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Nitrous Oxide Emissions and Methane Uptake from Organic and Conventionally Managed Arable Crop Rotations on Farms in Northwest Germany

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Abstract: Land-use extensification by shifting from conventional to organic arable farming is often discussed as a measure for reducing greenhouse gas (GHG) emissions from agricultural land. Doubts about the benefits arise when emissions are calculated per product unit, particularly where high yields are possible under conventional management. Among the non-CO_2 GHG emissions, nitrous oxide (N_2O) is the main contributor from arable land and is controlled by soil type, environmental conditions and management. In order to investigate how land-use change from conventional to organic farming would perform under highly productive site conditions in northwest Germany, and how this would affect the important greenhouse gases N_2O and methane (CH_4), an on-farm field research was conducted over two experimental years. Two site-specific organic crop rotations, (i) with 25% legumes (grass + clover–winter wheat–winter rye–oats) and (ii) with 40% legumes (grass + clover–winter wheat–winter rye–spring field peas–winter rye), were compared with (iii) a conventional arable rotation (winter oilseed rape–winter wheat–winter wheat–sugar beet–winter wheat) and two reference systems, (iv) extensive grassland and (v) a beech forest), which were chosen as the baseline. The results showed that organic farming had lower N_2O emissions of 0.7 N_2O–N ha^{-1} $year^{-1}$ than the conventional rotation, with 2.1 kg N_2O–N ha^{-1} $year^{-1}$ ($p < 0.05$), but higher emissions than the extensive grassland (0.3 kg N_2O ha^{-1} $year^{-1}$) and beech forest (0.4 kg N_2O ha^{-1} $year^{-1}$). CH_4 emissions were a negligible part of total GHG emissions (as CO_2 equivalents) in the two arable systems, and considerable uptake of CH_4 from the forest soils showed this was a GHG sink in the first experimental year. Organic systems produced up to 40% lower crop yields, but the emissions per product unit in rotation (iii) was not superior to (ii) during the two experimental years. Thus, arable organic farming showed the ability to produce agricultural commodities with low N_2O emissions per unit area, and no differences in product-related emissions compared with conventional farming. Conventional and organic systems both showed potential for further mitigation of N_2O emissions by controlling the field level nitrogen surplus to a minimum, and by the optimized timing of the removal of the grass–clover ley phase.

Keywords: eco-efficiency; greenhouse gas (GHG) emissions; on-farm research; system comparison; crop yield

1. Introduction

Nitrous oxide (N_2O) is an important greenhouse gas (GHG) in agriculture and has a global warming potential (GWP) 265 times that of carbon dioxide (CO_2) over a 100-year time horizon [1]. Agricultural N_2O emissions are therefore considered to contribute significantly to climate change [2] and, additionally, are involved in the degradation of the protective ozone layer [3,4]. At a global scale, about 62% of atmospheric N_2O emissions are attributed to soils [5], which is the equivalent of 13 Mt N_2O–N annually [6]. Soil-derived N_2O emissions occur mainly as an intermediate stage during the denitrification and nitrification processes [7–11]. Its production is controlled by available nitrogen (N) in the soil and different abiotic factors that stimulate or inhibit biological activity [12]. These factors comprise the availability of easily degradable organic matter [13,14], temperature, pH [15,16] and soil moisture [14,17]. The magnitude of N_2O emissions is often characterized by high spatial and temporal variation [12,18], and in agricultural systems increased use of N fertilization is associated with increased N_2O emissions [19–22]. Thus, in mitigating the causes of global warming associated with agriculture and land use, adopting management changes that can help to reduce N_2O emissions are increasingly important [23].

Agricultural systems can also act as a considerable GHG sink. In addition to carbon sequestration in soils and plants, additional soil GHG consumption can take place by methane (CH_4)-oxidizing bacteria that use CH_4 (GWP of 28) as an energy source. However, in relation to the most important direct non-CO_2 GHG emissions (N_2O and CH_4), the sink strength of soils is comparatively low when expressed in CO_2 equivalents on an area basis [24,25], but its value has to be considered in GHG inventories.

In Germany, agriculture contributes 80% to the total N_2O emissions [26]. One measure proposed by the government for reducing agricultural GHG emissions is to increase the share of organic farming to 20% of the total utilized agricultural area [27]. On the area basis, it appears likely that emissions will be reduced by this measure, as conventional farming systems are usually characterized by high inputs of mineral N fertilizers and, therefore, are often associated with high N losses to the environment [28]. Conversely, on farms managed by organic farming methods, there is a dependency on internal N-supply from biological nitrogen fixation (BNF), which can lead to reduced N_2O emissions because external N inputs are lower [29]. However, this situation may also have the unintended consequence of stimulating considerable N_2O releases from agricultural soils. This can arise due to the high quantities of easy decomposable organic matter coincident with narrow C/N ratios of plant material, creating favorable conditions for N_2O release in soils. Several authors have found increased N_2O emissions after ploughing grass–clover swards [30,31]. Consequently, some authors have concluded that organically managed fields can lead to higher N_2O emissions compared with conventional systems [32]. The prevailing farm management practices could also have diverse effects on N_2O emissions, depending on the types and timing of manure and other types of fertilizer application [33], and the composition and quality of incorporated crop residues [34].

A further important consideration is that, in organic arable farming, crop yields per hectare are often much lower than in conventional systems [35,36]. This is highly relevant because the climate impact of agricultural production is often not only evaluated as emissions per unit of area, but as emissions per product unit [37]. Recent studies have recommended that N_2O emissions from agricultural production should be expressed as a function of dry matter yield, e.g., grain equivalents in arable farming systems [38]. Against this background, it is important to question whether an increased share of farmland used for organic farming would result in reduced emissions per product unit. However, comparative datasets about the environmental impacts of the systems are lacking. Beyond this, there are further questions about the extent of arable agricultural production's contributions to national GHG inventories, in comparison with extensive grassland or reserve systems. For the latter, it is not only the question of the amount of GHG emissions produced, but also the question of the productivity needed to maintain the economic resilience of farmers and ensure food security. In the forestry sector in particular, there are current considerations on how to increase the sector's land-use share as a possible climate change mitigation strategy. However, it can take several years before forests can provide a timber harvest and thereby support livelihoods.

The ongoing public discussion about the efficiency and resilience of future production systems has focused attention on the need for studies beyond academic approaches for measuring N_2O emissions from arable conventional and organic systems, under common and current agricultural management, to examine whether GHG mitigation targets are fulfilled. In this context, a more detailed understanding of N losses is also crucial for further lifecycle impact assessments, as inaccurate prediction of N cycles when comparing organic and conventional production can result in large errors, which often show disadvantages for organic systems [36]. Moreover, typical regional production systems have to be chosen, rather than simple comparisons of identical crop rotations. Organic farming has different requirements for crops, such as greater reliance on N-fixing legumes compared with conventional systems that use external synthetic N, leading to different agricultural commodities and environmental impacts [39].

Accordingly, in this paper, we present results from an on-farm study to address these important questions: (1) Can stockless arable organic farming reduce direct GHG (N_2O and CH_4) emissions from soils per ha and per product unit relative to a conventional stockless arable system, on highly productive sites in northern Germany? (2) What are the main drivers for the important field-level N_2O emissions in the investigated systems, and how they can be improved? (3) To what extent do the examined arable systems behave in terms of GHG emissions compared with other referenced land-use systems (extensive grassland and beech forest)?

2. Materials and Methods

2.1. Research Area

The region where the study was conducted in Schleswig-Holstein, northern Germany, has a hilly terminal-moraine landscape and supports approximately 400,000 ha of agricultural land that is mainly used for intensive arable cropping. The weather is characterized by a temperate oceanic climate. Mean (1981–2010) annual temperature is 8.9 °C and annual precipitation is 737 mm. The region, together with similar areas along the Baltic Sea coast of Denmark, southern Sweden and north-east Germany, is one of the most suitable regions in Europe for cereals, oilseed rape and winter wheat; e.g., yields for winter wheat on commercial farms typically exceed 9.0 t ha^{-1} (based on 6-year average 2008–2013). Organic farming is currently practiced on only 3.7% of the utilized agricultural area in this region. In view of current policy aims to increase land under organic management, we considered on-farm investigations under these favorable climate and soil conditions to be appropriate to answering the question of to what extent a shift from conventional to organic arable farming would affect productivity and GHG emissions from sandy loam soils used for arable cropping.

2.2. On-Farm Research Design

The two-year field study (1st year: October 2012–September 2013; 2nd year: October 2013–September 2014) was conducted on 400 ha land comprised of arable land, beech forest and permanent grassland (53°39′ N, 10°34′ E). All land-use types were arranged completely randomized within this area. The arable land was managed under the common agricultural practice for that area. The study area included two organically managed arable crop rotations and a typical conventional stockless cropping system with a history of more than ten years of continuous management. The crop rotations were established randomly on fields of 5–23 ha, and every crop in each crop rotation was present during both experimental years. There were four replicates for investigations established in each field to provide a representative average for each field. Soils were classified as sandy loam (10.6–11.4% clay, 26.3–29.4% silt and 59.4–63.0% sand in the 0–30 cm topsoil), with dominant soil types being Luvisols, Cambisols and Haplic Stagnosols.

Total precipitation during the experimental years (1st year: 645 mm; 2nd year: 615 mm) was less than the long-term average (737 mm). Compared with the long-term average of 8.9 °C, air temperature during the study period was similar in the first experimental year (8.4 °C) but higher in the second year (10.4 °C). The differences between the two experimental years are also shown by numbers of days with temperatures below freezing point. In the first winter period, there were 62 days with daily average temperatures below 0 °C, whereas only 16 days with sub-zero temperatures were recorded in the second year (Figure 1).

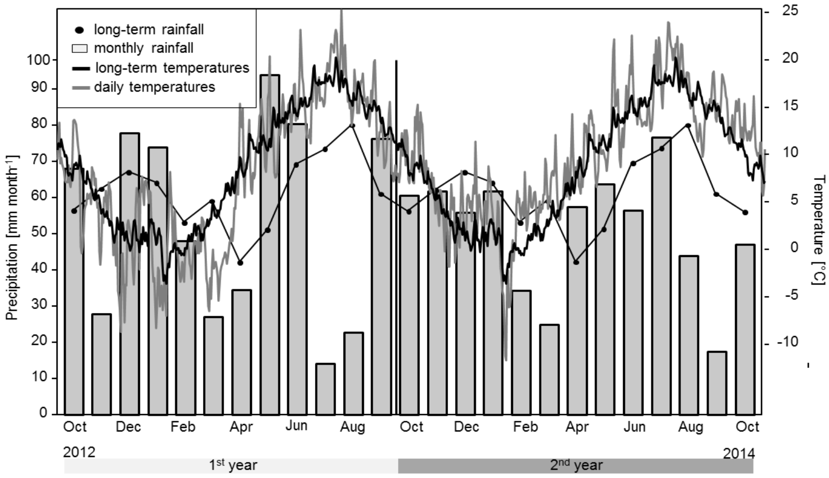

Figure 1. Monthly sum of precipitation in mm month^{-1} (grey bars) compared with the long-term averages (black dots), and daily mean air temperature (grey line) compared with the long-term average (black line) during the experimental years (October 2012–October 2014).

2.3. Crop Rotations, N Inputs and Crop Management

The N intensity of the two tested organic systems was controlled by the proportion of legumes in the crop rotations. We distinguished between "organic-low-BNF" as a four-crop rotation (i. grass–clover (*Trifolium pratense*, / *Lolium multiflorum*), ii. winter-wheat (*Triticum aestivum*, cv. 'Naturastar'), iii. winter-rye (*Secale cereale*, cv. 'Dukato'), iv. oats (*Avena sativa*, cv. 'Ivory')) with a share of 25% legumes; and "organic-semi-BNF" as five-crop rotation (i. grass–clover (*T. pratense*/*L. multiflorum*), ii. winter-wheat (*T. aestivum*, cv. 'Naturastar'), iii. winter-rye (*S. cereale*, cv. 'Dukato'), iv. peas (*Pisum sativum*, cv. 'Santana'), v. winter-rye (*S. cereale*, cv. 'Dukato')) with a share of 40% legumes. The conventional system comprised an intensive five-crop rotation (i. winter oilseed rape (*Brassica napus*, cv. 'Visby')), ii. winter-wheat (*T. aestivum*, cv. 'Tobak'), iii. winter-wheat (*T. aestivum* cv. 'JB Asano'), iv. sugar beets (*Beta vulgaris*, cv. 'Sabrina'), v. winter-wheat (*T. aestivum*, cv. 'Touareg')).

In order to account for the N inputs in the organic systems, the determination of BNF by forage legumes was based on the empirical model of Høgh-Jensen et al. [40]. This approach is calculated from measures of dry matter (DM) yield of legumes and their N%. The relevant default parameters were set according to the assumptions of a 1–2-year-old grass-red clover sward given by Høgh-Jensen et al. [40]. To determinate the amount of N_2 fixation of grain legumes, we used the extended difference method (reference crop oat) according to Wichmann et al. [41]. Based on these models the calculated average BNF for the crop rotation elements was 50 and 65 kg N ha^{-1} year^{-1} for organic-low-BNF and organic-semi-BNF, respectively, with no further N amendments. In the conventional system, however, N inputs were primarily as mineral fertilizers with small amendments of digestates and pig slurry, with total N rates of 224 kg N ha^{-1} year^{-1}. The N rates were consistent with the agricultural practice recommended for each specific crop (Table 1).

Soil cultivation was adapted to the specific weed infestations and therefore varied between systems, years and crops. In the organic systems the primary soil tillage was carried out by ploughing, whereas a cultivator was used in the conventional system. In the organic systems the arable land for summer crops was managed mechanically in autumn and was not sown or covered with any plant residues during winter, with bare ground until seeding in spring (see Supplementary Information Table S1). Grass–clover mixtures in the first year were established by under-sowing in spring. The grass–clover mixtures in the second year were sown in August, shortly after harvesting. Grass–clover swards were cut three times a year in organic-low-BNF and twice a year in organic-semi-BNF.

Table 1. Application dates and amounts of N fertilizer, manure and digestate on different crops within the conventional arable system.

Rotation Element	Year	Amount [kg N ha^{-1}], Fertilizer Type and Date of N Application					Total Input [kg N ha^{-1}]
Treatment		1.	2.	3.	4.	5.	
Oilseed rape	1st	31 [b] (09-Sep-12)	138 [c] (28-Feb-13)	42 [d] (08-Apr-13)			211
	2nd	106 [f] (30-Aug-13)	130 [c] (18-Feb-14)	74 [f] (26-Feb-14)	31 [d] (01-Mar-14)		341
Winter wheat	1st	83 [c] (06-Mar-13)	25 [d] (08-Apr-13)	38 [b] (22-Apr-13)	32 [d] (21-May-13)		178
	2nd	80 [c] (25-Feb-14)	31 [d] (08-Mar-14)	55 [c] (06-Apr-14)	52 [c] (18-May-14)		218
Winter wheat	1st	88 [c] (06-Mar-13)	61 [e] (06-Mar-13)	29 [d] (12-Apr-13)	52 [c] (17-Apr-13)	37 [c] (21-May-13)	267
	2nd	95 [c] (25-Feb-14)	82 [e] (13-Mar-14)	32 [d] (10-Mar-14)	21 [b] (07-Apr-14)	55 [c] (17-May-14)	285
Sugar beet	1st	36 [b] (07-Mar-13)	78 [f] (17-Apr-13)	38 [a] (07-May-13)			152
	2nd	36 [b] (11-Mar-14)	100 [c] (23-Apr-14)				136
Winter wheat	1st	101 [c] (06-Mar-13)	25 [d] (08-Apr-13)	36 [b] (22-Apr-13)	60 [c] (21-May-13)		222
	2nd	92 [c] (25-Feb-14)	31 [d] (08-Mar-14)	55 [c] (03-Apr-14)	55 [c] (18-May-14)		233

[a]: CAN, Calcium ammonium nitrate; [b]: DAP, Diammonium Phosphate; [c]: Urea; [d] SSA ammonium sulphate; [e]: pig slurry; [f]: biogas digestate; the total N amount is shown for organic fertilizers.

In addition, the study included two permanent land use types as reference systems. These were permanent extensive grassland and beech forest, each of which occupied more than 5 ha. The permanent grassland has been cut twice each year since its establishment in 2004. The sward composition showed typical patterns for extensively used permanent grassland, with 49% *L. perenne*, 13% *T. repens*, 12% *P. trivialis*, 8% *T. sect. Ruderalia*, 6% *D. glomerata*, 5% *P. pratensis* and 7% other species. The beech forest has existed in its current form for more than 50 years. Only beech trees were present in the study area. Both treatments were included in the research area to provide additional information for either extensively managed agricultural land (grassland) or reserve area (beech forest) as a baseline scenario for comparison with land-use for arable agriculture. Both reference systems were on land with the same soil conditions as described above.

2.4. Flux-Measurements

Fluxes of N_2O and CH_4 were measured with the static closed chamber method [42]. The minimum sampling frequency was once a week in all rotations and crop, taken between 10:00 a.m. and 12:00 p.m. during the two experimental years (01 October 2012–14 October 2014). In a pre-treatment, four collars (d = 60 cm, h = 15 cm) per crop (n = 4) made from polyvinyl chloride (PVC) were installed in the 10 cm soil depth at an angle of 45° with uniform distances in order to capture a representative area across each field. The collars were removed briefly during harvesting and tillage operations. During the flux measurements, the collars were closed gas tight with white PVC chambers (d = 60 cm, h = 35 cm). The area between the basal ring and chamber was tightened with a taut butyl rubber band. The chambers were closed for 60 min on each measurement day, when a fan within each chamber allowed homogenized air conditions before sampling. Gas samples were taken at 0, 20, 40 and 60 min after closure through a gas-tight septum on the top of the chamber using a 30 ml syringe. Samples were directly transferred into 12 ml pre-evacuated septum-capped vials (Labco, High Wycombe, UK). After fertilizer application events, measurements were conducted more frequently over two weeks at irregular intervals. The size of the collars and chambers allowed for undisturbed plant growth within the collars. In the later growth stages, when plant growth exceeded chamber height, extensions (h:35 cm) were used. Gas samples were analyzed for N_2O and CH_4 through a gas chromatograph (SCION 456-GC, Bruker, Leiderdorp, Netherlands). Samples were injected using an autosampler (model 271 LH, Gilson Inc., Middleton, WI, USA). Data were processed using the software Compass CDS (Version 3.0.1). The change of gas concentration in the chamber headspace during the measurement was calculated by linear regression. In order to calculate the CO_2-equivalents (CO_{2eq}), the GWP values of 265 for N_2O and 28 for CH_4 were used [1]. In order to calculate the product carbon footprint (PCF), derived from the two non-CO_2 trace gases, namely N_2O and CH_4, the accumulated annual CO_{2eq} fluxes were divided by the functional unit grain equivalents (see Section 2.5.) and is referred to as $PCF_{NON\text{-}CO2}$ in the remainder of the paper.

2.5. Determination of Yield and N-Balance

The fresh-matter (FM) yields of crops were recorded at a field scale by weighing the commercial machinery with the harvested crops. To calculate the dry matter yields, additional subsamples of aboveground biomass were taken with shears at random in the field, with five replicates prior to each harvest. In the permanent grassland and grass–clover swards, subsamples were taken at a stubble height of 5 cm from an area of 0.25 m^2. In crops, the biomass samples were taken on 1.0 m^2 directly above the soil surface, although for sugar beet, samples were subdivided into primary and secondary products from yield samples. The dry matter contents of the subsamples were estimated after oven drying for 24 h at 58°C. After drying, the samples were milled to pass a 2 mm sieve (Cyclotec mill, Foss, Hillerød, Denmark). Subsequently, the N and C contents of all samples were measured using a C/N analyzer (Vario Max CN, Elementar, Hanau, Germany). The values for the N balance (kg N ha^{-1} $year^{-1}$)

were calculated after deduction of the N yields (N output) from the amounts of applied N and BNF (N-input):

$$\text{N balance [kg N ha}^{-1}\text{ year}^{-1}] = \text{N input} - \text{N output} \quad (1)$$

For product unit comparisons, the functional unit "grain equivalents" (GE) was chosen, using values obtained from the German Federal Ministry of Food and Agriculture [43], with GE = 1.04 for wheat, GE = 1.01 for rye, GE = 0.84 for oats, GE = 1.04 for peas, GE = 1.3 for oilseed rape, GE = 0.23 for sugar beets and GE = 0.58 for grass–clover. One GE was defined as the feeding value of 100 kg barley. The GE factors were multiplied by the DM yields of the different crops.

2.6. Statistical Analysis

The statistical software R (2015) was used to evaluate the data. The data evaluation started with the definition of an appropriate statistical model. The data were normally distributed and heteroscedastic due to the experimental year and single crop rotation elements within the system. For comparison of crops within a single system the statistical model included the experimental year (first and second) and crops, as well as their interaction terms as fixed factors. For management system comparison, the experimental year and crops were considered as random factors. Based on these mixed effect models an analysis of variance (ANOVA) was conducted to answer the questions of the trial. Multiple contrast tests were then conducted in order to compare the several levels of the influence factors. In addition, simple linear regression models were developed to test whether N_2O emissions can be estimated from other measured variables (e.g., N balance, CN ratio of crop residues). The significance of the tested factors, comparisons of means, and regression equations (intercepts and slopes) were declared when $p < 0.05$.

3. Results

3.1. GHG-Fluxes

The experimental year showed an interaction with crop and the tested systems on GHG fluxes per ha^{-1} and $PCF_{NON\text{-}CO2}$ (Table 2).

Table 2. Level of significance for the impact of period, crop within a system, as well as the impact of the interaction of both factors on N_2O, CH_4, CO_{2eq} ha^{-1} and product carbon footprints (PCFNON-CO_2).

Effect	N_2O	CH_4	CO_{2eq} ha^{-1}	$PCF_{NON\text{-}CO2}$
Period	ns	$p < 0.001$	ns	ns
Crop	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$
Period*Crop	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$

Daily GHG fluxes measured during the two experimental years in the arable crop rotations and reference system are presented in Figures 2–4. Measured N_2O fluxes in all systems ranged from 0.0 to 0.13 kg N_2O–N ha^{-1} day^{-1}. The fluxes showed a high variability during the year and among the systems. The lowest N_2O fluxes during the two years were recorded in the beech forest and in the extensively managed grassland (see Supplementary Figure S1). In the comparison of the tested arable crops, grass–clover showed the lowest N_2O fluxes (Figures 2 and 3). In general, there were clearly detectable emission peaks for conventional crops when fertilizer dressings took place (Figure 4). In all systems, there were additional N_2O peaks after soil-tillage and frost-thaw events during winter. The organically managed crops showed low emissions on average, with the highest figures for cereals, of 1 kg N_2O–N ha^{-1} $year^{-1}$ or less. In comparison, the conventional system had the highest emissions, with annual emissions of > 1.5 kg N_2O–N ha^{-1} (Table 3). Annual N_2O-emissions in the two reference systems (grassland and beech forest) were lowest on average and were below 0.5 kg N_2O–N ha^{-1} $year^{-1}$ in both experimental years.

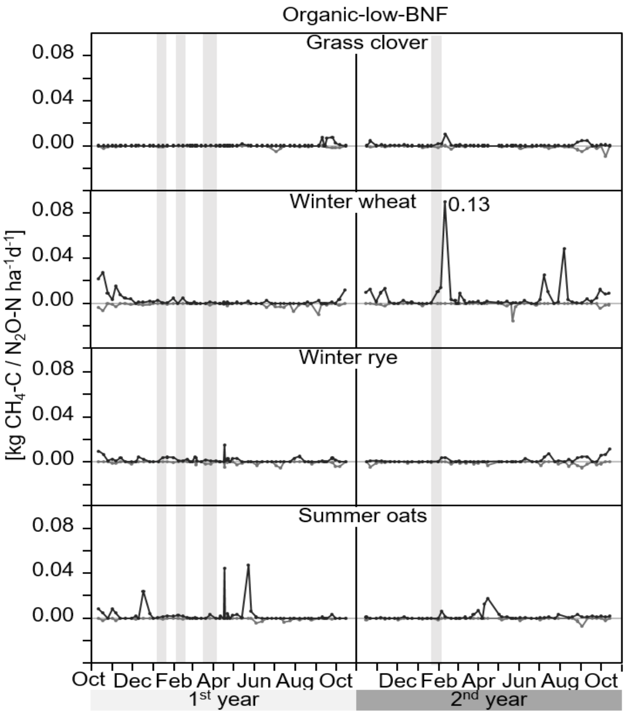

Figure 2. Daily N_2O- (black line and dots) and CH_4-fluxes (grey line and dots) for each crop within the crop rotation of the organic-low-BNF during the study period (October 2012–October 2014). The grey background represents periods with mean daily temperatures below 0 °C. Mean values are shown (n = 4).

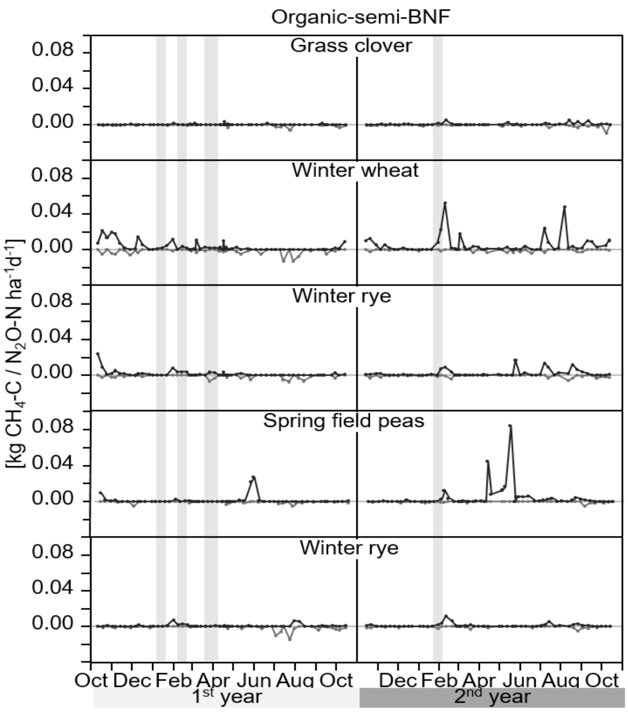

Figure 3. Daily N_2O- (black line and dots) and CH_4-fluxes (grey line and dots) for each crop within the crop rotation of the organic system organic-semi-BNF during the study period (October 2012–October 2014). The grey background represents periods with mean daily temperatures below 0 °C. Mean values are shown (n = 4).

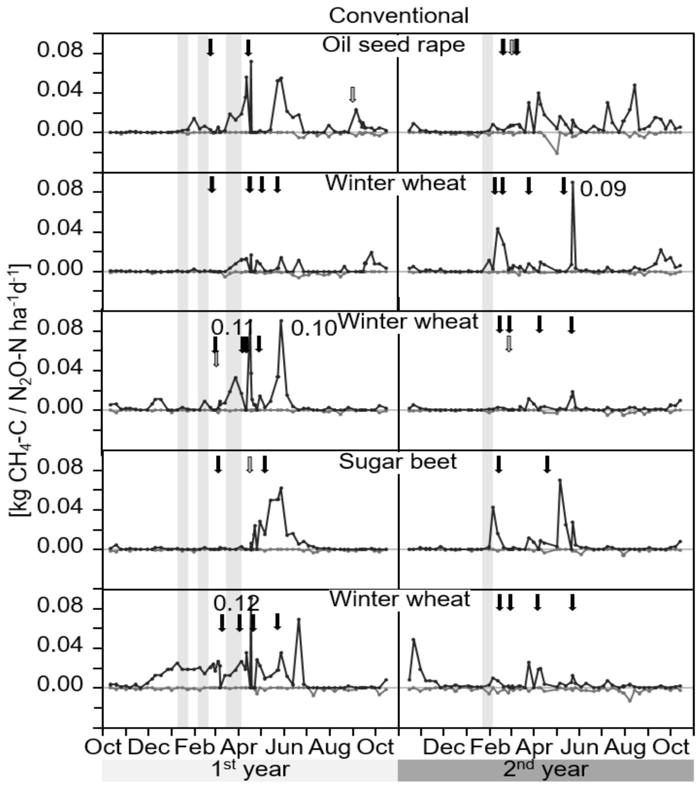

Figure 4. Daily N_2O- (black line and dots) and CH_4-fluxes (grey line and dots) for each crop within the crop rotation of the conventional system during the study period (October 2012–October 2014). The black arrows indicate applications of mineral N fertilizer. The grey arrows indicate applications of organic manure. The grey background represents periods with mean daily temperatures below 0 °C. Mean values are shown (n = 4).

Table 3. Accumulated N_2O, CH_4 and global warming potential (GWP) per ha within one crop rotation (system). Yields were estimated at farm level by weighing the harvested crops. Product carbon footprints (PCFNON-CO_2) were derived from GWP and grain equivalents (GE). Different uppercase letters show significant differences between the two experimental years within each crop. Different lowercase letters show significant differences between crops within each system and experimental year ($p < 0.05$).

System	Crop	First Experimental Year						Second Experimental Year					
		N_2O [kg N ha^{-1}]	CH_4 [kg C ha^{-1}]	GWP [kg CO_{2eq} ha^{-1}]	Yield [t DM ha^{-1}]	Yield [GE ha^{-1}]	$PCF_{NON\text{-}CO2}$ [kg CO_{2eq} GE^{-1}]	N_2O [kg N ha^{-1}]	CH_4 [kg C ha^{-1}]	GWP [kg CO_{2eq} ha^{-1}]	Yield [t DM ha^{-1}]	Yield [GE ha^{-1}]	$PCF_{NON\text{-}CO2}$ [kg CO_{2eq} GE^{-1}]
Organic-low-BNF	Grass–clover	0.2 ^{Ab}	−0.1 ^{Aa}	79.5 ^{Ab}	7.1	40.9	1.9 ^{Ac}	0.3 ^{Ab}	−0.3 ^{Ba}	113.6 ^{Ab}	6.7	38.7	2.9 ^{Ab}
	Winter wheat	0.9 ^{Aa}	−0.3 ^{Aa}	363.2 ^{Aa}	3.2	33.6	10.8 ^{Aa}	2.3 ^{Aa}	−0.1 ^{Ba}	953.2 ^{Aa}	3.0	31.1	30.6 ^{Aa}
	Winter rye	0.5 ^{Aa}	−0.3 ^{Aa}	196.8 ^{Aa}	4.1	41.4	4.8 ^{Ab}	0.4 ^{Ab}	−0.2 ^{Aa}	159.0 ^{Ab}	4.4	44.4	3.6 ^{Ab}
	Summer oats	1.0 ^{Aa}	−0.1 ^{Ab}	412.3 ^{Aa}	3.1	25.9	15.9 ^{Aa}	0.6 ^{Ab}	−0.1 ^{Aa}	245.9 ^{Ab}	3.7	31.1	7.9 ^{Ab}
Organic-semi-BNF	Grass–clover	0.0 ^{Ad}	−0.1 ^{Ac}	12.9 ^{Ad}	6.3	36.4	0.4 ^{Ad}	0.2 ^{Bc}	−0.2 ^{Aa}	75.7 ^{Bc}	5.8	33.4	2.3 ^{Bc}
	Winter wheat	1.1 ^{Aa}	−0.5 ^{Aa}	439 ^{Aa}	3.5	36.8	11.9 ^{Aa}	1.7 ^{Aa}	−0.2 ^{Ba}	699.8 ^{Aa}	4.0	41.6	16.8 ^{Aa}
	Winter rye	0.5 ^{Ab}	−0.3 ^{Aabc}	196.8 ^{Ab}	4.1	41.4	4.8 ^{Ab}	0.7 ^{Ab}	−0.2 ^{Aa}	283.8 ^{Ab}	4.9	49.5	5.7 ^{Ab}
	Spring field peas	0.4 ^{Aab}	−0.1 ^{Aab}	162.7 ^{Abc}	3.5	36.1	4.5 ^{Aab}	1.6 ^{Aabc}	−0.1 ^{Aa}	661.9 ^{Aabc}	3.1	32.2	20.6 ^{Aabc}
	Winter rye	0.3 ^{Ac}	−0.3 ^{Ab}	113.6 ^{Ac}	4.6	46.0	2.5 ^{Ac}	0.4 ^{Abc}	−0.1 ^{Ba}	162.7 ^{Abc}	3.0	30.3	5.4 ^{Ab}
Conven-tional	Oil seed rape	2.6 ^{Ab}	−0.2 ^{Aab}	1074.3 ^{Ab}	4.3	55.9	19.2 ^{Aa}	2.6 ^{Aa}	−0.4 ^{Bab}	1066.8 ^{Aa}	4.9	63.7	16.7 ^{Aa}
	Winter wheat	0.9 ^{Ac}	−0.3 ^{Aa}	363.2 ^{Ac}	10.8	112.3	3.2 ^{Ac}	1.7 ^{Bb}	−0.2 ^{Ac}	699.8 ^{Bb}	11.4	118.6	5.9 ^{Bb}
	Winter wheat	2.7 ^{Ab}	−0.1 ^{Ab}	1119.6 ^{Ab}	9.8	101.8	11.0 ^{Ab}	0.7 ^{Bc}	−0.2 ^{Aac}	283.8 ^{Bc}	10.3	107.1	2.6 ^{Bc}
	Sugar beet	2.0 ^{Aab}	−0.2 ^{Aab}	824.6 ^{Aabc}	75.0	172.5	4.8 ^{Ac}	1.6 ^{Ab}	−0.2 ^{Ac}	658.2 ^{Ab}	89.0	204.7	3.2 ^{Ac}
	Winter wheat	4.1 ^{Aa}	−0.3 ^{Aa}	1694.6 ^{Aa}	9.8	101.8	16.6 ^{Aa}	1.7 ^{Bb}	−0.5 ^{Ba}	688.6 ^{Bb}	11.2	116.5	5.9 ^{Bb}
Grass-land		0.2	−0.1	82.2	2.4	14.2	5.8	0.3	−0.1	122	2.4	14.1	8.6
Beech forest		0.2	−3.9	−51.3		-	-	0.5	−2.5	114		-	-

The amounts of N input, N balance and the C/N ratio were statistically tested against amounts of released annual N_2O emissions. In general, the conventional crop rotations showed higher N inputs and positive N balance in all crops and experimental years. With regard to the N balance levels in the organic systems, the values in the subsequent crop after cereals were negative. The highest surpluses were measured in the winter wheat after incorporation of the preceding grass–clover swards.

The regression analysis showed, on average, a positive linear relationship between N input and annual N_2O emissions for the tested arable systems (intercept: 0.44, slope:0.007, $R^2 = 0.6$, $p < 0.01$). The relationship of N surplus on annual N_2O emissions was also significant (intercept: 0.86, slope:0.01, $R^2 = 0.6$, $p < 0.01$). Comparing this N surplus to the N surplus of the preceding crop, the effect of the preceding crop was a more reliable indicator to predict annual N_2O emissions (intercept: 0.4, slope:0.03, $R^2 = 0.8$, $p < 0.01$) (Figure 5). This fact was also confirmed by a strong effect of the C/N ratio of plant residues on the measured N_2O losses in the subsequent months after incorporation into soil. The measured emissions of N_2O declined exponentially, with increasing C/N ratio from incorporated straw and stubble residues (Figure 6). On average, for the different arable crop rotations, the C/N ratios of crop residues differed in the order of conventional (65 (SE 5.9)) < organic-semi-BNF (102 (SE 6.4)) < organic-low-BNF (125 (SE 5.9)) ($p < 0.05$).

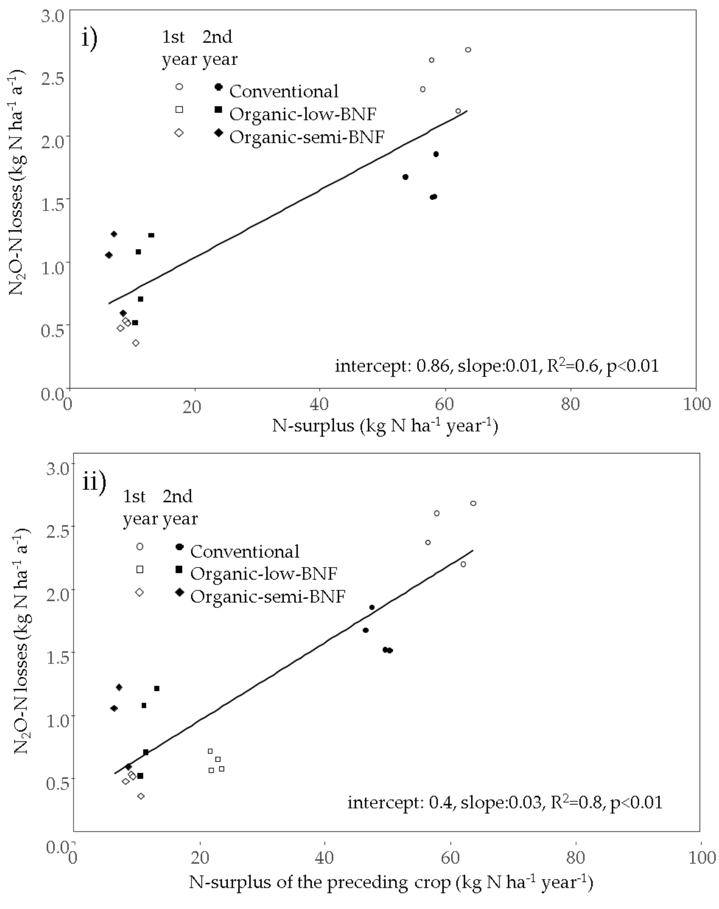

Figure 5. The relationship between annual N_2O–N emissions and (i) N surplus, as well as (ii) N surplus of the preceding crop, average of the different crop rotations.

According to the performed regression analysis, the emission factor for N inputs was 0.7%. However, taking the N balance of the main crop and the preceding crop into account, the emissions factors are 1% and 3% of each kg N in surplus, respectively.

For CH_4, the tested arable systems showed only negligible exchange (values close to zero), or they acted as a sink. The measured sink of the beech forest, as a natural reference ecosystem, exceeded those of the agriculturally managed systems. Thus, annual accumulated emissions showed a notable CH_4 sink capacity of 3.9 and 2.5 kg CH_4-C ha^{-1} in the first and second year, respectively.

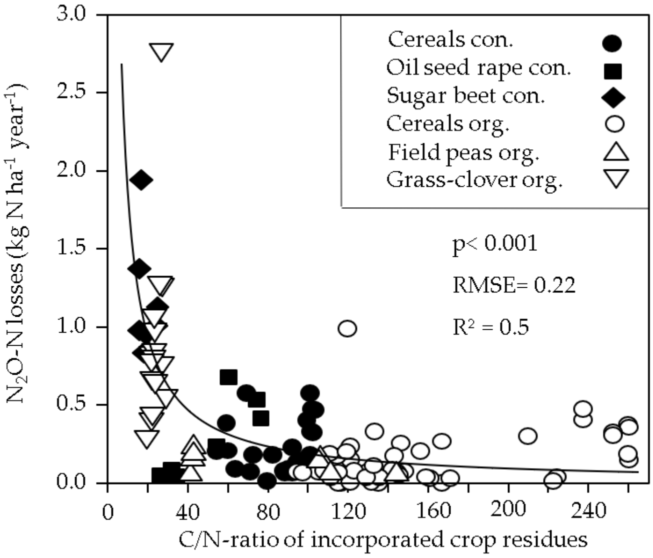

Figure 6. Relationship between C/N ratio of incorporated aboveground crop residues and N2O–N emissions for a period of 4 months (Oct–Feb) post-harvest: [kg N_2O–N ha^{-1} $year^{-1}$ = 18.8 * 1/CN-ratio].

3.2. *Crop Yields and Yield-Related GHG Emissions*

The crop yields of the organic crop rotations were lower in both experimental years compared with the conventional farming system. On average, the GE output from organic farming was 0.32 of the yield of the conventional system. The total average yields of the two organic systems were similar for dry matter but slightly higher for energy expressed as GE in the organic-semi-BNF (38 vs. 36 GE $year^{-1}$). Within the organically managed systems the grass–clover leys showed the lowest emissions per GE. Slightly higher emissions of 5 kg CO_2e GE^{-1} were observed for winter rye. A higher $PCF_{NON\text{-}CO2}$ was found in the organically managed winter wheat due to the effect of grass–clover removal as a preceding crop. Due to the relatively low yield level of spring oat and spring field peas, the product-related emissions of these crops exceeded all other crops in the organically managed systems. In the conventional system the highest $PCF_{NON\text{-}CO2}$ values were found in oilseed rape and winter wheat with sugar beet as a preceding crop (Table 3).

The annual emissions of CO_{2eq} averaged over each of the whole crop rotations are given in Table 4. All the studied agricultural systems were sources of GHG emissions, whereas the beech forest functioned as a sink for CO_{2eq} because of its negligible N_2O losses and significant CH_4 consumption during the first year. The highest CO_{2eq} emissions were observed for the conventional system, whereas emissions from the two organically managed systems, on average over the two years, were 0.35 that of the conventional system. The differences in the $PCF_{NON\text{-}CO2}$ values showed no clear trend between the different systems. Both the organic-low-BNF and conventional cropping systems showed differences between the experimental years, but these were not different between organic-semi-BNF and grassland. During the first experimental year, the $PCF_{NON\text{-}CO2}$ values were lowest for the reference grassland and highest for conventional farming. However, during the second year $PCF_{NON\text{-}CO2}$ was lowest for conventional farming and highest for organic-low-BNF, with no differences to organic-semi-BNF (Table 4). Thus, comparing the two organic systems, the organic-semi-BNF showed an advantage, with either lower emissions than the organic-low-BNF or no differences to the conventional system.

Table 4. Average global warming potential (GWP) and product carbon footprint ($PCF_{NON\text{-}CO2}$) per system and experimental year (first and second). Yields were estimated at farm level by weighing the harvested crops. Product carbon footprints were derived from GWP and grain equivalents (GE). Different uppercase letters show significant differences between the two experimental years within each system. Different lowercase letters show significant differences between the systems within each experimental year. ($p < 0.05$). Standard errors (SE) are shown in brackets.

	First Experimental Year		Second Experimental Year	
System	GWP [kg CO_{2eq} ha^{-1}]	$PCF_{NON\text{-}CO2}$ [kg GE^{-1}]	GWP [kg CO_{2eq} ha^{-1}]	$PCF_{NON\text{-}CO2}$ [kg GE^{-1}]
Organic-low-BNF	263.0 (24.1) ^{Ab}	8.4 (0.58) ^{Ab}	367.9 (66.5) ^{Ab}	11.3 (2.13) ^{Aab}
Organic-semi-BNF	185.0 (10.5) ^{Ac}	4.8 (0.28) ^{Ac}	376.8 (98.4) ^{Bb}	10.1 (3.17) ^{Ba}
Conventional	1015.3 (60.1) ^{Aa}	11.0 (0.44) ^{Aa}	679.4 (31.5) ^{Ba}	6.9 (0.28) ^{Bb}
Grassland	82.2 (46.4) ^{Acd}	5.8 (3.3) ^{Aabc}	122.1 (29.2) ^{Ac}	8.6 (2.05) ^{Aab}
Beech forest	−51.3 (39.8) ^{Ad}		114.9 (8.9) ^{Ac}	

4. Discussion

4.1. GHG-Fluxes

In general, there were differences between the investigated organic and conventional systems in crop rotations, type of grown crops, and the intensity of N input. Even though the conventional system was managed in accordance with the German fertilizer ordinance (2017), it had a substantially higher N input than the organic systems and reference land-use systems. It is widely accepted that the N input from synthetic fertilizers and manure is the main source of N_2O emissions from agricultural soils [28,29]. The daily N_2O flux rates and the measured emission peaks generally followed the fertilizer applications, which is consistent with similar previous findings [10,44–46]. This also accords with GHG inventories using N input as the main variable to calculate direct N_2O emissions from agricultural soils. However, although the standard emission factor used was 1% for each kg of N-applied, we found emission factors of 0.2 and 1.8% in the conventional crop rotation. This may be explained partly by differences between different types of N fertilizer [47,48], as greater N_2O peaks after application of manure, in comparison with applications of mineral N fertilizer, are likely [46,49] due to the higher availability of easily accessible carbon triggering high N_2O peaks from soil denitrifiers, when oxygen is limiting. Other authors have also concluded that calculations of N_2O emissions based on N-fertilizer input are not reliable, as better correlations exist between N surplus and N_2O emissions [28]. Hence, to reduce N_2O emissions, the aim at the farm scale should be to increase N use efficiency [32]. This aim was also supported by the results of our study, with a significant correlation between N_2O emissions and the calculated N balance of the preceding crop showing a better relationship than only the N input. In addition to the fertilizer application and its management, the potential mineralizable N from crop residues is a further main driver of N_2O release in soil. This was particularly evident in the conventional systems for winter oilseed rape, as the N removed in the harvested seeds is low and the crop residues are within a narrow C/N ratio. This situation is also true after sugar beet harvest, due to the narrow C/N ratio of the leaves and tops with high moisture content compared with cereal straw and rape straw [50]. In the case of cereal straw, low N_2O emissions can be further explained by temporary immobilization of soil N due to the high C/N ratios. This may have occurred in the case of incorporated cereal straw material, potentially explaining the comparatively low emissions of N_2O.

In the organic systems, grass–clover swards are utilized as the main source of N [32,51], but with different findings of the effect on N_2O emissions. According to the results of Benoit et al. [52], intact legume swards were shown to be the crops with the lowest N_2O emissions in agricultural systems because of a high potential for effective N cycling [53]. However, in the year of grassland removal, high N surpluses and annual N_2O emissions were measured in the following crop of winter wheat.

Agriculture has also been identified as a major anthropogenic source of CH_4 due to microbial activity in ruminant digestion and in livestock slurry, while wetlands are considered as a major natural source of emission [54]. However, under aerobic conditions, soils under agricultural management act as a sink for CH_4 [55,56]. According to Boeckx and Van Cleemput [55], arable soils have a lower mean CH_4 sink function (−1.5 kg CH_4 ha^{-1} $year^{-1}$) than grassland soils (−2.5 kg CH_4 ha^{-1} $year^{-1}$). Mineral N fertilizer application can also have a negative effect on the CH_4 oxidation capacity of arable soils [57]. The highest CH_4 oxidation rates have been reported for near-natural forest soils [25,58,59]. In support of this, in our study the beech-forest showed the highest average CH_4 sink capacity. However, our findings did not confirm higher CH_4 uptake of grassland soils compared with the arable systems, and only a minimal higher CH_4 oxidation capacity of the unfertilized organic rotations.

4.2. Crop Yields and Yield-Related GHG Emissions

In the conventional system the relatively high but site-typical application rate of mineral fertilizers, combined with appropriate plant protection measures, led to three-fold higher crop yields compared with the organic systems. Such distinct differences in yields have generally not been documented in other studies [35,60]. Lower yields from organic systems are due partly to lower supplies of nutrients, including possible deficits of phosphorus, potassium, magnesium and micronutrients, and the comparatively lower soil pH values of the organically managed fields (see Supplementary Table S1). Nutrient deficits are a particular problem in pure arable organic systems without animal husbandry and external N supplies, leading to negative N balance levels and decreasing soil N stocks.

In the context of increasing demand for organic farming products and addressing the environmental problems associated with agriculture, including the challenges of climate change, there is a need to also consider emissions per unit of product when conventional and organic farming systems are compared [61]. This, and the concept of sustainable intensification in agriculture, are today often considered with the twin aims of maximizing the production of agricultural goods and minimizing the negative impacts on the environment [62]. In the present study, we considered the product unit (grain equivalents) as an appropriate measure relevant at a global environmental scale, and here we found that the three systems showed comparable values. Similar results were reported by Flessa et al. [63], where organic farming did not result in different emissions per unit of crop yield. Controversially, in a meta-analysis, Skinner et al. [23] found 29% higher yield-related N_2O emissions for arable organic systems. Also, Chirindaa et al. [64] and Knudsen et al. [65] showed higher product-related N_2O emissions for organic arable systems on average for all investigated crop rotations. However, comparisons in the literature between organic and conventional arable production depend strongly on the types of crops under investigation [29,31,52,64] with different crops affecting yield-related emissions differently. Further, in the present study, we also included non-food crops in the analysis, in this case red clover as fodder legumes, and, as they were sold, they have a market value as a product.

Beyond the measured field-level emissions, the different systems should be further evaluated in terms of a full lifecycle assessment (LCA) analysis that also includes any emissions produced from external resources [65]. With regard to our field measurement results and the expected (though unmeasured) higher energy use in the conventional system, we assume that the tested organic systems would show further net benefit, relative to intensive conventional systems, under the perspective of an LCA analysis. However, other impact categories like eutrophication, acidification and cumulative energy use should be taken into account in order to avoid negative implication on other environmental resources. In our study, associated research groups from the project group Hof Ritzerau identified significant benefits from the conversion to organic farming over a ten-year period, e.g., regarding soil biota and birds [66,67]. Thus, when modelling options for optimized land use systems in terms of sustainable intensification on high fertility arable soils, an optimal share of the area under organic farming has to be predicted in order to ensure long-term ecosystem functioning in landscapes. Finally, the economic consequences for producers, and for wider society and its welfare, also need to be considered in relation to any large-scale land-use change, either to organic farming or other extensive land-use systems like grassland and forestry.

4.3. Further Mitigation Potential

An important finding of this study was the increased N_2O emissions after soil tillage in arable systems. This was mainly controlled by the chemical composition of the crop residues, particularly when narrow C/N ratios were present, and was clearly shown over a wide range of arable crops and use intensities. A serious weakness in the conventional as well as the two organic systems was the inefficient use of nutrients in crop residues, as well as the failure not to take mineralizable N into account for fertilizer planning and to ensure optimal timing of ploughing in the grass–legume ley phase. Thus, further mitigation could be achieved through reduced input of mineral N fertilizer in the conventional systems. If organic fertilizers are applied, as manure or digestates, their N contents should be also taken into account for optimal N management strategies to avoid highly positive N surpluses. With regards to the replacement of the grass–clover ley in the rotation, one efficient measure would be to postpone ploughing until spring to reduce emissions, as high N_2O fluxes occur during winter at locations where freeze-thaw cycles are expected [30]. Generally, there is a need to reduce N_2O emissions in organic farming systems after the incorporation of N rich crop residues [31] such as those which occur after the replacement of a grass–clover ley. Organic arable farms especially should seek opportunities to enter into cooperation agreements with livestock farms to exchange herbage harvested from grass–clover crops (as silage bales) for livestock manure. This would not only support increased yields but also help maintain soil fertility. Another option is to remove the mown grass–clover biomass for biogas production and apply the digestate to the soil the following spring. Möller et al. [68] concluded that this option leads to additional energy yields, a lower risk of nitrate leaching, and lower N_2O emissions compared with swards that are just mown and mulched. Manure application has been shown to be one of the main factors in arable organic farming needed to generate competitive crop yields. Nevertheless, the use of organic fertilizer can cause additional N_2O emissions [65], and, thus, has to be further evaluated.

Finally, the aim of all arable systems should be to maintain or increase crop yields without causing additional environmental problems [69,70]. In this context, the key factor in organic farming is an efficient method of converting the grass–clover ley phase for a subsequent crop, without significant N losses, and efficiently managing the use of legumes in the crop rotation. In our study, the use of 25% of legumes in the arable crop rotation was not enough to achieve adequate yield to compete with the arable conventional systems in terms of emissions per product unit. This was only given with a share of 40% legumes in the crop rotation.

5. Conclusions

Organically managed arable rotations showed advantages in comparison with intensive conventional arable cropping through reduced GHG emissions and N surplus expressed on a per-ha basis. Hence, an increase in the area of organic farming on farmland with arable rotation systems is a suitable measure for reducing GHG emissions and nutrient loads at a national or landscape level. On the global scale, however, emissions have to be considered also on a per-unit of product basis, as climate is a global environmental good. Taking that into account, and the strongly limited productivity of organic all arable farming as well, it becomes evident that improved strategies beyond specialized conventional and organic approaches focusing on mixed farming systems with enhanced eco-efficiency are needed to ensure both global food demand and ecosystem services.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/12/8/3240/s1, Figure S1: Daily N_2O- (black line and dots) and CH_4-fluxes (grey line and dots) of the reference systems (grassland and beech forest) during the study period (October 2012–October 2014). The grey background represents periods with mean daily temperatures below 0 °C. Mean values are shown (n = 4)., Table S1: Observed systems and crops within the different arable rotations, soil properties of the experimental sites, selected agronomic practices and sowing date during the experimental years (1st year: October 2012–October 2013, 2nd year: October 2013–October 2014).

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