – such as Daisy (Hansen et al., 2012) – offer a great opportunity to evaluate the performances of BBFs. The Daisy model has been successfully used to evaluate the impact and performance of cropping systems under variable environmental factors and management practices, especially under European conditions (Manevski et al., 2016; Manevski et al., 2019; Gyldengren et al., 2020; Duan et al., 2021; Rashid et al., 2022a), and is thus a viable approach to use for the evaluation of BBFs. To our knowledge, no studies have thoroughly assessed different BBFs across the diverse conditions of Europe, including variations in cropping systems, soil types, climates, and management practices. This work was designed to fill these gaps by providing an in-depth, quantitative assessment across Europe.

This study aimed to evaluate the agronomic and environmental performance of BBFs through the modelling approach and provide an assessment of BBFs in comparison with conventional fertilization choices. The specific objectives were, i) to simulate and quantify N yield, nitrogen uptake efficiency (NUE) and N losses with current and biobased fertilization approaches under diverse European cropping systems, ii) to quantify the benefits of using BBFs for soil quality (changes in soil organic N (SON) stocks).

2. Materials and methods

This study followed these steps: (i) development of baseline crop production scenarios representing typical European cropping systems with conventional fertilization, (ii) setting up the Daisy model for baseline scenarios using site-specific data to simulate crop and soil N dynamics, (iii) characterization and parameterization of biobased fertilizers (DIG, LFDIG, AS) according to Daisy model requirements using lab incubation data on N turnover, (iv) selection of three N fertilizer replacement scenarios to replace baseline fertilization with BBFs, (v) simulation of baseline crop production scenarios with three BBFs (DIG, LFDIG, AS) and three N fertilizer replacement scenarios, (vi) using simulated outputs with BBFs and baseline fertilization to calculate N yield, NUE, N losses, and SON stock changes.

2.1. Baseline crop production scenarios

A baseline scenario reflects the regional geoclimatic conditions and current agricultural management practices of a specific farming system. To encompass Europe, six geoclimatic regions were chosen: Atlantic north (ATN), Atlantic central (ATC), Mediterranean east (MDE), Mediterranean west (MDW), continental east (CTE), and continental west (CTW). Representative cropping systems were chosen for each region (Table 1), following the rationale and procedures outlined by Duan et al. (2021). Briefly, a number of selection principles and methods were followed, such as baselines reflecting current agricultural practices, focusing on intensive farming systems with high environmental emissions, and well-characterized farming systems (e.g., supported by field trials). Scenarios were chosen considering several geoclimatic regions based on environmental stratification, with each region represented by several predominant farming systems and variations in topsoil texture and/or organic carbon content. Firstly, six geoclimatic regions were selected based on Europe's environmental zones (Metzger et al., 2012), mainly characterized by climate and soil type. Secondly, one or two farming types (dairy, pig, or arable production) were chosen per region (Table S1). This combination resulted in ten baseline scenarios, termed as cropping systems. Detailed principles and methods for the selection of baseline scenarios are given in Duan et al. (2021).

2.2. Model and modelling setup

2.2.1. The Daisy model

Daisy is a process-based model that simulates crop production and carbon, nitrogen, and water cycling (Hansen et al., 2012). In Daisy, solute and water transport follow convection-dispersion and Richards' equations. Organic matter turnover is managed by the soil organic matter sub-modules, influenced by clay, soil moisture, and temperature. Crop production relies on sub-modules considering photosynthesis, leaf area, radiation, temperature, nitrogen, water stresses, and senescence. Plant root water uptake is calculated using the Darcy equation, limited by potential evapotranspiration. Nitrogen uptake involves mass flow and diffusion. Daisy's groundwater component manages groundwater table, with options for aquitard or free drainage. In this study, Daisy version 6.22 (released 19-08-2021; https://daisy.ku.dk/) was utilized for simulating baseline crop production and fertilization scenarios.

2.2.2. Data, model calibration and simulation of baseline cropping systems

Input data to the Daisy model include weather, soil characterizations, and field management activities. The weather data for each geoclimatic region were obtained either from the EU Agri4Cast's gridded agrometeorological dataset (for CTE and MDE regions) or from the local weather stations (for all other regions). Weather data included daily means of temperature, precipitation, wind speed, humidity, global radiation, and evapotranspiration. The weather data were available for CTE, MDE, and MDW from 1979 to 2019, for ATN from 1992 to 2012, for CTW from 1992 to 2017, and for ATC from 1995 to 2015. For ATN, ATC, CTW and MDW regions, soil characterizations were retrieved from previous studies (Styczen et al., 2006; Groenendijk et al., 2017; Ehlert et al., 2018; Montemayor et al., 2019). CTE and MDE regions shared identical subsoil characterizations with CTW and MDW, respectively, whereas topsoil texture and organic carbon content in these two regions were obtained

Table 1
Cropping systems with crop sequences and fertilization in the baseline setup.

Cropping system	Crop sequences						Manure-total N (kg N ha^{-1} y^{-1})[d]	Manure type and FRV (%)[e]	Average mineral N input (kg N ha^{-1} y^{-1})	Average total N input (kg N ha^{-1} y^{-1})
	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6				
ATN-Dairy[a]	Silage maize (S 125, M 63)[b]	Silage spring barley (S 200, M 33) Grass-clover (67 M)	Ryegrass-clover ley (S 200, M 100)	Ryegrass-clover ley (S 200, M 100)	Ryegrass-clover ley (S 200, M 100)	Spring barley-usrg [c] (S 43, M 22)	230	Dairy, 70	219	425
ATN-Pig	Spring barley (S 101, M 44)	Winter rapeseed (S 150, M 65)	Winter wheat (S 131, M 81)	Winter wheat (S 147, M 65)	Spring barley (S 101, M 44)	Winter barley-FR (S 135, M 59)	170	Pig, 75	187	247
ATC Dairy	Perennial ryegrass (S 103, M 246)	Perennial ryegrass (S 103, M 246)	Perennial ryegrass (S 103, M 246)	Perennial ryegrass (S 103, M 246)	Silage maize (S 100)		170	Cattle, 60	277	394
ATC-Arable	Silage maize (S 181, M 17)	Potato (S 181, M 101)	Sugar beet (S 181, M 131)	Winter wheat (M 150)			170	Pig, 80	187	273
MDW-Maize	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	0		217	235
MDE-Maize	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	Silage maize (M 210)	0		219	235
CTE-Pig	Spring barley (S 101, M 44)	Winter rapeseed (S 150, M 65)	Winter wheat (S 131, M 81)	Winter wheat (S 147, M 65)	Spring barley (S 101, M 44)	Winter barley FR (S 135, M 59)	170	Pig, 75	187	244
CTE-Arable	Spring barley (M 145)	Winter rapeseed (M 215)	Winter wheat (M 212)	Winter wheat (M 212)	Spring barley (M 145)	Winter barley-FR (M 194)	0		187	202
CTW-Pig	Spring barley (S 101, M 44)	Winter rapeseed (S 150, M 65)	Winter wheat (S 131, M 81)	Winter wheat (S 147, M 65)	Spring barley (S 101, M 44)	Winter barley-FR (S 135, M 59)	170	Pig, 75	187	246
CTW-Arable	Spring barley (M 145)	Winter rapeseed (M 215)	Winter wheat (M 212)	Winter wheat (M 212)	Spring barley (M 145)	Winter barley-FR (M 194)	0		187	203

[a] Cropping-system names are based on geoclimatic regions and farm types. The geoclimatic regions are Atlantic north (ATN), Atlantic central (ATC), Mediterranean east (MDE), Mediterranean west (MDW), continental east (CTE) and continental west (CTW). The farm types are arable, dairy, pig and maize monoculture.
[b] S and M followed by numbers inside the brackets indicate the amount of available slurry-N and mineral-N respectively, applied for each crop, (kg N ha^{-1} y^{-1}).
[c] Cultivation of catch crops (if any) is indicated by abbreviations after the main crop viz. spring barley-usrg which is undersown ryegrass and winter barley-FR, which is fodder radish, sown after harvest of the main crop.
[d] Total manure-N amount that was allowed and applied according to the limits set under the EU Nitrates Directive.
[e] Manure type with standard fertilizer replacement values (FRV).

from the European Soil Database (ESDB) and the Land Use and Land Cover Survey (LUCAS). Soil hydraulic (Mualem van Genuchten) parameters were obtained from Styczen et al. (2006) (for ATN and CTW), Heinen et al. (2021) (for ATC), or the Hypres Database of Hydraulic Properties of European Soils (for CTE, MDE, and MDW).

Table 1 lists crop management details, which were obtained from field trials in each geoclimatic region (Styczen et al., 2006; Groenendijk et al., 2017; Ehlert et al., 2018 Montemayor et al., 2019). The standard fertilization applied in the cropping systems of the baseline scenarios was termed 'baseline fertilization,' and the amount of baseline N fertilizer for each scenario was based on the recommended or regulatory N fertilization rate for the respective crop and regions/sites (Table 1). The type of N fertilizers used as baseline fertilization was either mineral or a combination of mineral and manure N, where the manure was either cattle or pig slurry depending on the farm type. The manures and slurries were applied using specific fertilizer replacement values (FRVs) as shown in Table 1, which indicate the relative proportion of total N in an applied organic fertilizer (e.g., manure, digestate) that can substitute synthetic fertilizer N, based on nitrogen content, plant availability and expected volatilization losses. Manure N was always applied at the maximum rate allowed under the EU Nitrates Directive, i.e., 170 kg total N ha^{-1} for arable crops. For the ATN-Dairy scenario, manure N was applied at 230 kg total N ha^{-1} due to a national derogation. The fertilizer replacement values (FRV) of cattle and pig slurries were 70 and 75% (slurry-1; https://lbst.dk/landbrug/goedning/vejledning-om-goedsknings-og-harmoniregler) in ATN and CTW, and 60% and 80% in ATC, respectively (slurry-2; https://www.rvo.nl/onderwerpen/mest/tabellen). For all other regions, the FRV was set at 70% (Table 1). Irrigation was applied in scenarios ATC-Dairy, MDE-Maize and MDW-Maize during the summer (May to Sep), at a rate of 30 mm/h for 1 h when the soil water pressure head dropped below −600 cm in the top 30 cm soil profile. Arable crops were harvested once at maturity, whereas ryegrass and grass-clover leys were mowed four times from June to September (Rashid et al., 2022b). For cereal crops, potato, and sugar beet, dry matter/N yield refer to storage organ only, whereas for silage maize and grass, yields consisted of all harvested aboveground biomass.

The Daisy crop modules were calibrated against field observations for crop dry matter and nitrogen yields (Table S2). Briefly, existing crop modules from the Daisy model library were used as the starting point and further calibrated to reduce the percentage (%) difference between simulated and observed dry matter and N yields from field trials in the respective regions. The calibration aimed to retain the original parameter sets for each crop as much as possible, ensuring the Daisy model's applicability for scenario analysis. It focused on a small number of key parameters, including photosynthetic rate, N concentrations in plant parts based on development stages, and the maximum and minimum N uptake rate through roots. The process was carried out manually using a trial-and-error approach. Calibration outcomes are presented in Table S2, which resulted in simulated yields closely matching targets, typically around a 10% margin. For scenarios ATC-Arable and ATC-Dairy, the model was further calibrated for temporal dynamics of the groundwater table using field measurements following the method outlined by Duan et al. (2021), involving adjustments to soil column parameters affecting aquitard thickness, water conductivity, and pressure potential in the aquifer (Fig. S1). The organic matter module of the Daisy model utilized parameterization based on long-term field experimental data (Bruun et al., 2003). Given Daisy's recurrent evaluation and widespread use, re-evaluation of other model processes was omitted, aligning with common practice in well-tested model studies for scenario analysis (Ozturk et al., 2018). This is also because this study aimed to use a well-tested model to evaluate the performance of BBFs against a mineral baseline, rather than to calibrate and test the model itself under various soil-climate-management conditions. However, the model was calibrated specifically to simulate N mineralization from the novel BBFs evaluated in this study (see details in section 2.3.1).

For simulations, the model underwent a 30-year spin-up period, followed by 60-year simulations for scenario results. Due to limited weather data availability (approximately 20 years – ATN, ATC, CTW), weather data were reused twice after the initial 20 years of simulation. Year sequences in the later 40 years were randomized to reduce repeated weather-crop combinations. This approach enabled continuous utilization of a real weather dataset representing current climatic conditions. Soil N balance components were logged annually on February 28, spanning the rooting depth or at least 50 cm below the drainage depth if shallower (Table S1). Logging depths varied as 190 cm for ATN-Pig, CTE-Arable, CTE-Pig, CTW-Arable, and CTW-Pig; 160 cm for ATC-Dairy; 180 cm for ATC-Arable; 150 cm for ATN-Dairy; and 100 cm for MDE-Maize and MDW-Maize, reflecting differences in maximum rooting depth and drain depth.

2.3. Fertilization scenarios

The fertilization scenarios were based on three biobased fertilizers (BBFs) and three fertilizer replacement schemes (FRS).

2.3.1. Biobased fertilizers and their characterization

Three biobased products were chosen: digestate (DIG), liquid fraction of digestate (LFDIG), and ammonium sulphate (AS). Manure-derived products are subject to EU N application limits, but some may be exempt if they meet RENURE (Recovered Nitrogen from Manure) criteria, such as a mineral N to total N ratio >90% or a TOC:TN ratio <3 (Huygens et al., 2020). AS and LFDIG were selected for likely meeting these conditions, with DIG as their primary source. These were characterized, and Daisy model's organic matter module was parameterized using lab incubation data from Sigurnjak et al. (2017). The dry matter, total C, total N, and $NH4^{+}$-N fractions (of dry matter) were 0.07, 0.386, 0.061, 0.512 for DIG, and 0.023, 0.309, 0.204, 0.766 for LFDIG, respectively (Table 2). NO_3^--N in both was negligible. In Daisy, organic matter turnover is simulated by dividing added matter into slow and fast decomposing pools (AOM-slow and AOM-fast), which are mainly characterized by initial fractions, turnover rates, and CN ratios. These parameters were calibrated using N mineralization data from Sigurnjak et al. (2017), applying an automated optimization routine with the Nelder-Mead algorithm (SciPy ver. 1.7.3; Virtanen et al., 2020) to minimize RMSE between simulated and observed N mineralization fractions (Fig. 1; Table 2).

For the slurries in the baseline, standard parametrization available from the Daisy library was used. The Daisy model also needs information on the ammonia volatilization losses from field-applied nitrogenous fertilizers, which were estimated for both DIG (10.8%) and LFDIG (7.1%) using the ALFAM2 model, assuming application with trailing shoe, no rainfall, 15 °C temperature, and a wind speed of 2 m/s (Hafner et al., 2019). The ammonium sulphate (AS) was characterized to contain 100% NH_4^+-N.

Table 2
Physico-chemical characteristics and parametrization of added organic matter pools of Daisy for digestate and liquid fraction of digestate.

Added organic matter pools[a]	Parameters	Digestate	Liquid fraction of digestate
AOM-slow[a]	Turnover rate (h^{-1})	7.56 e^{-5}	6.54 e^{-5}
	Initial fraction	0.89	0.89
	CN ratio	12.97	5.85
AOM-fast[a]	Turnover rate (h^{-1})	0.038	0.051
	Initial fraction	0.01	0.01
	CN ratio	12.97	2.32
Physico-chemical characteristics[b]			
	Dry matter fraction	0.071	0.023
	C fraction	0.386	0.309
	N fraction	0.061	0.204
	NH_4-N fraction	0.512	0.766

[a] The parameter values of AOM-slow and AOM-fast pools were calibrated based on N mineralization data from laboratory incubation experiments.
[b] The physicochemical characteristics are measured values.

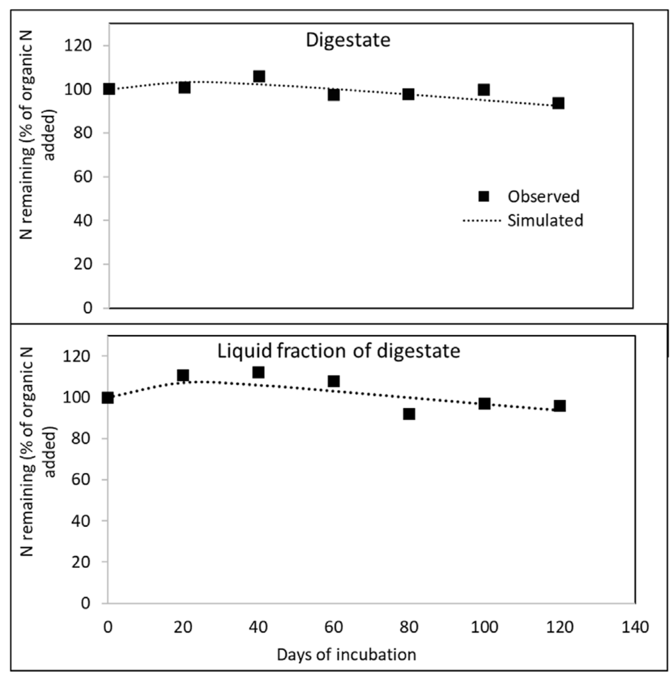

Fig. 1. Simulated and observed values of N remaining (percentage of total organic-N added). The observed values are calculated from the lab incubation experiment with digestate (DIG) and liquid fraction of digestate (LFDIG). Values higher than 100 indicate immobilization, while values lower than 100 indicate mineralization.

2.3.2. Fertilizer replacement scenarios

Three fertilizer-replacement scenarios (FRS) were selected (Table 3). In FRS-1, only manure-N from baseline fertilization was replaced with DIG, LFDIG, and AS, following current legal restrictions (European Comission, 2013). FRS-2 and FRS-3 fully replaced both mineral and manure components of baseline fertilization with DIG, LFDIG, and AS. FRS-2 and FRS-3 assumed anaerobic digestion-derived products might eventually be exempt from these restrictions, allowing higher application rates (Huygens et al., 2020). In FRS-1 and FRS-2, the FRV of BBFs matched that of slurries in baseline scenarios. In FRS-3, FRV of BBFs was adjusted to match total applied N as in baseline. This created nine fertilization scenarios (3 BBFs: AS, DIG, LFDIG, and 3 FRS: FRS-1, FRS-2, FRS-3) for evaluation.

Table 3
Fertilizer replacement scenarios (FRS) with varying degrees of substitution in the baseline fertilization with biobased fertilizers (BBFs).

Scenarios	Nitrogen replacement with BBFs	Fertilizer replacement value (FRV)	Total N applied
Baseline	Reference: crop N demand supplied as manure N, mineral N or combination.	FRV of slurries depended on slurry type and country (Table 1)	Reference amount: depending on manure FRV and cropping system
FRS-1	Partial replacement: only manure-N proportion was replaced with BBFs	FRV of BBFs assumed same as of slurries in the baseline	Same as in the baseline scenarios
FRS-2	Full replacement: both mineral and manure N proportions were replaced with BBFs	FRV of BBFs assumed same as of slurries in the baseline	Higher than the baseline scenarios
FRS-3	Full replacement: both mineral and manure N proportions were replaced with BBFs	FRV of BBFs was adjusted to maintain an equal total N application as in the baseline scenario	Same as in the baseline scenarios

2.4. Simulation of fertilization scenarios

A total of 78 scenarios were evaluated, consisting of nine fertilization scenarios (three biobased fertilizers and three fertilizer replacement schemes) applied to ten baseline-cropping systems (Table 1). The cropping systems CTE-Arable, CTW-Arable, MDE-Maize, and MDW-Maize, which were only fertilized with mineral N (HB-N) in the baseline setup, were not evaluated under FRS-1 (partial substitution).

2.5. Calculations and compilation of results

The simulated crop and soil N dynamics were used to calculate i) harvested N, ii) total N losses (iii + iv), iii) gaseous N losses, iv) non-gaseous N losses (leaching), v) NUE, and vi) SON stock changes. The total N in the harvested (removed) plant biomass (excluding the crop residues/stubble that remained on the soil) was considered as the N output. Total N losses were calculated as the sum of gaseous (ammonia volatilization, N_2O nitrification, total denitrification) and non-gaseous (soil matrix leaching and drain flow) N losses. The NUE was calculated as a ratio of N output and total N input, where total N input was equal to the sum of N in organic and inorganic fertilizers, atmospheric N deposition, seed N, and N-fixation by legumes. The SON stock changes were calculated as the annual change in the organic N stocks in the target soil profile.

For the baseline scenarios, the average annual values of the aforementioned variables are presented. To evaluate the effect of biobased fertilizers, the results for individual fertilization scenarios are presented as a response relative to the respective baselines. The relative responses were calculated as follows

$$\textbf{Relative response} = \frac{\textbf{value with biobased fertilization}}{\textbf{value with baseline fertilization}}$$

A relative response value of 1.10 means a 10% increase compared to the baseline, while a value of 0.90 indicates a 10% decrease.

3. Results

3.1. Model calibration for N mineralization of added bio-based fertilizers

Fig. 1 shows the results for DIG and LFDIG, demonstrating good agreement between simulated and observed N remaining over time (expressed as % of total organic N added). The evaluation metrics, RMSE and Pearson's correlation, were 2.67 and 0.72 for DIG, and 4.55 and 0.85 for LFDIG, respectively, further confirming this agreement. Both DIG and LFDIG showed initial N immobilization (N remaining >100% of initial) followed by mineralization. Differences between simulated and observed patterns may occur due to the model's limited ability to capture multimodal N release dynamics, likely influenced by variations in degradability and C/N content of different organic matter pools. The slightly higher RMSE for LFDIG was likely due to fast mineralization followed by immobilization around day 80, which was not captured by the model.

3.2. Nitrogen dynamics with baseline fertilization

Simulated harvested N (N yield), NUE, ΔSON and N losses under baseline fertilization indicated large variability among cropping systems (Table 4). The ATC-Dairy was the most productive system with an average annual harvest of 326 kg N ha^{-1}, while ATN-Pig was the least productive (156 kg N ha^{-1} y^{-1}). The harvested N was about 10 kg N (6%) higher for the CTW cropping systems as compared to CTE cropping systems. Likewise, the harvested N was up to 3 % higher for CTW/CTE-Pig as compared to CTW/CTE arable systems, respectively (Table 4). Generally, the N inputs and outputs were highly correlated ($R^2 > 0.73$). The highest (83%) and lowest (58%) NUE values were observed for ATC-Dairy and ATN-Dairy, respectively (Table 4). The N losses through leaching were highest for ATN-Dairy followed by ATC-Arable, while ATC-Dairy and CTW arable cropping systems had the lowest losses through leaching. The leaching losses were around 33% (16.5 kg) lower for CTW compared to CTE. Similarly, the N leaching was ca. 15 kg (32%) lower for CTE/CTW-Arable compared to CTE/CTW-Pig cropping systems. The gaseous N losses were generally highest for cropping systems on Dairy and Pig farms compared to arable or maize monoculture (Table 4). The highest and lowest gaseous N losses were observed for CTW-Pig and CTE-Arable, respectively (Table 4).

The SON stocks increased in three out of ten cropping systems, with the highest accumulation observed under ATN-Dairy (18 kg N ha^{-1} y^{-1}). Depletion of SON stocks was observed in seven cropping systems, with the highest depletion recorded under ATC-Arable (−21 kg N ha^{-1} y^{-1}). The changes in SON stocks were primarily influenced by crop selection, particularly cropping systems involving ryegrass and grass-clover-based leys (ATC-Dairy, ATN-Dairy), which had a strong positive effect on stock accumulation. The impact of soil-climate (comparing CTE-Arable/Pig with CTW-Arable/Pig) and fertilizer type (comparing CTE/CTW-Arable with CTE/CTW-Pig) on the reduction of SON stocks was 4 kg (west > east) and 3 kg (arable > pig), respectively (Table 4). The results for MDE-Maize and MDW-Maize showed minor differences, with slightly higher N leaching (4 kg) and SON stock depletion (2 kg) observed under a west-climate (MDW).

3.3. Performance of biobased, recovered N fertilizers compared to baseline fertilization

The results for the effects of BBFs use are presented relative to the baseline fertilization (Figs. 2–4), except for the SON stocks, for which actual values of stock changes with biobased and baseline fertilization are presented (Fig. 5).

The simulated harvested N with the three biobased fertilizers averaged over cropping systems was comparable (change <2%) to baseline under the 'equivalent total N input' scenarios (FRS-1 and FRS-3), and ca. 7% higher under FRS-2 that received comparatively higher total N input (Fig. 2). There were some differences among cropping systems for the effects of BBFs and FRS. The harvested N was up to 15% lower for arable/maize cropping systems under FRS-3, and the highest reduction was with DIG, followed by LFDIG and AS (Fig. 2). Likewise, the increase in harvested N under FRS-2 was not similar for all cropping systems, it was highest for MDE/MDW-maize (up to 18%) followed by ATC-Dairy (up to 16%). The increase in harvested N under BBFs under FRS-2 was in the order of AS > LFDIG > DIG.

The average change in NUE (calculated as harvested N/total N input) under BBFs and cropping systems compared to baseline fertilization was

Table 4
Simulated nitrogen dynamics of ten cropping systems with baseline fertilization.

Cropping system	Total N input (kg ha^{-1} y^{-1})	Harvested N (kg ha^{-1} y^{-1})	NUE (%)	ΔSON (kg ha^{-1} y^{-1})	N leaching (kg ha^{-1} y^{-1})	Gaseous N losses (kg ha^{-1} y^{-1})	Total N losses (kg ha^{-1} y^{-1})
ATC Dairy	394	326	83	11	25	31	56
ATC Arable	273	176	65	−21	94	23	117
ATN Dairy	425	247	58	18	128	32	160
ATN Pig	247	156	63	−2	64	28	92
CTE Arable	202	157	78	−1	41	7	48
CTE Pig	244	162	66	2	58	26	84
CTW Arable	203	167	82	−5	26	14	40
CTW Pig	246	173	70	−2	40	34	74
MDE Maize	235	190	81	−5	29	21	50
MDW Maize	235	188	80	−7	33	21	54

Total N input is not simulated, it's an input to the model.

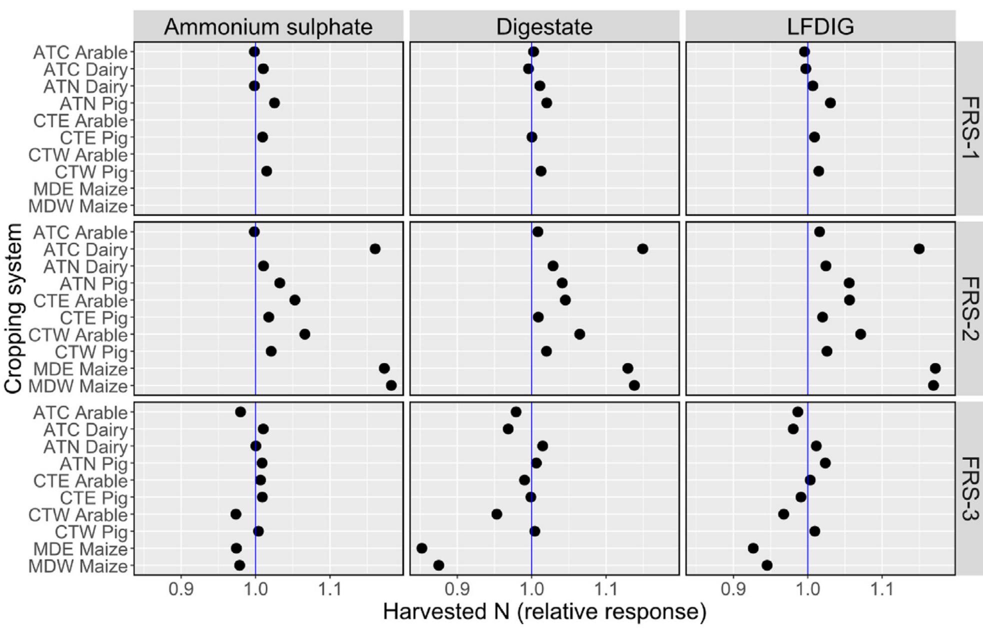

Fig. 2. Simulated effects on harvested N with biobased fertilizers (ammonium sulphate, digestate, liquid fraction of digestate-LFDIG) relative to the baseline fertilization. The effects of biobased fertilizers were evaluated for ten cropping systems using three fertilizer replacement scenarios (FRS-1 = partial replacement where only manure N was replaced; FRS-2 and FRS-3 = full replacement where both mineral and manure N were replaced). The total N input under FRS-1 and FRS-3 was equivalent to the baseline, but higher under FRS-2. Results are presented relative to baseline, representing baseline fertilization. A relative response of 0.9 and 1.1 indicates harvested N 10% lower and 10% higher, respectively, than baseline.

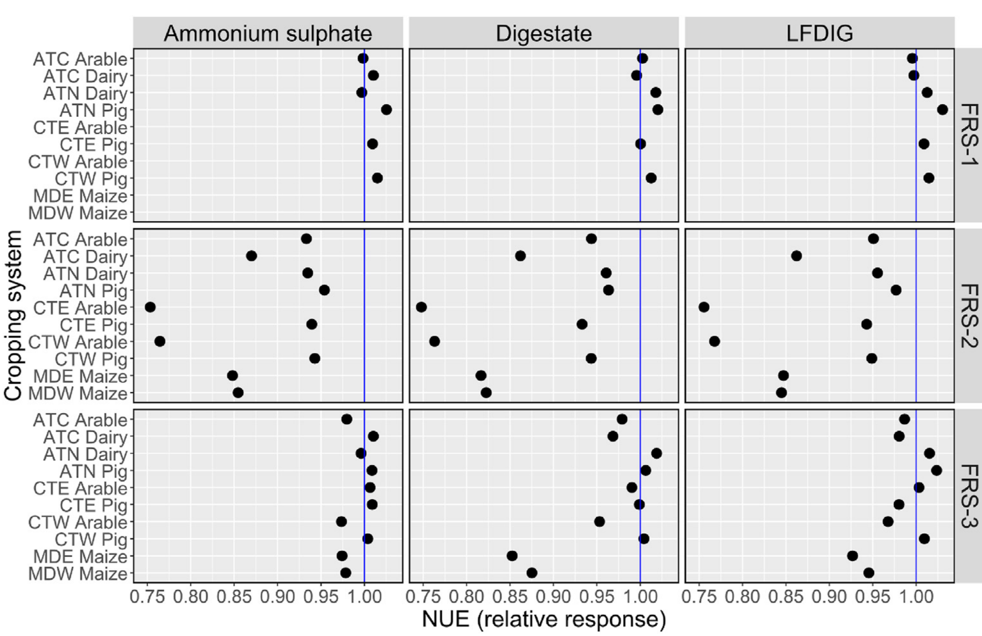

Fig. 3. Simulated effects on NUE with biobased fertilizers (ammonium sulphate, digestate, liquid fraction of digestate-LFDIG) relative to the baseline fertilization. The effects of biobased fertilizers were evaluated for ten cropping systems using three fertilizer replacement scenarios (FRS-1 = partial replacement where only manure N was replaced; FRS-2 and FRS-3 = full replacement where both mineral and manure N were replaced). The total N input under FRS-1 and FRS-3 was equivalent to the baseline, but higher under FRS-2. The results are presented relative to the baseline, which represents the current fertilization.

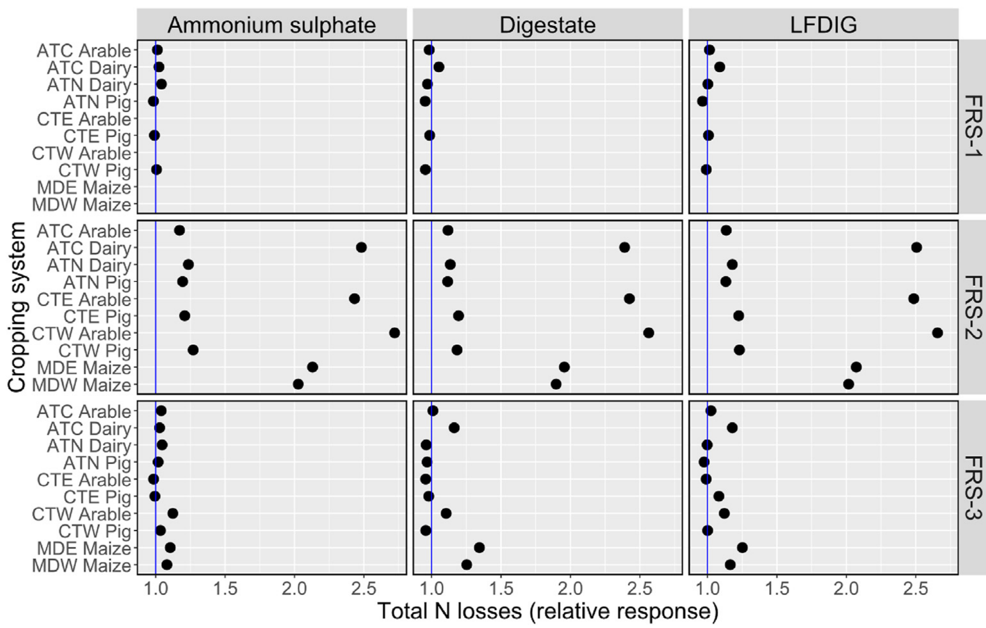

Fig. 4. Simulated effects on total N losses with biobased fertilizers (ammonium sulphate, digestate, liquid fraction of digestate-LFDIG) relative to the baseline fertilization. The effects of biobased fertilizers were evaluated for ten cropping systems using three fertilizer replacement scenarios (FRS-1 = partial replacement where only manure N was replaced; FRS-2 and FRS-3 = full replacement where both mineral and manure N were replaced). The total N input under FRS-1 and FRS-3 was equivalent to the baseline, but higher under FRS-2. The results are presented relative to the baseline, which represents the current fertilization.

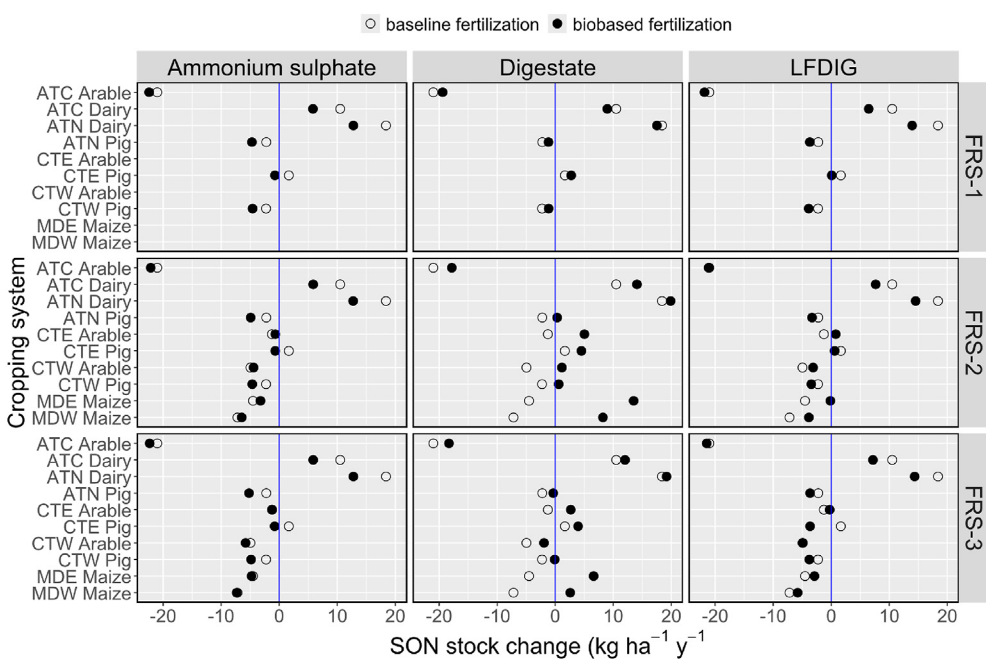

Fig. 5. Simulated soil organic N stock (SON) changes with biobased fertilizers (ammonium sulphate, digestate, liquid fraction of digestate-LFDIG) (filled circles) and baseline fertilization (empty circles). The BBFs were evaluated for ten cropping systems according to three fertilizer replacement scenarios (FRS-1 = partial replacement where only manure N was replaced); FRS-2 and FRS-3 = full replacement where both mineral and manure N was replaced). The amount of 'total N input' under FRS-1 and FRS-3 was equivalent to the baseline but higher under FRS-2. The vertical blue line indicates a steady-state level (no change in stocks), while positive and negative values indicate stock accumulation and depletion, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

small under FRS-1 (increase <1%) and FRS-3 (decrease <2%), but considerably higher under FRS-2 (decrease ca. 12%) (Fig. 3), emphasizing the impact of higher total N input under FRS-2. Within FRS-2, the reduction in NUE compared to baseline was higher for arable/maize cropping systems, reaching up to 25% for CTE-Arable (Fig. 3). Among BBFs, there was a suggestion of lower NUE with DIG compared to AS and LFDIG, specifically for MDE/MDW-Maize under FRS-2; however, the differences among BBFs for most cropping systems were minimal.

Compared to the baseline fertilization, total N losses (gaseous + leaching) averaged over BBFs and cropping systems were ca. 1%, 76% and 8% higher under FRS-1, FRS-2 and FRS-3, respectively (Fig. 4), which mainly highlighted the effect of higher total N input under FRS-2. The differences among three BBFs (averaged across cropping systems) were rather small (difference <3 %) under FRS-1 and FRS-3; however, there was a tendency for a higher increase in N losses with AS (79%) compared to LFDIG (76%) and DIG (70%) under FRS-2. Irrespective of BBFs, the increase in N losses was generally higher for arable/maize-based cropping systems under FRS-2, i.e. CTW-Arable with 265% average increase compared to baseline (Fig. 4). Overall, there was a tendency of lower N losses with DIG compared to AS and LFDIG.

Total N losses were divided into leaching and gaseous losses. Compared to the baseline, the N leaching values averaged across BBFs and cropping systems were 11%, 98% and 1% higher under FRS-1, FRS-2 and FRS-3, respectively (Fig. S2). The order of N leaching losses with BBFs was AS > LFDIG > DIG. Gaseous N losses decreased by 22% under FRS-1 but increased by 47% and 22% under FRS-2 and FRS-3 (Fig. S3). Gaseous losses were highest with DIG and least with AS. Gaseous N losses decreased on Pig farms (ATN-Dairy/Pig and CTE/CTW-Pig), regardless of BBF and FRS, with a higher decrease observed with AS (Fig. S3).

The results for the effects of different fertilization scenarios on the rate of change in SON stocks are presented in Fig. 5. In comparison with the baseline fertilization, the SON stocks decreased or increased at a lower rate with AS and LFDIG, especially under FRS-1 and FRS-3 (Fig. 5), and generally these effects were consistent across cropping systems. Even under a high total N input scenario (FRS-2), the AS and LFDIG negatively affected SON stocks for six cropping systems, and had only a small positive effect for the remaining four cropping systems (filled vs empty circles in Fig. 5). In contrast, the DIG positively affected SON stocks compared to the baseline fertilization under all FRS and cropping systems, expect for Dairy rotations that received high amounts of manure in the baseline setup.

The positive effect of DIG on SON stocks was more pronounced under the FRS-2. Compared to the baseline (-1.3 kg N ha^{-1} y^{-1}), the average rate of change in SON stocks with DIG was 4.9 kg N ha^{-1} y^{-1} under FRS-2. Among cropping systems, the highest positive effect of DIG was observed in the MDE-Maize system, where the rate of change in SON stocks increased from -4.5 kg N ha^{-1} y^{-1} in the baseline to 13.5 kg N ha^{-1} y^{-1} under DIG and FRS-2. As expected, the magnitude of the positive effect of DIG varied depending on the farm type and fertilizer type. For instance, the effects were most significant for arable farms that only received mineral N in the baseline setup (e.g., CTE/CTW-Arable and MDE/MDW-Maize). Overall, only three cropping systems built SON stocks under the baseline fertilization, but with DIG under FRS-2, all cropping systems, except ATC-Arable, started to build SON stocks (comparing the number of filled and empty points on the right side of the blue vertical line in Fig. 5).

4. Discussion

4.1. Analysis approach

The modelling approach taken in this analysis is suitable and commonly used when the aim is to evaluate the impact and/or potential of different products and practices under a diverse set of cropping systems and biophysical conditions (Manevski et al., 2019; Duan et al., 2021). An alternative approach would be to carry out multiple experiments, which despite being able to produce reliable and realistic empirical results, is cumbersome, expensive and time-consuming (Reynolds et al., 2018). Nonetheless, the modelling approach is not meant to replace field trials entirely, but rather, model results can be used to select the most important treatments and to target key processes and mechanisms worthy of further investigation (Reynolds et al., 2018).

The reliability of model-based assessments depends on how well a model can simulate the relevant processes, generally tested through calibration/validation. This study relied on a well-tested process-based model with proven records to simulate crop production and cycling of C, N and water in the plant-soil-environment-continuum with high confidence (Hansen et al., 2012; Manevski et al., 2016, 2019). Although the Daisy model and its submodules have already been extensively calibrated and tested over the years (Bruun and Jensen, 2002; Bruun et al., 2003; Manevski et al., 2016; Gyldengren et al., 2020), further efforts were also made to improve model calibration for key outputs (Fig. S1; Table S2). In this study, the crop N yield (harvested N) was considered a key output for calibration because it ensures the correct amount of N removal from the soil, which ultimately improves the simulation of N losses (Table S2). Secondly, calibration for groundwater was also important, as water fluxes over the lower boundary strongly influence nitrate leaching. While N yields and dry matter yield were calibrated for all cropping systems, calibration of groundwater was limited to only two systems due to lack of relevant data. Despite this limitation, Daisy is built on sound physico-chemical processes with the capability to produce reliable results even for less-characterized systems. However, despite model abilities and model calibration, it is generally recommended to always exercise caution when interpreting model-based scenario results because large variations can occur in N dynamics across small spatiotemporal scales (Børgesen et al., 2019). Moreover, Daisy does not simulate pest attacks or abiotic/biotic stress-induced physiological feedbacks that can affect plant N dynamics. Additionally, management practices may vary at smaller scales (e.g., individual farms), while this study considered only one set per region. Future studies could explore these smaller-scale variations in BBF performance. The results could also be uncertain due to (i) the model's inability to capture biophysical processes, (ii) uncertainty in input data, and (iii) unincorporated processes. To minimize uncertainty, two approaches were used: (i) BBF results were presented relative to baseline results, which were simulated using well-calibrated modules, and (ii) simulation results were averaged over 30 years to reduce uncertainty from temporal variations, weather year-crop combinations, and carry-over effects.

4.2. Performance of biobased, recovered N fertilizers compared to the baseline fertilization

The simulation of soil and crop N dynamics with baseline (manure + synthetic N) and novel BBFs (AS, DIG, LFDIG) provided insights into the agronomic (N yield, NUE) and environmental (N losses, SON stock changes) performance of these products. The BBFs are primarily encouraged for nutrient recycling from farm-based waste streams, eventually to replace manure and synthetic mineral fertilizers (Huygens et al., 2020; Reuland et al., 2021). The results from this study indicated that using BBFs (AS, DIG, LFDIG) to replace manure-N proportion of baseline fertilization (FRS-1; Table 3) had a negligible effect on N yield, NUE and total N losses (FRS-1 in Fig. 2–4). In a similar study, Hendriks et al. (2022) also reported no significant difference in potato yield comparing the mineral fertilizer with a combination of manure and BBF. These findings suggest that the plant available N under BBFs most likely remained comparable to the baseline fertilization. Other studies have also found that evaluating products based on equivalent total plant available N may result in similar performance (Gunnarsson et al., 2010; Sigurnjak et al., 2019; Huygens et al., 2020).

Conversely, the lack of a clear difference in total N losses under BBFs compared to baseline (FRS-1, Fig. 4) was unexpected because generally processed organic fertilizers (e.g. DIG) are susceptible to, i) higher nitrate leaching due to asynchrony between mineral-N release and crop demand (Schröder et al., 2013; Svoboda et al., 2013), ii) higher gaseous (ammonia volatilization) losses due to increased pH and dry matter content (Pain et al., 1990; Bell et al., 2016) and higher denitrification due to higher content of easily degradable carbon (Surey et al., 2020). The variation in total N losses between baseline and BBFs was less than 5 % (up to 5 kg N ha^{-1} y^{-1}; FRS-1 in Fig. 4); however, since leaching and gaseous N losses can occur independently, looking at only total N losses may be misleading. Our results showed that although total N losses were similar (FRS-1, Fig. 4), the N leaching was generally slightly higher under BBFs compared to baseline, especially for cereal-based cropping systems (CTE-Pig, CTW-Pig, Fig. S2). This might be due to asynchrony between crop demand and release of organically bound N (De Boer, 2008; Kandel et al., 2020). The effect was larger for cereal-based cropping systems, most likely due to a shorter active N uptake period, compared to semi-perennial systems, which continuously take up N from spring to late autumn (de Boer, 2017). This is likely due to the physico-chemical properties of BBFs and the growth patterns of annual and perennial crops. Mineral-like BBFs, such as AS and LFDIG, likely provide readily available N, aligning with the early spring uptake of annuals, while perennials likely benefit from more prolonged nutrient release (e.g., from DIG) throughout the entire growing season, including autumn and even winter (in warmer climates). This highlights the need to optimize BBF application timing to maximize N uptake and reduce losses. These findings may have implications for crop management, especially for the time of application of these BBFs in different systems, which must be investigated in a separate study. In contrast, higher gaseous N losses are expected from processed biobased products due to the reasons mentioned above, but it is important to consider the reference that is being used for comparison (here slurries). The estimated ammonia volatilization losses for DIG, LFDIG, slurry-1 and slurry-2 were 10.8, 7.1, 15 and 5%, respectively. This is one reason why the gaseous N losses with BBFs were lower compared to the baseline for all cropping systems, except for systems where slurry-2 was used, namely ATC-Dairy and ATC-Arable (FRS-1 in Fig. S3).

Partial substitution of baseline fertilization (manure-N only) with AS and LFDIG depleted SON stocks at a greater rate (FRS-1, Fig. 5), which was expected because these products certainly add less organic material to soil than manure in the baseline. The DIG, however, maintained SON stocks at the same level as in the baseline (FRS-1, Fig. 5) and increased SON stocks in the full replacement scenario (FRS-2, FRS-3; Fig. 5). These results imply that DIG may be used as a replacement for manure without affecting the SON stocks and total N losses (FRS-1 in Fig.4; Fig. 5). However, the net increase in SON stocks in the full replacement scenario was mainly the consequence of higher total N input, and it may also increase the total N losses in the long run upon mineralization of organically bound N (Buckwell and Nadeu, 2018), as evidenced by higher gaseous and leaching N losses (FRS-2; Fig. S2; Fig. S3).

4.3. Importance and implications of partial and full replacement of baseline fertilization with biobased fertilizers

In this study, the “partial replacement” of baseline fertilization involved substituting only the manure-N proportion with BBFs (Table 3), aligning with current EU Nitrates Directive regulations that limit N application from manures to 170 (or 230) kg N ha^{-1} y^{-1} (European Commission, 1991). This regulation applies to all manures and manure-derived products, including AS, DIG and LFDIG, posing challenges for both the profitability of the biogas sector (Alburquerque et al., 2012) and enhanced nutrient recycling from farm waste streams on a larger scale (Sigurnjak et al., 2019). However, products meeting specific criteria as RENURE products may be exempt from this N application ceiling (Huygens et al., 2020). While the criteria for such exemption are pending EU adoption, it's crucial to assess the implications of full replacement. Alongside partial replacement, this study also investigated full replacement scenarios (FRS-2 & FRS-3; Table 3) for potential (AS, LFDIG) RENURE products (Huygens et al., 2020). Although DIG is a less favorable RENURE candidate, it was included in the full replacement analysis. The results from this study showed that under the (FRS-2, the total N input was about 22% higher compared to the baseline, FRS-1 and FRS-3, which considerably affected plant and soil N dynamics (Fig. 2; Fig. 3; Fig.4; Fig. 5; Fig. S2; Fig. S3). The N harvest was only marginally higher (ca. 8%); however, average total N losses were much higher (ca. 50%) compared to the other three scenarios (FRS-2; Fig. 4).

Assuming that AS and LFDIG could potentially qualify as RENURE products (with DIG mainly serving as their source), they might ideally replace the portion of N not supplied by manure (>170 kg N ha^{-1} y^{-1}), i.e., mineral fertilizer N. However, while this was not tested in our study, some RENURE implications can still be inferred from our results. Moreover, regardless of RENURE implementation, if these BBFs, being more mineral-like, are permitted for extensive application, they could serve as the sole N source (thus full replacement in our study). However, if these products fail to meet RENURE criteria or if such criteria aren't adopted by the EU, BBFs produced by adding manure as feedstock might be subject to N application limits set under the EU Nitrates Directive (e.g., 170 kg N ha^{-1}). Nonetheless, whether BBFs are used to replace only mineral N under anticipated RENURE implementation, manure N under EU Nitrate Directive guidelines, or the entire plant N demand with no limitation, their performance may largely hinge on having a higher FRV. Otherwise, there is a risk of increased N losses.

The underlying prerequisite for a product to be considered a RENURE product is that it should exhibit a more “mineral-like” nature, potentially having FRV close to 100%. The implications of fully replacing the baseline N with BBFs are significant if these products have a lower (<100%) FRV. The full replacement of baseline fertilization with BBFs poses challenges due to the perceived lower and more variable FRV of these products (e.g., LFDIG) compared to synthetic N fertilizers (Schröder, 2014; Sigurnjak et al., 2017). Consequently, higher application rates of BBFs are required to compensate for the lower FRV, which may have implications for agronomic and environmental objectives (Huygens et al., 2020). The comparison of results between FRS-2 and FRS-3 confirmed that the increased N losses and lower NUE under FRS-2 were primarily due to the higher total N application.

The results suggest that the challenges associated with full replacement are likely to persist unless: i) the FRV of BBFs is considered as 100%, and ii) the N release from these products is immediately taken up by plants. Another consequence of applying higher total N input due to lower FRV could be the building of SON stocks under DIG, ultimately increasing long-term soil fertility (as evidenced in this study: FRS-2, Fig. 5). These findings suggest that, in general, products like AS and LFDIG are likely to maintain SON stocks less than DIG or slurry for two main reasons: (i) higher FRVs, which lead to relatively lower total N application amounts, and (ii) lower organic N content. In contrast, products like DIG and slurry may contribute to long-term soil fertility and enhanced N supply capacity due to their higher organic N content. Their higher application rates, resulting from lower FRVs, may also promote a gradual buildup of SON stocks. This highlights the environmental trade-offs between mineral-like products, which synchronize nutrient supply with plant uptake to reduce N losses but do not benefit long-term soil fertility, and DIG-like products, which may lead to a slightly more asynchronized nutrient supply, but enhance soil fertility more over time. These considerations are crucial for understanding the long-term benefits and implications of using these products. However, further research is needed to establish the quantitative relationships and long-term benefits of continuous digestate application on soil fertility.

Additionally, local regulations for other nutrients, such as phosphorus (P), which may also limit the N application rate from BBFs (e.g., 80 kg P_2O_5 ha^{-1} in the Flemish region) (Sigurnjak et al., 2017), were not considered in this study and warrant further investigation.

The study's findings hold implications for stakeholders. Farmers aim to apply these products with a realistic FRV for favorable yields, but if the FRV of BBFs is not genuinely higher, N losses may increase. Policy makers seek to mitigate environmental emissions by permitting only products with a higher FRV or enforcing farmers to assume a higher FRV to minimize overall N input increase. If the FRV of BBFs is lower, required assumptions of a 100% FRV may decrease adoption due to anticipated lower yields, especially in the short term. Overcoming this challenge requires countries to establish systems promoting adoption through effective advisory services. The use of BBFs, particularly RENURE products such as AS and LFDIG, if approved, would also have significant implications for larger-scale policies. The EU Nitrates Directive currently restricts manure and manure-derived products (even those with 100% mineral N) to application rates below 170 kg total N ha^{-1}. Approval of RENURE products could relax these restrictions, enhancing nutrient circularity and recycling on farms and across regions, while potentially reducing reliance on synthetic fertilizers. Furthermore, converting manures and slurries into relatively less bulky products could help address challenges related to geographically uneven livestock production and need for manure transport, improving usability and supporting nutrient recycling policy goals.

5. Conclusions

This study examined full (manure + mineral N) and partial (only manure N) replacement of current fertilization (baseline) with biobased fertilizers (BBFs) like ammonium sulphate (AS), digestate (DIG), and liquid fraction of digestate (LFDIG) in European cropping systems. The main conclusions drawn from this study are:

- AS, DIG, and LFDIG can replace the manure-N portion of baseline fertilization (partial substitution) without negatively affecting crop N yield or NUE. However, SON stocks may be slightly reduced under AS

and LFDIG. Gaseous N losses are expected to decrease slightly, while leaching losses may increase correspondingly, keeping total N losses relatively unchanged. Annual cereal-based systems showed a tendency for higher N leaching under BBFs compared to semi-perennial grass-based systems.
- Overall, the BBFs were ranked as DIG > LFDIG > AS for their potential to build SON stocks and reduce N losses.
- Assuming BBFs as RENURE products, full replacement of baseline fertilization may increase N losses if they are applied assuming a FRV below 100%. If applied above 170 kg N ha^{-1}, they must have or be assumed to have an FRV of 100% to prevent increased N losses.
- Farmers should apply BBFs based on expected FRV, physico-chemical properties, and crop needs—AS and LFDIG may suit annuals better due to more synchronized N supply, while DIG can benefit perennials with its slower release.
- Policymakers should consider establishing support systems through advisory services to address farmer adoption barriers, especially for lower FRV products.
- Future research should explore a wider range of BBFs with diverse properties, FRVs, and nutrient profiles (N, P, K) to better understand their full replacement potential. Further work is needed to optimize application rates and timing of BBFs for both productivity and environmental sustainability.

CRediT authorship contribution statement

Muhammad Adil Rashid: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Formal analysis. **Yun-Feng Duan:** Writing – review & editing, Software, Methodology, Formal analysis. **Jan Peter Lesschen:** Writing – review & editing, Resources, Methodology. **Piet Groenendijk:** Writing – review & editing, Resources. **Sander Bruun:** Writing – review & editing, Methodology. **Lars Stoumann Jensen:** Writing – review & editing, Supervision, Resources, Project administration, Methodology, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.farsys.2025.100141.

References

Alburquerque, J.A., de la Fuente, C., Bernal, M.P., 2012. Chemical properties of anaerobic digestates affecting C and N dynamics in amended soils. Agric. Ecosyst. Environ. 160, 15–22. https://doi.org/10.1016/j.agee.2011.03.007.

Aso, S.N., Achinewhu, S.C., Iwe, M.O., 2022. Global fertilizer contributions from specific biogas coproduct. In: Abomohra, A.E., Salama, E. (Eds.), Biogas: Basics, Integrated Approaches, Case Studies. IntechOpen, p. 71. https://doi.org/10.5772/intechopen.101543.

Bell, M.W., Tang, Y.S., Dragosits, U., Flechard, C.R., Ward, P., Braban, C.F., 2016. Ammonia emissions from an anaerobic digestion plant estimated using atmospheric measurements and dispersion modelling. J. Waste Manag. 56, 113–124. https://doi.org/10.1016/j.wasman.2016.06.002.

Børgesen, C.D., Sørensen, P., Blicher-Mathiesen, G., Kristensen, M.K., Pullens, J.W.M., Zhao, J., Olesen, J.E., 2019. NLES5-an Empirical Model for Predicting Nitrate Leaching from the Root Zone of Agricultural Land in Denmark. DCA Report No. 163. Aarhus University, DCA - Danish Centre for Food and Agriculture. https://dcapub.au.dk/djfpublikation/index.asp?action=show&id=1313.

Bruun, S., Christensen, B.T., Hansen, E.M., Magid, J., Jensen, L.S., 2003. Calibration and validation of the soil organic matter dynamics of the Daisy model with data from the Askov long-term experiments. Soil Biol. Biochem. 35 (1), 67–76. https://doi.org/10.1016/S0038-0717(02)00237-7.

Bruun, S., Jensen, L.S., 2002. Initialisation of the soil organic matter pools of the Daisy model. Ecol. Model. 153 (3), 291–295. https://doi.org/10.1016/S0304-3800(02)00017-0.

Buckwell, A., Nadeu, E., 2018. What is the safe operating space for EU livestock? RISE Foundation, Brussels. Retrieved from. https://risefoundation.eu/publications/.

Chojnacka, K., Moustakas, K., Witek-Krowiak, A., 2020. Bio-based fertilizers: a practical approach towards circular economy. Bioresour. Technol. 295, 122223. https://doi.org/10.1016/j.biortech.2019.122223.

De Boer, H., 2008. Co-digestion of animal slurry can increase short-term nitrogen recovery by crops. J. Environ. Qual. 37 (5), 1968–1973. https://doi.org/10.2134/jeq2007.0594.

De Boer, H., 2017. Nitrate leaching from liquid cattle manure compared to synthetic fertilizer applied to grassland or silage maize in The Netherlands. Wageningen Livest. Res. Rapp., No. 1055. Wageningen Livest. Res. https://doi.org/10.18174/425920.

Duan, Y.F., Bruun, S., Jensen, L.S., van Gerven, L., Hendriks, C., Stokkermans, L., Groenendijk, P., Lesschen, J.P., Prado, J., Fangueiro, D., 2021. Mapping and characterization of CNP flows and their stoichiometry in main farming systems in Europe. https://edepot.wur.nl/547940.

Ehlert, P., van Middelkoop, J., van Geel, W., de Haan, J., Regelink, I., 2018. Veeljarige fosfaatveldproeven op gras- en bouwland: syntheserapport. Wageningen Environ. Res. Rapp., No. 2906, 113. https://doi.org/10.18174/460816. ISSN 1566-7197. (In Dutch).

Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z., Winiwarter, W., 2008. How a century of ammonia synthesis changed the world. Nat. Geosci. 1 (10), 636–639. https://doi.org/10.1038/ngeo325.

European Comission, 2013. Report from the commission to the council and the European parliament on the implementation of council directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on member state reports for the period 2008–2011. Retrieved from. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:52013DC0683.

European Commission, 1991. Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EC). https://eur-lex.europa.eu/eli/dir/1991/676/oj.

Groenendijk, P., Schipper, P., Hendriks, R., van den Akker, J., Heinen, M., 2017. Effecten van verbetering bodemkwaliteit op waterhuishouding en waterkwaliteit: deelstudies Goede Grond voor een duurzaam watersysteem. Wageningen Environ. Res. Rapp., No. 2811, 127. https://doi.org/10.18174/418283. ISSN 1566-7197. (In Dutch).

Gunnarsson, A., Bengtsson, F., Caspersen, S., 2010. Use efficiency of nitrogen from biodigested plant material by ryegrass. J. Plant Nutr. Soil Sci. 173 (1), 113–119. https://doi.org/10.1002/jpln.200800250.

Gyldengren, J.G., Abrahamsen, P., Olesen, J.E., Styczen, M., Hansen, S., Gislum, R., 2020. Effects of winter wheat N status on assimilate and N partitioning in the mechanistic agroecosystem model DAISY. J. Agro Crop Sci. 206, 784–805. https://doi.org/10.1111/jac.12412.

Hafner, S.D., Pacholski, A., Bittman, S., Carozzi, M., Chantigny, M., Génermont, S., Häni, C., Hansen, M.N., Huijsmans, J., Kupper, T., Misselbrook, T., Neftel, A., Nyord, T., Sommer, S.G., 2019. A flexible semi-empirical model for estimating ammonia volatilization from field-applied slurry. Atmos. Environ. 199, 474–484. https://doi.org/10.1016/j.atmosenv.2018.11.034.

Hansen, S., Abrahamsen, P., Petersen, C., Styczen, M., 2012. Daisy: model use, calibration, and validation. Trans. ASABE (Am. Soc. Agric. Biol. Eng.) 55 (4), 1315–1333. https://doi.org/10.13031/2013.42244.

Heinen, M., Brouwer, F., Teuling, C., Walvoort, D., 2021. BOFEK2020-Bodemfysische schematisatie van Nederland: update bodemfysische eenhedenkaart. Wageningen Environ. Res. Rapp., No. 3056, 83. https://doi.org/10.18174/541544. ISSN 1566-7197. (In Dutch).

Hendriks, C.M.J., Shrivastava, V., Sigurnjak, I., Lesschen, J.P., Meers, E., Noort, R.v., Yang, Z., Rietra, R.P.J.J., 2022. Replacing mineral fertilisers for bio-based fertilisers in potato growing on sandy soil: a case study. Appl. Sci. 12, 341. https://doi.org/10.3390/app12010341.

Huygens, D., Orveillon, G., Lugato, E., Tavazzi, S., Comero, S., Jones, A., Gawlik, B., Saveyn, H., 2020. Technical Proposals for the Safe Use of Processed Manure above the Threshold Established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC), EUR 30363 EN. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/984729,JRC121636. ISBN 978-92-76-21540-0.

Kandel, T.P., Gowda, P.H., Northup, B.K., Rocateli, A.C., 2020. Winter wheat yield and nitrous oxide emissions in response to cowpea-based green manure and nitrogen fertilization. Exp. Agric. 56 (2), 239–254. https://doi.org/10.1017/S0014479719000334.

Kovačić, Đ., Lončarić, Z., Jović, J., Samac, D., Popović, B., Tišma, M., 2022. Digestate management and processing practices: a review. Appl. Sci. 12, 9216. https://doi.org/10.3390/app12189216.

Manevski, K., Børgesen, C.D., Li, X., Andersen, M.N., Abrahamsen, P., Hu, C., Hansen, S., 2016. Integrated modelling of crop production and nitrate leaching with the Daisy model. MethodsX 3, 350–363. https://doi.org/10.1016/j.mex.2016.04.008.

Manevski, K., Børgesen, C.D., Li, X., Andersen, M.N., Zhang, X., Shen, Y., Hu, C., 2019. Modelling agro-environmental variables under data availability limitations and scenario managements in an alluvial region of the North China Plain. Environ. Model. Softw. 111, 94–107. https://doi.org/10.1016/j.envsoft.2018.10.001.

Metzger, M., Shkaruba, A., Jongman, R., Bunce, R., 2012. Descriptions of the European environmental zones and strata. Alterra Report, p. 152. No. 2281, Alterra. ISSN:1566-7197. https://edepot.wur.nl/197197.

Möller, K., Müller, T., 2012. Effects of anaerobic digestion on digestate nutrient availability and crop growth: a review. Eng. Life Sci. 12 (3), 242–257. https://doi.org/10.1002/elsc.201100085.

Möller, K., Stinner, W., Deuker, A., Leithold, G., 2008. Effects of different manuring systems with and without biogas digestion on nitrogen cycle and crop yield in mixed organic dairy farming systems. Nutr. Cycl. Agroecosyst. 82 (3), 209–232. https://doi.org/10.1007/s10705-008-9196-9.

Montemayor, E., Bonmatí, A., Torrellas, M., Camps, F., Ortiz, C., Domingo, F., Riau, V., Antón, A., 2019. Environmental accounting of closed-loop maize production scenarios: manure as fertilizer and inclusion of catch crops. Resour. Conserv. Recycl. 146, 395–404. https://doi.org/10.1016/j.resconrec.2019.03.013.

Ozturk, I., Sharif, B., Baby, S., Jabloun, M., Olesen, J.E., 2018. Long-term simulation of temporal change of soil organic carbon in Denmark: comparison of three model performances under climate change. J. Agric. Sci. 156 (2), 139–150. https://doi.org/10.1017/S0021859617000971.

Pain, B., Misselbrook, T.H., Clarkson, C.R., Rees, Y.J., 1990. Odour and ammonia emissions following the spreading of anaerobically-digested pig slurry on grassland. Biol. Wastes 34 (3), 259–267. https://doi.org/10.1016/0269-7483(90)90027-P.

Rashid, M.A., Bruun, S., Styczen, M.E., Borgen, S.K., Hvid, S.K., Jensen, L.S., 2022a. Adequacy of nitrogen-based indicators for assessment of cropping system performance: a modelling study of Danish scenarios. Sci. Total Environ. 842, 156927. https://doi.org/10.1016/j.scitotenv.2022.156927.

Rashid, M.A., Bruun, S., Styczen, M.E., Ørum, J.E., Borgen, S.K., Thomsen, I.K., Jensen, L.S., 2022b. Scenario analysis using the Daisy model to assess and mitigate nitrate leaching from complex agro-environmental settings in Denmark. Sci. Total Environ. 816, 151518. https://doi.org/10.1016/j.scitotenv.2021.151518.

Reuland, G., Sigurnjak, I., Dekker, H., Michels, E., Meers, E.J.A., 2021. The potential of digestate and the liquid fraction of digestate as chemical fertiliser substitutes under the RENURE criteria. Agronomy 11 (7), 1374. https://doi.org/10.3390/agronomy11071374.

Reynolds, M., Kropff, M., Crossa, J., Koo, J., Kruseman, G., Molero Milan, A., Rutkoski, J., Schulthess, U., Balwinder-Singh, Sonder, K., Tonnang, H., Vadez, V., 2018. Role of modelling in international crop research: overview and some case studies. Agronomy 8 (12), 291. https://doi.org/10.3390/agronomy8120291.

Schröder, J.J., 2014. The position of mineral nitrogen fertilizer in efficient use of nitrogen and land: a review. Nat. Resour. 5 (15), 936. https://doi.org/10.4236/nr.2014.515080.

Schröder, J.J., de Visser, W., Assinck, F.B.T., Velthof, G.L., 2013. Effects of short-term nitrogen supply from livestock manures and cover crops on silage maize production and nitrate leaching. Soil Use Manag. 29 (2), 151–160. https://doi.org/10.1111/sum.12027.

Sigurnjak, I., Brienza, C., Snauwaert, E., De Dobbelaere, A., De Mey, J., Vaneeckhaute, C., Michels, E., Schoumans, O., Adani, F., Meers, E., 2019. Production and performance of bio-based mineral fertilizers from agricultural waste using ammonia (stripping-) scrubbing technology. Waste Manag. 89, 265–274. https://doi.org/10.1016/j.wasman.2019.03.043.

Sigurnjak, I., De Waele, J., Michels, E., Tack, F.M.G., Meers, E., De Neve, S., 2017. Nitrogen release and mineralization potential of derivatives from nutrient recovery processes as substitutes for fossil fuel-based nitrogen fertilizers. Soil Use Manag. 33 (3), 437–446. https://doi.org/10.1111/sum.12366.

Stewart, W.M., Dibb, D.W., Johnston, A.E., Smyth, T.J., 2005. The contribution of commercial fertilizer nutrients to food production. Agron. J. 97 (1), 1–6. https://doi.org/10.2134/agronj2005.0001.

Styczen, M., Hansen, S., Jensen, L.S., Svendsen, H., Abrahamsen, P., Børgesen, C.D., Thirup, C., Østergaard, H.S., 2006. Standardopstillinger til Daisy-modellen. Vejledning og baggrund. Version 1.2 DHI Institut for Vand Og Miljø, p. 62 (in Danish).

Surey, R., Lippold, E., Heilek, S., Sauheitl, L., Henjes, S., Horn, M.A., Mueller, C.W., Merbach, I., Kaiser, K., Böttcher, J., Mikutta, R., 2020. Differences in labile soil organic matter explain potential denitrification and denitrifying communities in a long-term fertilization experiment. Appl. Soil Ecol. 153, 103630. https://doi.org/10.1016/j.apsoil.2020.103630.

Sutton, M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H., Grizzetti, B. (Eds.), 2011. The European Nitrogen Assessment: Sources, Effects and Policy Perspectives. Cambridge University Press, Cambridge. https://doi.org/10.1017/CBO9780511976988.

Svoboda, N., Taube, F., Wienforth, B., Kluß, C., Kage, H., Herrmann, A., 2013. Nitrogen leaching losses after biogas residue application to maize. Soil Till. Res. 130, 69–80. https://doi.org/10.1016/j.still.2013.02.006.

Virtanen, P., Gommers, R., Oliphant, T.E., Haberland, M., Reddy, T., Cournapeau, D., et al., 2020. SciPy 1.0: fundamental algorithms for scientific computing in Python. Nat. Methods 17, 261–272. https://doi.org/10.1038/s41592-019-0686-2.