Effect of preceding integrated and organic farming on ^{15}N recovery and the N balance, including emissions of NH_3, N_2O, N_2 and leaching of NO_3^-

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Abstract

Ecological intensification strategies in agriculture, including organic fertilization and diversified crop rotations, 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 ± 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 ± 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 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 within 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 ^{15}N gas flux method (^{15}NGF) is well-established for quantifying N_2 losses, and considering 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 assesses how preceding management effects N balance and fertilizer N allocation by comparing sites with different management history. On the organic farming site (OF), legumes had been cultivated frequently, and external N was only occasionally added in the form of cattle slurry, whereas a combination of synthetic and organic fertilizers had been used on the integrated farming site (IF).

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 management history and initial soil sampling

The study was conducted in 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 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, aiming at increasing soil organic carbon (SOC) levels (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 [cm]	pH	SOC [%]	TN [%]	C:N [-]	BD [g cm^{-3}]	Texture [%]		
							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_2O 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.
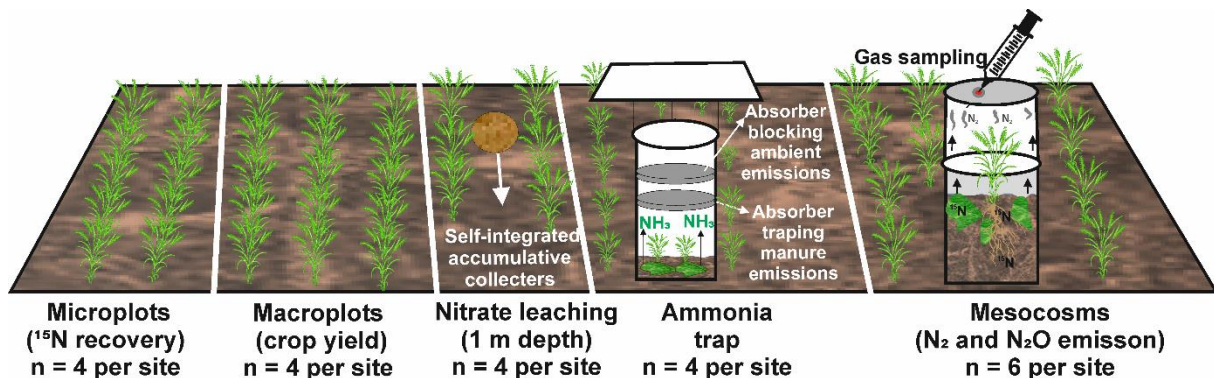
Figure 1: Schematic representation of the experimental units and associated measurements to assess the N balance and fertilizer N allocation. Microplots were established to assess ^{15}N recovery from soil and plants, macroplots to measure crop yield, self-integrated accumulative collectors (at 1-meter depth) to monitor nitrate leaching, semi open chambers with absorbers for ammonia, and mesocosms to quantify gaseous N emissions (N_2 and N_2O)

153   through gas sampling. The experiment included 4 replicates per site for microplots, macroplots, nitrate leaching,

154   and ammonia traps, as well as 6 replicates per site for mesocosms.

2.4 Crop biomass yield

156   At each harvest, aboveground biomass was harvested in each of the macroplots by harvesting the whole

157   area using a plot harvester (Haldrup, 1.5 m harvesting width). Biomass dry matter was determined in subsamples

158   from each macroplot by oven-drying at 60 °C for several days until a stable mass was reached. To determine total

159   biomass N content, samples were milled and analyses using an elemental analyser (DIN/ISO 13878:1998,

160   Elementar Analyser system GmbH, Langenselbold, Germany).

2.5 Determination of dinitrogen emissions from soil-plant mesocosms

162   To determine N_2 emission, the ^{15}N gas flux method (^{15}NGF) was applied where N_2 emission is calculated

163   from the isotopic compositions of N_2 in atmospheric background and in the headspace of a soil chamber that

164   previously had received ^{15}N labelled fertilizer (Hauck, 1958; Mulvaney et al., 1986; Spott et al., 2006). The

165   mesocosms were managed in the same manner as the field plots. To this end, six mesocosms from OF and IF were

166   collected in steel cylinders (0.26 m height and 0.16 m diameter) a month before fertilization and buried in a trench

167   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-

168   urea to obtain an enrichment of 85at% (Table A2). The ^{15}N-labelled cattle slurry was applied to each mesocosm

169   during green rye cultivation, by simulating a drag hose application. In the silage maize cultivation period, the target

170   application rate was 74.8 kg N ha^{-1}, and labelled slurry was applied by mimicking drag hose application, and

171   incorporated into the soil using manual tillage, following the legal regulation in Germany. Subsequently, corn

172   seeds were planted. In the same way, labelled cattle slurry, with 85at% was prepared for the perennial ryegrass

173   first (86 kg N ha^{-1}) and second (65 kg N ha^{-1}) application and applied by simulating a drag hose application.

174   For the N_2 gas measurements, 0.16 m high chambers were tightly affixed to the mesocosms using a steel

175   tension clasp, and gaseous samples were collected using a syringe directly after fixing the chamber and after 2

176   hours chamber closure time. The syringe was flushed three times with chamber headspace air and 20 ml was

177   transferred to a pre-evacuated 12 ml exetainer (Model 837W, Labco Limited, United Kingdom), that had been

178   flushed with helium gas three times. Gas samples were collected daily from day D1 to D7, and additionally on

179   days D9, D11, D13, D21, and D28, following fertilization.

180   Gas samples were analyzed using an isotope ratio mass spectrometer (Isoprime PrecisION, Elementar

181   UK Ltd., Stockport, UK), coupled to an isoFLOW GHG GasBench (Elementar UK Ltd.). This setup allows for

182   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-

183   N_2O after cryogenic pre-concentration of N_2O of the remaining vial content by measurement of m/z 44, 45, and

184   46. Gas handling and preparation followed the protocol described in Arah, (1997); Stevens and Laughlin, (2001b);

185   Spott et al. (2006). In-run uncertainty, determined from repeated analysis of reference gases for isotope ratios r29,

186   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

187   fertilizer added to the mesocosms (85% at), this standard deviation values relate to 87 µg N_2-N m^{-2} h^{-1}. The fraction

188   of fertilizer-derived N_2 was determined by calculating the ratio between the ^{15}N atom excess percentage of the

189   emitted N_2 and the ^{15}N atom excess percentage of the applied N fertilizer, following the procedure given in

190   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).

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

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

268   **2.11 Data processing and statistical analysis**

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

278   **3.   Results**

279   **3.1 Crop yield**

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

285   **Table 2:** Average aboveground biomass (AGB) and aboveground biomass-N (AGB-N) with standard error (SE), expressed as
286   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 ± 0.21 b	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

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

288   **3.2 Dinitrogen (N_2) and nitrous oxide (N_2O) emissions**

289   Following ^{15}N fertilizer applications, increased emissions of total N_2 and fertilizer-derived N_2 were
290   observed within two weeks, with fertilizer-derived emissions accounting for 70%, 63% and 76% for OF and
291   80%, 55%, 70% for IF of the total emissions during the green rye, silage maize and perennial ryegrass cultivation

period, respectively. There was no clear difference in total N_2 flux levels between OF and IF, and flux rates ranged from close to 0 to 1065 µg N m^{-2} h^{-1} (Fig. 2a-q, Fig. B1). In contrast, fluxes of total N_2O and fertilizer-derived N_2O for OF were higher after slurry application in the green rye and perennial ryegrass cultivation period, compared to silage maize cultivation period. The fertilizer-derived N_2O emissions accounted for 33, 38, and 33% for OF and 29, 37, and 38% for IF in the green rye, silage maize and perennial ryegrass cultivation period, respectively. However, no clear pattern of elevated total or fertilizer-derived N_2O flux levels was observed for silage maize. Flux rates for total N_2O were low with emissions peaks of approximately 40 µg N m^{-2} h^{-1} in the green rye and silage maize periods and 94 µg N m^{-2} h^{-1} after the first slurry application to perennial ryegrass (Fig. 2c-s), respectively.

The ratio of total N_2O : (N_2+N_2O) showed a similar progression over time for both sites, ranging from 0.01 to 0.45. Since the N_2O and N_2 flux levels were close to the detection limit from day 11 after fertilization, only values before this period were considered in green rye and silage and silage maize cultivation period (Fig. 2e, k). On average, across crops and sites, the N_2O : (N_2+N_2O) was 0.14, and the median amounted to 0.03.
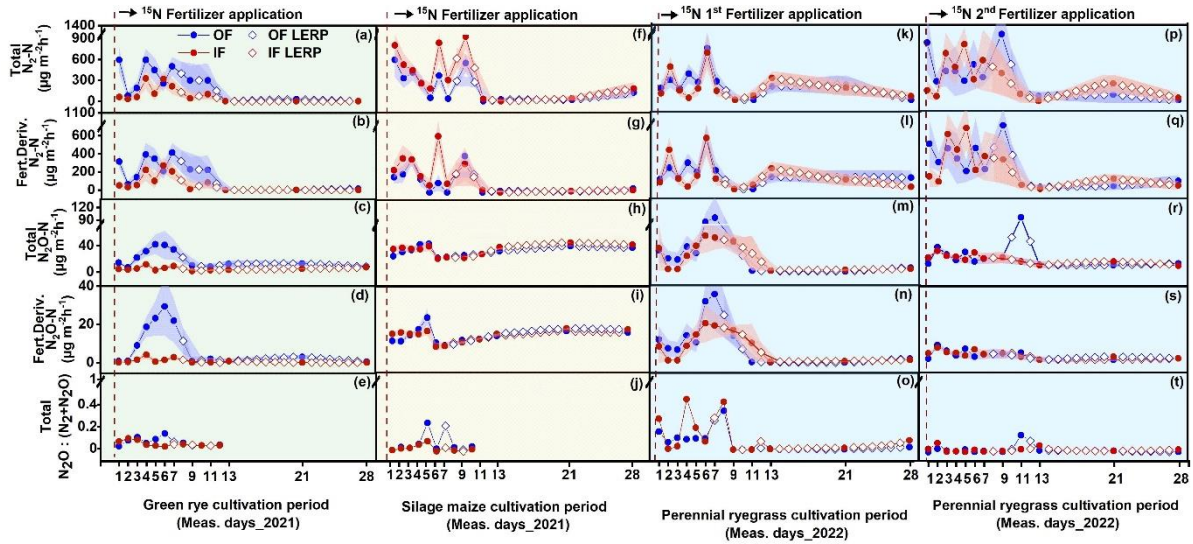
Figure 2: Results of the ^{15}N gas flux method using mesocosms. From top to bottom, the panels show total N_2 flux, fertilizer-derived (fert. deriv.) N_2 flux, total N_2O flux, fertilizer-derived (fert. deriv.) N_2O flux, total N_2O : (N_2+N_2O) ratio, following fertilizer application. Measurement periods 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. 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. 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. 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. 3b). However, fertilizer-derived N_2O emissions were not significantly different and amounted to 0.10 kg N ha^{-1} (Fig. 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. 3c). Similarly, cumulative total (IF: OF: 0.11 kg N ha^{-1}, Fig. 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. 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. 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. 3e).
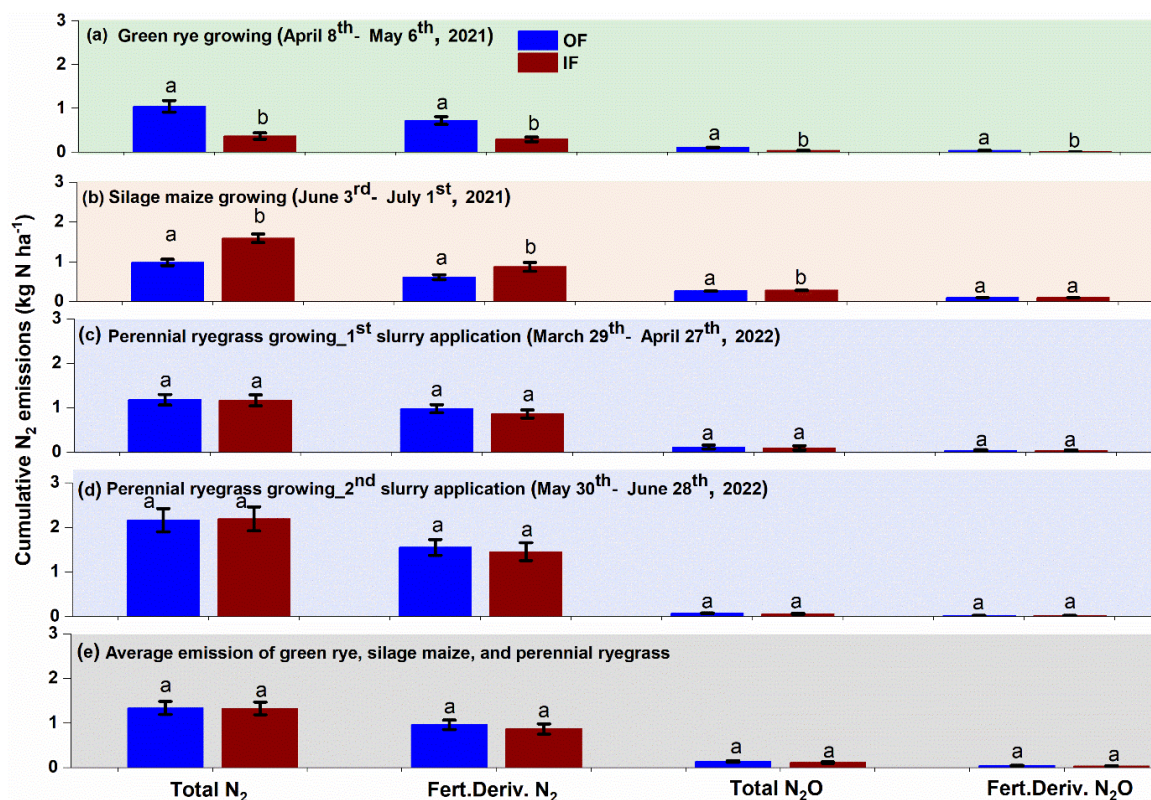
Figure 3: Cumulative total and fertilizer-derived N_2 and N_2O emission for a measurement period of one month during (a) green rye cultivation period, (b) silage maize cultivation period, (c) perennial ryegrass 1st ^{15}N-labelled cattle slurry application (d) perennial ryegrass 2nd ^{15}N-labelled cattle 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. 4a-c, Table A3). The highest average recovery for all cultivation periods was typically found in soil (OF: 44%, IF: 46%, Fig. 4d), followed with slightly lower average recovery in plant above and belowground biomass (OF: 33%, IF: 39%, Fig. 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 35% for IF, while during silage maize, it was 59% for IF and 44% for OF (Fig. 4a, b). For perennial ryegrass cultivation the soil ^{15}N recovery was slightly higher for IF (44%) than for OF (36% Fig. 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. 4a-c). Recovery from roots was nearly identical for both sites (OF, IF, Fig. 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. 4a-c). Similarly, recovery in N_2O was below 0.1% for both OF and IF across all cultivation periods (Fig. 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. 4a-d).

The total ^{15}N recovery considering soil, plant, and measured N loss components, was comparable in the green rye (OF: 87%, IF: 84%, Fig. 4a), silage maize (OF: 86%, IF: 95%, Fig. 4b), and perennial ryegrass (OF: 84% and IF: 100%; Fig. 4c) cultivation period. On average ^{15}N recovery for all cultivation periods was 85% for OF and 93% for IF (Fig. 4d). The unrecovered portion of ^{15}N 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 ^{15}N 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.
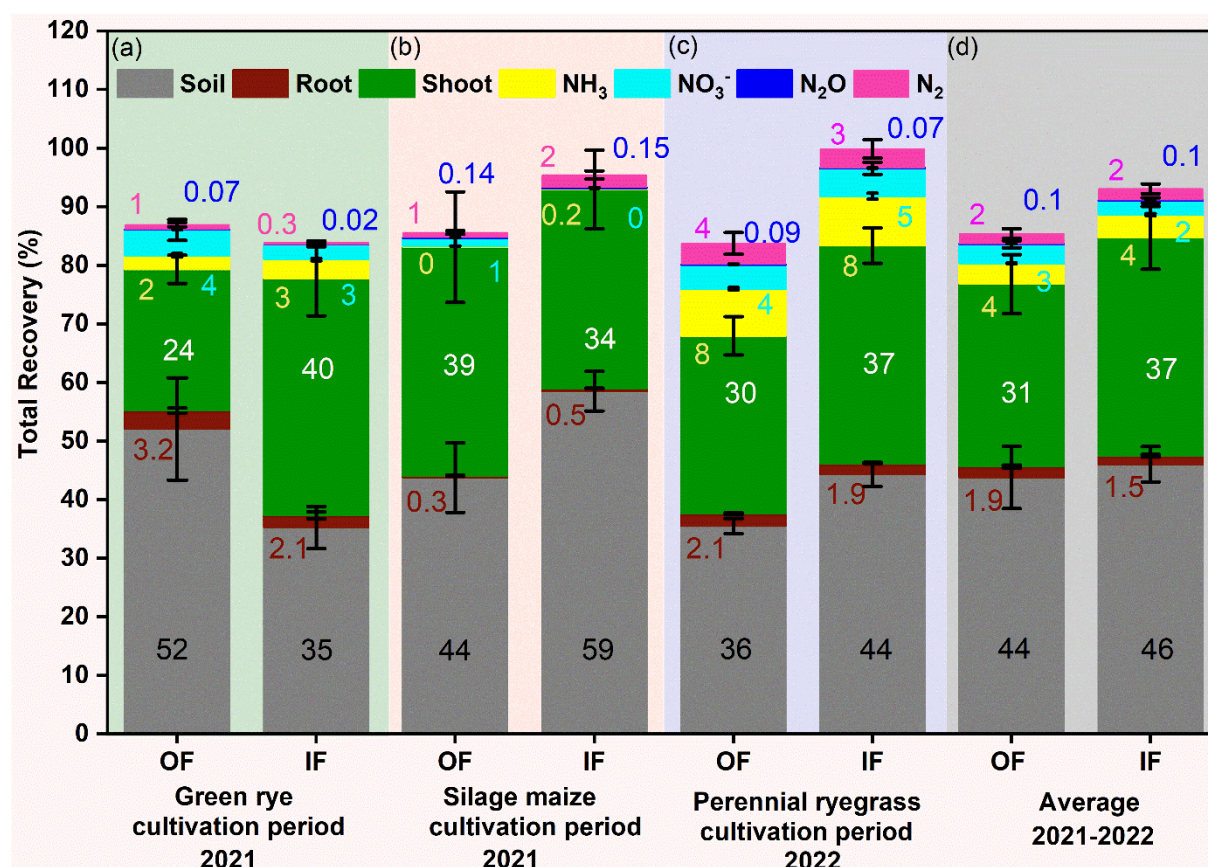
Figure 4: Recovery of fertilizer ^{15}N in soil (0–90 cm depth; grey), aboveground biomass (green), root (vine), ammonia (NH_3, yellow), nitrous oxide (N_2O, blue), dinitrogen (N_2, magenta) and nitrate leaching (NO_3^-, cyan) for (a) green rye, (b) silage maize, (c) perennial ryegrass cultivation periods 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 cattle slurry was applied as top dressing in OF and IF (Fig. 5), while for silage maize 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. 5a-c), calculated based on data from the German Environmental Protection Agency. In accordance with the ^{15}N recovery (section 3.3), IF exhibited significantly higher plant N uptake (98 vs. 57 kg N ha^{-1} in OF, Fig. 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. 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. 5b). Since total N losses were minor compared to yield for both sites (2 kg N ha^{-1}, Fig. 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. 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. 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. 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. 5d), compared to IF, which had a small negative value (-8 kg N ha^{-1}; Fig. 5d).
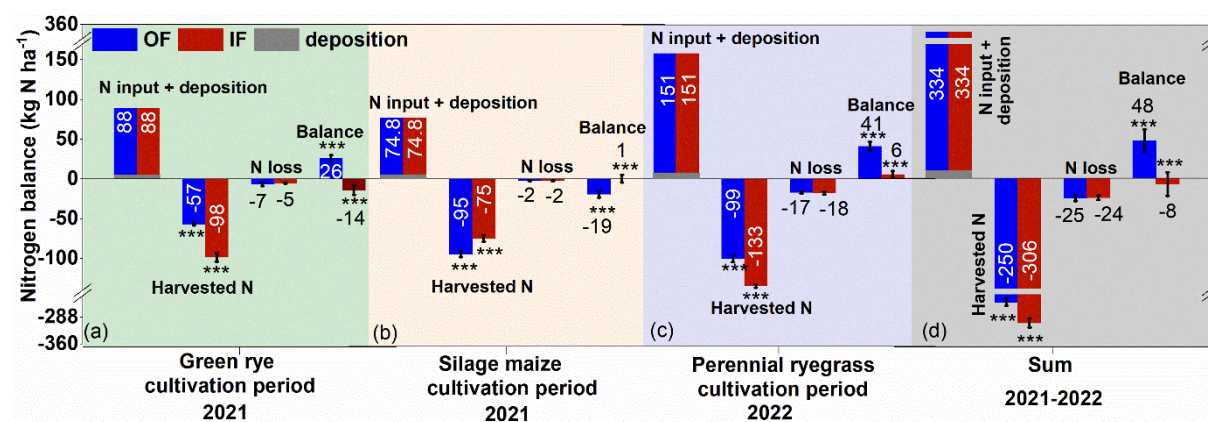
Figure 5: Cultivation period N balance for (a) green rye (b) silage maize (c) perennial ryegrass and (d) cumulative N balance for the years 2021 and 2022. N loss combines NH_3, N_2, N_2O and NO_3^- leaching. The bars with whiskers represent the mean ± standard error values. Organic farming (OF), integrated 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. 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. 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

436 and N cycling dynamics may have adjusted to the unified management, leading to more comparable emissions.
437 Additionally, the relatively high N input applied during the experimental period compared to the historically low
438 N input particularly under OF may have intensified the denitrification response, particularly in the early cropping
439 periods. Thus, the complex interplay of historical management, time since management harmonization, crop type,
440 and soil N cycle processes like plant N demand, the mineralization-immobilization cycle controlling N availability,
441 and the effect of environmental conditions on the denitrification process (Butterbach-Bahl et al., 2002; Chen et al.,
442 2019) led to variable N_2 emissions from both sites.

443   As mentioned above, N_2O and N_2 fluxes are closely linked as both are produced and consumed in the
444 process of denitrification, converting nitrate to nitrite, nitric oxide, nitrous oxide, and finally to N_2 (Butterbach-
445 Bahl and Dannenmann, 2011). For this reason, we expected to observe an increase in both N_2O and N_2 emission
446 after fertilizer application as was reported by Häfner et al. (2021), Herr et al. (2019) and Bizimana et al. (2022). In
447 this study, we observed a distinct N_2 emission following every manure application event, but the N_2O emission
448 hardly increased for IF during green rye cultivation and for OF and IF during silage maize cultivation (Fig. 3,
449 Table A3). The other N_2O emission pulses were moderate with emission rates below 140 µg m^{-2} h^{-1} (Fig. 2). The
450 lack of N_2O peaks after fertilizer application was also observed by Buchen-Tschiskale et al. (2023) and
451 Dannenmann et al. (2024) and can be explained by conditions conducive to full denitrification, i.e., low oxygen
452 levels, high soil moisture, good availability of labile carbon sources and high levels of mineral N (Smith and Arah,
453 1990; Morley et al., 2014; Rohe et al., 2021; Wang et al., 2021). Such conditions can result from the application
454 of manure, which is a mixture of water, organic substances, and ammonium.

455   With regard to the magnitude of N_2 emissions, few field studies have measured N_2 emissions from cattle
456 slurry application. The level of cumulative N_2 emission in this study was between 0.36 and 2.19 kg N ha^{-1} (Fig. 3)
457 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)
458 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
459 120 kg N ha^{-1} in form of cattle slurry (Stevens and Laughlin, 2001a). Despite the amount of N_2 losses quantified
460 in this study agreeing with the past studies, the direct field measurement of N_2 fluxes in addition to all other
461 relevant loss pathways did not result in a closed ^{15}N balance, with the average imbalance being 15 and 7 kg N ha^{-1}
462 for OF and IF, respectively. While some part of the unrecovered ^{15}N may be due to an underestimation of NH_3
463 loss, N_2 emissions may be underestimated as well (Yankelzon et al., 2024a). Heterogeneous ^{15}N distribution in the
464 microplot soil resulting from surface or slit application of the slurry and $^{15}N_2$ diffusion and storage in subsoil layers
465 contributes to the underestimation of N_2 flux rates. These complications associated with ^{15}N labelling were
466 discussed in previous studies which indicate that fluxes may be underestimated by up to 30-50% (Vanden Heuvel
467 et al., 1988; Arah, 1997; Well et al., 2018; Well et al., 2019; Friedl et al., 2020; Micucci et al., 2023; Dannenmann
468 et al., 2024). Even assuming a 50% underestimation, N_2 emissions after fertilization are in the range of 2 kg N ha^{-1}
469 , indicating that the method-inherent potential underestimation alone cannot close the N-balance. However, the
470 short coverage of the direct N_2 measurements, which is due to the short period of time during which the isotopic
471 enrichment of the N pool subject to denitrification is sufficiently high so that the N_2 flux can be detected, may be
472 an additional source for underestimation. Consequently, we cannot exclude that additional ^{15}N in N_2 is emitted in
473 the months following fertilizer application, particularly during rewetting events or towards the end of the growing
474 season when plants decrease their water uptake and water content increases, as observed by Almaraz et al. (2024).

475   At the same time, however, it must be recognized that the spatial variability of soil properties,
476 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. Thus, the apparent mismatch in the N budget may also be due to measurement uncertainties rather than actual N losses.

The $N_2O : (N_2 + N_2O)$ ratio in this study ranged from 0.01 to 0.45 across both sites using different slurry application techniques. This aligns with the findings of Fangueiro et al. (2008) who incorporated cattle slurry fractions (solid and liquid) into the soil, observing that the $N_2O : (N_2 + N_2O)$ ratio varied between 0.32 and 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 cattle slurry application (97 kg N ha^{-1}) under conditions favouring full denitrification, yielding a low $N_2O : (N_2 + N_2O)$ ratio of 0.03.

4.2 Nitrate leaching

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

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

Overall, it must be noted that the exact travel time of NO_3^- at the site is unknown and could exceed the duration of a cultivation period. In addition, total nitrate instead of ^{15}N-NO_3^- was determined in this study. Consequently, the observed nitrate leaching for the respective cultivation periods may include some contributions from preceding fertilizer applications, not fully represent the leaching rate due to the most recent fertilizer application and overestimate the contribution of NO_3^- leaching to the ^{15}N balance.

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

In a single cultivation period, the recovery of ^{15}N in plant biomass ranged from 24-39% for OF and 34-40% for IF, with averages of 31% for OF and 37% for IF (Fig. 4a-d). Soil ^{15}N recovery over a single cultivation period showed a range of 36-52% for OF and 35-59% for IF, averaging 44% for OF and 46% for IF across all cultivation periods (Fig. 4a-d). To our knowledge, there are only a few studies that used ^{15}N-labeled cattle slurry in field experiments on arable land, such as Jensen et al. (2000), who recovered 32% of ^{15}N in the aboveground biomass and 45% in the soil for winter wheat. Similarly, Paul et al. (1995) reported an average recovery of 43% of applied ^{15}N-labeled cattle slurry in above- and belowground biomass of corn. In recent work, Frick et al. (2023) applied ^{15}N-labeled cattle slurry (produced by feeding a heifer ^{15}N-enriched ryegrass hay) to a grass-clover system and observed annual recovery rates of 17-22% in plant biomass and 32-52% in soil. Buchen-Tschiskale et al. (2023) recovered 32-47% in soil, while plant recovery was between 25-33% using different application techniques (drag hose and slit injection) for ^{15}N-labeled cattle slurry to winter wheat, indicating that the recoveries of our study align well with published data.

Total soil and plant recoveries were highest during the silage maize cultivation period, where manure was incorporated into the soil. In contrast, recoveries of soil and plant were lower during the cultivation periods of green rye and perennial ryegrass, where manure application was performed using a drag-hose system. Manure incorporation distributes N in the soil column, reduces surface N concentration and thus reduces NH_3 volatilization and facilitates microbial immobilization (Sørensen and Thomsen, 2005; Lyu et al., 2024), resulting in overall recoveries close to 100%. While this trend was not significant in this study, Buchen-Tschiskale et al. (2023) observed significantly lower NH_3 emissions and a significantly higher soil plus plant recovery (79%) and overall recovery of 99% for slit injection compared to trailing hose application (57 and 78%). Simultaneously, significantly lower NH_3 emissions for the slit injection treatment in Buchen-Tschiskale et al. (2023) and the incorporation in this study must have resulted in higher availability of mineral N for soil microorganisms which could lead to increased emission of N_2O or N_2. However, in both studies, N_2O and N_2 emissions were not significantly higher, demonstrating that reducing NH_3 emissions does not necessarily result in pollution swapping from NH_3 to N_2O or N_2 when microbes efficiently immobilize N and prevent direct stimulation of nitrification and denitrification, which produce N_2O and N_2. In general, there was a trend towards lower recoveries in soil and plant biomass for manure application using a drag-hose which means that other ^{15}N loss pathways were more important. It is well established that the measurement of NH_3 emission in the field has remained a challenge and is associated with large uncertainties (Loubet et al., 2018), suggesting that underestimation of NH_3 emissions could explain this trend. However, since an overall recovery of close to 100% was observed for treatments with NH_3 emissions of 8%, i.e., the perennial ryegrass cultivation period of this study and the slit injection treatment of Buchen-Tschiskale et al. (2023), a significant underestimation of NH_3 emission seems unlikely since the underestimated NH_3 emission would imply a recovery greater than 100%. For the same reason, i.e., observation of approximately 100% recovery for certain treatments, a systematic underestimation of leaching losses is also unlikely. The only measurements that don't cover the whole cultivation period are those of N_2O and N_2, suggesting that underestimation of these N losses due to coverage of measurements of only a fraction of the whole cultivation period could explain the 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 $\mu g\ m^{-2}\ h^{-1}$, which is equivalent to an emission if measured isotope ratios are increased by

556 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,

557 maize and ryegrass cultivation, respectively. Such a background emission together with slightly underestimated

558 NH_3 emissions could explain the unrecovered N_2 losses for IF. Overall recovery for OF was close to or lower than

559 that of IF, suggesting that for the OF, additional N_2 was emitted during the cultivation period, which could be due

560 to more frequent denitrification events caused by higher soil bulk density (Table 1, Luo et al., 2000; Hamonts et

561 al., 2013). Furthermore, large error margins in soil and plant recovery rates, flux measurements, and spatial

562 variation introduce uncertainty, indicating that some of the apparent losses may also be due to measurement

563 limitations rather than actual N loss.

564   However, differences in recovery between OF and IF were not significant for single cultivation periods

565 and the full study period. The different fertilizer application techniques, as well as the different crop types, affect

566 the interannual variability of recovery rates, which complicates the detection of significant differences on the

567 interannual scale. In contrast, this suggests for the different cultivation periods that the management history either

568 didn't affect the allocation of ^{15}N to the different components of the ^{15}N balance, or that the differences are too

569 subtle to be determined compared to the spatial variability of soil properties on this specific site and measurement

570 uncertainty. The highest absolute uncertainties, which eventually also control the overall uncertainty of the

571 recovery were observed for ^{15}N recovery in soil and plants. Though in this study, the area of the microplots was

572 0.225 m², corresponding to approximately 30 kg of soil for the 0-10 cm layer, only few milligrams of finely ground

573 soil are eventually used to determine the ^{15}N enrichment. Consequently, the excavated soil and plants were

574 carefully crushed and homogenized in mixing vessels, but the ball mills usually used to prepare the finely ground

575 material are limited to 15 to 20 g of homogenised material. Since soil aggregates are not entirely destroyed during

576 homogenization, samples transferred to the ball mill may still show distinct variability which conceals significant

577 differences between sites. Obviously, increase of replicate amount or analysis of several subsamples (e.g., of soil

578 of a given layer) could reduce the uncertainty, but only by increasing the already high workload and costs by

579 several factors. For this reason, our study shows that research into protocols aiming at reducing the uncertainty

580 arising from incomplete sample homogenisation to the measurement instrument uncertainty by including for

581 instance mills with much larger sample capacities is required, especially since - to our knowledge - there is no

582 publication on different homogenisation and mixing protocols for ^{15}N balances. Following the same line of

583 argumentation, reduction in the uncertainty of the direct measurement of N balance components, i.e., losses

584 through leaching, NH_3, N_2 and N_2O emissions could help resolving difference more accurately. While recent

585 advancements, such as the improved ^{15}N gas flux method (^{15}NGF+) demonstrate potential for enhanced sensitivity

586 in quantifying N_2 emissions under field conditions (Eckei et al., 2025), such approaches are expensive and

587 technically challenging which will delay their use in studies targeting complete N balances. Since the same applies

588 to significant reduction of uncertainty in determining leaching losses, NH_3 volatilization, and N_2O emissions, it

589 appears like these loss pathways were not markedly influenced by management history given the current

590 measurement frameworks.

591 **4.4 Nitrogen balance and management history**

592   Previous studies showed that intensive agricultural systems often experience substantial N losses through

593 gaseous emissions and NO_3^- leaching, which can match or exceed crop N uptake (Ju et al., 2009; Zhou et al., 2016).

594 In this study, cumulative N losses were not significantly different between OF and IF and can be classified as low

595 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. 5d). 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. 5d). 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 N balance to some degree points towards unaccounted losses. Since positive N balances were observed for OF, where the ^{15}N recovery also showed higher unrecovered shares, we assume that not the full N balance can be attributed to an increase in soil N stocks. In this context, it is noteworthy that crop N export and balance values from different cultivation periods align with findings from other studies (Thompson et al., 1987; Dell et al., 2011; Lin et al., 2016; Duncan et al., 2019), which applied cattle slurry using various techniques, including surface application and incorporation.

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

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

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

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

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

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

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

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

645   the design of resilient and sustainable farming systems.

646   **4.5 Impact of experimental homogenization of management on legacy effects**

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

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

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

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

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

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

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

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

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

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

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

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

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

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

661   of OF and IF on N cycling.

662   **5. Conclusion**

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

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

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

666   farming systems under homogenized management conditions.

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

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

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

670   significantly different, but could only be determined for two weeks after fertilizer application. Both OF and IF

671   practices demonstrated minimal N losses, indicating that both approaches effectively mitigate N leaching and

672   emissions when management is standardized.

673   Integrated farming increased productivity by improving plant N supply through higher soil organic matter

674   and higher rates of mineralization, thereby maintaining a balanced N budget. The positive N balance of organic

675   farming demonstrates that increased N input may result in soil N and C accumulation, potentially improving

676   productivity on the long-term. Consequently, a higher level of organic fertilizer additions or a higher share of
677   residue returns for this specific, legume-cantered N management strategy may be beneficial with regard to
678   productivity and soil fertility. Finally, this study demonstrates that multi-year ^{15}N tracing and N balance studies
679   are powerful tools to quantitatively assessing the environmental and agronomic impacts of different management
680   strategies. Future studies that maintain distinct management systems during the experimental phase would help
681   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

687

688   **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

689

Table A3: Fertilizer ^{15}N input, ^{15}N recovery, and unrecovered amounts in green rye, silage maize, and perennial ryegrass 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_2O	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

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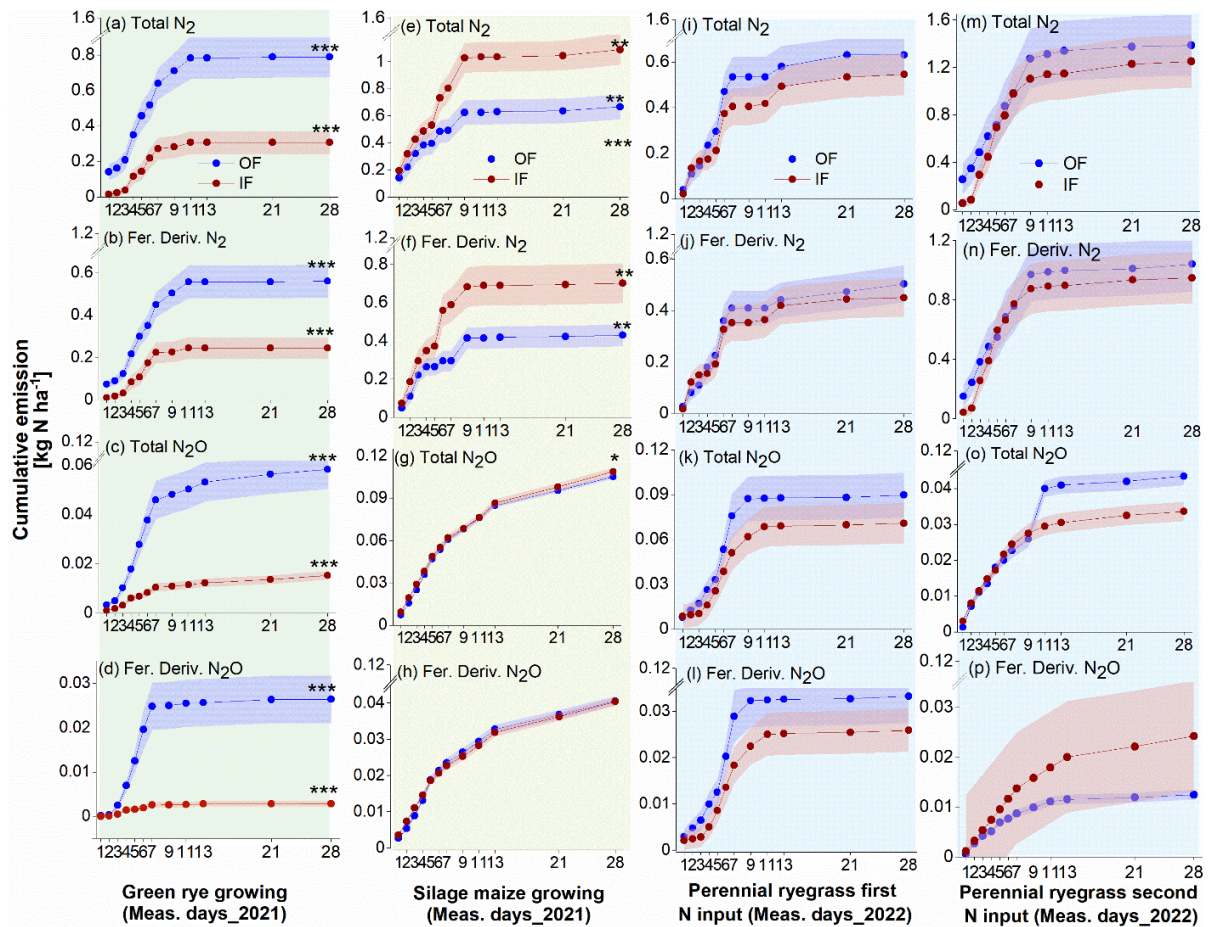
Figure B1: The cumulative total and fertilizer-derived emissions of N_2 and N_2O across different cultivation period. Subplots (a, b, c, and d) depict the total N_2, fertilizer-derived N_2, total N_2O, and fertilizer-derived N_2O emissions, respectively, during the green rye cultivation period and subplots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