Effect of preceding integrated and organic farming on ^{15}N recovery and the N balance, including emissions of NH$_3$, N$_2$O, N$_2$ and leaching of NO$_3$

~~Previous integrated or organic farming affects productivity and ecosystem N balance rather than fertilizer ^{15}N allocation to plants and soil, leaching, or gaseous emissions (NH$_3$, N$_2$O, and N$_2$)~~

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Abstract

Ecological intensification strategies in agriculture, including organic fertilization and diversified crop rotations, ~~Legumes in crop rotations are considered an ecological intensification management practice to~~ aim to reduce nitrogen (N) losses to the environment. However, studies on N allocation and loss on adjacent sites with the same pedoclimatic conditions but different management histories, i.e. organic farming (OF) with frequent legume cultivation and occasional organic fertilizer input, compared to integrated farming (IF) with synthetic and organic fertilizers, have remained scarce. Understanding how these systems differ in their N dynamics is essential for improving nutrient management strategies, mitigating environmental impacts, and guiding sustainable agricultural practices. Here, we quantified field N losses (ammonia, nitrous oxide, dinitrogen, and nitrate leaching), total N balances, and ^{15}N labelled cattle slurry allocation to soil and plants of two adjacent sites over a two-years cropping sequence. While IF had resulted in significantly higher pH and soil organic carbon and N content, the emissions of ammonia, nitrous oxide and dinitrogen after cattle slurry application as well as nitrate leaching were not significantly different across the two farming techniques. Ammonia losses were low for all cultivation periods, indicating that drag hose application and manure incorporation successfully mitigates ammonia emissions. High ^{15}N fertilizer recovery in plants and soil, along with a low share of unrecovered ^{15}N agreed well with the low directly measured N losses. On average, ^{15}N recovery was lower for OF (85% versus 93% in IF), likely due to unaccounted N_2 emissions which could only be measured within two weeks after fertilizer application, but the high spatial variability of ^{15}N recovery may have turned this difference insignificant. Significantly higher harvest biomass N for IF demonstrated that management history affected productivity through increased soil organic matter mineralization. Due to the higher productivity, the cumulative N balance across all cultivation period was neutral within the limits of the measurement uncertainty for IF (-8 $\pm$ 15 kg N ha^{-1}), indicating an optimized N management. For OF, the N balance across single cultivation period ranged from -19 to 41 kg N ha^{-1}, thus, the observations of a single cultivation period were inconclusive. The cumulative positive N balance (48 $\pm$ 14 kg N ha^{-1}) across all cultivation periods for OF suggests that more frequent organic fertilizer additions could increase soil N (and carbon) stocks, and finally improve yield. However, the positive N balance, coupled with lower ^{15}N recovery for OF, also points to a higher likelihood of unaccounted N losses, which would, in turn, slow down the accumulation of soil N and C over time.

Keywords: ^{15}N tracing, ^{15}N gas flux, ecosystem nitrogen losses, nitrogen balance, farming systems comparison, green rye**,** silage maize, perennial ryegrass

1. Introduction

The excessive and inadequate use of nitrogen (N) fertilizers in agriculture has led to N pollution worldwide (Good and Beatty, 2011; Cárceles et al., 2022; Pomoni et al., 2023) since gaseous compounds such as ammonia (NH_3), nitrous oxide (N_2O) and nitric oxide (NO) are released from agricultural fields into the atmosphere and dissolved nitrate (NO_3^-) enters water bodies through leaching and surface runoff. Adverse effects of N loss to the environment include eutrophication, biodiversity loss, global warming and air pollution (Liu et al., 2022; Abdo et al., 2022).

To reduce N losses, the 4R nutrient stewardship concept is one of the most prominent approaches providing advice on the right fertilizer source, at the right rate, at the right time, and at the right place (Bryla, 2011;

Fixen, 2020; De et al., 2024; Nigon, 2024). The adoption of this concept has led to improved nutrient use efficiency, increased crop yields, and reduced N loss (Surekha et al., 2016; Snyder, 2017; Costa et al., 2020). As a complementary strategy, legume cultivation is considered an "agro-ecological" intensification practice because it reduces reliance on external inputs, such as synthetic N fertilizers and associated fossil fuel consumption, due to their symbiotic ~~nitrogen~~N_2-fixing ability. Simultaneously, it enhances biodiversity, improves soil fertility, and their provision of nutrient-rich food (Reza and Sabau, 2022). Similarly, residue return and the combined use of organic and mineral fertilizers further enhance soil organic matter, improve soil structure, retain N in the soil, and increase N availability to plants (Gardner and Drinkwater, 2009; Iheshiulo et al., 2024; Khan et al., 2024). While these strategies offer clear benefits, studies on arable land that has been managed long-term according to these practices and which assess balances, allocation, and losses of fertilizer N have remained scarce.

The allocation of N with~~-~~in the soil-plant system can be determined using the ^{15}N tracing method which utilizes fertilizers containing the rare isotope nitrogen-15 (Kramer et al., 2002; Heng et al., 2014; Chalk, 2015). This method has been successfully applied in arable and grassland systems (Kramer et al., 2002; Zhang et al., 2012; Quan et al., 2020; Pearsons et al., 2023; Dannenmann et al., 2024). A comprehensive meta-analysis by Gardner and Drinkwater (2009), which reviewed 217 field-scale ^{15}N tracing studies, along with a recent global meta-analysis by Xu et al. (2024) of 79 studies on arable land, showed that the ^{15}N tracing approach gives useful information on the ability of management practices to improve N retention and N use efficiency. However, this method exclusively addresses the allocation of N in soil and plants, while neglecting to account for N losses in the form of ammonia (NH_3), nitrate (NO_3^-), nitrous oxide (N_2O), and dinitrogen (N_2). Nevertheless, such information is beneficial for fostering effective N management and providing an accurate assessment of N inputs and their environmental impacts (Zhou et al., 2016). Among the different N loss pathways, the emission of N_2 is the least known since the huge background of atmospheric N_2 complicates the determination of the – in comparison – small soil N_2 emissions. Available methods for in-situ measurement of N_2 fluxes, include natural abundance isotope approaches (Lewicka-Szczebak et al., 2017, 2020) and ^{15}N labelling (Micucci et al., 2023). The ~~only method for in-situ measurement of N_2 is the~~ [15]N gas flux method ([15]NGF) is well-established for quantifying N_2 losses, and ~~so that~~ considering~~ation of~~ this loss pathway together with the more easily accessible pathways enhances our understanding of N allocation in agroecosystems (Kulkarni et al., 2017; Friedl et al., 2020; Dannenmann et al., 2024).

Previous ^{15}N tracing studies have focused on mineral fertilizers, so there are only a few studies using ^{15}N-labelled cattle slurry on arable land (Paul et al., 1995; Buchen-Tschiskale et al., 2023). In addition, there is a lack of multi-year ^{15}N tracing studies in current literature directly comparing adjacent farming sites with similar pedo-climatic conditions but different long-term management histories. Specifically, this study ~~includes comparisons between~~ assesses how preceding management effects N balance and fertilizer N allocation by comparing sites with different management history. On the organic farming site (OF)~~, which incorporates frequent~~, legumes had been cultivated frequently, and external N was only ~~cultivation and~~ occasionally added in the form of cattle slurry, ~~organic fertilizer inputs in a crop rotation,~~ ~~and integrated farming (IF), which received~~ whereas a combination of synthetic and organic fertilizers had been used on the integrated farming site (IF).~~synthetic fertilizers in combination with organic fertilizers in a crop rotation.~~

In this context, the aims of this study were (i) to quantify in-situ N losses (NH_3, N_2O, N_2, and NO_3^- leaching); (ii) to determine ^{15}N fertilizer recovery in soil and plant; (iii) to calculate field N balances; (iv) compare

the results of (i) to (iii) obtained from two adjacent sites (OF, IF) and (v) relate the differences to the effect of management history on soil properties.

2. Materials and Methods

2.1 Study site ~~and, historical~~ management **history** and initial soil sampling

The study was conducted in ~~65618~~ Selters, Germany (50°21'28.8"N, 8°15'47.4"E, elevation 310 m a.s.l.), where the average annual temperature and precipitation amount is 9.3 °C and 655 mm, respectively. There are two adjacent sites, each approximately 1.0 ha in size, which were selected due to their differing long-term management histories, identical pedo-climatic conditions, and access to mains power. The organic farming (OF) site was managed organically for the past decade, with an emphasis on reducing external N inputs so that the majority of the N input is generated via legumes belowground biomass through ~~biological~~ symbiotic N_2 fixation in the crop rotation. However, nitrogen inputs from symbiotic N_2 fixation were not measured prior to our study. Therefore, these inputs are not included in the reported N input values. Periodically, N supply in OF was supplemented with organic cattle slurry (Khan et al., 2024, Table A1). The integrated farming (IF) site was managed using a combination of synthetic and organic fertilizers ~~under integrated farming practices~~, aiming at increasing soil organic carbon (SOC) levels ~~through the combination of synthetic and organic fertilizers~~ (Table A1). For the period 2012-2020, on average, IF received 410 kg C ha^{-1} y^{-1} and 184 kg N ha^{-1} y^{-1}, while OF received 200 kg C ha^{-1} y^{-1} and 23 kg N ha^{-1} y^{-1} (Table A1). The basic soil preparation with regular plowing (20 cm plow depth) and seedbed preparation was the same at both sites.

To assess the differences in soil physicochemical properties resulting from historical management practices, a soil sampling campaign was carried out prior to the field trial at four randomly selected locations at OF and IF sites. At each location, a soil profile was uncovered and sampled at three depths (0–10 cm, 10–30 cm, and 30–60 cm). The physiochemical analyses included pH, SOC, total nitrogen (TN), soil organic carbon to total nitrogen ratio (C:N), bulk density (BD), and texture analysis. Soil pH was determined using a pH metre (Metrohm, Inolab 7310, wtw, Germany) after the soil samples were diluted in distilled water (1:5 soil to water, Dannenmann et al., 2006). To determine the SOC and TN content, soil samples were ground in a mixer mill (Retsch, MM301, Haan, Germany), sealed in a tin capsule, and subjected to isotope ratio mass spectrometry analysis (IRMS: Delta Plus XP; Thermo, Bremen, Germany) according to the protocols of Dannenmann et al. (2016) and Couto-Vázquez et al. (2020). For BD, additional soil samples were collected using a soil core cutter with a volume of 100 cm³ and oven-dried at 105 °C for 24 hours (Khan et al., 2020). For texture analysis, soil samples were randomly collected from ten different locations in each site, pooled, sieved, air-dried, and analysed by a commercial laboratory (AGROLAB, Agrar GmbH, Sarstedt, Germany).

Due to different management histories, pH, SOC and TN were significantly lower on OF while bulk density was significantly higher for OF in 0 to 10 cm depth (Table 1). The soil of both sites has a silty loam texture with almost identical percentages of silt, sand, and clay.

Table 1: Soil properties of initial sampling from the organic farming (OF) and integrated farming (IF) sites. All values are given as mean ± standard error (SE, n=4), except for C:N and texture analysis values (only the mean value). In the case of texture analysis, NA represents the samples that were not measured at the corresponding depth.

Site	Depth	pH	SOC	TN	C:N	BD	Texture [%]		

	[cm]	[%]	[%]	[-]	[g cm^{-3}]	sand	silt	clay
OF	0-10	5.43 ± 0.05 a	1.04 ± 0.12 a	0.13 ± 0.01 a	8.00 a	1.45 ± 0.01 a	10.20	68.30	21.50
IF	0-10	6.84 ± 0.07 b	1.58 ± 0.07 b	0.18 ± 0.01 b	8.77 b	1.36 ± 0.02 b	8.20	67.10	24.70
OF	10-30	5.33 ± 0.06 a	0.84 ± 0.19	0.11 ± 0.02	7.63	1.43 ± 0.03 a	11.60	65.10	23.30
IF	10-30	6.34 ± 0.25 b	1.08 ± 0.08	0.14 ± 0.01	7.71	1.32 ± 0.04 b	9.20	68.30	22.50
OF	30-60	5.90 ± 0.08 a	0.37 ± 0.03	0.06 ± 0.01	6.19	1.47 ± 0.05	NA	NA	NA
IF	30-60	6.70 ± 0.09 b	0.62 ± 0.28	0.05 ± 0.01	11.24	1.41 ± 0.05	NA	NA	NA

The letters (a, b) specify the significant difference (p < 0.05) between OF and IF sites

2.2 Agricultural management during field experiment

The trial period spanned from October 2020 to the end of September 2022, during which fertilizer amount, type, and crop cultivation were identical on both sites to avoid bias in N loss and allocation due to fertilizer amount. However, we acknowledge that this management homogenization may mask legacy effects and introduce potential biases in interpreting differences between sites. Using a seed drilling device, 300 kernels per square metre of green rye were sown at a depth of 3 cm with a row distance of 15 cm in October 2020. On April 8th, 25 m^3 ha^{-1} (88 kg N ha^{-1}, 750 kg C ha^{-1}) of cattle slurry was applied as a top-dressing application for green rye using a drag hose. Green rye was harvested on May 10th, 2021. On May 31st, 2021, 20 m³ of cattle slurry (74.8 kg N ha^{-1}, 600 kg C ha^{-1}) was applied as a pre-sowing fertilization for silage maize. It was incorporated with one pass of a disc harrow followed by one pass of a rotary tiller. Subsequently, silage maize was sown at a plant density of 90,000 plants ha^{-1} with a row spacing of 75 cm. Silage maize was harvested on September 30th, 2021. In October 2021, perennial ryegrass was seeded on both sites at a density of 45 kg ha^{-1} with 12.5 cm row spacing. On March 29th and May 30th, 2022 cattle slurry was applied using a drag hose at a rate of 25 m^3 (86 kg N ha^{-1}) and 20 m^3 (65 kg N ha^{-1}), respectively. Perennial ryegrass was harvested on May 28th, July 5th, and September 22nd, 2022.

2.3 Experimental units and associated measurements

Since several parameters were quantified, an overview of the associated experimental units is given here (Fig.1) while the detailed method is presented in the dedicated sections. Crop yield was determined on macroplots (4 m by 4 m), 4 replicates at each site. Dinitrogen and N$_2$O emissions were measured on mesocosms (0.16 m diameter) with 6 mesocosms at each site. Recovery of ^{15}N in soil and plants was measured on microplots (0.30 m by 0.75 m) with 4 replicates at each site. Ammonia emissions were determined using semi-open chambers (0.16 m diameter) with 4 replicates at each site and NO$_3$$^-$ leaching was determined using self-integrated accumulative (SIA) collectors which were buried on both sites at four locations in 1 m depth, with three replicates per location for each cultivation period.

2.4 Crop biomass yield

At each harvest, aboveground biomass was harvested in each of the macroplots by harvesting the whole area using a plot harvester (Haldrup, 1.5 m harvesting width). Biomass dry matter was determined in subsamples from each macroplot by oven-drying at 60 ºC for several days until a stable mass was reached. To determine total biomass N content, samples were milled and analyses using an elemental analyser (DIN/ISO 13878:1998, Elementar Analyser system GmbH, Langenselbold, Germany).

2.5 Determination of dinitrogen emissions from soil-plant mesocosms

To determine N_2 emission, the ^{15}N gas flux method (^{15}NGF) was applied where N_2 emission is calculated from the isotopic compositions of N_2 in atmospheric background and in the headspace of a soil chamber that previously had received ^{15}N labelled fertilizer (Hauck, 1958; Mulvaney et al., 1986; Spott et al., 2006). The mesocosms were managed in the same manner as the field plots. To this end, six mesocosms from OF and IF were collected in steel cylinders (0.26 m height and 0.16 m diameter) a month before fertilization and buried in a trench at the site. For the target application rate of 88 kg N ha^{-1}, cattle slurry was properly mixed with ^{15}N–NH_4^+ and ^{15}N-urea to obtain an enrichment of 85at% (Table A2). The ^{15}N-labelled cattle slurry was applied to each mesocosm during green rye cultivation, by simulating a drag hose application. In the silage maize cultivation period, the target application rate was 74.8 kg N ha^{-1}, and labelled slurry was applied by mimicking drag hose application, and incorporated into the soil using manual tillage, following the legal regulation in Germany. Subsequently, corn seeds were planted. In the same way, labelled cattle slurry, with 85at% was prepared for the perennial ryegrass first (86 kg N ha^{-1}) and second (65 kg N ha^{-1}) application and applied by simulating a drag hose application.

For the N_2 gas measurements, 0.16 m high chambers were tightly affixed to the mesocosms using a steel tension clasp, and gaseous samples were collected using a syringe directly after fixing the chamber and after 2 hours chamber closure time. The syringe was flushed three times with chamber headspace air and 20 ml was transferred to a pre-evacuated 12 ml exetainer (Model 837W, Labco Limited, United Kingdom), that had been flushed with helium gas three times. Gas samples were collected daily from day D1 to D7, and additionally on days D9, D11, D13, D21, and D28, following fertilization.

Gas samples were analyzed using an isotope ratio mass spectrometer (Isoprime PrecisION, Elementar UK Ltd., Stockport, UK), coupled to an isoFLOW GHG GasBench (Elementar UK Ltd.). This setup allows for subsampling of gas volumes (30 µL) for ^{15}N-N_2 analysis by measuring m/z 28, 29, and 30, and for analysis of ^{15}N-N_2O after cryogenic pre-concentration of N_2O of the remaining vial content by measurement of m/z 44, 45, and 46. Gas handling and preparation followed the protocol described in Arah, (1997); Stevens and Laughlin, (2001b); Spott et al. (2006). In-run uncertainty, determined from repeated analysis of reference gases for isotope ratios r29, r30, r45, and r46, amounted to 1×10^{-6}, 5×10^{-7}, 7×10^{-5}, and 4×10^{-5}, respectively. At the enrichment of the fertilizer added to the mesocosms (85% at), this standard deviation values relate to 87 µg N_2-N m^{-2} h^{-1}. The fraction

of fertilizer-derived N_2 was determined by calculating the ratio between the ^{15}N atom excess percentage of the emitted N_2 and the ^{15}N atom excess percentage of the applied N fertilizer, following the procedure given in Yankelzon et al. (2024b).

2.6 Determination of ammonia losses

The measurement of NH_3 emissions into the atmosphere was conducted using the semi-open chamber approach, as described by Jantalia et al. (2012), using polyvinyl chloride (PVC) cylinders containing two polyurethane foams. The foams were pre-soaked in 40-ml 1 M sulfuric acid solution containing 4% (v/v) glycerol until they absorbed all the solution, and were then placed inside the PVC cylinder on fixtures 5 cm and 20 cm above the ground. The top foam was used to protect the lower foam disc of NH_3 deposition, while the bottom foam was used to detect NH_3 emissions from soil fertilizer N application. Subsequently, the PVC cylinders were placed onto frames previously installed in the soil of OF and IF. Foam discs were exchanged at daily frequency during the first week after fertilization and every second day up to a maximum of 14 days of measurement after fertilization. The NH_3 released was captured as NH_4^+ on the acidified foams and subsequently extracted using 150-ml of 2 molar potassium chloride. Extracts of the bottom foam discs were analysed for ammonium (NH_4^+) using an indophenol colorimetry approach based on Kempers and Zweers (1986), and converted to an emission rate in NH_4^+-N by considering the duration of the exposition of the foam disc and the area of the chamber. The quantified NH_3 emissions were considered in the ^{15}N balance by calculating the share of volatilized N of the mineral N amount applied in slurry for each cultivation period.

2.7 Determination of nitrous oxide fluxes

Nitrous oxide emissions were determined simultaneously with the N_2 emissions by sampling ^{15}N enriched soil plant mesocosms headspace air into 10 ml vials by manual syringe sampling as described above for dinitrogen fluxes and subsequently determining the mixing ratio using gas chromatographic analysis. The flux was calculated from the change in N_2O mixing ratio over time during chamber closure using the following equation (Eq. 1).

$$F = \frac{\Delta C_v \, p \, M \, h}{\Delta t \, A \, R \, T} \tag{1}$$

Where F is N_2O flux ($\mu g \, m^{-2} \, h^{-1}$), p is the atmospheric pressure ($N \, m^{-2}$), M is the molar mass of N_2O-N ($\mu g \, mol^{-1}$), h is chamber height (m), ΔC_v is the change in volume mixing ratio (ppm), R is the ideal gas constant ($J \, mol^{-1} \, K^{-1}$), T is the temperature (K), Δt is the duration of the chamber closure (hours), and A (m^2) is the surface area of the mesocosms. The obtained value was used to calculate the product ratio of denitrification by dividing F by the sum of F and the N_2 flux.

2.8 Determination of nitrate leaching

Losses in the form of NO_3^- leaching were determined using the SIA method (self-integrating accumulators, TerrAquat-GmbH; Grahmann et al., 2018). SIAs are absorber materials that were installed at a depth of 1.0 m in an undisturbed soil section of both sites (OF, IF) and can, thus, collect nitrate during each cultivation period. In the laboratory, the bound NO_3^- was extracted from the SIAs using a concentrated 2 molar potassium chloride solution and the nitrate concentration in the extracts was measured using an indophenol colorimetry

approach (Kempers and Zweers, 1986; Grahmann et al., 2018). Nitrate leaching was considered in the ^{15}N balance by calculating the share of leached N of the N amount applied in the labelled slurry for each cultivation period, implicitly assuming that leaching is due to fertilizer application and occurs within one cultivation period.

2.9 ^{15}N fertilizer tracing on microplots

2.9.1 Preparation and application of ^{15}N-labelled cattle slurry fertilizer

To determine ^{15}N recovery in soil and plants, microplots were established by inserting four steel frames each (length × width × height: 0.75 m × 0.30 m × 0.15 m; Table A2) into the soil at OF and IF. These microplots contained two rows of green rye in 2021, following by one row of silage maize in 2021, and two rows of perennial ryegrass in 2022. For fertilization, cattle slurry was mixed with ^{15}N-NH$_4^+$ and ^{15}N-urea to achieve an atomic enrichment of 10 at%, and the ^{15}N-labelled cattle slurry was applied manually, mimicking a drag hose fertilizer application. The amounts of cattle slurry, ^{15}N-salts and milli-q water for production of spiked slurry were calculated based on slurry analysis and in a way that the total N content and amount of spiked slurry equalled that of the original slurry and that the total N amount was 88 kg N ha^{-1} for the green rye cultivation period. Similarly, ^{15}N–NH$_4^+$ and ^{15}N-urea were properly mixed with cattle slurry to achieve a total N amount of 74.8 kg N ha^{-1} for the silage maize application. Following the legal regulation in Germany, the ^{15}N-labelled cattle slurry applied before planting silage maize and incorporated into the soil using a hoe. In a similar manner to green rye and silage maize, ^{15}N-labelled cattle slurry was prepared for the perennial ryegrass cultivation period in 2022, yielding 86 kg N ha^{-1} and 65 kg N ha^{-1} for the initial and second application, respectively. The slurry was applied manually, simulating a drag hose application.

2.9.2 Calculation of ^{15}N recovery in plants and soil

At the end of each cultivation period, all aboveground biomass was harvested from the microplots, and belowground biomass was separated from the soil in a vessel. On the date of the first and second perennial ryegrass harvest in 2022, only aboveground biomass was cut. Plant samples were dried at 60 °C for a week to reach a constant dry weight, after which they were crushed in a mixer mill (Retsch, MM301, Haan, Germany), and 2 mg samples were packed in tin capsules for determination of TN, SOC, and the ^{15}N atom fraction using an isotopic ratio mass spectrometer (IRMS: Delta Plus XP; Thermo, Bremen, Germany). The recovery of ^{15}N in plants was calculated from the ^{15}N excess compared to natural abundance and the amount of ^{15}N applied with the cattle slurry, with a detailed description being given in Dannenmann et al. (2016).

Similarly, the soil samples from each microplot (0.225 m^2) were collected at different depths (0–10 cm, 10–30 cm, 30–60 cm, and 60–90 cm). The soil samples were thoroughly homogenised, sieved (2 mm), and oven-dried at 60 °C until they reached a constant dry weight. After that, the soil samples were ground in a mixer mill (Retsch, MM301, Haan, Germany) and packed in tin capsules for measuring SOC, TN, and atom ^{15}N fraction by elemental analysis coupled to mass spectrometry (Delta Plus XP; Thermo, Bremen, Germany). In analogy to the plant recovery, total soil recovery was calculated from the ^{15}N excess compared to natural abundance and the amount of ^{15}N applied with the cattle slurry (Dannenmann et al., 2016).

2.10 Nitrogen balance

The N balance was calculated to quantify the difference of N applied to the sites and lost from the sites (Eq. 2).

$$N\ balance = N_{input} + N_{deposition} - N_{harvest} - N_{loss} \tag{2}$$

Here, N_{input} refers to the N added through fertilizer, $N_{deposition}$ is N from atmospheric deposition (taken from German Environmental Protection Agency), $N_{harvest}$ accounts for N removed through crop harvest, and N_{loss} includes N lost via leaching as well gaseous losses in form of NH_3, N_2O and N_2. To calculate N deposition for specific crop periods, the annual deposition rate was divided by 365 days and scaled according to the duration (in days) of each crop cultivation period.

2.11 Data processing and statistical analysis

The total ^{15}N recovery, losses, and N balance were calculated in the Microsoft Excel software program (Microsoft Office 2019, Microsoft, Seattle, WA, USA). The statistical package of the Social Sciences (SPSS version 27.0, IBM Crop., Armonk, NY, USA) was used for statistical analysis. Normality was tested using the Shapiro-Wilk test and based on the result, either the sample t-test or the Wilcoxon test was carried out to test significant differences at a 95% confidence interval between the OF and IF sites. The soil recovery of ^{15}N was calculated as the sum of the recoveries in different soil depths (0–10, 10–30, 30–60, and 60–90 cm). To calculate cumulative N_2 and N_2O emissions, a linear interpolation was made for the days where no measurement was conducted. Furthermore, OriginPro 2020b (OriginLab Corporation, Northampton, Massachusetts) was used for illustrations.

3.   Results

3.1 Crop yield

In the green rye cultivation period, significantly higher harvested aboveground biomass (AGB) and above ground biomass N (AGB-N) were recorded for IF (Table 2). In contrast, OF showed significantly higher AGB and AGB-N in the silage maize cultivation period. During the perennial ryegrass cultivation period, the AGB and AGB-N were significantly higher for IF, and average IF values across the three crops were significantly higher for AGB-N (20%) but not significantly higher for AGB (13%, Table. 2).

Table 2: Average aboveground biomass (AGB) and aboveground biomass-N (AGB-N) with standard error (SE), expressed as absolute dry matter, were reported for the different cultivation periods in organic farming (OF) and integrated farming (IF).

Period	Treatment	AGB ± SE [Mg ha⁻¹]	AGB–N ± SE [kg ha⁻¹]
Green rye 2021	OF	3.03 ± 0.17 a	56.84 ± 1.78 a
	IF	5.66 ± 0.21b	98.47 ± 5.58 b
Silage maize 2021	OF	9.57 ± 0.53 a	94.94 ± 3.96 a
	IF	7.14 ± 0.31 b	75.24 ± 4.21 b
Perennial ryegrass 2022	OF	8.50 ± 0.13 a	98.66 ± 3.01 a
	IF	11.13 ± 0.39 b	132.72 ± 9.86 b
Average (2021-2022)	OF	7.03 ± 0.28	83.48 ± 2.92 a
	IF	7.98 ± 0.30	102.14 ± 6.55 b

296   Lower-case letters (a, b) depict statistically significant differences (p < 0.05)

3.2 Dinitrogen (N₂) and nitrous oxide (N₂O) emissions

298   Following ^{15}N fertilizer applications, increased emissions of total N_2 and fertilizer-derived N_2 were
299   observed within two weeks, with fertilizer-derived emissions accounting for 70%, 63% and 76% for OF and
300   80%, 55%, 70% for IF of the total emissions during the green rye, silage maize and perennial ryegrass cultivation
301   period, respectively. There was no clear difference in total N_2 flux levels between OF and IF, and flux rates
302   ranged from close to 0 to 1065 µg N m^{-2} h^{-1} (Fig. ~~1a~~2a-q, Fig. B1). In contrast, fluxes of total N_2O and fertilizer-
303   derived N_2O for OF were higher after slurry application in the green rye and perennial ryegrass cultivation period,
304   compared to silage maize cultivation period. The fertilizer-derived N_2O emissions accounted for 33, 38, and 33%
305   for OF and 29, 37, and 38% for IF in the green rye, silage maize and perennial ryegrass cultivation period,
306   respectively. However, no clear pattern of elevated total or fertilizer-derived N_2O flux levels was observed for
307   silage maize. Flux rates for total N_2O were low with emissions peaks of approximately 40 µg N m^{-2} h^{-1} in the
308   green rye and silage maize periods and 94 µg N m^{-2} h^{-1} after the first slurry application to perennial ryegrass
309   (Fig. ~~1e~~2c-s), respectively.

310   The ratio of total $N_2O : (N_2+N_2O)$ showed a similar progression over time for both sites, ranging from 0.01 to
311   ~~1.0~~0.45. Since the N_2O and N_2 flux levels were close to the detection limit from day 11 after fertilization, only
312   values before this period were considered in green rye and silage and silage maize cultivation period (Fig. ~~1e~~2e,
313   k). On average, across crops and sites, the $N_2O : (N_2+N_2O)$ was 0.~~16~~14, and the median amounted to 0.03.

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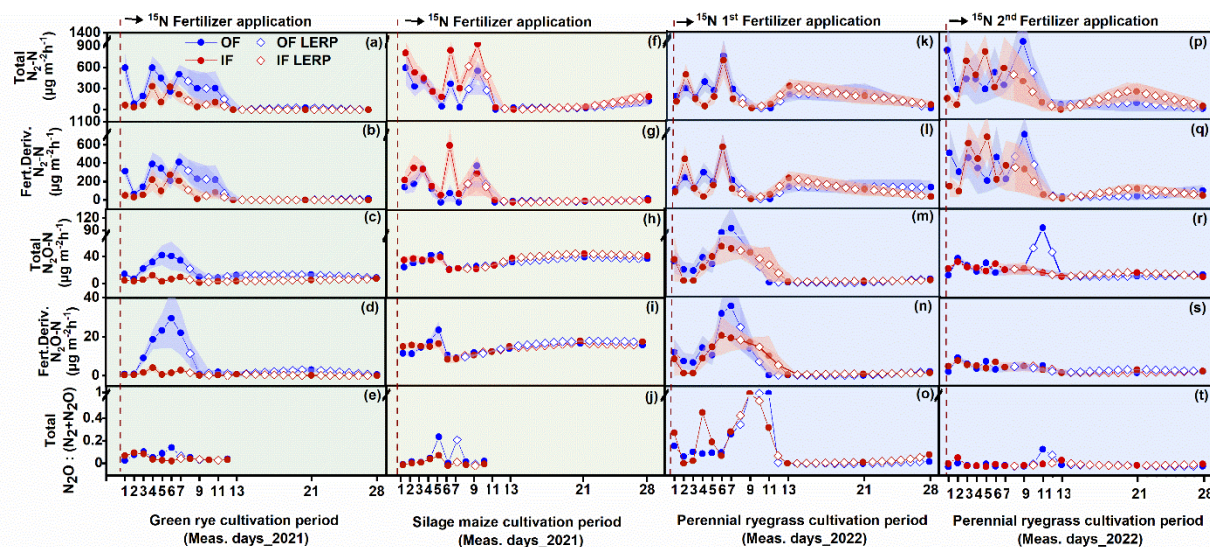
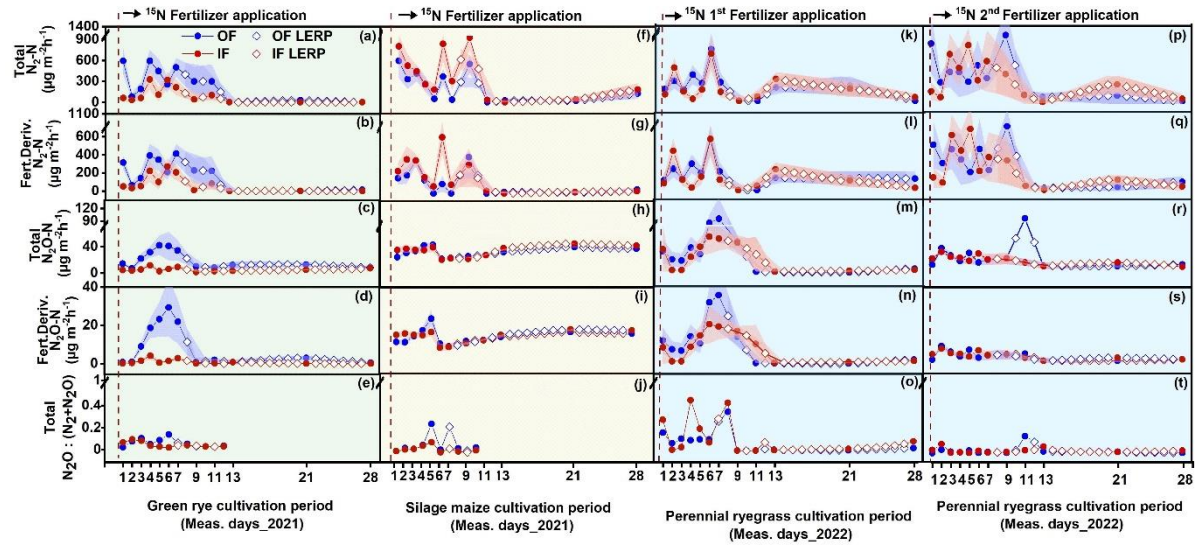


Figure ~~1~~2: Results of the ^{15}N gas flux method using mesocosms. From top to bottom, the panels show total N_2 flux, fertilizer-derived (fert. deriv.) N_2 flux, total N_2O flux, fertilizer-derived (fert. deriv.) N_2O flux, total N_2O : ($N_2 + N_2O$) ratio, following fertilizer application. Measurement periods for fertilization of green rye, silage maize, perennial ryegrass 1, and perennial ryegrass 2 are shown in subplots a–e, f–j, k–o, and p–t, which correspond to green rye, silage maize, the first perennial ryegrass application, and the second perennial ryegrass application, respectively. Filled circles and surrounding bands show mean values and standard errors for OF (blue) and IF (red). Linearly interpolated (LERP) values are marked by white-filled squares.

Cumulative total and fertilizer-derived N_2 emissions over the green rye measurement period were significantly higher for OF (1.04 and 0.72 kg N ha^{-1}) compared to IF (0.37 and 0.29 kg N ha^{-1}, Fig. ~~2a~~3a, B1). Similarly, cumulative total and fertilizer-derived N_2O emissions were significantly higher for OF (0.10 and 0.04 kg N ha^{-1}) than for IF (0.04 and 0.01 kg N ha^{-1}, Fig. ~~2a~~3a). During the silage maize cultivation period, cumulative total and fertilizer-derived N_2 emissions were with 1.60 and 0.88 kg N ha^{-1} significantly higher for IF than for OF with 0.98 and 0.62 kg N ha^{-1} (Fig. ~~2b~~3b), respectively. For the same period, total N_2O emissions were significantly different between the two sites (OF: 0.26 kg N ha^{-1}, IF: 0.29 kg N ha^{-1}; Fig. ~~2b~~3b). However, fertilizer-derived N_2O emissions were not significantly different and amounted to 0.10 kg N ha^{-1} (Fig. ~~2b~~3b).

After the first fertilizer application in the perennial ryegrass period, cumulative total and fertilizer-derived N_2 emissions were not significantly different, with emissions values of 1.18 and 0.98 kg N ha^{-1} as well as 1.17 and 0.87 kg N ha^{-1} for OF and IF, respectively (Fig. ~~2c~~3c). Similarly, cumulative total (IF: OF: 0.11 kg N ha^{-1}, Fig. ~~2c~~3c) and fertilizer derived (IF: OF: 0.04 kg N ha^{-1}) N_2O were also not significantly different and comparable for both sites. After the second fertilization of perennial ryegrass, cumulative total (OF: 2.16, IF: 2.20 kg N ha^{-1}, Fig. ~~2d~~3d) and fertilizer-derived (OF:1.55, IF: 1.46 kg N ha^{-1}) N_2 emissions were not significantly different, and N_2O emissions were below 0.05 kg N ha^{-1} for both sites, with no significant differences observed (Fig. ~~2d~~3d). Average emission of total and fertilizer-derived N_2 and N_2O for both sites during the green rye, silage maize and perennial ryegrass were comparable and not significantly different (Fig. ~~2e~~3e).

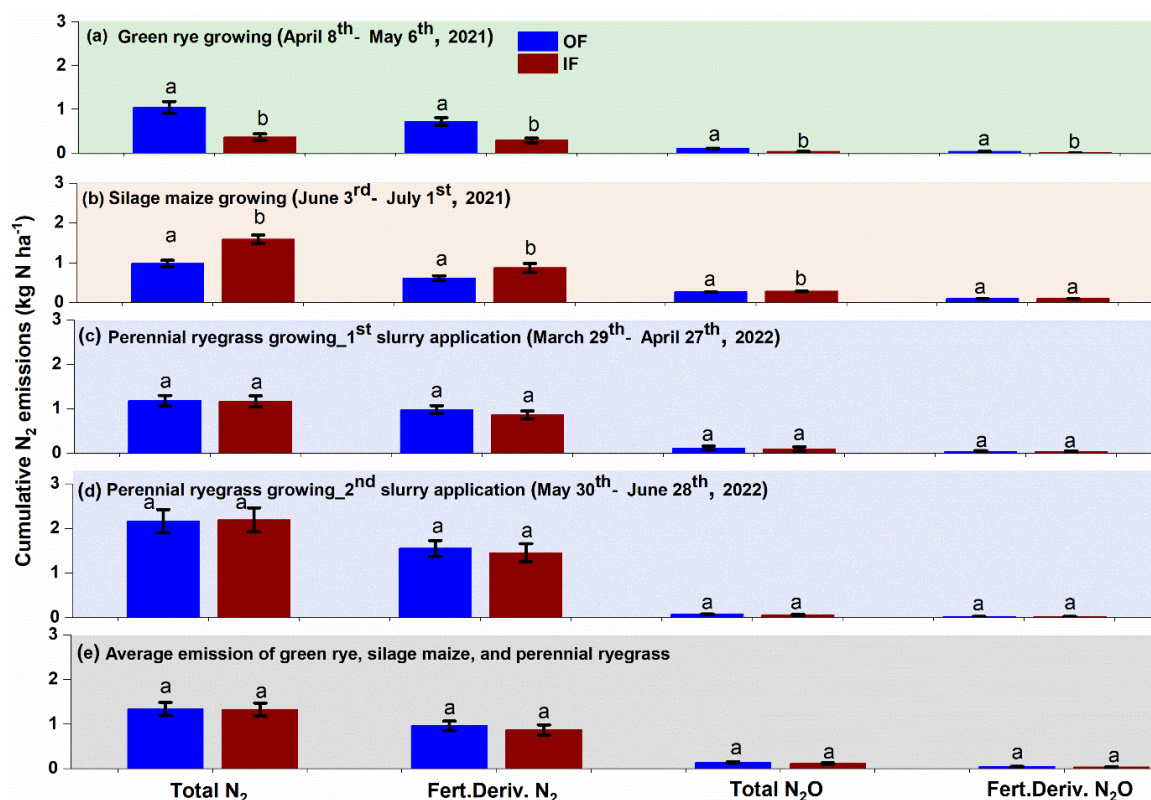



Figure 2~~3~~: Cumulative total and fertilizer-derived N_2 and N_2O emission for a measurement period of one month during (a) green rye cultivation period, (b) silage maize cultivation period, (c) perennial ryegrass 1st ^{15}N-labelled cattle slurry application (d) perennial ryegrass 2nd ^{15}N-labelled cattle slurry application and (e) average emissions across the crops. Bars with whiskers represent mean values and standard error, respectively.

3.3 ^{15}N fertilizer recovery

Across the treatments and cultivation periods, total ^{15}N fertilizer recovery considering plant, soil and directly measured N loss pathways ranged from 84 to 100% (Fig. ~~3a~~4a-c, Table A3). The highest average recovery for all cultivation periods was typically found in soil (OF: 44%, IF: 46%, Fig. ~~3d~~4d), followed with slightly lower average recovery in plant above and belowground biomass (OF: 33%, IF: 39%, Fig. ~~3d~~4d). Total N losses were dominated by NH_3 (0.2 – 8%), followed by nitrate leaching (0 – 5%), and denitrification losses (0.3 – 4%). Average ^{15}N allocation in the microplots and to N loss pathways did mostly not greatly differ between IF and OF.

During green rye cultivation, the ^{15}N recovery in the soil was 52% for OF and ~~for IF (~~35%~~)~~ for IF, while during silage maize, it was 59% for IF and 44% for OF (Fig. ~~3a~~4a, b). For perennial ryegrass cultivation the soil ^{15}N recovery was slightly higher for IF (44%) than for OF (36% Fig. ~~3c~~4c).

Recovery in the plant followed an opposite trend to soil recovery for green rye and maize, i.e., higher plant recovery was associated with lower soil recovery, except for perennial ryegrass, where both soil and plant recovery were higher for IF (Fig. ~~3a~~4a-c). Recovery from roots was nearly identical for both sites (OF, IF, Fig. ~~3a~~4a-d) across all cultivation period and contributed only marginally to total plant recovery.

Ammonia emission accounted for 3% and 2% of total fertilizer application on IF and OF, respectively during the green rye cultivation period, was less than 1% for silage maize cultivation and amounted to 8% for IF and OF during perennial ryegrass cultivation, with the no significant differences between the both sites (Fig. ~~3a~~4a-c). Similarly, recovery in N_2O was below 0.1% for both OF and IF across all cultivation periods (Fig. ~~3a~~4a-d).

Nitrate leaching was in the range of 0 to 5% for the different cultivation periods, but differences between sites were not significant (Fig. ~~3a~~4a-d).

The total ¹⁵N recovery considering soil, plant, and measured N loss components, was comparable in the green rye (OF: 87%, IF: 84%, Fig. ~~3a~~4a), silage maize (OF: 86%, IF: 95%, Fig. ~~3b~~4b), and perennial ryegrass (OF: 84% and IF: 100%; Fig. ~~3c~~4c) cultivation period. On average ¹⁵N recovery for all cultivation periods was 85% for OF and 93% for IF (Fig. ~~3d~~4d). The unrecovered portion of ¹⁵N was 13% and 16% in green rye, 14% and 4% in silage maize, and 17% and 0.1% in perennial ryegrass for OF and IF, respectively, with no significant differences (Table A3). The average unrecovered part of ¹⁵N for all cultivation periods was 15% for OF and 7% for IF (Table A3), both within the standard error range of 7-16% and 8-11%, respectively, and, thus, within the uncertainty of measurements.

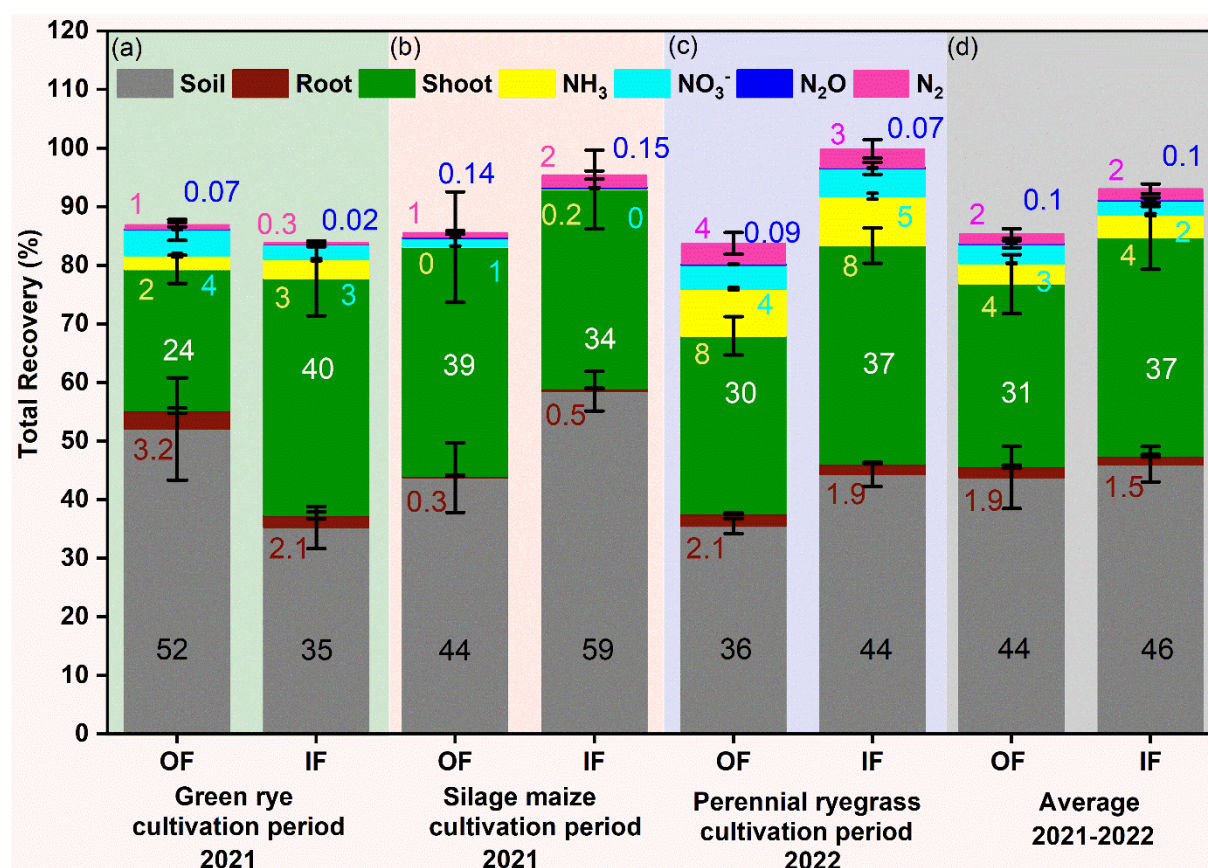



Figure ~~3~~4: Recovery of fertilizer ¹⁵N in soil (0–90 cm depth; grey), aboveground biomass (green), root (vine), ammonia (NH₃, yellow), nitrous oxide (N₂O, blue), dinitrogen (N₂, magenta) and nitrate leaching (NO₃⁻, cyan) for (a) green rye, (b) silage maize, (c) perennial ryegrass cultivation periods and (d) average of all cultivation periods in the organic (OF) and integrated (IF) farming sites. Vertical bars represent mean values, and whiskers represent the standard error (SE) in percentages. All values are presented in rounded decimals to the nearest whole number.

3.4 Nitrogen balance

In green rye, silage maize and perennial ryegrass cultivation, 88, 74.8 and 151 kg N ha⁻¹ cattle slurry were applied. For green rye and perennial ryegrass ~~in the~~ cattle slurry was applied as top dressing in OF and IF (Fig. ~~4~~5), ~~except~~ while for silage maize ~~where it~~ the slurry was incorporated. Nitrogen deposition was 1.60 kg N ha⁻¹

for green rye, 2.93 kg N ha⁻¹ for silage maize, and 5.01 kg N ha⁻¹ for perennial ryegrass (Fig. ~~4a~~5a-c), calculated based on data from the German Environmental Protection Agency. In accordance with the ¹⁵N recovery (section 3.3), IF exhibited significantly higher plant N uptake (98 vs. 57 kg N ha⁻¹ in OF, Fig. ~~4a~~5a) for green rye cultivation, while OF showed slightly higher N losses (7 vs. 5 kg N ha⁻¹ in IF) combining NH_3, N_2O, N_2, NO_3^- leaching. Consequently, the N balance was positive for OF (26 ± 4 kg N ha⁻¹, Fig. ~~4a~~5a), while IF had a small negative N balance (-14 ± 6 kg N ha⁻¹) with the difference being significant.

In the silage maize cultivation period, OF showed with 95 kg N ha⁻¹ significantly higher N in harvest than the 75 kg N ha⁻¹ of IF (Fig. ~~4b~~5b). Since total N losses were minor compared to yield for both sites (2 kg N ha⁻¹, Fig. ~~4b~~5b), this resulted in a small negative and positive N balance for OF and IF, respectively (-19 ± 4 and 1 ± 5 kg N ha⁻¹), with the difference being significant. Harvest N export during the cultivation of perennial ryegrass was significantly higher for IF than for OF (133 ± 3 and 99 ± 4 kg N ha⁻¹, Fig. ~~4c~~5c), with a comparable N loss for both sites. The N balance was significantly different for both sites, amounting to a surplus of 41 ± 6 for OF and 6 ± 4 kg N ha⁻¹ for IF (Fig. ~~4c~~5c).

The cumulative harvested N across all cultivation periods of 2021-2022 showed that a significantly higher amount of N was exported from IF (306 ± 12 kg N ha⁻¹, Fig. ~~4d~~5d) compared to OF (250 ± 10 kg N ha⁻¹). In contrast, the cumulative N losses were almost similar and not significantly different (OF: 25, IF: 24 kg N ha⁻¹). Consequently, the N surplus was significantly higher on OF (48 kg N ha⁻¹; Fig. ~~4d~~ 5d), compared to IF, which had a small negative value (-8 kg N ha⁻¹; Fig. ~~4d~~5d).

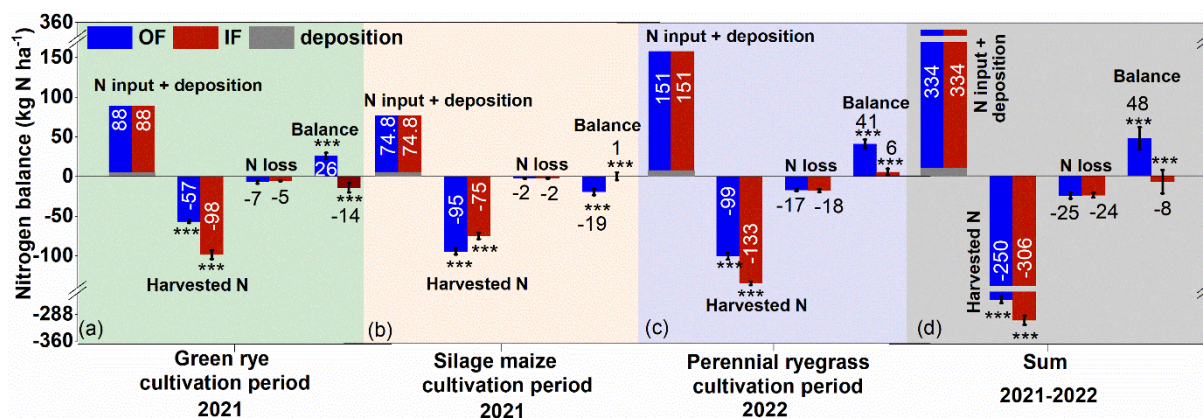



Figure ~~4~~5: Cultivation period N balance for (a) green rye (b) silage maize (c) perennial ryegrass and (d) cumulative N balance for the years 2021 and 2022. N loss combines NH_3, N_2, N_2O and NO_3^- leaching. The bars with whiskers represent the mean ± standard error values. Organic farming (OF), integrated farming (IF), and soil N deposition are shown in dark blue, red, and grey colour bars, respectively.

4. Discussion

4.1 Gaseous N loss (ammonia, nitrous oxide, dinitrogen)

A recent summary of experiments on ammonia emissions after slurry application showed that NH_3 loss amounted to 10 to 47% of ammoniacal N applied (Häni et al., 2018). The NH_3 loss in our study was at the lower end of the observed range, with 2.4 and 3.2%, 0.2 and 0.2%, and 8.0 and 8.4% (Fig. ~~3~~4, Table A3) for OF and IF, respectively, during green rye, silage maize, and perennial ryegrass cultivation periods. While the above range includes all spreading techniques, in this study manure was applied on every occasion using a technique known to significantly reduce NH_3 emissions. The complete suppression of NH_3 emission observed in the silage maize

cultivation period was achieved through the immediate incorporation of the slurry. This agrees with studies observing reduction of NH_3 emission after incorporation and confirms that slurry incorporation is among the most efficient NH_3 mitigation methods if incorporation takes place promptly (Sommer and Hutchings, 2001; Dell et al., 2011; Sherman et al., 2022). Though manure was applied using the same drag hose applicator during green rye and perennial ryegrass cultivation, NH_3 emissions were variable. Thus, other factors were responsible for the variability of emissions during green rye and perennial ryegrass cultivation period. Ammonia emission depends among other factors on ammonium concentration at the manure surface, wind speed, precipitation, solar radiation and temperature (Sommer and Hutchings, 2001). Consequently, the low temperature and light precipitation on the date of manure application to green rye reduced NH_3 emissions compared to the perennial ryegrass cultivation period. This agrees with Ni et al. (2015), who reported reduced NH_3 emissions following rainfall after fertilization, and the observation of higher NH_3 emissions under warmer and more windy conditions (Sommer et al., 1993; Häni et al., 2016; Nyameasem et al., 2022; Bleizgys and Naujokienė, 2023). Ammonia emissions of the drag hose applications (green rye and perennial ryegrass) were much lower than in previous studies using similar trailing hose application methods, which reported emission rates of 16-45% (Herr et al., 2019; Buchen-Tschiskale et al., 2023). Consequently, the low levels of NH_3 emission in our study were due to efficient NH_3 mitigation measures and the environmental conditions, while the different management histories did not have an effect on the emissions.

In contrast, cumulative N_2 emissions were significantly different between OF and IF for the green rye and silage maize cultivation periods, with higher emissions under OF for green rye and under IF for silage maize. It is well established that pH has a distinct effect on N_2O and N_2 emissions since at low pH, the synthesis of N_2O reductase, the enzyme catalysing the reduction of N_2O to N_2, is inhibited (Russenes et al., 2016; Zhang et al., 2021). From this perspective, lower N_2 emissions would be expected for OF, where the pH was significantly lower than that of IF (Table 1). However, this was only the case for silage maize cultivation, suggesting that other factors have contributed to the differences in cumulative N_2 emission. One such factor could be the time elapsed since management homogenization. The same crop and fertilizer management were applied to both sites starting in October 2020, but legacy effects from the prior organic or integrated farming histories may still have influenced emissions, especially during the early cropping periods like green rye and silage maize. Over time, these legacy effects may have diminished, which could explain the lack of significant differences in N_2 emissions observed during the later crop (perennial ryegrass). Similarly, crop-specific factors may have interacted with site history. For example,Alternatively, lower plant N uptake on OF compared to IF in the green rye cultivation period may have led to a higher mineral N availability for microbial processes in the soil. Higher nitrate availability combined with the carbon sources of the organic fertilizer may have stimulated denitrification, particularly N_2 production (Senbayram et al., 2012; Samad et al., 2016). In contrast, by the time of the perennial ryegrass phase, soil microbial communities and N cycling dynamics may have adjusted to the unified management, leading to more comparable emissions. Additionally, the relatively high N input applied during the experimental period compared to the historically low N input particularly under OF may have intensified the denitrification response, particularly in the early cropping periods. Thus, the complex interplay of historical management, time since management harmonization, crop type, and soil N cycle processes like plant N demand, the mineralization-immobilization cycle controlling N availability, and the effect of environmental conditions on the denitrification process (Butterbach-Bahl et al., 2002; Chen et al., 2019) led to variable N_2 emissions from both sites.

As mentioned above, N_2O and N_2 fluxes are closely linked as both are produced and consumed in the process of denitrification, converting nitrate to nitrite, nitric oxide, nitrous oxide, and finally to N_2 (Butterbach-

Bahl and Dannenmann, 2011). For this reason, we expected to observe an increase in both N_2O and N_2 emission after fertilizer application as was reported by Häfner et al. (2021), Herr et al. (2019) and Bizimana et al. (2022). In this study, we observed a distinct N_2 emission following every manure application event, but the N_2O emission hardly increased for IF during green rye cultivation and for OF and IF during silage maize cultivation (Fig. 2~~3~~, Table A3). The other N_2O emission pulses were moderate with emission rates below 140 µg m^{-2} h^{-1} (Fig. ~~1~~2). The lack of N_2O peaks after fertilizer application was also observed by Buchen-Tschiskale et al. (2023) and Dannenmann et al. (2024) and can be explained by conditions conducive to full denitrification, i.e., low oxygen levels, high soil moisture, good availability of labile carbon sources and high levels of mineral N (Smith and Arah, 1990; Morley et al., 2014; Rohe et al., 2021; Wang et al., 2021). Such conditions can result from the application of manure, which is a mixture of water, organic substances, and ammonium.

With regard to the magnitude of N_2 emissions, few field studies have measured N_2 emissions from cattle slurry application. The level of cumulative N_2 emission in this study was between 0.36 and 2.19 kg N ha^{-1} (Fig. ~~2~~3) for both sites which is in the same range as the 1.3 to 2.20 kg N ha^{-1} reported by Buchen-Tschiskale et al. (2023) for an agricultural site receiving 71 kg N ha^{-1} of cattle slurry, and the 0.52 to 0.78 kg N ha^{-1} for a grassland receiving 120 kg N ha^{-1} in form of cattle slurry (Stevens and Laughlin, 2001a). Despite the amount of N_2 losses quantified in this study agreeing with the past studies, the direct field measurement of N_2 fluxes in addition to all other relevant loss pathways did not result in a closed ^{15}N balance, with the average imbalance being 15 and 7 kg N ha^{-1} for OF and IF, respectively. While some part of the unrecovered ^{15}N may be due to an underestimation of NH_3 loss, N_2 emissions may be underestimated as well (Yankelzon et al., 2024a). Heterogeneous ^{15}N distribution in the microplot soil resulting from surface or slit application of the slurry and $^{15}N_2$ diffusion and storage in subsoil layers contributes to the underestimation of N_2 flux rates. ~~One reason for the underestimation of $^{15}N_2$ is the short coverage of the measurements which is due to the short period of time during which the isotopic enrichment of the N pool subject to denitrification is sufficiently high so that the enrichment in the chamber headspace air exceeds the detection limit of the mass spectrometers. Consequently, we cannot exclude that additional ^{15}N in N_2 is emitted in the months following fertilizer application, particularly during rewetting events or towards the end of the growing season when plants decrease their water uptake and water content increases, as observed by Almaraz et al. (2024). Additionally, heterogeneous ^{15}N distribution in the microplot soil resulting from surface or slit application of the slurry and $^{15}N_2$ diffusion and storage in subsoil layers contributes to the underestimation of N_2 flux rates.~~ These complications associated with ^{15}N labelling were discussed in previous studies which indicate that fluxes may be underestimated by up to 30-50% (Vanden Heuvel et al., 1988; Arah, 1997; Well et al., 2018; Well et al., 2019; Friedl et al., 2020; Micucci et al., 2023; Dannenmann et al., 2024). Even assuming a 50% underestimation, N_2 emissions after fertilization are in the range of 2 kg N ha^{-1}, indicating that the method-inherent potential underestimation alone cannot close the N-balance. However, the short coverage of the direct N_2 measurements, which is due to the short period of time during which the isotopic enrichment of the N pool subject to denitrification is sufficiently high so that the N_2 flux can be detected, may be an additional source for underestimation. Consequently, we cannot exclude that additional ^{15}N in N_2 is emitted in the months following fertilizer application, particularly during rewetting events or towards the end of the growing season when plants decrease their water uptake and water content increases, as observed by Almaraz et al. (2024).

At the same time, however, it must be recognized that the spatial variability of soil properties, environmental conditions, microbial activity and plant growth leads to a large uncertainty in the determination of the ^{15}N balance and therefore the residual element of the balance may be at least partly due to this uncertainty.

497 Thus, the apparent mismatch in the N budget may also be due to measurement uncertainties rather than actual N
498 losses.

499 The $N_2O : (N_2 + N_2O)$ ratio in this study ranged from 0.01 to 0.45~~1.00~~ across both sites using different
500 slurry application techniques. This aligns with the findings of Fangueiro et al. (2008) who incorporated cattle
501 slurry fractions (solid and liquid) into the soil, observing that the $N_2O : (N_2 + N_2O)$ ratio varied between 0.32 and
502 0.73. Similarly, Dannenmann et al. (2024) observed very low N_2O emissions (0.1 kg N_2O-N ha^{-1}) from a drag hose
503 cattle slurry application (97 kg N ha^{-1}) under conditions favouring full denitrification, yielding a low $N_2O : (N_2 +$
504 $N_2O)$ ratio of 0.03.

4.2 Nitrate leaching

506 Nitrate leaching is influenced by several factors, such as nitrate concentration, soil texture, timing of
507 manure application, meteorological conditions, and cropping system (Van et al., 2006; Maguire et al., 2011;
508 Wangari et al., 2024). In this study, nitrate leaching rates ranged from 0 to 4 kg N ha^{-1}, corresponding to 4 and 3%,
509 1 and 0%, and 4 and 5% of the applied NH_4^+-N (Fig. ~~3~~4, Table A3) for OF and IF, respectively, during the green
510 rye, silage maize and perennial ryegrass cultivation period. There was ~~on~~ no significant difference between OF
511 and IF, which indicates that management history had no impact on the NO_3^- leaching. The observed values are at
512 the lower end of the range of 4-107 kg N ha^{-1} reported for arable cropping systems in a review article (Di and
513 Cameron, 2002) and the IPCC-based model estimates reported by Eysholdt et al. (2022), averaging to 23.1 kg N
514 ha^{-1} for Germany during the reference period of 2014–2016. Similarly, comparable NO_3^- leaching values on
515 medium-texture soil were reported by Buchen-Tschiskale et al. (2023), who observed average nitrate leaching of
516 4% for trailing hose application or slurry injection of cattle slurry at an application rate of 71 kg N ha^{-1} for arable
517 cropping systems. Thus, the medium soil texture of the sites in this study (Table 1) may have contributed to the
518 observed low leaching rates.

519 Among the different crops, nitrate leaching was lowest for the silage maize cultivation period when
520 manure was incorporated. While preceding studies indicated that incorporation or injection of slurry increases
521 nitrate concentrations in the soil and increases the risk of N leaching, differences between application methods
522 vanished at lower slurry manure application rates or if the application rate was close to N demand (Kramer et al.,
523 2006; Maguire et al., 2011; Dannenmann et al., 2024). The latter was the case in this study since cumulative
524 aboveground N uptake was close to N application rate (75 to 92%, Fig. ~~4~~5), which may have also caused the low
525 N leaching rates of between 0-1 kg N ha^{-1}. Low nitrate leaching associated with cattle slurry application was also
526 observed in a 4 years study on grassland with trailing hose or cattle slurry injection of 80 kg N ha^{-1} (Kayser et al.,
527 2015) and by Dannenmann et al., (2024) who reported negligible nitrate leaching (0.2 kg N ha^{-1}) from the cattle
528 slurry application of 97 kg N ha^{-1} to carbon-rich pre-alpine grassland soils where aboveground N uptake exceeded
529 fertilization rate.

530 ~~But low leaching rates are not limited to grassland since Buchen-Tschiskale et al. (2023) reported average~~
531 ~~nitrate leaching of 4% for trailing hose application or slurry injection of cattle slurry at an application rate of 71~~
532 ~~kg N ha^{-1} for arable cropping systems on medium-textured soil, which is close to the values observed in this study.~~
533 ~~Thus, the medium soil texture of the sites in this study (Table 1) additionally contributed to the low leaching rates.~~

534 Overall, it must be noted that the exact travel time of NO_3^- at the site is unknown and could exceed the
535 duration of a cultivation period. In addition, total nitrate instead of ^{15}N-NO_3^- was determined in this study.
536 Consequently, the observed nitrate leaching for the respective cultivation periods may include some contributions

537 from preceding fertilizer applications, not fully represent the leaching rate due to the most recent fertilizer
538 application and overestimate the contribution of NO_3^- leaching to the ^{15}N balance.

4.3 Recovery of ^{15}N from soil and plants

540 In a single cultivation period, the recovery of ^{15}N in plant biomass ranged from 24-39% for OF and 34-
541 40% for IF, with averages of 31% for OF and 37% for IF (Fig. 3a4a-d). Soil ^{15}N recovery over a single cultivation
542 period showed a range of 36-52% for OF and 35-59% for IF, averaging 44% for OF and 46% for IF across all
543 cultivation periods (Fig. 3a4a-d). To our knowledge, there are only a few studies that used ^{15}N-labeled cattle slurry
544 in field experiments on arable land, such as Jensen et al. (2000), who recovered 32% of ^{15}N in the aboveground
545 biomass and 45% in the soil for winter wheat. Similarly, Paul et al. (1995) reported an average recovery of 43%
546 of applied ^{15}N-labeled cattle slurry in above- and belowground biomass of corn. In recent work, Frick et al. (2023)
547 applied ^{15}N-labeled cattle slurry (produced by feeding a heifer ^{15}N-enriched ryegrass hay) to a grass-clover system
548 and observed annual recovery rates of 17-22% in plant biomass and 32-52% in soil. Buchen-Tschiskale et al.
549 (2023) recovered 32-47% in soil, while plant recovery was between 25-33% using different application techniques
550 (drag hose and slit injection) for ^{15}N-labeled cattle slurry to winter wheat, indicating that the recoveries of our
551 study align well with published data.

552 Total soil and plant recoveries were highest during the silage maize cultivation period, where manure was
553 incorporated into the soil. In contrast, recoveries of soil and plant were lower during the cultivation periods of
554 green rye and perennial ryegrass, where manure application was performed using a drag-hose system. Manure
555 incorporation distributes N in the soil column, reduces surface N concentration and thus reduces NH_3 volatilization
556 and facilitates microbial immobilization (Sørensen and Thomsen, 2005; Lyu et al., 2024), resulting in overall
557 recoveries close to 100%. While this trend was not significant in this study, Buchen-Tschiskale et al. (2023)
558 observed significantly lower NH_3 emissions and a significantly higher soil plus plant recovery (79%) and overall
559 recovery of 99% for slit injection compared to trailing hose application (57 and 78%). Simultaneously,
560 significantly lower NH_3 emissions for the slit injection treatment in Buchen-Tschiskale et al. (2023) and the
561 incorporation in this study must have resulted in higher availability of mineral N for soil microorganisms which
562 could lead to increased emission of N_2O or N_2. However, in both studies, N_2O and N_2 emissions were not
563 significantly higher, demonstrating that reducing NH_3 emissions does not necessarily result in pollution swapping
564 from NH_3 to N_2O or N_2 when microbes efficiently immobilize N and prevent direct stimulation of nitrification and
565 denitrification, which produce N_2O and N_2. In general, there was a trend towards lower recoveries in soil and plant
566 biomass for manure application using a drag-hose which means that other ^{15}N loss pathways were more important.
567 It is well established that the measurement of NH_3 emission in the field has remained a challenge and is associated
568 with large uncertainties (Loubet et al., 2018), suggesting that underestimation of NH_3 emissions could explain this
569 trend. However, since an overall recovery of close to 100% was observed for treatments with NH_3 emissions of
570 8%, i.e., the perennial ryegrass cultivation period of this study and the slit injection treatment of Buchen-Tschiskale
571 et al. (2023), a significant underestimation of NH_3 emission seems unlikely since the underestimated NH_3 emission
572 would imply a recovery greater than 100%. For the same reason, i.e., observation of approximately 100% recovery
573 for certain treatments, a systematic underestimation of leaching losses is also unlikely. The only measurements
574 that don't cover the whole cultivation period are those of N_2O and N_2, suggesting that underestimation of these N
575 losses due to coverage of measurements of only a fraction of the whole cultivation period could explain the
576 unrecovered N losses. Since N_2O emissions are approximately a factor of 10 lower than N_2 emissions (Scheer et

al., 2020), N_2 emissions may have contributed the main part to the unrecovered losses. Assuming a background N_2 emission rate of 87 µg m^{-2} h^{-1}, which is equivalent to an emission if measured isotope ratios are increased by one standard deviation compared to the background, 0.7, 2.5 and 3.7 kg N_2-N ha^{-1} are released during green rye, maize and ryegrass cultivation, respectively. Such a background emission together with slightly underestimated NH_3 emissions could explain the unrecovered N_2 losses for IF. In this study, overall Overall recovery for OF was close to or lower than that of IF, suggesting that for the OF, additional N_2 was emitted during the cultivation period, which could be due to more frequent denitrification events caused by higher soil bulk density (Table 1, Luo et al., 2000; Hamonts et al., 2013). Furthermore, large error margins in soil and plant recovery rates, flux measurements, and spatial variation introduce uncertainty, indicating that some of the apparent losses may also be due to measurement limitations rather than actual N loss.

However, differences in recovery between OF and IF were not significant for single cultivation periods and the full study period. The different fertilizer application techniques, as well as the different crop types, affect the interannual variability of recovery rates, which complicates the detection of significant differences on the interannual scale. In contrast, this suggests for the different cultivation periods that the management history either didn't affect the allocation of ^{15}N to the different components of the ^{15}N balance, or that the differences are too subtle to be determined compared to the spatial variability of soil properties on this specific site and measurement uncertainty. The highest absolute uncertainties, which eventually also control the overall uncertainty of the recovery were observed for ^{15}N recovery in soil and plants. Though in this study, the area of the microplots was 0.225 m², corresponding to approximately 30 kg of soil for the 0-10 cm layer, only few milligrams of finely ground soil are eventually used to determine the ^{15}N enrichment. Consequently, the excavated soil and plants were carefully crushed and homogenized in mixing vessels, but the ball mills usually used to prepare the finely ground material are limited to 15 to 20 g of homogenised material. Since soil aggregates are not entirely destroyed during homogenization, samples transferred to the ball mill may still show distinct variability which conceals significant differences between sites. Obviously, increase of replicate amount or analysis of several subsamples (e.g., of soil of a given layer) could reduce the uncertainty, but only by increasing the already high workload and costs by several factors. For this reason, our study shows that research into protocols aiming at reducing the uncertainty arising from incomplete sample homogenisation to the measurement instrument uncertainty by including for instance mills with much larger sample capacities is required, especially since - to our knowledge - there is no publication on different homogenisation and mixing protocols for ^{15}N balances. At any rate, Following the same line of argumentation, reduction of in the uncertainty for of the direct measurement of N balance components, i.e., determination of leaching losses through leaching, NH_3, N_2 and N_2O emissions could help resolving difference more accurately is not in view, so that it appears like the latter loss pathways were not affected by management history. While recent advancements, such as the improved ^{15}N gas flux method (^{15}NGF+) demonstrate potential for enhanced sensitivity in quantifying N_2 emissions under field conditions (Eckei et al., 2025), such approaches are expensive and technically challenging which will delay their use in studies targeting complete N balances. Since the same applies to significant reduction of uncertainty in determining leaching losses, NH_3 volatilization, and N_2O emissions, it appears like these loss pathways were not markedly influenced by management history given the current measurement frameworks.

4.4 Nitrogen balance and management history

Previous studies showed that intensive agricultural systems often experience substantial N losses through gaseous emissions and NO_3^- leaching, which can match or exceed crop N uptake (Ju et al., 2009; Zhou et al., 2016). In this study, cumulative N losses were not significantly different between OF and IF and can be classified as low since they amounted to approximately 8 to 10% of plant N uptake. This was due to a combination of the medium textured soil attenuating NO_3^- leaching and the low NH_3 loss resulting from efficient NH_3 mitigation measures (see section 4.1). In contrast, cumulative green rye and perennial ryegrass N uptake were significantly higher on IF than on OF. Thus, IF was more productive, particularly given that the lower yield during silage maize cultivation was likely due to delayed herbicide application in response to weather conditions.

Since crop rotation, fertilizer type and fertilizer amount were the same during the two-years experiment of this study, and climatic conditions were the same for the two adjacent sites during the historical management from 2012-2020, differences in soil properties due to management must have affected productivity. The comparison of initial sampling showed that OF soil was significantly more acidic (Table 1), likely due to repeated legumes cultivation in the crop rotation as legumes take up more cations than anions (Msimbira and Smith 2020, Chaoui et al., 2023). Additionally, our results show that preceding management resulted in significantly higher average C and N content down to 30 cm for IF than for OF. On the one hand, the increased N application on IF during the preceding management of 2012 to 2020 (Table A1) likely boosted above- and belowground biomass production compared to OF, which resulted in higher C input through above- and belowground plant residues as well as root exudates on IF which eventually increased SOC levels, in agreement with other long-term studies (Gai et al., 2018; Böhm et al., 2020). On the other hand, the significantly higher input of a combination of synthetic and organic fertilizers and returned crop residues on IF (Table A1) has contributed to SOC build-up, and increased mineralization and microbial biomass (Ramirez et al., 2012; Dai et al., 2017; Tang et al., 2018; Marliah et al., 2020; Li et al., 2021; Peng et al., 2023; Khan et al., 2024). Consequently, the significantly higher yield can be related to improved supply of nutrients to the crops between fertilization events through higher mineralization of organic matter and probably improved water holding capacity (Dai et al., 2017; Manns and Martin 2018; Khan et al., 2024). In conjunction with the N input through deposition and fertilizer, which was close to the N application rates for IF from 2012-2020 (average of 184 kg N ha^{-1}, Table A1) and in accordance with the national fertilizer legislation for use of manure (170 kg N ha^{-1} year^{-1}), the cumulative and single cultivation period N balances were neutral within the limits of the measurement uncertainty for IF (-8 ± 15, Fig. 4d5d). This indicates effective N management for IF, as a balanced plant N demand with N supply will prevent the mining of soil N and maintain soil N and C levels through the coupling of the N and C cycles (Zistl-Schlingmann et al., 2020). However, the N balances across single cultivation periods for OF ranged from -19 to 41 kg N ha^{-1} and the cumulative N balance for all cultivation periods was positive (48 ± 14 kg N ha^{-1}, Fig. 4d5d). In a situation in which the boundary of the N balance is the agricultural field, and N loss pathways (NO_3^- leaching, NH_3, N_2O, N_2) were explicitly determined, a positive N balance indicates a surplus of N that remains on the field. In other words, this N is available for soil organic matter build-up, i.e., an increase in soil organic N stocks. This indicates that a more balanced fertilization approach compared to the management from 2012 to 2020 has the potential to restore C and N levels, which may increase the productivity of the site in the long-term. However, since the C or N stocks change cannot be determined on the time scale of 2 years (Küstermann et al., 2013), and – as discussed above – some N loss pathways do not cover the whole cultivation period and may, thus, be underestimated, it cannot be excluded that the positive

655 N balance to some degree points towards unaccounted losses. Since positive N balances were observed for OF,

656 where the ^{15}N recovery also showed higher unrecovered shares, we assume that not the full N balance can be

657 attributed to an increase in soil N stocks. In this context, it is noteworthy that crop N export and balance values

658 from different cultivation periods align with findings from other studies (Thompson et al., 1987; Dell et al., 2011;

659 Lin et al., 2016; Duncan et al., 2019), which applied cattle slurry using various techniques, including surface

660 application and incorporation.

661 Furthermore, the annual variability in N balances of OF was also observed by other studies due to

662 microbial N immobilization, soil environmental conditions, slurry application technique, crop N export and the

663 impact of historical management practices (De Jager et al., 2001; Dell et al., 2011; He et al., 2018; Chmelíková et

664 al., 2021; Winkhart et al., 2022). This underscores the need for repeated N balance measurements and cumulative

665 N balances to fully capture the effects of different N management practices on N dynamics, ultimately supporting

666 the development of more sustainable agricultural practices. To improve our understanding of long-term nutrient

667 cycling, future studies should explicitly address both the legacy effects of past management practices and the direct

668 impacts of current interventions. Such research will be crucial for disentangling time-dependent effects and guiding

669 the design of resilient and sustainable farming systems.

670 **4.5 Impact of experimental homogenization of management on legacy effects**

671 While the study aimed to investigate the legacy effects of organic farming (OF) and integrated farming

672 (IF) systems on nitrogen cycling, it is important to recognize that both systems were managed identically during

673 the experimental period (2021–2022). This homogenization of management practices, which included uniform

674 crop rotations, tillage methods, and high cattle slurry application rates, may have limited the ability to draw direct

675 comparisons between OF and IF systems in the experimental phase. As such, the observed differences in N cycling

676 may be more reflective of past management legacies rather than current practices. Additionally, the timing of the

677 crop assessments—green rye immediately after management homogenization and ryegrass two years later—could

678 have further influenced the results, as the green rye was likely more influenced by previous management practices.

679 Furthermore, the high rates of cattle slurry application, typical of IF systems but higher than those generally applied

680 in OF systems, may have masked potential differences between the two systems in terms of nutrient cycling.

681 Finally, the absence of legumes in the experimental crop rotations meant that their role in N cycling could only be

682 inferred as a legacy effect, rather than being directly assessed. These factors should be considered when

683 interpreting the findings, and future studies that include a more direct comparison of management systems with

684 consistent practices and crop rotations, including legumes, would be valuable in further understanding the impacts

685 of OF and IF on N cycling.

686 **5. Conclusion**

687 This study is the first to quantify fertilizer N balances, including directly measured emissions of N_2, N_2O,

688 NH_3, and NO_3^- leaching losses, as well as ^{15}N balances and total N balances, over three consecutive cultivation

689 periods (green rye, silage maize and perennial ryegrass) as affected by either legacy effects of integrated or organic

690 farming systems history under homogenized management conditions.

691 Ammonia losses were low due to efficient NH_3 mitigation measures, soil texture, and fertilization close

692 to plant N demand, resulting in low nitrate leaching, irrespective of the management history. Emissions of N_2O

693 were negligible compared to the N balance, and average N_2 emissions for all cultivation periods were not

694 significantly different, but could only be determined for two weeks after fertilizer application. Both OF and IF
695 practices demonstrated minimal N losses, indicating that both approaches effectively mitigate N leaching and
696 emissions when management is standardized.

697 Integrated farming increased productivity by improving plant N supply through higher soil organic matter
698 and higher rates of mineralization, thereby maintaining a balanced N budget. The positive N balance of organic
699 farming demonstrates that increased N input may result in soil N and C accumulation, potentially improving
700 productivity on the long-term. Consequently, a higher level of organic fertilizer additions or a higher share of
701 residue returns for this specific, legume-cantered N management strategy may be beneficial with regard to
702 productivity and soil fertility. Finally, this study demonstrates that multi-year ^{15}N tracing and N balance studies
703 are powerful tools to quantitatively assessing the environmental and agronomic impacts of different management
704 strategies. Future studies that maintain distinct management systems during the experimental phase would help
705 clarify the direct impacts of OF and IF on N dynamics under realistic farming conditions.

Appendix A: Tables

Table A1: Annual C and N input of organic farming (OF) and integrated farming (IF) sites for the period 2012-2020, i.e., prior to the field experiment. All crops were mowed from both sites with same machinery. Crop residues were returned to IF site but not to OF site. Tillage to a depth of 20 cm was uniformly executed at both sites (OF, IF). In case of missing documentation, gaps were filled based on recommended and typical practice for the area and literature values.

Year	Organic farming (OF)				Integrated farming (IF)			
	Crop	Input type	kg C ha^{-1} y^{-1}	kg N ha^{-1} y^{-1}	Crop	Input type	kg C ha^{-1} y^{-1}	kg N ha^{-1} y^{-1}
2012	Silage maize	cattle slurry	900	105	winter wheat	mineral N		180
2013	winter spelt				winter wheat	mineral N		180
2014	red clover & grass				winter wheat	mineral N		110
						sewage sludge	795	123
2015	red clover & grass				silage maize	mineral N		110
						separated slurry	837	57
2016	winter wheat				winter wheat	mineral N		180
2017	silage maize	cattle slurry	900	105	winter wheat	mineral N		110
						sewage sludge	373	60
2018	winter wheat				winter rye	mineral N		100
						separated slurry	837	57
2019	alfalfa and grass				winter wheat	mineral N		180
2020	alfalfa and grass				silage maize	mineral N		140
						sewage sludge	848	71
Mean			200	23			410	184

711

712 **Table A2:** Preparation of the ^{15}N labelled manures for different cultivation period.

Parameters	Green rye cultivation period		Silage maize cultivation period		Perennial ryegrass cultivation period 1st application		Perennial ryegrass cultivation period 2nd application	
	microplot	mesocosms	microplot	mesocosms	microplot	mesocosms	microplot	mesocosms
Area (m^2)	0.225	0.16	0.225	0.16	0.225	0.16	0.225	0.16
Replicate/field	4	6	4	6	4	6	4	6
Total unlabeled manure (L)	4.02	0.07	3.98	0.06	4.02	0.07	3.98	0.06
(^{15}NH$_4$)$_2$SO$_4$ (g)	5.75	5.61	5.06	4.93	5.75	5.61	5.06	4.93
^{15}N urea (g)	1.33	1.30	1.17	1.14	1.33	1.30	1.17	1.14
Milli-Q water (ml)	534	720	470	633	534	720	470	633
Manure + ^{15}N + mill-Q / replicate (ml)	503	50	495	50	503	50	495	50
Target enrichment level (%)	10	85	10	85	10	85	10	85

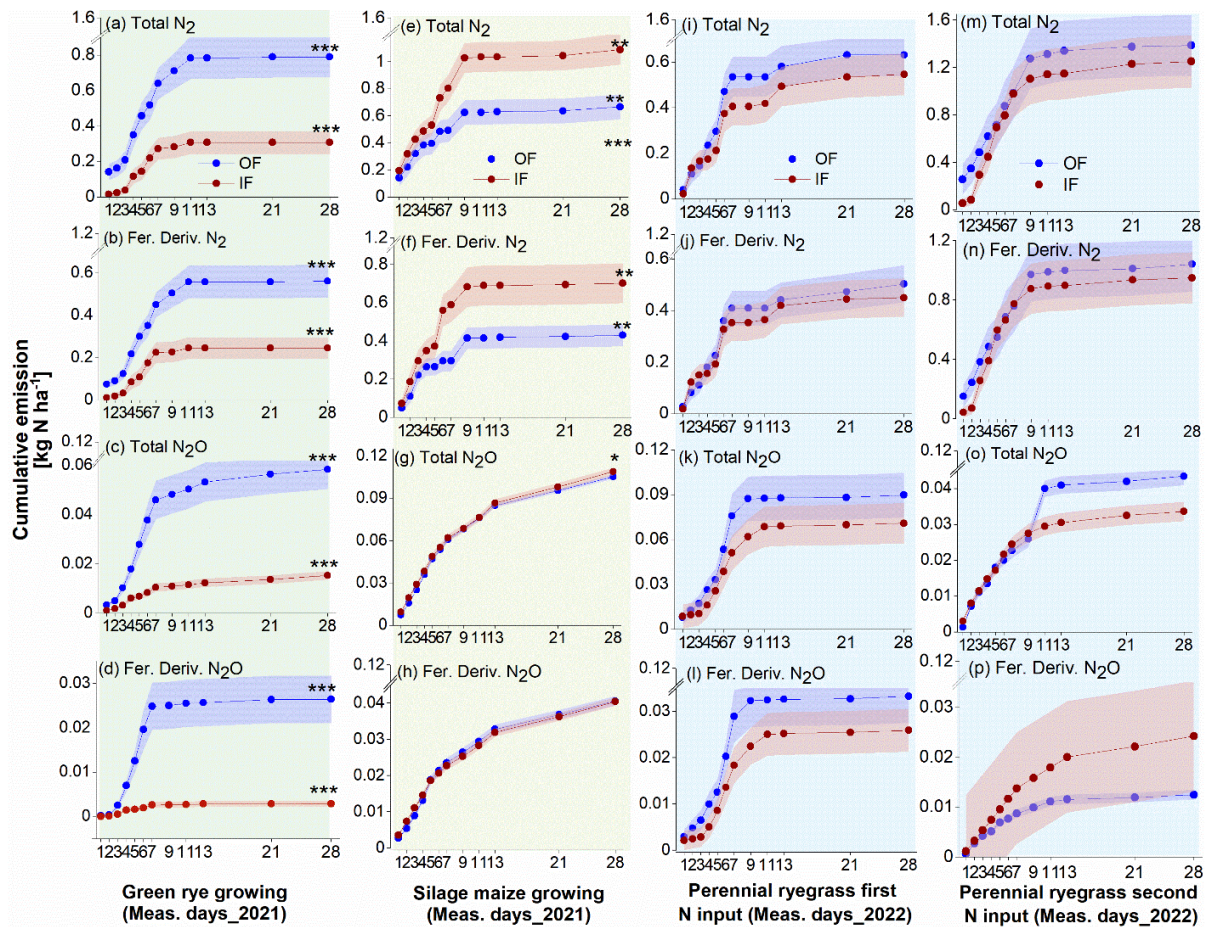
713

714 **Table A3:** Fertilizer ^{15}N input, ^{15}N recovery, and unrecovered amounts in green rye, silage maize, and perennial ryegrass
715 cultivation period.

[15]N input, loss and recovery	Segment	Green rye_2021 (mean value ± SE)		Silage maize_2021 (mean value ± SE)		Perennial ryegrass_2022 (mean value ± SE)		Average (mean value ± SE)	
		OF (%)	IF (%)	OF (%)	IF (%)	OF (%)	IF (%)	OF (%)	IF (%)
[15]N input	Microplots	100	100	100	100	100	100	100	100
Soil depth	0-90 cm	52.10 ± 8.73	35.27 ± 3.59	43.78 ± 5.95	58.53 ± 3.40	35.54 ± 1.30	44.32 ± 2.05	43.81 ± 5.33	46.04 ± 3.01
Plant	Shoot	24.06 ± 2.41	40.42 ± 6.36	39.00 ± 9.46	33.96 ± 6.71	30.34 ± 3.28	37.16 ± 3.01	31.13 ± 5.05	37.18 ± 5.36
	Root	3.17 ± 0.42	2.08 ± 0.56	0.34 ± 0.07	0.46 ± 0.04	2.09 ± 0.17	1.89 ± 0.02	1.87 ± 0.22	1.48 ± 0.21
	NH$_3$	2.39 ± 0.30	3.20 ± 0.20	0.19 ± 0.01	0.20 ± 0.01	8.03 ± 0.20	8.45 ± 0.51	3.54 ± 0.09	3.95 ± 0.19
Nitrogen loss through emissions and leaching	N$_2$O	0.07 ± 0.01	0.02 ± 0.00	0.14 ± 0.00	0.15 ± 0.00	0.09 ± 0.01	0.07 ± 0.01	0.10 ± 0.01	0.08 ± 0.00
	N$_2$	0.82 ± 0.36	0.33 ± 0.18	0.82 ± 0.30	2.13 ± 0.70	3.55 ± 1.85	3.26 ± 1.57	1.73 ± 0.84	1.91 ± 0.82
	NO$_3^-$	4.34 ± 1.79	2.56 ± 0.35	1.33 ± 0.00	0.00 ± 0.00	4.09 ± 0.00	4.74 ± 1.02	3.26 ± 0.60	2.43 ± 0.46
Recovery	Sum	86.83 ± 13.74	83.88 ± 11.24	85.60 ± 15.80	95.43 ± 10.87	83.73 ± 6.81	99.89 ± 8.19	85.42 ± 12.13	93.07 ± 10.05
Unrecovered	Input-sum	13.17	16.12	14.40	4.57	16.27	0.11	14.58	6.93

716

 Appendix B: Figure

718

719 **Figure B1:** The cumulative total and fertilizer-derived emissions of N_2 and N_2O across different cultivation period. Subplots
720 (a, b, c, and d) depict the total N_2, fertilizer-derived N_2, total N_2O, and fertilizer-derived N_2O emissions, respectively, during
721 the green rye cultivation period and subplots (e, f, g, and h) show the corresponding emissions for the silage maize cultivation
722 period. For the perennial ryegrass cultivation period, subplots (i, j, k, and l) illustrate emissions of total N_2, fertilizer-derived
723 N_2, total N_2O, and fertilizer-derived N_2O emissions, respectively, following the 1st fertilizer application, while subplots (m, n,
724 o, and p) represent corresponding emissions after the 2nd fertilizer application. Asterisk showed the significant difference (*:
725 $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$). Symbols and shade represent the mean values with standard error, respectively. The
726 background shading in light green, pale yellow, and light blue represents measurements during the green rye, silage maize, and
727 perennial ryegrass cultivation period, respectively.

Data availability

The data used in this manuscript is available from the Karlsruhe Institute of Technology, Garmisch-Partenkirchen, Germany, database. Currently, the dataset can be accessed upon request from the corresponding author and will be made open access upon the final publication of the manuscript.

Author contributions

Conceptualization: FK carried out field- and laboratory work, data analysis, and drafted the manuscript. SFL carried out field- and laboratory work. FH carried out field management, data collection and laboratory work. MD, CS and RK supervised, contributed to the study design and revised the manuscript. RG supervised field work. WN carried out the field management and collected data and revised the manuscript. EGW and RMM helped in laboratory work. BW and AG acquired the funding, supervised and revised the manuscript.

Conflict of interest

The authors declare no conflict of interest.

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References

Abdo, A. I., Sun, D., Li, Y., Yang, J., Metwally, M. S., Abdel-Hamed, E. M. W., Wei, H., and Zhang, J.: Coupling the environmental impacts of reactive nitrogen losses and yield responses of staple crops in China, Front. Plant Sci., 13, 927935, https://doi.org/10.3389/fpls.2022.927935, 2022.

Arah, J. R. M.: Apportioning nitrous oxide fluxes between nitrification and denitrification using gas-phase mass spectrometry, Soil Biol. Biochem., 29, 1295–1299, https://doi.org/10.1016/S0038-0717(97)00009-6, 1997.

Almaraz, M., Wong, M. Y., Zhang, Y., and Silver, W. L.: Biotic regulation of nitrogen gas emissions in temperate agriculture, Biogeochemistry, https://doi.org/10.1007/s10533-024-01157-9, 2024.

Bizimana, F., Luo, J., Timilsina, A., Dong, W., Gaudel, G., Ding, K., Qin, S., and Hu, C.: Estimating field N_2 emissions based on laboratory-quantified $N_2O/(N_2O + N_2)$ ratios and field-quantified N_2O emissions, J. Soils Sediments, 22, 2196–2208, https://doi.org/10.1007/s11368-022-03212-0, 2022.

Bleizgys, R., and Naujokienė, V.: Ammonia emissions from cattle manure under variable moisture exchange between the manure and the environment, Agronomy, 13, 1555, https://doi.org/10.3390/agronomy13061555, 2023.

Böhm, H., Dauber, J., Dehler, M., Gallardo, D. A. A., de Witte, T., Fuß, R., Höppner, F., Langhof, M., Rinke, N., and Rodemann, B.: Fruchtfolgen mit und ohne Leguminosen: ein Review, J. Kulturpflanzen, 72, 489–509, https://doi.org/10.5073/JfK.2020.10-11.01, 2020.

Bryla, D. R.: Application of the "4R" nutrient stewardship concept to horticultural crops: Getting nutrients in the "right" place, HortTechnology, 21, 674–682, https://doi.org/10.21273/HORTTECH.21.6.674, 2011.

Buchen-Tschiskale, C., Well, R., and Flessa, H.: Tracing nitrogen transformations during spring development of winter wheat induced by 15N-labeled cattle slurry applied with different techniques, Sci. Total Environ., 871, 162061, https://doi.org/10.1016/j.scitotenv.2023.162061, 2023.

Butterbach-Bahl, K., and Dannenmann, M.: Denitrification and associated soil N_2O emissions due to agricultural activities in a changing climate, Curr. Opin. Environ. Sustain., 3, 389–395, https://doi.org/10.1016/j.cosust.2011.08.004, 2011.

Butterbach-Bahl, K., Willibald, G., and Papen, H.: Soil core method for direct simultaneous determination of N_2 and N_2O emissions from forest soils, Plant Soil, 240, 105–116, https://doi.org/10.1023/A:1015870518723, 2002.

Cárceles Rodríguez, B., Durán-Zuazo, V. H., Soriano Rodríguez, M., García-Tejero, I. F., Gálvez Ruiz, B., and Cuadros Tavira, S.: Conservation agriculture as a sustainable system for soil health: A review, Soil Syst., 6, 87, https://doi.org/10.3390/soilsystems6040087, 2022.

Chalk, P. M., Craswell, E. T., Polidoro, J. C., and Chen, D.: Fate and efficiency of 15N-labelled slow- and controlled-release fertilizers, Nutr. Cycl. Agroecosyst., 102, 167–178, https://doi.org/10.1007/s10705-015-9697-2, 2015.

Chaoui, R., Boudsocq, S., Taschen, E., Sentenac, H., Farissi, M., and Lazali, M.: Intercropping durum wheat and chickpea increases nutrient availability and use efficiency under low phosphorus soils, J. Plant Nutr., 46, 4125–4139, https://doi.org/10.1080/01904167.2023.2221677, 2023.

Chen, T., Oenema, O., Li, J., Misselbrook, T., Dong, W., Qin, S., Yuan, H., Li, X., and Hu, C.: Seasonal variations in N_2 and N_2O emissions from a wheat–maize cropping system, Biol. Fertil. Soils, 55, 539–551, https://doi.org/10.1007/s00374-019-01373-8, 2019.

Chmelíková, L., Schmid, H., Anke, S., and Hülsbergen, K.-J.: Nitrogen-use efficiency of organic and conventional arable and dairy farming systems in Germany, Nutr. Cycl. Agroecosyst., 119, 337–354, https://doi.org/10.1007/s10705-021-10126-9, 2021.

Costa, C. M., da Costa, A. B. G., Theodoro, G. F., Difante, G. S., Gurgel, A. L. C., Santana, J. C. S., Camargo, F. C., and de Almeida, E. M.: The 4R management for nitrogen fertilization in tropical forage: A review, Aust. J. Crop Sci., 14, 1834–1837, https://doi.org/10.21475/ajcs.20.14.11.p2646, 2020.

Couto-Vázquez, A., and Gonzalez-Prieto, S. J.: Fate of 15N-fertilizers in the soil-plant system of a forage rotation under conservation and plough tillage, Soil Tillage Res., 161, 10–18, https://doi.org/10.1016/j.still.2016.02.011, 2016.

Dai, S., Jing, W., Cheng, Y., Zhang, J., and Cai, Z.: Effects of long-term fertilization on soil gross N transformation rates and their implications, J. Integr. Agric., 16, 2863–2870, https://doi.org/10.1016/S2095-3119(17)61673-3, 2017.

Dannenmann, M., Bimüller, C., Gschwendtner, S., Leberecht, M., Tejedor, J., Bilela, S., Gasche, R., Hanewinkel, M., Baltensweiler, A., and Kögel-Knabner, I.: Climate change impairs nitrogen cycling in European beech forests, PLoS ONE, 11, e0158823, https://doi.org/10.1371/journal.pone.0158823, 2016.

Dannenmann, M., Gasche, R., Ledebuhr, A., and Papen, H.: Effects of forest management on soil N cycling in beech forests on calcareous soils, Plant Soil, 287, 279–300, https://doi.org/10.1007/s11104-006-9077-4, 2006.

Dannenmann, M., Yankelzon, I., Wähling, S., Ramm, E., Schreiber, M., Ostler, U., Schlingmann, M., Stange, C. F., Kiese, R., and Butterbach-Bahl, K.: Fates of slurry-nitrogen applied to mountain grasslands: the importance of dinitrogen emissions versus plant N uptake, Biol. Fertil. Soils, https://doi.org/10.1007/s00374-024-01826-9, 2024.

De Jager, A., Onduru, D., Van Wijk, M. S., Vlaming, J., and Gachini, G. N.: Assessing sustainability of low-external-input farm management systems with the nutrient monitoring approach: a case study in Kenya, Agric. Syst., 69, 99–118, https://doi.org/10.1016/S0308-521X(01)00020-8, 2001.

De Sousa, R. N., and Moreira, L. A.: Plant nutrition optimization: integrated soil management and fertilization practices, https://doi.org/10.5772/intechopen.114848, 2024.

Dell, C. J., Meisinger, J. J., and Beegle, D. B.: Subsurface application of manure slurries for conservation tillage and pasture soils and their impact on the nitrogen balance, J. Environ. Qual., 40, 352–361, https://doi.org/10.2134/jeq2010.0069, 2011.

Di, H. J., and Cameron, K. C.: Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies, Nutr. Cycl. Agroecosyst., 64, 237–256, https://doi.org/10.1023/A:1021471531188, 2002.

Duncan, E. W., Kleinman, P. J. A., Beegle, D. B., and Dell, C. J.: Nitrogen cycling trade-offs with broadcasting and injecting dairy manure, Nutr. Cycl. Agroecosyst., 114, 57–70, https://doi.org/10.1007/s10705-019-09975-2, 2019.

Eckei, J., Well, R., Maier, M., Matson, A., Dittert, K., and Rummel, P. S.: Determining N2O and N2 fluxes in relation to winter wheat and sugar beet growth and development using the improved 15N gas flux method on the field scale, Biol. Fertil. Soils, 61, 489–505, https://link.springer.com/article/10.1007/s00374-024-01806-z 2025.

822 Fangueiro, D., Pereira, J., Chadwick, D., Coutinho, J., Moreira, N., and Trindade, H.: Laboratory assessment of
823   the effect of cattle slurry pre-treatment on organic N degradation after soil application and N_2O and N_2
824   emissions, Nutr. Cycl. Agroecosyst., 80, 107–120, https://doi.org/10.1007/s10705-007-9124-4, 2008.

825 Fixen, P. E.: A brief account of the genesis of 4R nutrient stewardship, Agron. J., 112, 4511–4518,
826   https://doi.org/10.1002/agj2.20315, 2020.

827 Frick, H., Oberson, A., Cormann, M., Wettstein, H.-R., Frossard, E., and Bünemann, E. K.: Similar distribution of
828   15N labeled cattle slurry and mineral fertilizer in soil after one year, Nutr. Cycl. Agroecosyst., 125, 153–
829   169, https://doi.org/10.1007/s10705-022-10205-5, 2023.

830 Friedl, J., Scheer, C., Rowlings, D. W., Deltedesco, E., Gorfer, M., De Rosa, D., Grace, P. R., Müller, C., and
831   Keiblinger, K. M.: Effect of the nitrification inhibitor 3,4-dimethylpyrazole phosphate (DMPP) on N-
832   turnover, the N_2O reductase-gene nosZ and $N_2O:N_2$ partitioning from agricultural soils, Sci. Rep., 10, 2399,
833   https://doi.org/10.1038/s41598-020-59249-z, 2020.

834 Gai, X., Liu, H., Liu, J., Zhai, L., Yang, B., Wu, S., Ren, T., Lei, Q., and Wang, H.: Long-term benefits of
835   combining chemical fertilizer and manure applications on crop yields and soil carbon and nitrogen stocks in
836   North China Plain, Agric. Water Manag., 208, 384–392, https://doi.org/10.1016/j.agwat.2018.07.002, 2018.

837 Gardner, J. B., and Drinkwater, L. E.: The fate of nitrogen in grain cropping systems: A meta-analysis of ^{15}N field
838   experiments, Ecol. Appl., 19, 2167–2184, https://doi.org/10.1890/08-1122.1, 2009.

839 Good, A. G., and Beatty, P. H.: Fertilizing nature: a tragedy of excess in the commons, PLoS Biol., 9, e1001124,
840   https://doi.org/10.1371/journal.pbio.1001124, 2011.

841 Grahmann, K., Verhulst, N., Palomino, L. M., Bischoff, W.-A., Govaerts, B., and Buerkert, A.: Ion exchange resin
842   samplers to estimate nitrate leaching from a furrow irrigated wheat-maize cropping system under different
843   tillage-straw systems, Soil Tillage Res., 175, 91–100, https://doi.org/10.1016/j.still.2017.08.013, 2018.

844 Häfner, F., Ruser, R., Claß-Mahler, I., and Möller, K.: Field application of organic fertilizers triggers N_2O
845   emissions from the soil N pool as indicated by ^{15}N-labeled digestates, Front. Sustain. Food Syst., 4, 614349,
846   https://doi.org/10.3389/fsufs.2020.614349, 2021.

847 Hamonts, K., Balaine, N., Moltchanova, E., Beare, M., Thomas, S., Wakelin, S. A., O'Callaghan, M., Condron,
848   L. M., and Clough, T. J.: Influence of soil bulk density and matric potential on microbial dynamics, inorganic
849   N transformations, N_2O and N_2 fluxes following urea deposition, Soil Biol. Biochem., 65, 1–11,
850   https://doi.org/10.1016/j.soilbio.2013.05.006, 2013.

851 Häni, C., Flechard, C., Neftel, A., Sintermann, J., and Kupper, T.: Accounting for field-scale dry deposition in
852   backward Lagrangian stochastic dispersion modelling of NH_3 emissions, Atmosphere, 9, 146,
853   https://doi.org/10.3390/atmos9040146, 2018.

854 Hauck, R. D., Melsted, S. W., and Yankwich, P. E.: Use of N-isotope distribution in nitrogen gas in the study of
855   denitrification, Soil Sci., 86, 287–291, 1958.

856 He, W., Jiang, R., He, P., Yang, J., Zhou, W., Ma, J., and Liu, Y.: Estimating soil nitrogen balance at regional
857   scale in China's croplands from 1984 to 2014, Agric. Syst., 167, 125–135,
858   https://doi.org/10.1016/j.agsy.2018.09.002, 2018.

859 Heng, L. K., Sakadevan, K., Dercon, G., and Nguyen, M. L.: International Symposium on Managing Soils for
860   Food Security and Climate Change Adaptation and Mitigation, Food Agric. Organ. U. N., Rome, 331–348,
861   2014.

862 Herr, C., Mannheim, T., Müller, T., and Ruser, R.: Effect of cattle slurry application techniques on N_2O and NH_3

863 emissions from a loamy soil, J. Plant Nutr. Soil Sci., 182, 964–979, https://doi.org/10.1002/jpln.201800376,

864 2019.

865 Iheshiulo, E. M.-A., Larney, F. J., Hernandez-Ramirez, G., St. Luce, M., Chau, H. W., and Liu, K.: Crop rotations

866 influence soil hydraulic and physical quality under no-till crop management, Agric. Ecosyst. Environ., 14,

867 1774, https://doi.org/10.1016/j.agee.2023.108820, 2024.

868 Jantalia, C. P., Halvorson, A. D., Follett, R. F., Rodrigues Alves, B. J., Polidoro, J. C., and Urquiaga, S.: Nitrogen

869 source effects on ammonia volatilization as measured with semi-static chambers, Agron. J., 104, 1595–1603,

870 https://doi.org/10.2134/agronj2012.0210, 2012.

871 Jensen, L. S., Pedersen, I. S., Hansen, T. B., and Nielsen, N. E.: Turnover and fate of ^{15}N-labelled cattle slurry

872 ammonium-N applied in the autumn to winter wheat, Eur. J. Agron., 12, 23–35,

873 https://doi.org/10.1016/S1161-0301(99)00040-4, 2000.

874 Ju, X.-T., Xing, G.-X., Chen, X.-P., Zhang, S.-L., Zhang, L.-J., Liu, X.-J., Cui, Z.-L., Yin, B., Christie, P., and

875 Zhu, Z.-L.: Reducing environmental risk by improving N management in intensive Chinese agricultural

876 systems, Proc. Natl. Acad. Sci. USA, 106, 3041–3046, https://doi.org/10.1073/pnas.0813417106, 2009.

877 Kayser, M., Breitsameter, L., Benke, M., and Isselstein, J.: Nitrate leaching is not controlled by the slurry

878 application technique in productive grassland on organic–sandy soil, Agron. Sustain. Dev., 35, 213–223,

879 https://doi.org/10.1007/s13593-014-0220-y, 2015.

880 Kempers, A. J., and Zweers, A.: Ammonium determination in soil extracts by the salicylate method, Commun.

881 Soil Sci. Plant Anal., 17, 715–723, 1986.

882 Khan, F., Basak, J. K., Jaihuni, M., Lee, D. H., Lee, J. H., and Kim, H. T.: Forced aerated poultry compost effects

883 on soil physicochemical properties and lettuce plant growth, J. Biosyst. Eng., 45, 104–116,

884 https://doi.org/10.1007/s42853-020-00050-1, 2020.

885 Khan, F., Franco, S., Ulrich, M., Rainer, D., Andreas, G., Hartmann, F., Tobisch, B., Kiese, R., and Wolf, B.:

886 Integrated rather than organic farming history facilitates soil nitrogen turnover and N_2O reduction in a green

887 rye–silage maize cropping sequence, Biol. Fertil. Soils, https://doi.org/10.1007/s00374-024-01865-2, 2024.

888 Kramer, A. W., Doane, T. A., Horwath, W. R., and van Kessel, C.: Short-term nitrogen-15 recovery vs. long-term

889 total soil N gains in conventional and alternative cropping systems, Soil Biol. Biochem., 34, 43–50,

890 https://doi.org/10.1016/S0038-0717(01)00149-3, 2002.

891 Kramer, S. B., Reganold, J. P., Glover, J. D., Bohannan, B. J. M., and Mooney, H. A.: Reduced nitrate leaching

892 and enhanced denitrifier activity and efficiency in organically fertilized soils, Proc. Natl. Acad. Sci. USA,

893 103, 4522–4527, https://doi.org/10.1073/pnas.0600359103, 2006.

894 Kulkarni, M. V., Yavitt, J. B., and Groffman, P. M.: Rapid conversion of added nitrate to nitrous oxide and

895 dinitrogen in northern forest soil, Geomicrobiol. J., 34, 670–676,

896 https://doi.org/10.1080/01490451.2016.1238981, 2017.

897 Küstermann, B., Munch, J. C., and Hülsbergen, K.-J.: Effects of soil tillage and fertilization on resource efficiency

898 and greenhouse gas emissions in a long-term field experiment in Southern Germany, Eur. J. Agron., 49, 61–

899 73, https://doi.org/10.1016/j.eja.2013.02.012, 2013.

900 Lewicka-Szczebak, D., Augustin, J., Giesemann, A., and Well, R.: Quantifying N2O reduction to N2 based on

901 N2O isotopocules–validation with independent methods (helium incubation and 15 N gas flux method),

902 Biogeosciences, 14, 711–732, https://doi.org/10.5194/bg-14-711-2017, 2017.

903 Lewicka-Szczebak, D., Lewicki, M. P., and Well, R.: N2O isotope approaches for source partitioning of N2O
904   production and estimation of N 2 O reduction–validation with the 15 N gas-flux method in laboratory and
905   field studies, Biogeosciences, 17, 5513–5537, https://doi.org/10.5194/bg-17-5513-2020, 2020.

906 Li, Z., Reichel, R., Xu, Z., Vereecken, H., and Brüggemann, N.: Return of crop residues to arable land stimulates
907   N_2O emission but mitigates NO_3^- leaching: a meta-analysis, Agron. Sustain. Dev., 41, 66,
908   https://doi.org/10.1007/s13593-021-00715-x, 2021.

909 Lin, H.-C., Huber, J. A., Gerl, G., and Hülsbergen, K.-J.: Nitrogen balances and nitrogen-use efficiency of different
910   organic and conventional farming systems, Nutr. Cycl. Agroecosyst., 105, 1–23,
911   https://doi.org/10.1007/s10705-016-9770-5, 2016.

912 Liu, L., Xu, W., Lu, X., Zhong, B., Guo, Y., Lu, X., Zhao, Y., He, W., Wang, S., and Zhang, X.: Exploring global
913   changes in agricultural ammonia emissions and their contribution to nitrogen deposition since 1980, Proc.
914   Natl. Acad. Sci. USA, 119, e2121998119, https://doi.org/10.1073/pnas.2121998119, 2022.

915 Loubet, B., Carozzi, M., Voylokov, P., Cohan, J.-P., Trochard, R., and Génermont, S.: Evaluation of a new
916   inference method for estimating ammonia volatilisation from multiple agronomic plots, Biogeosciences, 15,
917   3439–3460, https://doi.org/10.5194/bg-15-3439-2018, 2018.

918 Luo, J., Tillman, R. W., and Ball, P. R.: Nitrogen loss through denitrification in a soil under pasture in New
919   Zealand, Soil Biol. Biochem., 32, 497–509, https://doi.org/10.1016/S0038-0717(99)00179-0, 2000.

920 Lyu, H., Li, Y., Wang, Y., Wang, P., Shang, Y., Yang, X., Wang, F., and Yu, A.: Drive soil nitrogen transformation
921   and improve crop nitrogen absorption and utilization—a review of green manure applications, Front. Plant
922   Sci., 14, 1305600, https://doi.org/10.3389/fpls.2023.1305600, 2024.

923 Maguire, R. O., Kleinman, P. J. A., Dell, C. J., Beegle, D. B., Brandt, R. C., McGrath, J. M., and Ketterings, Q.
924   M.: Manure application technology in reduced tillage and forage systems: a review, J. Environ. Qual., 40,
925   292–301, https://doi.org/10.2134/jeq2009.0228, 2011.

926 Manns, H. R., and Martin, R. C.: Cropping system yield stability in response to plant diversity and soil organic
927   carbon in temperate ecosystems, Agroecol. Sustain. Food Syst., 42, 724–750,
928   https://doi.org/10.1080/21683565.2017.1423529, 2018.

929 Marliah, A., Anhar, A., and Hayati, E.: Combine organic and inorganic fertilizer increases yield of cucumber
930   (Cucumis sativus L.), in: IOP Conf. Ser.: Earth Environ. Sci., 425, 012075, https://doi.org/10.1088/1755-
931   1315/425/1/012075, 2020.

932 Micucci, G., Sgouridis, F., McNamara, N. P., Krause, S., Lynch, I., Roos, F., Well, R., and Ullah, S.: The ^{15}N-Gas
933   flux method for quantifying denitrification in soil: Current progress and future directions, Soil Biol.
934   Biochem., 184, 109108, https://doi.org/10.1016/j.soilbio.2023.109108, 2023.

935 Morley, N. J., Richardson, D. J., and Baggs, E. M.: Substrate induced denitrification over or underestimates shifts
936   in soil N_2/N_2O ratios, PLoS ONE, 9, e108144, https://doi.org/10.1371/journal.pone.0108144, 2014.

937 Msimbira, L. A., and Smith, D. L.: The roles of plant growth promoting microbes in enhancing plant tolerance to
938   acidity and alkalinity stresses, Front. Sustain. Food Syst., 4, 106, https://doi.org/10.3389/fsufs.2020.00106,
939   2020.

940 Mulvaney, R. L., and Boast, C. W.: Equations for determination of nitrogen-15 labelled dinitrogen and nitrous
941   oxide by mass spectrometry, Soil Sci. Soc. Am. J., 50, 360–363,
942   https://doi.org/10.2136/sssaj1986.03615995005000020021x, 1986.

Ni, K., Köster, J. R., Seidel, A., and Pacholski, A.: Field measurement of ammonia emissions after nitrogen fertilization—A comparison between micrometeorological and chamber methods, Eur. J. Agron., 71, 115–122, https://doi.org/10.1016/j.eja.2015.09.006, 2015.

Nigon, L. L.: Integrating enhanced-efficiency fertilizers in 4R nutrient management, Crops Soils, 57, 32–37, https://doi.org/10.1002/crso.20329, 2024.

Nyameasem, J. K., Zutz, M., Kluß, C., Huf, M. T., Essich, C., Buchen-Tschiskale, C., Ruser, R., Flessa, H., Olfs, H.-W., Taube, F., and Reinsch, T.: Impact of cattle slurry application methods on ammonia losses and grassland nitrogen use efficiency, Environ. Pollut., 315, 120302, https://doi.org/10.1016/j.envpol.2022.120302, 2022.

Paul, J. W., and Beauchamp, E. G.: Availability of manure slurry ammonium for corn using 15N-labelled $(NH4)_2SO_4$, Can. J. Soil Sci., 75, 35–42, 1995.

Pearsons, K. A., Omondi, E. C., Zinati, G., Smith, A., and Rui, Y.: A tale of two systems: Does reducing tillage affect soil health differently in long-term, side-by-side conventional and organic agricultural systems?, Soil Tillage Res., 226, 105562, https://doi.org/10.1016/j.still.2022.105562, 2023.

Peng, G., Zhang, T., Lei, X., Cui, X., Lu, Y., Fan, P., Long, S., Huang, J., Gao, J., and Zhang, Z.: Improvement of soil fertility and rice yield after long-term application of cow manure combined with inorganic fertilizers, J. Integr. Agric., https://doi.org/10.1016/j.jia.2023.02.037, 2023.

Pomoni, D. I., Koukou, M. K., Vrachopoulos, M. G., and Vasiliadis, L.: A review of hydroponics and conventional agriculture based on energy and water consumption, environmental impact, and land use, Energies, 16, 1690, https://doi.org/10.3390/en16041690, 2023.

Quan, Z., Li, S., Zhang, X., Zhu, F., Li, P., Sheng, R., Chen, X., Zhang, L.-M., He, J.-Z., and Wei, W.: Fertilizer nitrogen use efficiency and fates in maize cropping systems across China: Field 15N tracer studies, Soil Tillage Res., 197, 104498, 2020.

Ramirez, K. S., Craine, J. M., and Fierer, N.: Consistent effects of nitrogen amendments on soil microbial communities and processes across biomes, Glob. Chang. Biol., 18, 1918–1927, https://doi.org/10.1111/j.1365-2486.2012.02639.x, 2012.

Reza, M. S., and Sabau, G.: Impact of climate change on crop production and food security in Newfoundland and Labrador, Canada, J. Agric. Food Res., 8, 100405, https://doi.org/10.1016/j.jafr.2022.100405, 2022.

Rohe, L., Apelt, B., Vogel, H.-J., Well, R., Wu, G.-M., and Schlüter, S.: Denitrification in soil as a function of oxygen availability at the microscale, Biogeosciences, 18, 1185–1201, https://doi.org/10.5194/bg-18-1185-2021, 2021.

Russenes, A. L., Korsaeth, A., Bakken, L. R., and Dörsch, P.: Spatial variation in soil pH controls off-season N_2O emission in an agricultural soil, Soil Biol. Biochem., 99, 36–46, https://doi.org/10.1016/j.soilbio.2016.04.019, 2016.

Samad, M. S., Bakken, L. R., Nadeem, S., Clough, T. J., de Klein, C. A. M., Richards, K. G., Lanigan, G. J., and Morales, S. E.: High-resolution denitrification kinetics in pasture soils link N_2O emissions to pH, and denitrification to C mineralization, PLoS ONE, 11, e0151713, https://doi.org/10.1371/journal.pone.0151713, 2016.

Scheer, C., Wassmann, R., Butterbach-Bahl, K., Lamers, J. P. A., and Martius, C.: The role of N_2O and N_2 emissions in nitrogen cycling in agroecosystems, Agric. Ecosyst. Environ., 293, 106848, https://doi.org/10.1016/j.agee.2020.106848, 2020.

984 Senbayram, M., Chen, R., Budai, A., Bakken, L., and Dittert, K.: N_2O emission and the $N_2O/(N_2O+N_2)$ product
985   ratio of denitrification as controlled by available carbon substrates and nitrate concentrations, Agric. Ecosyst.
986   Environ., 147, 4–12, https://doi.org/10.1016/j.agee.2011.06.022, 2012.

987 Sherman, J., Young, E., Jokela, W., and Kieke, B.: Manure application timing and incorporation effects on
988   ammonia and greenhouse gas emissions in corn, Agriculture, 12, 1952,
989   https://doi.org/10.3390/agriculture12111952, 2022.

990 Smith, K. A., and Arah, J. R. M.: Losses of nitrogen by denitrification and emissions of nitrogen oxides from soils,
991   Soil Res., 28, 95–120, 1990.

992 Snyder, C. S.: Enhanced nitrogen fertiliser technologies support the '4R' concept to optimise crop production and
993   minimise environmental losses, Soil Res., 55, 463–472, https://doi.org/10.1071/sr16335, 2017.

994 Sommer, S. G., and Hutchings, N. J.: Ammonia emission from field applied manure and its reduction, Eur. J.
995   Agron., 15, 1–15, https://doi.org/10.1016/S1161-0301(01)00112-5, 2001.

996 Sommer, S. G., Christensen, B. T., Nielsen, N. E., and Schjorring, J. K.: Ammonia volatilization during storage of
997   cattle and pig slurry: Effect of surface cover, J. Agric. Sci., 117, 91–100,
998   https://doi.org/10.1017/S0021859600066206, 1993.

999 Sørensen, P., and Thomsen, I. K.: Separation of pig slurry and plant utilization and loss of nitrogen-15-labeled
1000   slurry nitrogen, Soil Sci. Soc. Am. J., 69, 1644–1651, https://doi.org/10.2136/sssaj2004.0365, 2005.

1001 Spott, O., Russow, R., Apelt, B., and Stange, C. F.: A 15N-aided artificial atmosphere gas flow technique for
1002   online determination of soil N_2 release using the zeolite Köstrolith SX6®, *Rapid Commun. Mass Spectrom.*,
1003   20, 3267–3274, https://doi.org/10.1002/rcm.2722, 2006.

1004 Stevens, R. J., and Laughlin, R. J.: Lowering the detection limit for dinitrogen using the enrichment of nitrous
1005   oxide, Soil Biol. Biochem., 33, 1287–1289, https://doi.org/10.1016/S0038-0717(01)00036-0, 2001a.

1006 Stevens, R. J., and Laughlin, R. J.: Cattle slurry affects nitrous oxide and dinitrogen emissions from fertilizer
1007   nitrate, Soil Sci. Soc. Am. J., 65, 1307–1314, https://doi.org/10.2136/sssaj2001.6541307x, 2001b.

1008 Surekha, K., Kumar, R. M., Nagendra, V., Sailaja, N., and Satyanarayana, T.: 4R nitrogen management for
1009   sustainable rice production, Better Crop, 10, 16–19, 2016.

1010 Tang, H., Xiao, X., Tang, W., Li, C., Wang, K., Li, W., Cheng, K., and Pan, X.: Long-term effects of NPK
1011   fertilizers and organic manures on soil organic carbon and carbon management index under a double-
1012   cropping rice system in Southern China, Soil Tillage Res., 49, 1976–1989,
1013   https://doi.org/10.1016/j.still.2016.10.005, 2018.

1014 Thompson, R. B., Ryden, J. C., and Lockyer, D. R.: Fate of nitrogen in cattle slurry following surface application
1015   or injection to grassland, J. Soil Sci., 38, 689–700, https://doi.org/10.1111/j.1365-2389.1987.tb02166.x,
1016   1987.

1017 Van Es, H. M., Sogbedji, J. M., and Schindelbeck, R. R.: Effect of manure application timing, crop, and soil type
1018   on nitrate leaching, J. Environ. Qual., 35, 670–679, https://doi.org/10.2134/jeq2005.0192, 2006.

1019 Vanden Heuvel, R. M., Mulvaney, R. L., and Hoeft, R. G.: Evaluation of nitrogen-15 tracer techniques for direct
1020   measurement of denitrification in soil: II. Simulation studies, Soil Sci. Soc. Am. J., 52, 1322–1326, 1988.

1021 Wang, C., Amon, B., Schulz, K., and Mehdi, B.: Factors that influence nitrous oxide emissions from agricultural
1022   soils as well as their representation in simulation models: a review, Agronomy, 11, 770,
1023   https://doi.org/10.3390/agronomy11040770, 2021.

Well, R., Burkart, S., Giesemann, A., Grosz, B., Köster, J. R., and Lewicka-Szczebak, D.: Improvement of the 15N gas flux method for in situ measurement of soil denitrification and its product stoichiometry, Rapid Commun. Mass Spectrom., 33, 437–448, https://doi.org/10.1002/rcm.8363, 2019.

Well, R., Maier, M., and Müller, C.: Underestimation of denitrification rates from field application of the 15N gas flux method and its correction by gas diffusion modelling, Biogeosciences, 15, 1165–1180, https://doi.org/10.5194/bg-15-1165-2018, 2018.

Wangari, G., Mwanake, R. M., Houska, T., Kraus, D., Kikowatz, H.-M., Wolf, B., Gettel, G. M., Breuer, L., Ambus, P., Kiese, R., and Butterbach-Bahl, K.: Spatial-temporal patterns of foliar and bulk soil 15N isotopic signatures across a heterogeneous landscape: Linkages to soil N status, nitrate leaching, and N_2O fluxes, Soil Biol. Biochem., 199, 109609, https://doi.org/10.1016/j.soilbio.2024.109609, 2024.

Winkhart, F., Mösl, T., Schmid, H., and Hülsbergen, K.-J.: Effects of organic maize cropping systems on nitrogen balances and nitrous oxide emissions, Agriculture, 12, 907, https://doi.org/10.3390/agriculture12070907, 2022.

Xu, C., Zhu, H., Liu, H., Ji, C., Yuan, J., Li, G., Wang, J., and Zhang, Y.: Patterns of crop-specific fertilizer-nitrogen losses and opportunities for sustainable mitigation: A quantitative overview of 15N-tracing studies, Soil Ecol. Lett., 6, 230206, https://doi.org/10.1007/s42832-023-0206-2, 2024.

Yankelzon, I., Schilling, L., Butterbach-Bahl, K., Gasche, R., Han, J., Hartl, L., Kepp, J., Matson, A., Ostler, U., Scheer, C., Schneider, K., Tenspolde, A., Well, R., Wolf, B., Wrage-Moennig, N., and Dannenmann, M.: Lysimeter-based full fertilizer 15N balances corroborate direct dinitrogen emission measurements using the 15N gas flow method, Biol. Fertil. Soils, https://doi.org/10.1007/s00374-024-01801-4, 2024a.

Yankelzon, I., Willibald, G., Dannenmann, M., Malique, F., Ostler, U., Scheer, C., and Butterbach-Bahl, K.: A new incubation system to simultaneously measure N_2 as well as N_2O and CO_2 fluxes from plant-soil mesocosms, Biol. Fertil. Soils, 1–19, https://doi.org/10.1007/s00374-024-01809-w, 2024b.

Zhang, J. B., Zhu, T. B., Cai, Z. C., Qin, S. W., and Müller, C.: Effects of long-term repeated mineral and organic fertilizer applications on soil nitrogen transformations, Eur. J. Soil Sci., 63, 75–85, https://doi.org/10.1111/j.1365-2389.2011.01410.x, 2012.

Zhang, Y., Zhao, J., Huang, X., Cheng, Y., Cai, Z., Zhang, J., and Müller, C.: Microbial pathways account for the pH effect on soil N_2O production, Eur. J. Soil Biol., 106, 103337, https://doi.org/10.1016/j.ejsobi.2021.103337, 2021.

Zhou, M., Zhu, B., Brüggemann, N., Dannenmann, M., Wang, Y., and Butterbach-Bahl, K.: Sustaining crop productivity while reducing environmental nitrogen losses in the subtropical wheat-maize cropping systems: A comprehensive case study of nitrogen cycling and balance, Agric. Ecosyst. Environ., 231, 1–14, https://doi.org/10.1016/j.agee.2016.06.022, 2016.

Zistl-Schlingmann, M., Kwatcho Kengdo, S., Kiese, R., and Dannenmann, M.: Management intensity controls nitrogen-use-efficiency and flows in grasslands—a 15N tracing experiment, Agronomy, 10, 606, https://doi.org/10.3390/agronomy10040606, 2020.