Review of causes and sources for N₂O emissions and NO₃-leaching from organic arable crop rotations

Sissel Hansen[1], Randi Berland Frøseth[2], Maria Stenberg[3], Jarosław Stalenga[4], Jørgen E. Olesen[5], Maike Krauss[6], Paweł Radzikowski[4], Jordi Doltra[7], Shahid Nadeem[8], Torfinn Torp[9], Valentini Pappa[10], Christine A. Watson[11,12]

[1]Norwegian Centre for Organic Agriculture, NO-6630 Tingvoll, Norway
[2]Department of Grain and Forage Seed Agronomy, Norwegian Institute of Bioeconomy (NIBIO), NO-1431 Ås, Norway
[3]Department of Soil and Environment, Swedish University of Agricultural Sciences (SLU), Skara, Sweden
[4]Department of Systems and Economics of Crop Production, Institute of Soil Science and Plant Cultivation - State Research Institute, 24-100 Puławy, Poland
[5]Department of Agroecology, Aarhus University, DK-8830 Tjele, Denmark
[6]Department of Soil Sciences, Research Institute of Organic Agriculture (FiBL), 5070 Frick, Switzerland
[7]Cantabrian Agricultural Research and Training Centre, CIFA, 39600 Muriedas, Cantabria, Spain
[8]Faculty of Environmental Sciences and Natural Resource Management (MINA), Norwegian University of Life Sciences (NMBU), 1430 Ås, Norway
[9] Department of Research, Norwegian Institute of Bioeconomy (NIBIO), NO-1431 Ås, Norway
[10] Texas A&M, Energy Institute, College Station, TX 77845-3372, USA
[11]Department of Crop Production Ecology, Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden
[12]SRUC, Craibstone Estate, Aberdeen, AB21 9YA, UK

Correspondence to: Sissel Hansen (sissel.hansen@norsok.no)

Abstract.

The emissions of nitrous oxide (N$_2$O) and leaching of nitrate (NO$_3$) from agricultural cropping systems have considerable negative impacts on climate and the environment. Although these environmental burdens are on average less per unit area in organic than in non-organic production, they are roughly similar per unit of product. If organic farming is to maintain its goal of being environmentally friendly, these loadings must be mitigated. We discuss the impact of possible drivers of N$_2$O emissions and NO$_3$ leaching within organic arable farming practice under European climatic conditions, and potential strategies to reduce these. Organic arable crop rotations are generally diverse with frequent use of legumes, intercropping and organic fertilizers. The soil organic matter content and share of active organic matter, soil structure, microbial and faunal activity are higher in such diverse rotations, and yields lower, than in non-organic arable cropping systems based on less diverse systems and inorganic fertilizers. Soil mineral nitrogen (SMN), N$_2$O emissions and NO$_3$ leaching are low under growing crops, but there is the potential for SMN accumulation and losses after crop termination, harvest or senescence. The risk of high N$_2$O fluxes increases when large amounts of herbage or organic fertilizers with readily available nitrogen (N) and degradable carbon are incorporated into the soil or left on the surface. Freezing / thawing, drying / rewetting, compacted and/or wet soil and mechanical mixing of crop residues into the soil further enhance the risk of high N$_2$O fluxes. N derived from soil organic matter (background emissions) do, however, seem to be the most important driver for N$_2$O emission from organic arable crop rotations and the correlation between yearly total N-input and N$_2$O emissions is weak. Incorporation of N rich plant residues or mechanical weeding followed by bare fallow increases the risk of NO$_3$ leaching. In contrast, strategic use of deep-rooted crops with long growing seasons or effective cover crops in the rotation reduces NO$_3$ leaching risk. Enhanced recycling of herbage from green manures, crop residues and cover crops through biogas or composting may increase N efficiency and reduce N$_2$O emissions and NO$_3$ leaching. Mixtures of legumes (e.g., clover or vetch) and non-legumes (e.g., grasses or Brassica species) are as efficient cover crops for reducing NO$_3$ leaching as monocultures of non-legume species. Continued regular use of cover crops has the potential to reduce NO$_3$ leaching and enhance soil organic matter but may enhance N$_2$O emissions. There is a need to optimise the use of crops and cover crops to enhance the synchrony of mineralisation with crop N uptake to enhance crop productivity, and this will at the same time reduce long-term risks of NO$_3$ leaching and N$_2$O emissions.

We use the following abbreviations: *BNF*: biological nitrogen fixation, *C*: Carbon, *CC*: cover crops, *CH$_4$*: methane, *EF*: Emission factor = % of N applied emitted as N$_2$O-N, *N*: nitrogen, *N$_2$O*: nitrous oxide, *NO$_3$*: nitrate, *PMN*: Potentially mineralizable N, *SMN*: Soil mineral nitrogen, *SOC*: Soil organic carbon, *SOM*: Soil organic matter.

and terminology: *Background emissions*: N$_2$O emissions that derive from N released from SOM as opposed to N added through fertilization, soil amendments or plant residue applied in the current year, *Degradable C*: Easily degradable carbon compounds, *Hot moments of N$_2$O flux*: Brief and disproportionally high short-term N$_2$O flux event due to the combination of multiple influencing factors (Molodovskaya et al., 2012), *N$_2$O emission*: The cumulative flux reported for one field treatment during the actual measurement period, *NO$_3$ leached*: NO$_3$ transported below the root zone, *PMN***:** The amount of SMN released from organic matter under ideal soil environmental conditions, *SMN:* The content of mineral N in the form or ammonium (NH$_4$) and NO$_3$ in soil.

1 Introduction

Biologically available nitrogen (N) or reactive N is limited in most natural terrestrial ecosystems. In modern crop production, addition of N fertilizer has become crucial to achieve high crop yields. This has resulted in cropping systems where a substantial proportion of the N added is lost to the environment, and where the excess reactive N threatens the quality of air, water and ecosystems (Robertson and Vitousek, 2009). The emissions of N$_2$O have considerable environmental impacts through the contribution to global warming and ozone depletion (Ravishankara et al., 2009), and about 16 to 20 Tg N$_2$O-N are emitted annually to the atmosphere. Of this, close to 40% are anthropogenic, and agriculture accounts for 67–80% of the anthropogenic N$_2$O emissions (Ussiri and Lal, 2013). About half of the anthropogenic N$_2$O emissions originate from cultivated soils (Stehfest and Bouwman, 2006). In addition, agricultural soils are sources of indirect N$_2$O emissions resulting from downstream microbial turnover of N from NO$_3$ leaching or ammonia volatilization (IPCC, 2006). NO$_3$ lost by leaching may also contaminate drinking water and lead to eutrophication of freshwater and marine ecosystems (Dalgaard et al., 2014). The area under organic production increases worldwide (Willer and Lernoud, 2018). In Europe, 2.7% of the agricultural land is under organic farming, and in nine countries, 10% or more of the agricultural land is managed organically (Willer et al., 2018). In 2016, 43% (6 mill ha) of the organic farmed area in Europe was under arable crops.

Organic agriculture aims to be an environmentally friendly production system that sustains the health of soils, ecosystems and people. It should rely on ecological processes, biodiversity and nutrient cycles adapted to local conditions, rather than the use of inputs (IFOAM, 2019). Because of the serious consequences of N$_2$O emissions and NO$_3$ leaching, these environmental burdens are important issues also for organic farming, and there is a continued debate whether an organic mode of crop production enhances or reduces greenhouse gas (GHG) emissions and NO$_3$ leaching from agriculture (Lorenz and Lal, 2016; McGee, 2015). We have chosen to focus on arable systems because crop and soil management vary more between organic and non-organic production of arable crops than for grassland (Barbieri et al., 2017), and the yield gap is larger (De Ponti et al., 2012). These conditions will affect N$_2$O emissions and NO$_3$ from the systems. Previous reviews have compared the net N$_2$O emissions and NO$_3$ leaching from these systems. Lower area-scaled, but roughly similar yield-scaled emissions (slightly higher, similar or slightly lower) are commonly observed for N$_2$O emissions (Skinner et al., 2019; Skinner et al., 2014; Tuomisto et al., 2012,) and NO$_3$ leaching (Aronsson et al., 2007; Benoit et al., 2014; Kirchmann and Bergström, 2001; Stopes et al., 2002; Tuomisto et al., 2012) from organic versus non-organic arable crop production.

In this review we focus on the drivers for N$_2$O emissions and NO$_3$-leaching under organic arable crop rotations under European climatic conditions. There are insufficient robust field data on N$_2$O emissions and NO$_3$ leaching within organic

arable crop rotations to allow for a meta-analysis to quantify the impact of key causes. Thus, here we use the available data to identify sources and causes of N_2O emission and NO_3 leaching in these rotations, as a basis for suggesting targeted mitigation strategies. We define "organic arable crop rotations" as cropping systems with associated crop and soil management commonly used in European farms dominated by arable cropping and following the European Council

5   Regulation (EC) No 834/2007s on organic farming (Council of the European Union, 2007). Among others, the use of synthetic N-fertilizers and N-inhibitors are prohibited, but manure and/or short-term leys may be used in these rotations. We designate "non-organic crop systems" as arable cropping systems generally based on inorganic fertilizers, use of pesticides and often using narrow crop rotations, commonly called conventional farming.

Both globally and in Europe organic rotations are longer and more diversified than non-organic rotations (Barbieri et al.,

10   2017). This is essential for nitrogen supply, and for pest and weed control (Stockdale et al., 2001). Barbieri et al. (2017) found that catch crops and undersown cover crops are 2.4 and 8.7 times more frequent in organic compared to conventional systems, respectively. They further found that the share of pulses and temporary fodder crops (such as alfalfa, clover and ryegrass) were higher in organic than in non-organic crop rotations, and that the difference between organic and non-organic crop rotations was greater in this respect in Europe than in North America and globally. They further found that more

15   legumes are included in fodder crops, and in catch crops, under sown cover crops and intercropping than in non-organic rotations. In addition to plant derived N, organic N are applied in manure or other organic fertilizers and amendments. The great diversity of N mineralisation patterns among the organic fertilizers and crop residues is a challenge for farm management to synchronize the N release with plant N uptake. If N is released during periods with poor plant uptake, then the content of soil mineral nitrogen (SMN) and other easily available N can accumulate, creating a large risk of N losses

20   through gaseous emissions or through leaching. Because N is mainly applied through plant residues and a limited amount of organic fertilizers in arable organic systems, the N-turnover from biological activity is crucial for the content and type of SMN. Plants and organic fertilizers are also important sources of soil organic carbon (SOC). We address how supply and quality of organic matter in above- and belowground residues and organic amendments influence availability and type of SMN and degradable carbon.

25   Increasing the content of SOC enhances the risk of N_2O emissions (Li et al., 2005). This is true whether the soil has a high content of SOC or the content is increased by additions of organic matter to the soil and is caused by the tight link between SOC and microbial N_2O production (Sahrawat and Keeney, 1986). Because the impact of SOC on N_2O emission is dependent on NO_3 content in soil (e.g. Weier et al., 1992; Li et al., 2005), we address how supply of N and carbon through organic inputs drive N_2O emissions in organic arable rotations. Based on the IPCC (2006), most inventories and farm

30   models assume that 1% of total N-input by fertilizers, manure and plant residues are emitted as N_2O-N. Skinner et al. (2014) found no correlation between total N-input and N_2O emission in organic systems. If this is a general trend, the total N-input cannot be used to estimate N_2O emissions from organic crop rotations. In non-organic cropping systems, peak fluxes of N_2O are commonly observed shortly after fertilization with mineral fertilizers in moist soil (Smith et al., 2012), whereas in organic crop rotations the highest fluxes are often observed after incorporation of plant residues (e.g. Brozyna et al., 2013;

35   Krauss et al., 2017b; Nadeem et al., 2012; Pappa et al., 2011; Skinner et al., 2019). Because of enhanced content of SOM and thus a larger impact of background emissions of N_2O in organic versus non-organic cropping systems (Skinner et al., 2014), increased background emissions are likely to have a major impact in organic-crop rotations. In order to design good mitigation strategies, it is useful to know the relative importance of these two sources of N_2O emissions.

The main sources for NO_3 leaching are NO_3 from nitrification of plant residues and added organic matter, as NO_3-fertilizers

40   are prohibited in organic systems. The crop N requirement dependency on soil organic matter turnover may lead to an asynchrony between crop nutrient demand and the mineralization of soil organic N, which enhances the risk of NO_3 leaching (Crew et al., 2005; Di and Cameron, 2002). Furthermore, soil cultivation for weeding or incorporation of plant residues have been shown to influence NO_3 leaching (Askegaard et al., 2011). We discuss the main drivers for NO_3 leaching in organic

arable cropping systems and the associated preventative measures. We address the following questions for organic arable crop rotations:

1. How does supply and quality of organic matter in above- and below ground residues and organic amendments influence availability of easily available N and degradable C?

2. How does supply of easily available N and degradable C drive N_2O emissions and how can these be mitigated?

3. Is there a lack of correlation between total N-input and N_2O emission in organic arable crop rotations, and are total N_2O emissions primarily driven by background emissions or by episodes with high N_2O fluxes following N additions?

4. What are the main drivers for NO_3 leaching in organic arable systems and how can the leaching be reduced?

2. Methodology

10 Based on the authors own field trials, literature databases and searches through Google Scholar, we compiled data on agronomic management, soil properties and yield level of organic arable crop rotations and measurements of SMN, N_2O emissions and NO_3 leaching from field trials relevant to organic crop rotations, climate and soil conditions in Europe. For SMN and NO_3 leaching, we used the available literature and the data in Tables S1 and S3 to explore the importance of the determining factors for SMN and NO_3 leaching, and to identify factors of importance for NO_3 leaching in organic arable

15 crop rotations. For SMN and NO_3 leaching the structure of the available data did not allow meaningful statistical analyses. For N_2O we used data presented in Table S2 to explore the impact of background emissions and episodes with high N_2O fluxes.

We used the available literature to explore the impact of total N added and, N and C added through organic inputs from living plants, plant residues and organic fertilizers on N_2O emission as we did not have enough data on added N to be able to

20 include this in the regression analyses.

We aimed to analyse the impact of high emission events of N_2O fluxes on the total N_2O emission. However, we lacked daily measurements, and we lacked data for yearly periods. It was thus not possible to identify the full impact of hot moments as done by Molodovskaya et al. (2012). Because of the differences in measurement period, it was also not possible to make direct comparisons between the different field trials. To overcome this, we used a regression model based on N_2O emissions

25 in the actual period and peak N_2O flux within this period, resulting in the following fitted model:

$$\ln\left(\frac{C}{P}\right) = -1.34 + 0.83 \cdot \ln\left(F_{max} + 2\right), \quad R^2 = 0.66, \quad n = 97 \tag{Eq. 1}$$

F is the cumulated N_2O flux (emission) in the measurement period (-278 to 8566 g N_2O-N ha^{-1}); t is duration of the period (38 to 490 days); F/t express the average daily N_2O flux (-1.3 to 53.2 g N_2O-N ha^{-1} d^{-1}) and F_{max} is the highest N_2O-flux rate (0.1 to 605 g N_2O-N ha^{-1} d^{-1}) in the measurement period. The total analysis and data used is given in S4. One negative value

30 for average daily N_2O flux (-1.3) in a barley/pea crop was removed from the analyses, since it would not have had a large impact on the results.

We also calculated the percentage contribution of the highest daily N_2O flux of the total N_2O emissions in the measurement periods for all trials presented in Table S2 (n=97), and correspondingly the sum of the fluxes for the days with the five highest flux rates as a percentage of the total N_2O emissions. The choice of five days was to represent what typically

35 constitutes a peak emission event.

The impact of the following potential explanatory variables on the highest daily N_2O flux rates: clay (soil clay content %), pH (soil pH), SOC (soil organic C, g kg^{-1} dry soil), WFPS (soil water filled pore space, %), NO_3 (soil content of NO_3, kg NO_3-N ha^{-1}) and NH_4 (soil content of NH_4, kg NH_4-N ha^{-1}) and temp (soil temperature, °C) was calculated by stepwise regression (α to enter = 0.15; α to remove = 0.15) by Minitab 18.1, © 2017 Minitab, Inc. The selection of these variables was

40 based on expected impact on N_2O emissions (Sect. 4.1) and the data we were able to obtain from the following studies: Ball

et al. (2007a), Baral et al. (2017), Brozyna et al. (2013), Chirinda et al. (2010), Krauss et al. (2017b), Li et al. (2015a), Nadeem et al. (2012b) and Pugesgaard et al. (2017). The N$_2$O flux data were log-transformed to achieve near normality and variance homogeneity: ln (daily N$_2$O flux +2), where daily N$_2$O flux = g N$_2$O-N ha^{-1}d^{-1} is the highest N$_2$O flux rate during the actual measurement period (Table S2, Highest daily flux rate). The highest flux rates were chosen for analysis, since we wanted the extreme values to explore which factors are mostly influential for hot moments of N$_2$O emissions. The total analysis and data used is given in S5.

Based on the stepwise regression, we achieved the following fitted regression model:

$$\ln\left(\text{daily } N_2O \text{ flux} + 2\right) = -1.18 + 0.04 \cdot clay + 0.06 \cdot SOC + 0.03 \cdot WFPS + 0.01 \cdot NO_3 + 0.05 \cdot temp, \quad R^2 = 0.71, \quad n = 66$$

(Eq. 2)

3 Drivers of SMN and degradable C in organic arable crop rotations

3.1 Supply and quality of soil organic matter

Return of crop residues to the soil is standard practice in both organic and non-organic arable production; however, because of the more diverse crop rotations in organic production systems, larger and more diverse inputs of herbage from legume-based green manures, leys, cover crops (CC) and intercrops are returned to soil than in non-organic cropping systems commonly used in Europe (Gattinger et al., 2012). The N content in the crop residues of legume-based systems is typically higher than from non-legume systems (Watson et al., 2002). Commonly used external sources of organic inputs in organic cropping systems are animal manures and slurries, composts or biogas residues, and organic fertilizers based on animal manure or municipal waste (Løes et al., 2017). The great diversity of N mineralisation patterns among the organic fertilizers and crop residues result in a large variation in how much N and carbon (C) are rapidly degradable in the organic inputs.

Through the application of organic amendments and various crop residues from arable and forage crops, C and N is applied to soil, and the soil organic matter (SOM) content is often higher in organic than in non-organic arable crop rotations (Marinari et al., 2007: about 40% more total organic C short time after application of organic compared with mineral fertilizer; Marriott and Wander, 2006: Concentrations of SOC was about 14% higher in organic than non-organic systems; Gattinger et al., 2012: 3.5 Mg C more in SOC stocks in organic compared with non-organic production in a global meta-analysis; Aguilera et al., 2013: SOC concentration 19% higher in a meta-analysis from Mediterranean; Hu et al., 2018: 0.4 Mg C ha^{-1}yr^{-1} more SOC accumulated with organic than non-organic treatment at Foulum, but 0.4 Mg less C than non-organic at Flakkebjerg in long-term field trials; Pimentel et al., 2005: 15% higher SOC concentrations in legume-based organic versus non-organic crop rotation in a long-term field trial). The quality of SOM differs between non-organic and the more diversified organic, arable crop rotation, with a higher share of labile SOM (Lynch, 2015) and thus easily degradable organic matter (C and N) in soils in organic crop rotations (Marinari et al., 2007; Marriott and Wander, 2006; Martyniuk et al., 2016). The higher content of degradable SOM in organic crop rotations is a valuable soil fertility asset as it provides a short-term pool for plant nutrient supply (Marriott and Wander, 2006; Martyniuk et al., 2016). SOM turnover rates vary with soil texture and climate, being higher when organic carbon is less protected from decomposers (low clay content) and in warm climates with suitable moisture (Burke et al., 1989).

3.2 Soil biological activity

Soil microbes contribute directly to plant residue decomposition and to mineralization and turnover of SOM, as well as earthworms and other soil fauna (Lubbers et al., 2013, Kuiper et al., 2013). The inflow of degradable organic matter provides substrate for soil organisms, and application of organic matter increases the growth of microbial communities, their enzyme activities and the microbial diversity compared to an unfertilised control or soil fertilized with only mineral fertilizer

(Anderson and Domsch, 1989; Marinari et al., 2007; Thangarajan et al., 2013), although such changes in the topsoil may be slow processes (Petersen et al., 2013). Accordingly, higher biological activity has commonly been found in arable soils managed organically compared with non-organically (Mäder et al., 2002; Gomiero, 2013; Hartmann et al., 2015; Lori et al., 2017). In their meta-analysis, Lori et al. (2017) found that organic systems had 32% to 82% greater microbial biomass C, microbial biomass N, total phospholipid fatty-acids, and dehydrogenase, urease and protease activities than conventional systems. They found that when both organic and non-organic systems included legumes, the organic system displayed a higher microbial N content than the non-organic counterpart. In cases where only the organic systems contained legumes, the difference in microbial N between the two systems was even more pronounced. The abundance of earthworms can be twice as high in organic compared with non-organic systems (Pfiffner and Mäder., 1997; Filser et al., 1999; Hansen and Engelstad, 1999; Riley et al., 2008). More abundant earthworm populations are found when large amounts of animal manure or green manure are applied to soil (Hansen and Engelstad, 1999; Frøseth et al., 2014), when autumn ploughing is avoided (Pfiffner and Luka, 2007) and in the absence of tractor traffic (Hansen and Engelstad, 1999).

3.3 Soil N-dynamics within organic arable cropping systems

Crop N supply and SMN in organic farming relies largely on mineralisation of N in soil organic matter, N in organic amendments and crop residues and BNF of legume-based crops (Gattinger et al., 2012; Lorenz and Lal, 2016). A principal sketch for N-dynamics in organic systems is given in Fig. 1 and an overview of performance of selected drivers of SMN in organic and non-organic crop rotations are given in Table 1.
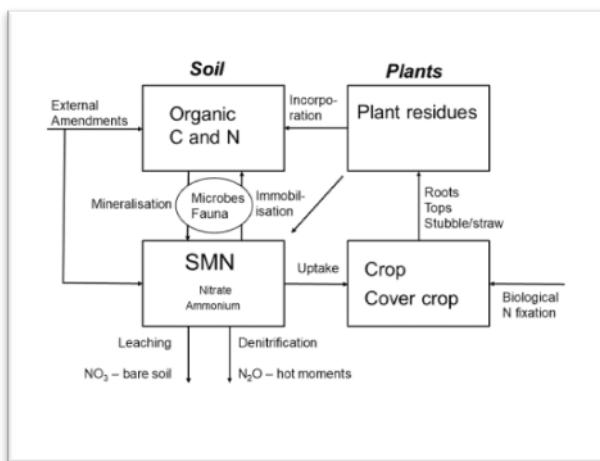
Fig.1. The principle of N-dynamic in organic crop rotations

Table 1. Performance of selected drivers of SMN in organic and non-organic cropping systems

Indicator	Organic arable systems	Conventional arable systems	Source
Main source of SMN	Mineralization of crop residues and soil organic matter	Synthetic fertilizers	*Lorenz and Lal (2016)*
BNF in SMN supply	High	Low	*Kayser et al. (2010); Pandey et al. (2017)*

Size of the potentially mineralizable N pool	Usually high	Usually medium	*Poudel et al. (2002)*
Microbial biomass nitrogen content	High	Low	*Lori et al. (2017)*
Release of SMN	Slow and more continuous	Rapid with clear peaks	*Poudel et al. (2002)*
Concentration of SMN during growing season	Usually low	Usually high	*Brozyna et al. (2013) Frøseth et al. (2014) Krauss et al. (2017b)*
Concentration of SMN off growing season	Usually low, high only after termination of legume-rich crops	Usually high	*Hansen et al. (2007) Jończyk and Martyniuk (2017)*

Despite substantial inputs of N from BNF (Kayser et al., 2010; Pandey et al., 2017) and from organic amendments, the N-supply is often below optimum for plant growth in arable organic farming (Berry et al., 2002; Tuomisto et al., 2012). BNF may be a more important N source, than N from organic manures and fertilizers in organic arable cropping systems (Pandey et al., 2018).

The major release of plant available N from added organic matter depends on the C:N ratio, the mineral N content and degradability of the C of added plant residues manure, compost and other decomposed amendments. Bhogal et al. (2016) showed that for pig slurry and poultry layer manure with C:N_{org} of 9-12:1, up to 70% of the organic N was mineralized after five growing seasons, whereas in cattle slurry and straw-based farmyard manure with C:N_{org} of 10-21:1, only 10-30% of N were net mineralised. For crop residues with a high C:N content there will be initial immobilisation and the net mineralisation may only start very late or after the growing season of the main crop (Li et al., 2015b). The mineralization process of the more stable N can continue over years to decades. Simulation modelling has shown that even over a 20-year period, only 10-15% of organic N in applied manure may be taken up by crops, the rest being lost or retained in soil organic matter (Berntsen et al., 2007). The high microbial activity and high content of organic matter affect N-cycling. Hu et al. (2018b) found a higher net N mineralisation of added organic matter in soils having a prehistory of use of CC, indicating positive legacy effects of CC use, which was attributed to a greater microbial biomass N in systems with use of CC.

The design of the rotation, as well as its management, influences the PMN pool. Working in three different organic arable systems, Spargo et al. (2011) showed that the PMN pool amounted on average for 315 kg N ha^{-1}. They showed that more diversified crop rotations resulted in larger PMN pools. Poudel et al. (2002) reported 112 and 56% greater PMN pool in the organic system in comparison to the conventional and low-input systems, respectively. Moreover, they observed slower and more continuous release of mineral N in the organic systems compared to the more rapid release of mineral N from synthetic N fertilizers applied in non-organic systems. Moyo et al. (2016) reported higher PMN in soils under wheat following a cut and mulched red clover ley compared to after a ley where the residues had been removed. This indicates the importance of total N input to the soil. The importance of total N input in plant residues for crop N uptake was also observed by Petersen et al. (2013).

SMN was found to be very low under grass-clover leys (Table S1, Watson et al., 1993; Nadeem et al., 2012; Brozyna et al., 2013; Frøseth et al., 2014; Krauss et al., 2017b), since grasses quickly take up soil NO_3 in the root zone (Brophy et al.,

1987). Frøseth et al. (2014) observed low levels of SMN in a 1-year grass-clover ley, irrespective of whether the herbage was mulched or removed. After termination of a ley, the concentration of SMN usually increases (Table S1, Ball et al., 2007a; Brozyna et al., 2013; Krauss et al., 2017b). Even in the year following termination of a ley, the content of SMN can still be high (Hansen et al., 2007; Jończyk and Martyniuk, 2017). Kayser et al. (2010) pointed out that N provided by spring

5 ploughing of both 1-year grass-clover ley and 3-year grassland ley resulted in high concentrations of SMN (0-90 cm, 61 kg N ha^{-1} and 95 kg N ha^{-1}, respectively) in the following autumn after harvest of spring triticale. Much of this SMN may not have been available for the crops during spring and is likely to have been mineralised after the end of the growing season. In contrast, low concentrations of SMN were observed in topsoil and subsoil after spring ploughing of a 1-year grass-clover ley in four field trials in Norway, in late spring and in autumn after harvest of a spring barley crop (Table S1) and in the

10 following year (Frøseth, 2014, 2016).

4 Drivers of N$_2$O emissions in organic arable crop rotations

4.1 Mechanisms for N$_2$O emissions

Many processes contribute to N$_2$O production in soils, but the dominant mechanisms for N$_2$O emissions from terrestrial agricultural soils are the microbial processes of nitrification, nitrifier denitrification (as a result of incomplete nitrification)

15 and denitrification (Firestone and Davidson, 1989; Butterbach-Bahl et al., 2013). Nitrification and denitrification are both biological processes, thus the same mechanisms will cause N$_2$O emissions in organic as in non-organic farming systems. However, fertilisation, crop and soil management practices differ substantially between these two systems (Sect. 3), and the relative importance of the relevant triggers therefore differ (Sect. 6).

Nitrification is the microbial oxidation of NH$_3$ to NO$_2$ and ultimately NO$_3$, where N$_2$O is produced as by-product through

20 some partially understood biotic and abiotic reactions of hydroxylamine (Anderson, 1964, Liu et al., 2017). Nitrifier denitrification occurs when NO$_2$ produced during nitrification is reduced to N$_2$O (by denitrifying organisms), instead of being oxidized to NO$_3$, under fluctuating oxic-anoxic conditions (Firestone and Davidson, 1989). Denitrification is the microbial anaerobic reduction of NO$_3$ via NO$_2$ to gaseous NO, N$_2$O and N$_2$, which are ultimately transported through the soil to the atmosphere. Denitrification is the main source of N$_2$O production in soils, as N$_2$O yield potential of denitrification is

25 much higher (1-100%) than that of nitrification (0.1-1%) (e.g., Andersson et al., 1993, Butterbach-Bahl et al., 2013).

The ratio between the gaseous products of denitrification depends on NO$_3$ availability, oxygen availability in the soil, amount of easily decomposable carbon as an energy source, soil pH and microbial community structure (Bakken et al., 2012). Oxygen availability depends on soil microbial activity and gas diffusivity, which depends on soil moisture content, texture and density. Gas diffusivity is a promising predictor for N$_2$O fluxes from soils with varying bulk density as observed

30 by Balaine et al. (2013), who found that the production of N$_2$O increased when the relative gas diffusivity was between 0.006 and 0.020 and the soil became anaerobic.

As explained above, risk of N$_2$O emissions increases as soil carbon content increases (Li et al., 2005). N$_2$O and N$_2$ production correlates with total organic C, water soluble C and mineralizable C in soil, but increased availability of C also decreases the ratio of N$_2$O/N$_2$ (Sahrawat and Keeney, 1986). Emissions from soil organic matter (background emission) will

35 vary between years because of variations in temperature and precipitation (Hansen et al., 2014; Brozyna et al., 2013) and between different categories of crops due to different time windows with high SMN (Dobbie and Smith, 2003).

Organic amendments and plant residues that provide carbon that is easily decomposable by microbes may enhance microbial activity and deplete soil oxygen through enhanced soil respiration. In addition, degradable carbon is an energy source for denitrifying bacteria. In accordance with this, Köster et al. (2011) concluded that bacterial denitrification was the main

40 process for producing N$_2$O during the first three weeks after application of biogas residues, and high carbon availability was an important cause for this. Li et al. (2016) concluded that denitrification was the main cause for N$_2$O emission after addition

of legume-based residues. Several studies have shown higher rates of N loss through denitrification from soils treated with organic amendments such as manure, composts, and plant residues when compared to unamended or mineral N treated soils (Thangarajan et al., 2013). In line with this, the incorporation of residues by tillage increases soil respiration and N_2O fluxes because of microbial stimulation (Krauss, 2017a).

5 Low soil pH inhibits the activity of the N_2O reductase enzyme and thus $N_2O:N_2$ ratio increases (Liu et al., 2010). At higher soil pH, the denitrification rate is higher, but the $N_2O:N_2$ ratio is lower as a greater part is completely denitrified to N_2. At low temperatures, nitrous oxide reductase is hampered (Holtan-Hartwig et al., 2002), but on the other hand, denitrification rates are also reduced (Butterbach-Bahl et al., 2013).

4.2 Legumes during active plant growth

10 In general, unfertilized legumes have small N_2O emissions during their growing period, particularly when grown in mixtures with non-legumes. Low N_2O emission are found during growth of grain legumes (Dusenbury et al., 2008; Jensen et al., 2012; Jeuffroy et al., 2013; Pappa et al., 2011; Rochette and Janzen, 2005), green manure crops and CC (Baggs et al., 2000b; Brozyna et al., 2013; Peyrard et al., 2016; Li et al., 2015a; Shelton et al., 2018) as well as for grass-clover leys (Baggs et al., 2000b; Ball et al., 2002; Brozyna et al., 2013; Skinner et al., 2019; Krauss et al., 2017b; Nadeem et al., 2012). This is 15 consistent with low SMN concentrations during growth (Sect. 3) and negligible N_2O emissions associated with BNF by the legume rhizobium symbioses (Rochette and Janzen, 2005; Carter and Ambus, 2006). However, following termination or senescence of the legume crops, reactive N released from dying roots and nodules may lead to enhanced N_2O emission (Rochette and Janzen, 2005).

Surface mulching of harvested herbage may theoretically enhance N_2O emissions due to mineral N released from the 20 herbage. However, several studies show that mulching of grass-clover herbage on the growing ley only causes a slight increase in N_2O emissions (Brozyna et al., 2013; Möller and Stinner, 2009; Nadeem et al., 2012). None of these studies measured ammonia volatilization from mulched herbage, which could have been a major loss of mineralized N corresponding to the findings of Larsson et al. (1998). Volatilized NH_3 will be redeposited elsewhere and may result in increased N_2O formation downstream (IPCC, 2006).

25 **4.3 Crop residues**

As outlined in Sect. 4.1, there is an enhanced risk of N_2O emissions from agricultural soils when easily degradable carbon and N are simultaneously available, and denitrification is probably the main source for this. Because legume-based crop residues also increase SMN (Sect. 3.3), increased N_2O emissions have been reported in field trials whether the residues are from grain legumes (Jeuffroy et al., 2013; Pappa et al., 2011), grass-clover (Baggs et al., 2000b; Ball et al., 2007a; Skinner et 30 al., 2019; Nadeem et al., 2012; Brozyna et al., 2013), intercropped clover (Pappa et al., 2011) or CCs (Baggs et al., 2000b; Peyrard et al., 2016, Pugesgaard et al., 2017). The increase of N_2O fluxes after incorporation of crop residues and other plant material might however, be small and a negligible part of the total N_2O emissions (Peyrard et al., 2016; Pugesgaard et al., 2017; Shelton et al., 2018).

The C/N ratio of incorporated herbage does affect N_2O emissions, with higher emissions expected from herbage having low 35 C/N-ratio (Chen et al., 2013). From this, one should expect higher N_2O emission from legume residues than from cereals or grasses (e.g. Rochette and Janzen, 2005). However, Larsson et al. (1998) observed the same N_2O-N EF (1% of applied-N) from mulched alfalfa (C/N-ratio 11) as from mulched grass with a C/N ratio of 21, but higher EF than from a mulched grass with a low N-content (C/N-ratio 36, EF = 0.1%). The N_2O fluxes might be high despite a high C/N-ratio when the carbon source is easily degradable as observed by fodder radish by Li et al. (2015a) (Table S2).

40 N_2O emissions may also be associated with previous incorporation of plant residues. In accordance with this, Skinner et al. (2019) observed enhanced N_2O fluxes after a maize crop succeeding a grass-clover ley. Measurements of N_2O fluxes shortly

after incorporation of plant material, or measurements in the following year, only tell part of the story. Enhanced content of various fractions of SOM derived from crop residues, ley and CC (Sect. 3.2) are likely to increase the long-term background emissions of N_2O (Sect. 4.1). In a ten-year-old field experiment with and without legume-rich CC in the crop rotation, Pugesgaard et al. (2017) concluded that crop residues were important source of N_2O, and that mineralizable C, rather than N input, was the main driver for N_2O emission. Contrary to this, Peyrard et al. (2016) observed in a three-year low-input field trial that although N_2O fluxes increased for a few days after incorporation of CC, the contribution of such events to cumulative N_2O emissions were negligible. In their study, however, the CC treatments started when the N_2O measurements started. More studies in long-term experiments with continuous use of CC are needed to verify the actual impact of crop residues in a long-term perspective in various field situations, because addition of plant material to soil also affect soil structure, soil biological activity and N turnover.

4.3.1 Freeze/thaw – dry/wet

The mechanisms behind freeze/thaw have been comprehensively reviewed by Congreves et al. (2018) showing that the causes for N_2O emissions are different for these two mechanisms and that freeze/thaw cycles have a larger impact on N_2O emissions in temperate agroecosystems than drying/rewetting. Wagner-Riddle et al. (2017) estimated that by neglecting freeze/thaw N_2O emissions global agricultural N_2O emissions are underestimated by 17 to 28%. Freezing/thawing of soil rich in organic matter and soil biota, or soil covered with plant residues may result in a N_2O boost as easily degradable C and N is released from cells through lysis after frost. As summarized in the introduction and in Sect. 3 these conditions are particularly relevant for organic crop rotations. Flessa et al. (1995) observed that 46% of total annual N_2O emissions from a sunflower crop, solely fertilized with farmyard manure (12 Mg ha^{-1}) occurred during December and January, mainly due to high N_2O peak fluxes (650 g N_2O-N ha^{-1}d^{-1}) after thawing of the first freezing period during winter. Correspondingly, Westphal et al. (2018) did not observe any enhanced N_2O fluxes after late summer incorporation of a ley dominated by alfalfa (0-10 g N_2O-N ha^{-1} d^{-1}), but fluxes were greatly enhanced during spring thaw in the following year (60 g N_2O-N ha^{-1}d^{-1}).

When CCs are killed by frost, N_2O fluxes will increase during thawing of the soil due to the release of easily degradable C and N in the plant material. Li et al. (2015a) observed significantly higher N_2O emissions from the frost-sensitive fodder radish rich in readily degradable carbon than from other less frost sensitive CCs. Winter emissions were even greater when fodder radish was harvested in late autumn (30 October), leaving only roots and stubble (Table S2). This suggests that N and C in roots of frost sensitive CCs can be an important source for N_2O emissions after thawing. Also, in leys frost may enhance N_2O emissions. Sturite et al. (2014) observed that enhanced N_2O emission during thawing of a frozen grass-clover ley correlated with clover content in the ley.

Under drought conditions, the nitrification process prevails and N_2O is produced at very low rates. However, with rewetting easily degradable N and C is mineralized, resulting in increased N_2O fluxes. Hansen et al. (2014) observed that the N_2O flux increased with increasing clover content during rewetting of a grass-clover ley after drought. Hence, both freezing/thawing and rewetting may have a large impact in organic systems.

4.3.2 Soil and tillage effects

N_2O emissions associated with crop residues are affected by tillage depth and soil type. Large N_2O emissions have been observed when crop residues are placed near the soil surface in heavy soil (Peyrard et al., 2016, Krauss et al., 2017b) whereas in lighter soil types the highest N_2O emissions have been observed either after rotary harrow (Baggs et al., 2000a) or after ploughing (Petersen et al., 2011). When the crop residues are squeezed and mixed with a rotary harrow, easily available N and degradable C become available for denitrifying bacteria in the soil and the potential for denitrification is large. In line with this, Krauss et al. (2017b) observed high N_2O fluxes few days after weeds and crop residues were superficially

incorporated with a rotary harrow in a moist calcareous clay soil (WFPS 80 %) (Table S2, highest observed peak in single plot 800 g N_2O-N ha^{-1}d^{-1}). Similarly, Peyrard et al. (2016) observed enhanced N_2O fluxes (max rates 60 g N_2O-N ha^{-1}d^{-1}) up to several days after crop destruction when crop residues (sunflower, wheat, faba bean) were mulched or placed near the soil surface of a calcareous-clay, but not by ploughing or mechanical weeding. Baggs et al. (2000a) observed higher N_2O fluxes when lettuce residues were incorporated by rotary harrow than by ploughing (peak of 67 g N_2O-N ha^{-1}d^{-1}).

Restricted gas diffusivity is another possible explanation for the observed lower N_2O fluxes with deep incorporation of crop residues in dense soil. With reduced gas diffusivity more N_2O is likely reduced to N_2, in accordance with a general trend of a larger ratio of N_2O-N/(N_2O-N+N_2-N) close to the soil surface and smaller fluxes deeper in the soil profile (Sahrawat and Keeney, 1986). Kuntz et al. (2016) observed decreased O_2 concentration at 8 cm soil depth and a corresponding reduction of N_2O to N_2 with surface application of carbon rich material. As another example, Petersen et al. (2011) found in their loamy sandy soil that the largest fluxes were observed when residues were incorporated by ploughing compared to reduced tillage. Possible explanations for this could be that residues came directly into contact with mineral N from the injected slurry after ploughing, thus fostering enhanced microbial turnover of C and N and that in this soil, the aeration with O_2 was still available at plough depth.

4.4 Organic fertilizers

Organic fertilizers vary widely in the content and types of N and C compounds causing large variations in N_2O emissions after application. Animal slurries have a higher content of NH_4-N and contain more easily degradable N and C than solid manures and composts and are thus stronger triggers for rapid N_2O emissions shortly after application (Charles et al., 2017, Sect. 3). In accordance with this, Krauss et al. (2017b) observed higher N_2O emissions shortly after application of cattle slurry than composted solid cattle manure. Correspondingly, in a field experiment with spring barley fertilized with various organic slurries, Baral et al. (2017) observed the highest N_2O EF in the treatment with highest application of organic matter, and thus highest content of easily degradable C. Meijide et al. (2007) and Chantigny et al. (2007) found that the use of digested slurry, with lower content of degradable C compared to untreated pig slurry, reduced soil N_2O emissions by 25 and 50% respectively.

The effect of organic fertilizers does depend on soil type and content of SOC. Degradable C applied with organic fertilizers will to a greater extent trigger microbial respiration and denitrification in a soil with low content of SOC than in a soil with high content of SOC (Chantigny et al., 2010, Pelster et al., 2012), whereas in a soil with high content of SOC the impact of easily available N is higher (Petersen et al., 2008). Because of a higher content of SOC and a higher share of labile SOM in organic crop rotations compared with non-organic crop rotations (Sect. 3.1), the short-term effect of organic fertilizers with a high content of degradable C on N_2O emission are likely lower in organic than in non-organic crop rotations.

Also, the absence of synthetic fertilizers containing easily available N means that organic fertilizers are likely to have a smaller short-term impact on N_2O emission in organic than in non-organic crop production. Charles et al. (2017) found in a meta-analysis that the N_2O EF was higher when soils received organic amendments in combination with synthetic fertilizers. They found EFs for liquid manures + synthetic fertilizers: 2.14 % ($\pm$ 0.53), composts + synthetic fertilizers: 0.37 % ($\pm$ 0.24), and the corresponding EFs for manure: 1.12 % ($\pm$0 .18) and compost: 0.00% ($\pm$ 0.17).

However, the long-term impact of manures is not included in these EFs. In contrast to the short-term fertilizer effect, a long-term fertilization with organic fertilizers may enhance N_2O emissions through enhanced background emissions. Chang et al. (1998) observed that annual N_2O emissions increased with manure rate when different rates of solid feedlot manure and thus N-application were applied for 21 years. Their manure rate was far greater than would be applied under organic farming conditions, but the possibility for enhanced background N_2O emissions after long-term input of organic matter through manuring and application of crop residues should be considered. Krauss et al. (2017b) found that fertilization with slurry and manure compost increased annual N_2O emissions during winter wheat after more than ten years of differentiated

management compared to sole slurry fertilization (mean values in the period (369 days), were 2.2 and 2.9 kg N_2O-N ha^{-1}, respectively). They related this to higher microbial biomass and SOC. Mean values for the upper 10 cm in soil were 28 and 30 Mg C ha^{-1} for fertilization with slurry and manure compost and sole slurry fertilization, respectively.

4.5 Contribution of total N-input and high emission events to N_2O emissions

Skinner et al. (2014) concluded in a review that soil characteristics (soil N content) had a greater impact on N_2O emissions from organic production than the total-N input by fertilization. Lack of correlation between N_2O emissions and N fertilization in organic production corresponds to the more recent findings of Krauss et al. (2017b) and Pugesgaard et al. (2017). Pugesgaard et al. (2017) observed no significant correlation between N_2O emissions and N input in fertilizer/manure, for either annual N_2O emissions or spring emissions, but N_2O emissions were correlated with N input in residues from the previous main crop and CC. This agrees with the findings of Bouwman et al. (2002), van Groenigen et al. (2010), Peyrard et al. (2016) and Shcherbak et al. (2014), who observed low EF's when N fertilization was below optimum as commonly found in organic production systems.

As discussed earlier, N_2O emission in organic crop rotations are driven by enhanced background emissions from long term input of organic matter, and episodes with enhanced N_2O emission after application of crop residues or organic fertilizer, but are N_2O emissions primarily driven by background emissions or by episodes with high N_2O fluxes? We used data from mainly organic field trials (Table S2) and one non-organic trial fertilized with organic fertilizers (Baral et al., 2017) to calculate the impact of the highest N_2O fluxes on the total N_2O emissions. In Frick (CH), Edinburgh (UK), Aberdeen (UK) and Ås (NO), the highest daily flux rates were 605, 211, 297 and 94 g N_2O-N ha^{-1}d^{-1}, respectively (Table S2). Because of the high flux rates, we hypothesised that high emission were responsible for a major part of the N_2O emissions from these systems. The single days with the highest fluxes correspond to 18% (65 days measurement period) in Frick, 2% in Edinburgh (161 days), 17 % in Aberdeen (38 days), and 2% in Ås (218 days) of the cumulated N_2O emissions in the measurement periods. The five highest daily N_2O fluxes corresponded to 22, 7, 55 and 5% of the N_2O emission in the measurement periods in these investigations, respectively. In field trials conducted on well-structured sandy loams at either Foulum or Flakkebjerg in Denmark (Table S2, 105 to 365 days), peak N_2O fluxes from one or five days, however, only constituted from <1-8% and 5-14% of the total emission in the periods, respectively. The highest daily flux rate in these trials was only 78 g N_2O-N ha^{-1} d^{-1}. This was in a non-organic treatment heavily fertilized with cattle slurry and digested sewage sludge (476 kg total N ha^{-1}, Table S2 (Baral et al., 2017)). From this we reject the hypothesis that high emission events were responsible for a major part of the N_2O emissions from these systems, rather background emissions seemed to be the major N_2O source. However, a simple regression model (Eq. 1) showed that the average daily N_2O flux correlated positively with episodes with high N_2O fluxes. From this we can conclude that when the conditions for high N_2O fluxes are met for one or more days, there is a large chance for high total N_2O emissions in the period. The small peaks in the Danish field trials reveals that well-drained sandy soils promote rapid water infiltration and good gas-diffusivity and in turn low N_2O emissions.

We wanted to explore the impact of soil conditions on the highest N_2O flux peak. In a stepwise regression (Eq. 2), the content of clay, SOC, NO_3-N in soil and soil temperature had significant positive impacts on peak N_2O fluxes in the selected investigations (Table S2). (P_{clay}, $P_{SOC} < 0.001$, P_{NO3}, $P_{temperature} < 0.01$, $P_{WFPS} < 0.05$). The content of NH_4-N in soil did not affect peak N_2O fluxes. These findings indicate that denitrification is the main cause for high N_2O-flux rates in these studies. To visualize the impact of different factors on the mean and the highest N_2O flux, and the contribution of the 5 highest daily fluxes to the cumulated N_2O emissions in the measurement period we grouped the investigations according to % clay in soil, N added with organic fertilizers, SOC, soil pH, experimental period, type of crop and mean daily precipitation in the period (Fig. 2). Because of lack of information, we were neither able to include total-N input nor soil porosity in the regression analyses and the box-plots.

Fig.2. Box-plot (mean -, median x, upper and lower quartile) for (a) mean N$_2$O flux in the period (g N$_2$O-N ha^{-1}d^{-1}) and the highest daily flux rate in the same period (g N$_2$O-N ha^{-1}d^{-1}) and (b) % contribution of the 5 highest daily fluxes to cumulated N$_2$O emissions in the period. N=number of sites, n=number of observations

In line with the findings of Skinner et al. (2014) we did not find a clear impact of N added with organic fertilizers, but the soils with the lowest content of SOC, showed the lowest mean and lowest peak N_2O emission. This supports the hypothesis that the SOC content of the soil, and thus the content of SOM, are important drivers for N_2O emission. At high SOC contents other factors seem to be more important. The impact of type of crop on N_2O emissions do mainly follow the trend we have seen in other investigations (Sect. 5.2, 5.3) with higher N_2O emissions when cereals succeed a short-term ley (grass-clover or clover) and that mixtures with monocotyledons and legumes do not have higher N_2O emissions than cultures with only monocotyledons. The high residual effects of previous leys support the idea that background emissions are the main driving force for N_2O emission in organic crop rotations. The high N_2O emissions (up to 600 g N_2O-N ha^{-1} d^{-1}) from one occasion in Switzerland with incorporation of weeds in wet soil have a large impact on all box-plots (Fig. 2). None of the other groupings in Fig. 2 showed any clear causal effect on N_2O emissions.

The low effect we observed of pH on N_2O emissions (Eq. 2, Fig. 2) is contradictory to enhanced N_2O emission caused by hampered N_2O reductase enzyme often observed in acid soils (Bakken et al., 2012). A reason for this could be that in these soils and with these managements, other factors meant more for N_2O emissions than pH. Although denitrification was likely to have been the main cause of N_2O fluxes at the highest N_2O flux rates, the soil NO_3 concentration are often low between the episodes with high N_2O flux rates, as observed by Brozyna et al. (2013), Chirinda et al. (2010), Li et al. (2015a), Nadeem et al. (2012) and Pappa et al. (2008).

4.6 Impacts of earthworms

Abundant earthworm populations in organic crop rotations (Sect. 3.3) are likely to influence N_2O fluxes as they significantly affect mineralization and reduction of N compounds to N_2O and N_2 (Prieto, 2011). N_2O is emitted from intestinal microbes but is also released from nitrates emitted in body fluids in the earthworm gut as well as from casts, middens and burrows (Prieto, 2011). On the other hand, earthworms improve soil porosity and aggregate stability (Bronick and Lal, 2005) and thus gas-diffusivity and water infiltration in soils, which will reduce N_2O emissions. Epigeic species (living near surface and feeding on surface litter) and anecic species (deep burrowing) are well known to enhance N_2O production, because they feed directly on decomposing herbage (Evers et al., 2010; Nebert et al., 2011; Lubbers et al., 2011). Endogeic earthworms that feed on SOM particles are most common in cultivated arable soils (Hansen and Engelstad, 1999), and they do not increase denitrification (Postma-Blaauw et al., 2006). There are too few published results to robustly predict the impact of earthworms in arable organic crop rotations on N_2O emissions as this will depend on local climatic and edaphic conditions.

5 Drivers of NO_3 leaching in organic arable crop rotations

5.1 Mechanisms for NO_3 leaching

NO_3 leaching is an abiotic process driven by diffusion and convection (e.g. Johnsson et al., 1987), where NO_3 is transported out of the root zone along with the downward water flow. In addition to soil water content, soil texture and structure are important in determining leaching rates. Fine textured soils have slower infiltration rates than coarse textured soils, and porous sandy soils are most vulnerable to leaching, also because these soils often have more shallow rooting depths than loamy soils (Askegaard et al., 2005). The impact of soil type was clearly demonstrated by Askegaard et al. (2011) who found that, depending on soil type (coarse sand>loamy sand>sandy loam) and precipitation, 20-100 kg N ha^{-1} yr^{-1} were leached on average for the crop rotations. In their study, the location on coarse sand had 200-300 more mm rainfall per year than the other locations. The leaching was considerably higher than for Swedish clay soils: 20 kg N ha^{-1} yr^{-1} (Stenberg et al., 2012).

Due to its high mobility in soil, NO_3 can easily be lost from the agroecosystem by leaching during periods of high drainage rates. A well-developed active root system enhances NO_3 uptake, while a poor root system will not utilize all the NO_3 within the soil profile (Dunbabin et al., 2003). NO_3 remaining in the soil after the growing season of crops, or mineralised

subsequently, will greatly increase the risk of leaching loss. This could happen outside the growing season, but also when there is poor crop establishment caused by unfavourable seedbed structure or from crop failure caused by diseases or pests (Stenberg et al., 2012). If crop failure coincides with rainy weather, the risk of severe NO_3 leaching is large. This was observed by Torstensson et al. (2006) and De Notaris et al. (2018) in organic farming systems, where potato growth was restricted due to early crop termination following disease outbreaks. Torstensson et al. (2006) determined the annual leaching in the potato year to be 75 kg N ha^{-1} after green manure and 98 kg N ha^{-1} after pea/barley (Table S3), whereas De Notaris et al. (2018) observed substantially higher leaching rates. They measured 213 kg N ha^{-1} yr^{-1} leached when the potato followed green manure and 133 kg N ha^{-1} yr^{-1} after grain legumes in a year with early occurring potato late blight. This was substantially higher than in previous years (140 and 78 kg N ha^{-1}, respectively).

Extreme rainfall events and/or periods with drought can significantly affect leaching for a variety of reasons. A field experiment over 13 years in the UK showed that N leaching in winter from fertilized grass (non-organic) was highly correlated with the preceding summer's soil moisture deficit, with the highest losses following dry summers (Tyson et al., 1997). In this case, poor grass growth due to drought led to a buildup of NO_3 from unused fertilizer present in the autumn. Prolonged mineralization of organic fertilizers or crop residues due to drought may also lead to a similar situation in organic farming systems. Tosti et al. (2016) found, under Mediterranean rainfed conditions, that the risk of NO_3 leaching was mainly at the onset of drainage due to rainfall, i.e. at the initial stage of growth, and being typically variable among years depending on timing of heavy rains. Thus, amendments applied at pre-crop stage would be a risky practice for NO_3 leaching. Most N leaching studies in organic farming in Mediterranean environments focused on row and vegetables crops (e.g. Campanelli and Campali, 2012), because these systems are most demanding in N inputs and thus have higher N applications and potential leaching than in common arable crops.
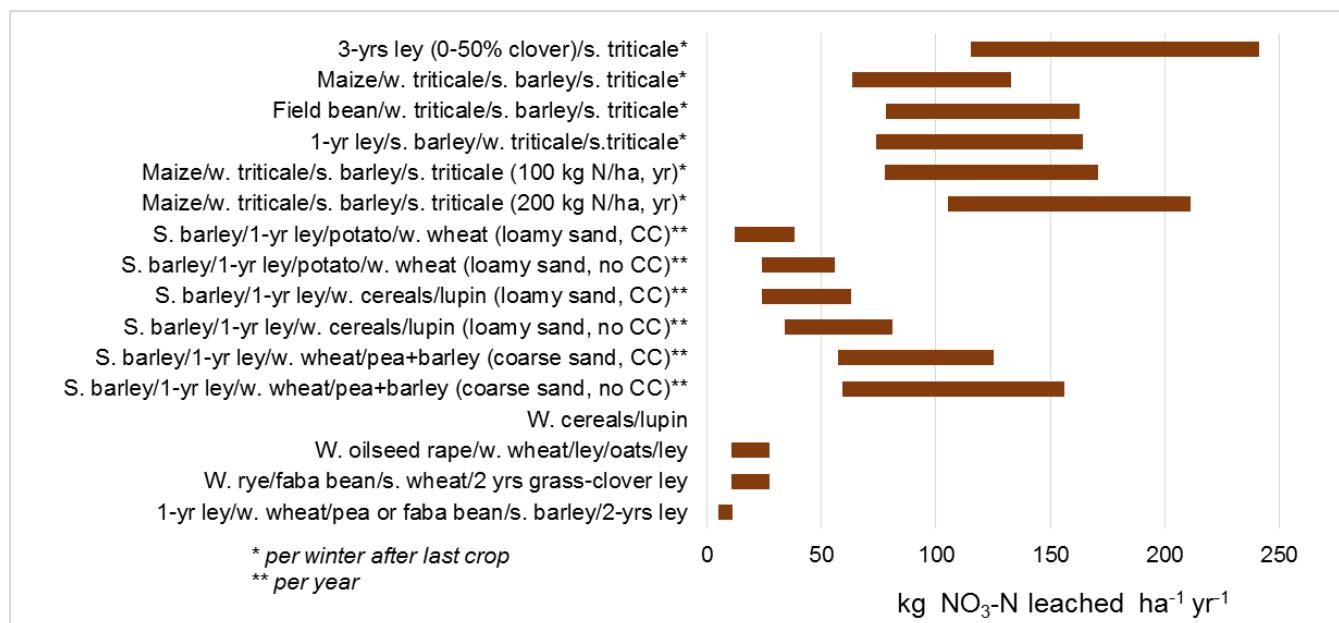
Fig. 3. Lower and upper level of NO_3 leached (kg NO_3-N ha^{-1} yr^{-1}) in various field investigations: with or without cover crops (CC), different amounts of N applied, and with or without grass-clover ley. The data used are given in Table S3.

5.2 Legumes

In a crop rotation with a large contribution through BNF, some of the N inputs will be retained in crop residues and in particular in mulched green manures (Frøseth et al., 2014). In their review, Crews and Peoples (2005) found that when the N input was based on BNF, the proportion of the N retained in the soil was higher (58% of legume N) than in the fertilized systems (31% of fertilizer N). From this, it is likely to assume that the risk of N release outside the growing season are high in rotations without legumes. In their meta-analyses of crop yield and N dynamics as influenced by CCs, Tonitto et al.

(2006) concluded, however, that on average, NO_3 leaching was reduced by 40% in legume-based systems relative to conventional fertilizer-based systems. The reason for this is probably the large difference in N input between legume-based systems relative to conventional fertilizer-based systems. The response of NO_3 leaching to N input in fertilizer, manure and residues may also differ between sites due to soil type and precipitation (Pandey et al., 2018).

5.2.1 Grain legumes

Nitrate leaching reported from crop residues of grain legumes vary. Highest values are found when grain legumes are grown in monoculture rather than in mixtures with e.g. cereals, and when CCs are not used (Plaza-Bonilla et al., 2015). Stenberg et al. (2012) observed higher NO_3 leaching after faba bean compared to after non-leguminous crops. On a clay soil in Sweden, they observed an average leaching of 20 kg N ha^{-1} yr^{-1}, which was twice that for spring cereals. On average over three years on loamy sand in Denmark, De Notaris et al. (2018) reported about twice as high NO_3 leaching following a barley/pea intercrop compared with spring wheat or spring barley. In a sandy soil in northwest Germany, Kayser et al. (2010) observed that 83 kg N ha^{-1} leached in triticale following field bean. In a worst-case scenario, Askegaard et al. (2011) observed annual NO_3 leaching of 270 kg N ha^{-1} during and after a lupin crop on a course sandy soil in a situation where the lupin crop did not ripen, leaving a large amount of N in crop residues (same experiment as De Notaris et al. (2018)). Pappa et al. (2008) observed very low N-leaching during and after a barley/pea intercrop, but they observed a significant effect of the pea cultivar on N leaching in the autumn and winter period.

5.2.2 Forage legumes

Many authors (Kayser et al., 2010; Neumann et al., 2011; Stalenga and Jończyk, 2008) emphasize that one of the most critical times for NO_3 leaching in organic crop rotations occur after soil incorporation of a grass-clover ley. N leaching is low during the growing period of grass-clover leys (Kayser et al., 2010), but because of the large amounts of mineralized N after termination of a grass-clover ley (Sect. 3.3), the risk of NO_3 leaching is large for one to two years after termination of these crops (Berntsen et al., 2005). The leaching may occur shortly after ley termination, during winter, or during the succeeding seasons, depending on time of incorporation, quality of the herbage, the weather and the crop sequence.

Stenberg et al. (2012) observed higher NO_3 leaching following termination of a grass-clover ley than following faba bean, but values were still low (4 kg N ha^{-1} higher on average). They found the highest leaching when the grass-clover ley lasted for two years (up to 40 kg N leached ha^{-1} yr^{-1}). This corresponds to the finding of Kayser et al. (2010), who observed greater NO_3 leaching during the winter after spring incorporation of a three-year ley than after a one-year ley (121 versus 83 kg N ha^{-1}, Fig. 3). However, the crop yield of triticale yielded much better after the three-year ley than after the one-year ley. The percentage share of clover (0-5, 30 and 50%) did not influence the amount leached after ley termination, neither the crop yield. Eriksen et al. (2008) measured NO_3 leaching after 1 to 8 year old grass/clover leys, but found that the length of the ley had no effect on NO_3 leaching. Stenberg et al. (2012) observed that cereals succeeding grass-clover ley had nearly double yearly N leaching compared to cereals with no legume pre-crop. The highest NO_3 leaching occurred after cultivation of a winter rye (48 kg N ha^{-1} yr^{-1}).

De Notaris et al. (2018) observed that NO_3 leaching during cultivation of spring wheat was about 50 kg N ha^{-1} higher when the spring wheat succeeded a two-year green manure crop (alfalfa or grass-clover) than when it succeeded a grain legume (107 versus 50 kg N ha^{-1}). Similarly, Askegaard et al. (2011) observed peaks in NO_3 leaching in autumn and winter after ploughing-in grass-clover ley. At crop rotation level, inclusion of grass-clover or alfalfa on 25% of the area increased NO_3 leaching rate by 6-12 kg N ha^{-1} (De Notaris et al., 2018). Forage legumes may also be undersown as intercrops to increase soil fertility in organic crop rotations. Pappa et al. (2008) found that clover intercropped in spring barley only increased annual NO_3 leaching by 1-2 kg NO_3-N ha^{-1}.

5.3 Cover crops

CCs are grown between main crops to minimize NO_3 leaching. Many field trials in non-organic systems have shown reduced leaching using CC (e.g., Rasse et al., 2000; Torstensson and Aronsson, 2000; Constantin et al., 2010; Valkama et al., 2015). This is also the case for organic crop rotations (Tonitto et al., 2006; Askegaard et al., 2011; Tosti et al., 2014; Tosti et al., 2016; De Notaris et al., 2018). The reduction in NO_3 leaching can be substantial. Studies in Nordic countries report reductions of 50-60% in N leaching (Askegaard et al., 2011; De Notaris et al., 2018; Torstensson and Aronsson, 2000, Fig. 3). If the cash crop fails, the effect of CCs on reduced leaching can be even higher. In a year where potato late blight caused crop failure in potato, the CCs reduced N-leaching by 95% when the potato succeeded a grain legume (from 133 to 6 kg N ha^{-1} leached), and by 92% when the potato succeeded a green manure ley (from 213 to 17 kg N ha^{-1} leached) (calculated from Table S3, De Notaris et al., 2018).

De Notaris et al. (2018) concluded that the use of CCs had a larger impact on leaching than a substantial variation in N surplus between alternative cropping systems. In three long-term field trials (13-17 years) in Northern France, Constantin et al. (2010) observed that CCs were the most efficient management option for reducing NO_3 leaching (from 36 to 62%). Good establishment and growth of the CC is essential to obtain sufficient uptake of SMN and thus reduce NO_3 leaching. Stenberg et al. (1999) found no significant reduction in NO_3 leaching during winter from a ryegrass CC that was undersown in spring. They explained this by poor CC establishment. De Notaris et al. (2018) also observed occasions with very small effect of CCs on NO_3 leaching. They related this to CC growth and identified threshold values in CC above-ground biomass determined in November, above which N leaching was reduced to a stable low level. NO_3 leaching from spring wheat averaged 15 kg N ha^{-1} yr^{-1} with CC biomass above 0.9 Mg ha^{-1}, and 41 kg N ha^{-1} yr^{-1} with CC biomass below 0.9 Mg ha^{-1}. In potatoes, the average N leaching was 11 and 80 kg N ha^{-1} yr^{-1} with CC biomass above and below 1.5 Mg ha^{-1}, respectively.

Including legumes in CC mixtures does not seem to reduce the ability of CCs to reduce NO_3 leaching (Tonitto et al., 2006; Tosti et al., 2014; De Notaris et al., 2018; Shelton et al., 2018). In a field trial with barley, hairy vetch and a 50:50 mixture of both species as CC, Tosti et al. (2014) found that, in all years, the barley/vetch mixture decreased N leaching to the same level of pure barley, both during its own growing cycle and after CC incorporation into the soil. De Notaris et al. (2018) concluded that the same degree of reduced NO_3 leaching was obtained with legume-based CCs as with non-legume CCs. The CC was either undersown in spring or after harvest of the main crop. The undersown legumes were white clover and red clover, and winter vetch was used in the mixture sown after harvest. Shelton et al. (2018) found greater NO_3 leaching from intercropped hairy vetch than from simultaneously grown wheat and wheat/hairy vetch mixture. When the CC is a pure stand of legumes, the CC does not necessarily reduce N leaching (Tosti et al., 2014; Valkama et al., 2015; Shelton et al., 2018). Tosti et al. (2014) concluded that hairy vetch sown as a pure crop in autumn showed high BNF, but no NO_3 leaching mitigation effect as compared to bare soil. Valkama et al. (2015) found in their meta-analysis of Nordic studies of undersown CCs that legumes (white and red clovers) in pure stand did not diminish the risk of NO_3 leaching.

5.4 Tillage

Tillage stimulates soil N mineralization, at least in the short term. Tillage is also often associated with soil incorporation of plant residues that may lead to net N immobilisation or mineralisation, depending on quality of the residues. Timing of tillage is therefore crucial for the fate of the mineralised N, whether SMN becomes available for crop uptake or subject to leaching. In general, incorporation should consider soil type, climate conditions and type of herbage (C/N ratio). Thorup-Kristensen and Dresbøll (2010) suggested late incorporation of CCs in high rainfall areas on sandy soils, and earlier in low rainfall areas on NO_3 retentive soils. Field studies have shown rapid N mineralization from N-rich plant material, even at low temperatures (Breland, 1994; Thorup-Kristensen and Dresbøll, 2010). Spring incorporation has therefore been recommended to increase N recovery by subsequent crops. However, under Scandinavian conditions, there may still be a deficit in crop-

available N, even after spring incorporation of a green manure ley (Frøseth et al., 2014; Känkänen et al., 1998). Under such conditions, Torstensson and Aronsson (2000) suggest that late autumn incorporation of CCs, instead of spring incorporation, will be preferable with respect to N availability for the subsequent crop and will not substantially increase NO_3 leaching. Under Mediterranean climatic conditions, characterized by mild rainy winters and warm to hot dry summers, there is a risk of NO_3 leaching if residues are incorporated prior to the wet season. No studies were found that measured NO_3 leaching in relation to timing of tillage in organic arable farming under these conditions.

In organic crop production, the timing of cultivation for mechanical control of perennial weeds may conflict with the aims of high N use efficiency (Melander et al., 2016). Askegaard et al. (2011) found, on sandy soils in Denmark, that the management of crop and soil during autumn was the main determinant of N leaching. Stubble harrowing in autumn for controlling perennial weeds, followed by bare soil during winter, led to an average of 25 kg N ha^{-1} more leached than for soils left untouched with a cover of weeds/volunteers. NO_3 leaching increased with increasing number of autumn soil cultivations.

Reducing tillage intensity may also enhance the need for weed management, and thereby the risk of NO_3 leaching. In their meta-analysis comparing different reduced tillage intensities in organic farming, Cooper et al. (2016) found that the weed incidence was consistently higher, by about 50 %, when tillage intensity was reduced, although this did not always lower the yields. Compared to conventional tillage, reduced tillage may reduce NO_3 leaching, but this depends on the establishment and growth of the succeeding crop (Känkänen et al., 1998).

Bare fallow has traditionally been a method to control perennial weeds by repeated tillage of superficial or deeper top soil layers. In practical organic farming, a bare fallow can sometimes be used before or after the main crop if conditions have promoted perennial weeds. However, if carried out in the growing season, soil temperature and moisture conditions favour soil microbial activity and therefore the build-up of SMN, which greatly increases the risk of NO_3 leaching (Borgen et al., 2012).

6. Key drivers of N$_2$O emissions and NO$_3$ leaching and suggested mitigation strategies

Easily available N and degradable C added through organic inputs enhances risks of high N_2O emissions in the short term because of enhanced biological activity (Sect. 3.3) and increased denitrification potential (Sect. 4.1), and in the longer term because of higher soil content of N and C in labile organic matter (Sect. 3.1) contributing to N_2O emissions from mineralized SOM (background emission, Sect. 4.1, 4.3, 4.4, 4.5). There is no strong correlation between total N-input and N_2O emissions in organic arable crop rotations (Sect. 4.5), which is another indication of the strong impact of background emissions in these systems (Sect. 4.5). The same is also true of the relatively low impact of episodes with high N_2O fluxes on the total N_2O emission in various investigations (Sect. 4.5, Fig. 2).

From this, we postulate that background emissions in most organic crop rotations are a more important driver for N_2O emissions than episodes with high N_2O flux rate (hot moments). Nevertheless, reducing periods with hot moments of denitrification and thus high N_2O fluxes rates are important for reducing total emissions. One mitigation measure is to avoid large applications of residues from crops comprising easily available N and degradable C like clover or Brassica (Sect. 4.3). Mulching of grass-clover on top of a ley seems to only slightly increase N_2O-emissiom compared with no mulch (section 4.2). Incorporating organic material in the soil surface does not reduce the risk for high N_2O fluxes and N_2O emissions are often higher after surface application than after ploughing (Sect. 4.3.2). Mixing by rotary harrow can enhance emissions, particularly in moist soil. If the crop residues are removed, and composted, used in a biogas plant or treated by other methods before targeted soil application, such events can be avoided. To avoid that these measures result in that GHG emissions only are moved to another place, this requires measures that minimise GHG emissions during and after treatment of the plant residues (see sect 7). The largest stimulation of denitrification by application of plant residues or other organic material with a high content of degradable C seems to be in soils with a low content of SOC, particularly if the soil has a high NO_3 content

of or easily available N is applied (Sect. 4.4). If crop residues consist of less degradable C and easily available N like straw from cereals and grain legumes, they do not stimulate rapid denitrification and enhanced N_2O flux rate (Sect. 4.3), and the N_2O flux is likely reduced (Xia et al., 2018).

Background emissions are highly influenced by content and mineralization patterns of SOM and release of SMN and degradable C (Sect. 3 and 4.1). Thus, weather conditions will have a large impact on the background emissions, which are likely to be higher in warm and moist years, than in cold or dry years. These conditions cannot be influenced by farmers, but it is important to develop strategies to decrease the content of SMN to reduce the risk of N_2O emissions and NO_3 leaching in periods with risk of SMN accumulation. The choice and sequence of crops and the use of CC is a central strategy for simultaneously mitigating N_2O emissions and NO_3 leaching in organic systems. Poor timing of N released from crop residues from preceding crops and crop N uptake (Sect. 3.3 and 5.2) is a major challenge. There is a need to reduce periods with bare soil, although this may conflict with the need for mechanical weed control (Melander et al., 2016).

CCs take up surplus SMN and protect soil from erosion, they reduce NO_3 leaching, and during growth CCs do not contribute to N_2O emissions (Sect. 4.2). However, in areas with frost, frost sensitive crops like fodder radish, will release easily available N and degradable C during thawing, and thus stimulate denitrification (Sect. 4.3.1). The choice of CC must be adapted to local conditions for ensuring good establishment and appropriate tolerance to frost and droughts. A mixture of legumes and non-legumes (for instance grasses or cereals) are just as efficient for reducing N leaching as sole non-legume CCs, whereas sole legumes are not as efficient (Sect. 5.3). CCs containing legumes have a lower C:N ratio than CC without legumes, which enhances the N fertilizer value for the following crop (Li et al., 2015b).

The knowledge about the rooting pattern of different crop species can be used as a tool for designing crop rotations that achieve higher N use efficiency and thereby reduces the risk of NO_3 leaching. Deep-rooted crops, especially tap rooted ones, can recover NO_3 from deeper soil layers before and after more shallow-rooted cash crops, such as leek (Thorup-Kristensen, 2006). As shown by Fan et al. (2016), the root distribution and rooting depth may differ between varieties, although plant breeders do not normally select crops based on the root system.

For the choice of crop, species and varieties should be well-adapted to the climate conditions on the farm and the soil fertility level. Crops in good conditions also compete better against weeds. This decreases the need for soil management to achieve weed control, and thereby reduces the risk of NO_3 leaching. Timing of release of N from residues and amendments and crop uptake are crucial for minimizing the risk of NO_3 leaching. This can be achieved by timing of soil tillage and incorporation of residues. In any case, the effect of mitigation strategy is highly dependent on soil type and precipitation.

Because of the large impact of poorly aerated soil on N_2O emissions (Sect. 4.1) and plant growth, measures should be taken to improve and maintain a good soil structure. In organic production, soil fauna, microorganisms and the development and maintenance of soil structure, are supported by crop rotations that include legumes or grass-clover leys, use of CCs and application of organic fertilizers. Even so, traffic and tillage under wet soil conditions are damaging to soil structure and should therefore be avoided.

A way to reduce N_2O emissions and NO_3 leaching per unit produced could be to increase yields in organic production as more land is commonly needed per unit product in organic than non-organic production (De Ponti et al., 2012, Meier et al., 2015). However, as discussed by Röös et al. (2018) this is not straight forward as many of the available measures have negative side effects. More targeted and thus efficient use of N applied through crop residues and organic fertilizers seems to have co-benefits in terms for higher productivity as well as reduced N_2O emissions and NO_3 leaching. This may be achieved by recycling of these residues through biogas and targeted application to crops according to their N demand (Brozyna et al., 2013; Knudsen et al., 2014).

7. Research and innovation needs

The knowledge of N cycling and losses in arable organic farming is constrained by few investigations that have adequately quantified the magnitude and timing of N_2O emissions and NO_3 leaching. Many of the reported results are from a few experiments, including a long-term experiment in Denmark (Olesen et al., 2000). Such long-term experiments are important
5   for quantifying N transformation and loss processes in organic cropping systems, where the legacy effects of N in plant residues and other organic amendments are often considerably larger than in non-organic systems. Lessons from these experiments can help to develop methods and technologies to improve synchronisation of N released from crop residues and soil organic matter and the N demand of cash crops. However, there are few long-term experiments with organic arable crop rotations, and even fewer have been systematically used for quantifying the N cycling processes. Therefore, research on
10   existing and new long-term experiments covering relevant soil and climatic conditions should be coordinated to make data and facilities available for research on N-dynamics, including N_2O emissions, NO_3 leaching, NH_3 volatilization and the fate of added N. In this way, useful lessons can be learned on the impact of long-term management on the N-dynamics. The availability of historical data on N-input and crop management is a prerequisite to achieve this, such data could be made available and structured through a coordination of long-term experiments.

15   CCs have been shown to efficiently reduce NO_3 leaching (Sect. 5.3), whereas they can reduce or increase N_2O emissions depending on the conditions (Sect. 4.2, 4.3). To optimise the use of CCs for crop productivity, environment and climate, improved knowledge is needed of locally adapted crop rotations and CC that maintain living plants that can take up available N and simultaneously reduce weed pressure.

Simulation models can be used to quantify N transformation and loss processes and how different management practices
20   affect N storage in SOM, as well and short-term effects of crop residues and organic fertilizers on N_2O emissions and NO_3 leaching. However, process-based models fail to accurately simulate the impact of grass/clover leys on turnover of N and C (Doltra et al., 2019, Frøseth, 2016). To our knowledge, no model of field scale does include all C and N turnover processes, such as phyllodeposition and rhizodeposition. These processes may be of greater importance in organic than non-organic systems, because of the greater emphasis on fertility building measures. This may lead to underestimation of the amount of
25   N and C returned to soil. Improving models on these aspects requires more comprehensive data on all C and N inputs and flows in the cropping systems. This can only be achieved through studies using isotopic labelling of C and N, and consideration should be given to conducting such studies also in the long-term experiments. There is further uncertainty on the release of N from legumes and thus the impact of legume residues on N_2O emission and NO_3 leaching (see Sect. 4.2, 4.3 and 5.2). More research on the mineralization pattern of various legume residues would therefore be useful.

30   Heterogeneity is another aspect that make it difficult to predict availability of N and C in organic systems and thus N_2O emission. In addition to the soil heterogeneity, variability is created from incorporated crop residues, CC, short term leys and organic fertilizer that are seldom evenly distributed. Uneven impact of freezing / thawing and drying / rewetting will add to this variability and make it difficult to predict hot spots and hot moments of N_2O emissions. There is also a need to develop technologies to better measure this heterogeneity and to take account for the heterogeneity in N dynamic models adapted to
35   organic crop rotations.

Composting of crop residues and manures is common on organic farms and is a way to make the organic matter more resistant and useful as a soil amendment. However, composting manure without emissions of N_2O and other greenhouse gases is challenging (Chadwick et al., 2011) and needs to be addressed to develop a more comprehensive picture of greenhouse gas emissions from organic arable production and how to mitigate them. Neither composting nor biogas
40   fermentation are always feasible. Improved understanding of different pathways for microbial degradation of crop residues is needed as well, to better manage crop residues and organic waste for C and N retention properties.

8. Conclusions

Organic arable production is based on diverse crop rotations, N inputs through BNF in legumes, external inputs of manure and compost, and recovery of excess N in grassland and cover crops. This results in considerable inputs on N in organic matter from plant residues, green manures and application of organic fertilisers and soil amendments that enhance the content of total and labile SOC in organic arable production compared to non-organic production. When the conditions for mineralization of SOC are met this will lead to a high availability of easily available N and degradable C, and an enhanced content of SMN, if available N is not taken up by growing plants. Conditions with high content of SMN and degradable C provide hot moments for high N_2O fluxes. Degradable C will increase microbial growth and thus O_2 consumption in soil leading to anaerobe conditions and provide suitable conditions for denitrification of available NO_3. In organic arable crop rotations background emissions are more important in most situations than episodes with high N_2O fluxes for the total N_2O emission from organic arable crop rotations. SMN from mineralization of SOM and plant residues in combination with periods of bare soil or sparse plant growth and precipitation surplus provide drivers for NO_3 leaching.

Main mitigation strategies are targeted use of growing plants to take up surplus soil mineral N from root zone, and targeted use of crop residues and organic fertilizers to synchronise the availability of N with the crop demand. Continued use of CCs has a proven ability to reduce NO_3 leaching from organic arable crop rotations but does increase N_2O emissions in some situations. A CC mixture of legumes and non-legumes (for instance grasses or cereals) is just as efficient as sole non-legumes and has a better impact on soil fertility than non-legumes. More research is needed to develop locally adapted crop rotations and CC mixtures that at the same time reduce N_2O emissions and NO_3 leaching.

Author contributions: Sissel Hansen: Prepared the manuscript with contributions from all co-authors, Supplementary material Table S2, Fig. S2; Randi Berland Frøseth: Structure and content of the MS, mainly N leaching, complementary data from Norwegian field trials in Fig. 3, Table S1, Supplementary material Table S3; Maria Stenberg: Content of the MS, mainly N leaching, Supplementary material Table 3 and complementary data from Swedish field trials to this Table; Jarosław Stalenga: Content of the MS, mainly SMN, Supplementary material Table S1 and Table 1; Jørgen E. Olesen: Overall content and structure of the MS, Fig. 1, complementary data from Danish field trials in Table S2; Maike Krauss: Content of the MS, complementary data from Swiss field trial in Table S2; Paweł Radzikowski: Content of the MS, mainly earthworms, main editing of references and reference list; Jordi Doltra: Content of the MS, mainly from Mediterranean perspective; Shahid Nadeem: Content of the MS, mainly N_2O, complementary data from Norwegian field trial in Table S2; Torfinn Torp: Statistical analyses on cause of N_2O fluxes based on Table S2; Valentini Pappa: Complementary data from field trials in Edinburgh and Aberdeen in Table S2; Christine Watson: Overall development and content of the MS.

Acknowledgments. The FertilCrop project was financially supported by transnational funding bodies of the FP7 ERA-Net project, CORE Organic Plus, and from the European Commission. The authors also acknowledge support from the Scottish Government Strategic Research Programme and from Møre og Romsdal county council. The text in this report is the sole responsibility of the authors and does not necessarily reflect the views of the national funding bodies having financed this project. The authors acknowledge Bruce Ball for complementary data from a field trial in Aberdeen used in table S2, and an anonymous referee for valuable and comprehensive comments.

References

Aguilera, E., Lassaletta, L., Gattinger, A., and Gimeno, B. S.: Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: A meta-analysis, Agr. Ecosyst. Environ., 168, 25–36, doi:10.1016/j.agee.2013.02.003, 2013.

Anderson, J. H.: The metabolism of hydroxylamine to nitrite by Nitrosomonas, Biochem. J., 91, 8–17, 1964.

Anderson, T.-H., and Domsch, K. H.: Ratios of microbial biomass carbon to total organic carbon in arable soils, Soil Biol. Biochem., 21(4), 471–479, doi:10.1016/0038-0717(89)90117-X, 1989.

Anderson, I. C., Poth, M., Homstead, J., and Burdige, D.: A comparison of NO and N2O production by the autotrophic nitrifier Nitrosomonas europaea and the heterotrophic nitrifier Alcaligenes faecalis, Appl. Environ. Microbiol., 59, 3525–3533, 1993.

Aronsson, H., Torstensson, G., and Bergstrom, L.: Leaching and crop uptake of N, P and K from organic and conventional cropping systems on a clay soil, Soil Use Manage., 23(1), 71–81, 2007.

Askegaard, M., Olesen, J. E., and Kristensen, K.: Nitrate leaching from organic arable crop rotations: effects of location, manure and catch crop, Soil Use Manage., 21(2), 181–188 [online] Available from: isi:000230816100003, 2005.

Askegaard, M., Olesen, J. E., Rasmussen, I. A., and Kristensen, K.: Nitrate leaching from organic arable crop rotations is mostly determined by autumn field management, Agr. Ecosyst. Environ., 142(3–4), 149–160, doi:10.1016/j.agee.2011.04.014, 2011.

Baggs, E. M., Rees, R. M., Smith, K. A., and Vinten, A. J. A.: Nitrous oxide emission from soils after incorporating crop residues, Soil Use Manage., 16, 82–87, 2000a.

Baggs, E. M., Watson, C. A., and Rees, R. M.: The fate of nitrogen from incorporated cover crop and green manure residues, Nutr. Cycl. Agroecosys., 56(2), 153–163 [online] Available from: wos:000086647600008, 2000b.

Bakken, L. R., Bergaust, L., Liu, B., and Frostegård, A.: Regulation of denitrification at the cellular level: a clue to the understanding of N2O emissions from soils., Philos. T. R. Soc. B., 367(1593), 1226–34, doi:10.1098/rstb.2011.0321, 2012.

Balaine, N., Clough, T.J., Beare, M.H., Thomas, S.M., Meenken, E.D., Ross, J.G.: Changes in Relative Gas Diffusivity Explain Soil Nitrous Oxide Flux Dynamics. Soil Sci. Soc. Am. J., 77, 1496. https://doi.org/10.2136/sssaj2013.04.0141, 2013.

Ball, B. C., Mctaggart, I. P., and Watson, C. A.: Influence of organic ley-arable management and afforestation in sandy loam to clay loam soils on fluxes of N2O and CH4 in Scotland, Agr. Ecosyst. Environ., 90(3), 305–317, [online] Available from: isi:000177795300008, 2002.

Ball, B. C., Watson, C. A., and Crichton, I.: Nitrous oxide emissions, cereal growth, N recovery and soil nitrogen status after ploughing organically managed grass/clover swards, Soil Use Manage., 23(2), 145–155, doi:10.1111/j.1475-2743.2006.00072.x, 2007a.

Baral, K. R., Labouriau, R., Olesen, J. E., and Petersen, S. O.: Nitrous oxide emissions and nitrogen use efficiency of manure and digestates applied to spring barley, Agr. Ecosyst. Environ., 239, 188–198, doi:10.1016/j.agee.2017.01.012, 2017.

Barbieri, P., Pellerin, S., Nesme, T.: Comparing crop rotations between organic and conventional farming. Sci. Rep-UK., 7, 1–10, https://doi.org/10.1038/s41598-017-14271-6., 2017.

Benoit, M., Garnier, J., Anglade, J., and Billen, G.: Nitrate leaching from organic and conventional arable crop farms in the Seine Basin (France), Nutr. Cycl. Agroecosys., 100(3), 285–299, 2014.

Berntsen, J., Petersen, B. M., Olesen, J. E., Eriksen, J. E., and Soegaard, K.: Simulation of residual effects and nitrate leaching after incorporation of different ley types, Eur. J. Agron., 23(3), 290–304, [online] Available from: isi:000233029100007, 2005.

Berntsen, J., Petersen, B. M., Sørensen, P., and Olesen, J. E.: Simulating Residual Effects of Animal Manures Using N-15 Isotopes, Plant Soil, 290, 173–87, 2007.

Berry, B. M., Sylvester-Bradley, R., Philipp, L., Hatch, D. J., Cuttle, S. P., Rayns, F. W., and Gosling, P.: Is the productivity of organic farms restricted by the supply of available nitrogen? Soil Use Manage., 18(3), 248–255, https://doi.org/10.1079/SUM2002129, 2002.

Bhogal, A., Williams, J.R., Nicholson, F.A., Chadwick, D.R., Chambers, K.H., and Chambers, B.J.: Mineralization of organic nitrogen from farm manure applications, Soil Use Manage., 32, 32-43, 2016.

Borgen, S. K., Lunde, H. W., Bakken, L. R., Bleken, M. A., and Breland, T. A.: Nitrogen dynamics in stockless organic clover-grass and cereal rotations, Nutr. Cycl. Agroecosys., 92(3), 363–378 [online] Available from: wos:000302576100009, 2012.

Bouwman, A. F., Boumans, L. J. M., and Batjes, N. H.: Modeling global annual N2O and NO emissions from fertilized fields, Global Biogeochem Cy., 16(4), [online] Available from: isi:000180874100028, 2002.

Breland, T. A.: Measured and predicted mineralization of clover green manure at low temperatures at different depths in two soils, Plant Soil, 166(1), 13–20, 1994.

Bronick, C. J., and Lal, R.: Soil structure and management: A review, Geoderma, 124(1–2), 3–22, doi:10.1016/j.geoderma.2004.03.005, 2005.

Brophy, L.S., Heichel, G. H., and Russelle, M. P.: Nitrogen transfer from forage legumes to grass in a systematic planting design, Crop Sci., 27, 753–758, 1987.

Brozyna, M. A., Petersen, S. O., Chirinda, N., and Olesen, J. E.: Effects of grass-clover management and cover crops on nitrogen cycling and nitrous oxide emissions in a stockless organic crop rotation, Agr. Ecosyst. Environ., 181, 115–126, doi:10.1016/j.agee.2013.09.013, 2013.

Burke, I. C., Yonker, C. M., Parton, W. J., Cole, C. V., Flach, K., and Schimel, D. S.: Texture, Climate, and Cultivation Effects on Soil Organic Matter Content in U.S. Grassland Soils, Soil Sci. Sos. Am. J., 53(i), 800–805, 1989.

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S.: Nitrous oxide emissions from soils: how well do we understand the processes and their controls?, Philos. T. R. Soc. B., 368(1621), 20130122, doi:10.1098/rstb.2013.0122, 2013.

Campanelli, G., and Canali, S.: Crop production and environmental effects in conventional and organic vegetable farming systems: The case of a long-term experiment in Mediterranean conditions (Central Italy), J. Sustain. Agr., 36(6), 599-619, 2012.

Carter, M. S. and Ambus, P.: Biologically fixed N_2 as a source for N_2O production in a grass-clover mixture, measured by N-15, Nutr. Cycl. Agroecosys., 74 (1), 13–26, wos:000235949200002, 2006.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B., and Misselbrook, T.: Manure management: Implications for greenhouse gas emissions, Anim. Feed Sci. Tech., 166–167, 514–531, 2011.

Chang, C., Cho, C. M., and Janzen, H. H.: Nitrous oxide emission from long-term manured soils, Soil Sci. Soc. Am. J., 62(3), 677–682, 1998.

Chantigny, M.H., Angers, D.A., Rochette, P., Bélanger, G., Massé, D., Côté, D.: Gaseous Nitrogen Emissions and Forage Nitrogen Uptake on Soils Fertilized with Raw and Treated Swine Manure. J. Environ. Qual., 36, 1864. https://doi.org/10.2134/jeq2007.0083,2007.

Chantigny, M. H., Rochette, P., Angers, D.A., Bittman, S., Buckley, K., Massé, D., Bélanger, G., Eriksen-Hamel, N., Gasser, M.-O.: Soil Nitrous Oxide Emissions Following Band-Incorporation of Fertilizer Nitrogen and Swine Manure. J. Environ. Qual., 39, 1545–1553. https://doi.org/10.2134/jeq2009.0482, 2010.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., and Bertrand, N.: Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis, Agr. Ecosyst. Environ., 236(3), 88–98, doi:10.1016/j.agee.2016.11.021, 2017.

Chen, H. H., Li, X. C., Hu, F., and Shi, W.: Soil nitrous oxide emissions following crop residue addition: a meta-analysis, Glob. Chang. Biol., 19(10), 2956–2964, doi:10.1111/gcb.12274, 2013.

Chirinda, N., Carter, M. S., Albert, K. R., Ambus, P., Olesen, J. E., Porter, J. R., and Petersen, S. O.: Emissions of nitrous oxide from arable organic and conventional cropping systems on two soil types, Agr. Ecosyst. Environ., 136(3–4), 199–208, [online] Available from: isi:000276275400003, 2010.

Congreves, K.A., Wagner-Riddle, C., Si, B.C., Clough, T.J.: Nitrous oxide emissions and biogeochemical responses to soil freezing-thawing and drying-wetting, Soil Biol. Biochem., 117, 5–15, doi:10.1016/j.soilbio.2017.10.040, 2018.

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P., and Beaudoin, N.: Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments, Agr. Ecosyst. Environ., 135(4), 268–278, doi:10.1016/j.agee.2009.10.005, 2010.

Cooper, J., Baranski, M., Stewart, G., Nobel-de Lange, M., Bàrberi, P., Fließbach, A., Peigné, J., Berner, A., Brock, C., Casagrande, M., Crowley, O., David, C., De Vliegher, A., Döring, T. F., Dupont, A., Entz, M., Grosse, M., Haase, T., Halde, C., Hammerl, V., Huiting, H., Leithold, G., Messmer, M., Schloter, M., Sukkel, W., van der Heijden, M. G. A., Willekens, K., Wittwer, R., and Mäder, P.: Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis, Agron. Sustain. Dev., 36(1), doi:10.1007/s13593-016-0354-1, 2016.

Council of the European Union.: Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91, Official Journal of the European Union, L 189/1, 2007. http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32007R0834&from=EN

Crews, T. E., and Peoples, M. B.: Can the synchrony of nitrogen supply and crop demand be improved in legume and fertilizer-based agroecosystems? A review, Nutr. Cycl. Agroecosys., 72(2), 101–120, doi:10.1007/s10705-004-6480-1, 2005.

Dalgaard, T., Hansen, B., Hasler, B., Hertel, O., Hutchings, N.J., Jacobsen, B., Kronvang, B., Olesen, J.E., Schjørring, J., Termansen, M., and Vejre, H.: Policies for agricultural nitrogen management - trends, challenges and prospects for improved efficiency in Denmark, Environ. Res. Lett., 9, 115002, 2014.

De Notaris, C., Rasmussen, J., Sørensen, P., and Olesen, J. E.: Nitrogen leaching: A crop rotation perspective on the effect of N surplus, field management and use of catch crops, Agr. Ecosyst. Environ., 255, 1–11, doi:10.1016/J.AGEE.2017.12.009, 2018.

De Ponti, T., Rijk, B., and Van Ittersum, M. K.: The crop yield gap between organic and conventional agriculture, Agr. Syst., 108, 1–9. https://doi.org/10.1016/j.agsy.2011.12.004, 2012.

Di, H. J., and Cameron, K. C.: Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies, Nutr. Cycl. Agroecosys., 64(3), 237–256, [online] Available from: isi:000179746500004, 2002.

Dobbie, K. E., Smith, K. A.: Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water filled pore space and other controlling variables. Glob. Chang. Biol., 99, 204–218, https://doi.org/10.1046/j.1365-2486.2003.00563.x, 2003.

Doltra, J., Gallejones, P., Olesen, J.E., Hansen, S., Frøseth, R.B., Krauss, M., Stalenga, J., Jończyk, K., Martínez-Fernández, A., Pacini, G.C.: Simulating soil fertility management effects on crop yield and soil nitrogen dynamics in field trials under organic farming in Europe. F. Crop. Res. 233, 1–11. https://doi.org/10.1016/j.fcr.2018.12.008, 2019.

Dunbabin, V., Diggle, A., and Rengel, Z.: Is there an optimal root architecture for nitrate capture in leaching environments?, Plant. Cell. Environ., 26(6), 835-844, 2003.

Dusenbury, M.P., Engel, R.E., Miller, P.R., Lemke, R.L., Wallander, R.: Nitrous Oxide Emissions from a Northern Great Plains Soil as Influenced by Nitrogen Management and Cropping Systems, J. Environ. Qual., 37, 542, https://doi.org/10.2134/jeq2006.0395, 2008.

Eriksen, J., Askegaard, M., and Søegaard, K.: Residual effect and nitrate leaching in grass-arable rotations: effect of grassland proportion, sward type and fertilizer history, Soil Use Manage., 24, 373–382, 2008.

Evers A. K., Demers T. A., Gordon A. M., and Thevathasan, N. V.: The effect of earthworm (Lumbricus terrestris L.) population density and soil water content interactions on nitrous oxide emissions from agricultural soils, Appl. Environ. Soil Sci., doi:10.1155/2010/737096, 2010.

Fan, J., McConkey, B., Wang, H., and Janzen, H.: Root distribution by depth for temperate agricultural crops, Field. Crop. Res., 189, 68–74, doi:10.1016/j.fcr.2016.02.013, 2016.

Filser J., Dette A., Fromm H., Lang A., Munch J.C., Winter K., and Beese F.: Reactions of soil organisms in site-specific management. The first long-term study at the landscape scale, Ecosyst. Suppl., 28, 139–147, 1999.

Firestone, M., and Davidson, E.: Microbiological basis of NO and N_2O production and consumption in soil, in Exchange of trace gases between terrestrial ecosystems and the atmosphere., (Andreac, M. O. and Schimel, D. S. editors) pp. 7–21, John Wiley & Sons Ltd., 1989.

Flessa, H., Dörsch, P., and Beese, F.: Seasonal variation of N_2O and CH_4 fluxes in differently managed arable soils in southern Germany, J. Geophys. Res., 100(23), 115–124, 1995.

Frøseth, R. B.: Nitrogen dynamics in an organic green manure – cereal rotation and mineralization of clover leaves at low temperature, Philosophiae Doctor (PhD) Thesis 2016:25, Norwegian University of Life Sciences, Ås, pp 58, 2016.

Frøseth, R. B., Bakken, A. K., Bleken, M. A., Riley, H., Pommeresche, R., Thorup-Kristensen, K., and Hansen, S.: Effects of green manure herbage management and its digestate from biogas production on barley yield, N recovery, soil structure and earthworm populations, Eur. J. Agron., 52, doi:10.1016/j.eja.2013.10.006, 2014.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N. E.-H., and Niggli, U.: Enhanced top soil carbon stocks under organic farming., P. Natl. Acad. Sci. USA, 109(44), 18226–18231, doi:10.1073/pnas.1209429109, 2012.

Gomiero, T.: Alternative Land Management Strategies and Their Impact on Soil Conservation, Agriculture, 3(3), 464–483, doi:10.3390/agriculture3030464, 2013.

Hansen, E. M., Eriksen, J., and Vinther, F. P.: Catch crop strategy and nitrate leaching following grazed grass-clover. Soil Use Manage., 23(4), pp.348-358., 2007.

Hansen, S., and Engelstad, F.: Earthworm populations in a cool and wet district as affected by tractor traffic and fertilisation, Appl. Soil Ecol., 13(3), doi:10.1016/S0929-1393(99)00037-2, 1999.

Hansen, S., Bernard, M.-E., Rochette, P., Whalen, J. K., and Dörsch, P.: Nitrous oxide emissions from a fertile grassland in Western Norway following the application of inorganic and organic fertilizers, Nutr. Cycl. Agroecosys., 98(1), 71–85, doi:10.1007/s10705-014-9597-x, 2014.

Hartmann, M., Frey, B., Mayer, J., Mäder, P., and Widmer, F.: Distinct soil microbial diversity under long-term organic and conventional farming, ISME J., 9(5), 1177–1194, doi:10.1038/ismej.2014.210, 2015.

Holtan-Hartwig, L., Dorsch, P., and Bakken, L. R.: Low temperature control of soil denitrifying communities: kinetics of N2O production and reduction, Soil Biol. Biochem., 34(11), 1797–1806, [online] Available from: isi:000179996900029, 2002.

Hu, T., Sørensen, P., and Olesen, J .E.: Soil carbon varies between different organic and conventional management schemes in arable agriculture, Eur. J. Agron., 94, 79-88, 2018a.

Hu, T., Olesen, J.E., Christensen, B.T. & Sørensen, P.: Release of C and N from fodder radish (Raphanus sativus) shoots and roots incubated in soils of different management history, Acta Agr. Scand. B.-S. P. (in press), 2018b.

IFOAM: The Principles of Organic Agriculture. https://www.ifoam.bio/sites/default/files/poa_english_web.pdf, downloaded April 2019.

IPCC: Guidelines for National Greenhouse Gas Inventories. Agriculture, Forestry and Other Land Use., International Panel oif Climate Change, http://www.ipcc-nggip.iges.or.jp/public/2006gl/index. [online] Available from: http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html, 2006.

Jensen, L.S., Salo, T., Palmason, F., Breland, T.A., Henriksen, T.M., Stenberg, B., Pedersen, A., Lundström, C., and Esala, M.: Influence of biochemical quality on C and N mineralisation from a broad variety of plant materials in soil, Plant Soil 273, 307-326, 2005.

Jeuffroy, M.H., Baranger, E., Carrouée, B., de Chezelles, E., Gosme, M., Hénault, C., Schneider, A., Cellier, P.: Nitrous oxide emissions from crop rotations including wheat, oilseed rape and dry peas, Biogeosciences 10, 1787–1797,https://doi.org/10.5194/bg-10-1787-2013, 2013.

Johnsson, H., Bergström, L., Jansson, P.-E., and Paustian, K.: Simulated nitrogen dynamics and losses in a layered agricultural soil, Agr. Ecosyst. Environ., 18, 333-356, 1987.

Jończyk, K., and Martyniuk, S.: Productivity of crop rotations and selected indicators of soil fertility in different types of ecological farms, J. Res. Appl. Agric. Eng., 62(3), 153-157, 2017.

Knudsen, M.T., Meyer-Aurich, A., Olesen, J.E., Chirinda, N., and Hermansen, J.E.: Carbon footprints of crops from organic and conventional arable crop rotations using a life cycle assessment approach. J. Cleaner Prod. 64, 609-618, 2014.

Kayser, M., Müller, J., and Isselstein, J.: Nitrogen management in organic farming: Comparison of crop rotation residual effects on yields, N leaching and soil conditions, Nutr. Cycl. Agroecosys., 87(1), 21–31, doi:10.1007/s10705-009-9309-0, 2010.

Kirchmann, H. and Bergström, L.: Do organic farming practices reduce nitrate leaching?, Commun. Soil Sci. Plan., 32(7–8), 997–1028, [online] Available from: isi:000170101800005, 2001.

Krauss, M., Krause, H. M., Spangler, S., Kandeler, E., Behrens, S., Kappler, A., Mäder, P., and Gattinger, A.: Tillage system affects fertilizer-induced nitrous oxide emissions, Biol. Fert. Soils, 53(1), 49–59., doi.org/10.1007/s00374-016-1152-2, 2017a.

Krauss, M., Ruser, R., Müller, T., Hansen, S., Mäder, P., and Gattinger, A.: Impact of reduced tillage on greenhouse gas emissions and soil carbon stocks in an organic grass-clover ley - winter wheat cropping sequence, Agr. Ecosyst. Environ., 239, 324–333, doi:10.1016/j.agee.2017.01.029, 2017b.

Kuiper, I., Deyn G. B., Thakur M. P., and Groenigen J. W.: Soil invertebrate fauna affect N2O emissions from soil, Glob. Chang. Biol., 19(9): 2814-2825, 2013.

Kuntz, M., Morely, N. J., Hallett, P. D., Watson, C., and Baggs, E. M.: Residue-C effects on dinitrification vary with soil depth, Soil Biol. Biochem., 103, 365–375, 2016.

Köster, J. R., Cárdenas, L., Senbayram, M., Bol, R., Well, R., Butler, M., Mühling, K. H., and Dittert, K.: Rapid shift from denitrification to nitrification in soil after biogas residue application as indicated by nitrous oxide isotopomers, Soil Biol. Biochem., 43(8), 1671–1677, doi:10.1016/j.soilbio.2011.04.004, 2011.

Känkänen, H., Kangas, A., Mela, T., Nikunen, U., Tuuri, H., and Vuorinen, M.: Timing incorporation of different green manure crops to minimize the risk of nitrogen leaching, Agric. Food Sci. Finland., 7(5–6), 553–567, 1998.

Larsson, L., Ferm, M., Kasimir-Klemedtsson, A., and Klemedtsson, L.: Ammonia and nitrous oxide emissions from grass and alfalfa mulches, Nutr. Cycl. Agroecosys., 51(1), 41–46, 1998.

Li, C. S., Frolking, S., and Butterbach-Bahl, K.: Carbon Sequestration in Arable Soils Is Likely to Increase Nitrous Oxide Emissions, Offsetting Reductions in Climate Radiative Forcing, Climatic Change, 72, 321–38, 2005.

Li, X., Petersen, S. O., Sørensen, P., and Olesen, J. E.: Effects of contrasting catch crops on nitrogen availability and nitrous oxide emissions in an organic cropping system, Agr. Ecosyst. Environ., 199, 382–393, doi:10.1016/j.agee.2014.10.016, 2015a.

Li, X., Sørensen, P., Li, F., Petersen, S.O. & Olesen, J.E.:. Quantifying biological nitrogen fixation of different catch crops, and the residual effects of roots and tops on nitrogen uptake in barley using in situ 15N labelling. Plant Soil, 395, 273-287, 2015b.

Li, X., Sørensen, P., Olesen, J. E., and Petersen, S. O.: Evidence for denitrification as main source of N_2O emission from residue-amended soil, Soil Biol. Biochem., 92, 153–160, doi:10.1016/j.soilbio.2015.10.008, 2016.

Liu, B., Mørkved, P. T., Frostegård, Å., and Bakken, L. R.: Denitrification gene pools, transcription and kinetics of NO, N_2O and N_2 production as affected by soil pH, FEMS Microbiol. Ecol., 72(3), 407–417, doi:10.1111/j.1574-6941.2010.00856.x, 2010.

Liu, S., Han, P., Hink, L., Prosser, J., Wagner M., and Brüggemann N.: Abiotic conversion of extracellular NH_2OH contributes to N_2O emission during ammonia oxidation. Environ. Sci. Technol., 51, 13122-13132. DOI:10.1021/acs.est.7b02360, 2017.

Lorenz, K., and Lal, R.: Environmental Impact of Organic Agriculture, In Adv. Agron. 139, 99-152, Academic Press, 2016.

Lori, M., Symnaczik, S., Mäder, P., De Deyn, G., Gattinger, A., and Dentener, F.: Organic farming enhances soil microbial abundance and activity—A meta-analysis and meta-regression, PLoS One, 12(7), 1–25, doi:10.1371/journal.pone.0180442, 2017.

Lubbers, I. M., Brussaard, L., Otten, W., and Van Groenigen, J. W.: Earthworm induced N mineralization in fertilized grassland increases both N2O emission and crop N uptake, Eur. J. Soil Sci., 62(1), 152-161, 2011.

Lubbers I. M., Van Groenigen K. J., Fonte S. J., Six J., Brussaard L., and Van Groenigen J. W.: Greenhouse-gas emissions from soils increased by earthworms, Nat. Clim. Chang., 3(3), 187-194, 2013.

Lynch, D. H.: Nutrient Cycling and Soil Health in Organic Cropping Systems - Importance of Management Strategies and Soil Resilience, Sustain. Agric. Res., 4(3), 80, doi:10.5539/sar.v4n3p80, 2015.

Løes, A.-K., Bünemann, E. K., Cooper, J., Hörtenhuber, S., Magid, J., Oberson, A., Möller, K.: Nutrient supply to organic agriculture as governed by EU regulations and standards in six European countries, Org. Agric., 7, 395–418. https://doi.org/10.1007/s13165-016-0165-3, 2017.

Marinari, S., Masciandaro, G., Ceccanti, B., and Grego, S.: Evolution of soil organic matter changes using pyrolysis and metabolic indices: A comparison between organic and mineral fertilization, Bioresource Technol., 98(13), 2495–2502, doi:10.1016/j.biortech.2006.09.001, 2007.

Marriott, E. E., and Wander, M. M.: Total and Labile Soil Organic Matter in Organic and Conventional Farming Systems, Soil Sci. Soc. Am. J., 70(3), 950, doi:10.2136/sssaj2005.0241, 2006.

Martyniuk, S., Koziel, M., Stalenga, J., and Jonczyk, K.: Labile factions of soil organic matter and microbial characteristics of soil under organic and conventional crop management systems, Biol. Agric. Hortic., 32, 1–6, doi:10.1080/01448765.2015.1015054, 2016.

McGee, J. A.: Does certified organic farming reduce greenhouse gas emissions from agricultural production?, Agric. Hum. Values, 32(2), 255–263, doi:10.1007/s10460-014-9543-1, 2015.

Meier, M. S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., and Stolze, M.: Environmental impacts of organic and conventional agricultural products - Are the differences captured by life cycle assessment?, J. Environ. Manage., 149, 193–208, doi:10.1016/j.jenvman.2014.10.006, 2015.

Meijide, A., Díez, J. A., Sánchez-Martín, L., López-Fernández, S., and Vallejo, A.: Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate, Agr. Ecosyst. Environ., 121(4), 383–394, doi:10.1016/j.agee.2006.11.020, 2007.

Melander, B., Rasmussen, I.A., and Olesen, J.E.: Incompatibility between fertility building methods and the management of perennial weeds in organic cropping systems, Agr. Ecosyst. Environ., 220, 184-192, 2016.

Molodovskaya, M., Singurindy, O., Richards, B. K., Warland, J., Johnson, M. S., Steenhuis, T. S.: Temporal Variability of Nitrous Oxide from Fertilized Croplands: Hot Moment Analysis, Soil Sci. Soc. Am. J., 76, 1728. https://doi.org/10.2136/sssaj2012.0039, 2012.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., and Niggli, U.: Soil fertility and biodiversity in organic farming., Science, 296(5573), 1694–7, doi:10.1126/science.1071148, 2002.

Moyo, H., Davies, W. P., Cannon, N. D., and Conway, J. S.: Influences of two year red clover-grass ley management on nitrogen economy and following wheat performance, Biol. Agric. Hort., 32:2, 98-109, DOI: 10.1080/01448765.2015.1057865, 2016.

Möller, K., and Stinner, W.: Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and on gaseous nitrogen losses (ammonia, nitrous oxides), Eur. J. Agron., 30(1), 1–16, doi:10.1016/j.eja.2008.06.003, 2009.

Nadeem, S., Hansen, S., Bleken, M. A., and Dörsch, P.: N_2O emission from organic barley cultivation as affected by green manure management, Biogeosciences, 9(7), doi:10.5194/bg-9-2747-2012, 2012.

Nebert L. D., Bloem J., Lubbers I. M., and van Groenigen J. W.: Association of earthworm-denitrifier interactions with increased emission of nitrous oxide from soil mesocosms amended with crop residue, Appl. Environ. Microbiol., 77(12), 4097-4104, 2011.

Neumann, A., Torstensson, G., and Aronsson, H.: Losses of nitrogen and phosphorus via the drainage system from organic crop rotations with and without livestock on a clay soil in southwest Sweden, Org. Agric., 1(4), 217–229, doi:10.1007/s13165-011-0017-0, 2011.

Olesen, J.E., Askegaard, M., and Rasmussen, I.A.: Design of an organic farming crop rotation experiment. Acta Agriculturae Scandinavica, Section B, Soil and Plant Science 50, 13-21, 2000.

Pandey, A., Li, F., Askegaard, M., and Olesen, J. E.: Biological nitrogen fixation in organic and conventional arable crop rotations in Denmark, Eur. J. Agron, 90, 87-95, 2017.

Pandey, A., Li, F., Askegaard, M., Rasmussen, I.A., and Olesen, J.E.: Nitrogen balances in organic and conventional arable crop rotations and their relations to nitrogen yield and leaching losses, Agr. Ecosyst. Environ., 265, 350-362, 2018.

Pappa, V. A., Rees, R. M., Walker, W. A., and Baddely, J. A.: Intercropping can reduce environmental impacts, Agric. Environ. VII L. Manag. a Chang. Environ. Proc. SAC SEPA Bienn. Conf., 287–290, 2008.

Pappa, V. A., Rees, R. M., Walker, R. L., Baddeley, J. A., and Watson, C. A.: Nitrous oxide emissions and nitrate leaching in an arable rotation resulting from the presence of an intercrop, Agr. Ecosyst. Environ., 141(1–2), 153–161, doi:10.1016/j.agee.2011.02.025, 2011.

Pelster, D.E., Chantigny, M.H., Rochette, P., Angers, D.A., Rieux, C., Vanasse, A.: Nitrous Oxide Emissions Respond Differently to Mineral and Organic Nitrogen Sources in Contrasting Soil Types. J. Environ. Qual., 41, 427, https://doi.org/10.2134/jeq2011.0261., 2012

Petersen, S.O., Schjonning, P., Thomsen, I.K., Christensen, B.T.: Nitrous oxide evolution from structurally intact soil as influenced by tillage and soil water content, Soil Biol. Biochem., 40, 967–977, https://doi.org/10.1016/j.soilbio.2007.11.017, 2008.

Petersen, S. O., Mutegi, J. K., Hansen, E. M., and Munkholm, L. J.: Tillage effects on N_2O emissions as influenced by a winter cover crop, Soil Biol. Biochem., 43(7), 1509–1517, doi:10.1016/j.soilbio.2011.03.028, 2011.

Petersen, S.O., Schjonning, P., Olesen, J.E., Christensen, S., and Christensen, B.T.: Sources of Nitrogen for Winter Wheat in Organic Cropping Systems, Soil Sci. Soc. Am. J., 77(1), 155-165, 2013.

Peyrard, C., Mary, B., Perrin, P., Vericel, G., Grehan, E., Justes, E., and Leonard, J.: N_2O emissions of low input cropping systems as affected by legume and cover crops use, Agr. Ecosyst. Environ., 224, 145–156, doi:10.1016/j.agee.2016.03.028, 2016.

Pfiffner L., Mäder P.: Effects of biodynamic, organic and conventional production systems on earthworm populations, Biol. Agric. Hortic., 15:1-4, 2–10, DOI: 10.1080/01448765.1997.9755177, 1997.

Pfiffner, L., Luka, H.: Earthworm Populations in Two Low-Input Cereal Farming Systems, Appl. Soil Ecol., 37, 184–91, doi:10.1016/j.apsoil.2007.06.005, 2007.

Plaza-Bonilla, D., Nolot, J.M., Raffaillac, D., and Justes, E.: Cover crops mitigate nitrate leaching in cropping systems including grain legumes: field evidence and model simulations, Agr. Ecosyst. Environ., 212,1-12, 2015.

Postma-Blaauw, M. B., Bloem, J., Faber, J. H., Van Groenigen, J. W., De Goede, R. G., and Brussaard, L.: Earthworm species composition affects the soil bacterial community and net nitrogen mineralization, Pedobiologia, 50(3), 243-256, 2006.

Poudel D. D., Horwath W. R., Lanini W. T., Temple S. R., Van Bruggen A. H. C.: Comparison of soil N availability and leaching potential, crop yields and weeds in organic, low-input and conventional farming systems in northern California, Agr. Ecosyst. Environ., 90,125–137, 2002.

Prieto, R. A.: The role of earthworms in nitrous oxide emissions from forage agroecosystems, In Masters Abstracts International, (Vol. 50, No. 02), 2011.

Pugesgaard, S., Petersen, S. O., Chirinda, N., and Olesen, J. E.: Crop residues as driver for N_2O emissions from a sandy loam soil, Agric. Forest. Meteorol., 233, 45–54, doi:10.1016/j.agrformet.2016.11.007, 2017.

Rasse, D., Ritchie, J., Peterson, W., Wei, J., and Smucker, A.: Rye cover crop and nitrogen fertilization effects on nitrate leaching in inbred maize fields, J. Environ. Qual., 29, 298–304, doi:10.2134/jeq2000.00472425002900010037x, 2000.

Ravishankara, A. R., Daniel, J. S., and Portmann, R. W.: Nitrous Oxide (N_2O): The Dominant Ozone-Depleting Substance Emitted in the 21st Century, Science 326, 123–125, doi:10.1126/science.1176985, 2009.

Riley, H., Pommeresche, R., Eltun, R., Hansen, S., and Korsaeth, A.: Soil structure, organic matter and earthworm activity in a comparison of cropping systems with contrasting tillage, rotations, fertilizer levels and manure use, Agr. Ecosyst. Environ., 124(3–4), 275–284, [online] Available from: isi:000254944900016, 2008.

Robertson, G. P., and Vitousek, P. M.: Nitrogen in Agriculture: Balancing the Cost of an Essential Resource, Annu. Rev. Envi. Resour., 34(1), 97–125, doi:10.1146/annurev.environ.032108.105046, 2009.

Rochette, P., and Janzen, H. H.: Towards a revised coefficient for estimating N2O emissions from legumes, Nutr. Cycl. Agroecosys., 73(2–3), 171–179, [online] Available from: wos:000233363200006, 2005.

Röös, E., Mie, A., Wivstad, M., Salomon, E., Johansson, B., Gunnarsson, S., Wallenbeck, A., Hoffmann, R., Nilsson, U., Sundberg, S., and Watson, C. A.: Risks and opportunities of increasing yields in organic farming. A review, Agron. Sustain. Dev., 38, 14, https://doi.org/10.1007/s13593-018-0489-3, 2018.

Sahrawat, K. L., and Keeney, D. R.: Nitrous Oxide Emissions from Soils, Adv. Soil Sci., 4, 103–47, 1986.

Shcherbak, I., Millar, N., and Robertson, G. P.: Global metaanalysis of the nonlinear response of soil nitrous oxide (N2O) emissions to fertilizer nitrogen, P. Natl. Acad. Sci. USA, 111(25), 9199–9204, doi:10.1073/pnas.1322434111, 2014.

Shelton, R. E., Jacobsen, K. L., and McCulley, R. L.: Cover Crops and Fertilization Alter Nitrogen Loss in Organic and Conventional Conservation Agriculture Systems, Front. Plant Sci., 8, 1–14, doi:10.3389/fpls.2017.02260, 2018.

Skinner, C., Gattinger, A., Muller, A., Mäder, P., Flieβbach, A., Stolze, M., Ruser, R., and Niggli, U.: Greenhouse gas fluxes from agricultural soils under organic and non-organic management — A global meta-analysis, Sci. Total Environ., 468–469, 553–563, doi:http://dx.doi.org/10.1016/j.scitotenv.2013.08.098, 2014.

Skinner, C., Gattinger, A., Krauss, M., Krause, H.-M., Mayer, J., van der Heijden, M. G. A., Mäder, P.: The impact of long-term organic farming on soil-derived greenhouse gas emissions, Sci. Rep. 9, 1702. https://doi.org/10.1038/s41598-018-38207-w, 2019.

Smith, K. A., Dobbie, K. E., Thorman, R., Watson, C. J., Chadwick, D. R., Yamulki, S., Ball, B. C.: The effect of N fertilizer forms on nitrous oxide emissions from UK arable land and grassland, Nutr. Cycl. Agroecosys., 93, 127–149, 2012.

Spargo, J. T., Cavigelli, M. A., Mirsky, S. B., Maul, J. E., and Meisinger, J. J.: Mineralizable soil nitrogen and labile soil organic matter in diverse long-term cropping systems, Nutr. Cycl. Agroecosys., 90, 253–266, 2011.

Stalenga, J., and Jończyk, K.: Nutrient management and soil organic matter balance in the organic crop production system evaluated by NDICEA model, J. Res. Appl. Agric. Eng., 53(4), 78-84, 2008.

Stehfest, E., and Bouwman, L.: N$_2$O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions, Nutr. Cycl. Agroecosys., 74(3), 207–228, [online] Available from: isi:000240619700001, 2006.

Stenberg, M., Aronsson, H., Linden, B., Rydberg, T., and Gustafson, A.: Soil mineral nitrogen and nitrate leaching losses in soil tillage systems combined with a catch crop, Soil Till. Res., 50(2), 115–125, doi:10.1016/S0167-1987(98)00197-4, 1999.

Stenberg, M., Ulén, B., Söderström, M., Roland, B., Delin, K., and Helander, C. A.: Tile drain losses of nitrogen and phosphorus from fields under integrated and organic crop rotations. A four-year study on a clay soil in southwest Sweden, Sci. Total Environ., 434, 79–89, doi:10.1016/j.scitotenv.2011.12.039, 2012.

Stockdale, E. A., Lampkin, N. H., Hovi, M., Keatinge, R., Lennartsson, E. K. M., MacDonald, D. W., Padel, S., Tattersall, F. H., Wolfe, M. S., and Watson, C. A.: Agronomic and Environmental Implications of Organic Farming Systems, Adv. Agron. 70, 261-327, 2001.

Stopes, C., Lord, E. I., Philipps, L., and Woodward, L.: Nitrate leaching from organic farms and conventional farms following best practice, Soil Use Manage., 18, 256–263, [online] Available from: isi:000178455700004, 2002.

Sturite, I., Rivedal, S., and Dörsch, P.: Nitrous oxide emissions from clover rich leys during the long Northern winter., in Cordovil C.M.d.S.(Ed.), Proceedings of the 18th Nitrogen Workshop – The nitrogen challenge: building a blueprint for nitrogen use efficiency and food security, 30th June – 3rd July 2014, Lisboa, Portugal, pp. 517–518., 2014.

Thangarajan, R., Bolan, N. S., Tian, G. L., Naidu, R., and Kunhikrishnan, A.: Role of organic amendment application on greenhouse gas emission from soil, Sci. Total Environ., 465, 72–96, doi:10.1016/j.scitotenv.2013.01.031, 2013.

Thorup-Kristensen, K.: Effect of deep and shallow root systems on the dynamics of soil inorganic N during 3-year crop rotations, Plant Soil, 288(1–2), 233–248, [online] Available from: isi:000243001400018, 2006.

Thorup-Kristensen, K., and Dresbøll, D. B.: Incorporation time of nitrogen catch crops influences the N effect for the succeeding crop, Soil Use Manage., 26(1), 27–35, [online] Available from: wos:000274666500005, 2010.

Tonitto, C., David, M. B., and Drinkwater, L.: Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics, Agr. Ecosyst. Environ., 112, 58–72, doi:10.1007/s10705-010-9354-8, 2006.

Torstensson, G., and Aronsson, H.: Nitrogen leaching and crop availability in manured catch crop systems in Sweden, Nutr. Cycl. Agroecosys., 56(2), 139–152, doi:10.1023/A:1009821519042, 2000.

Torstensson, G., Aronsson, H., and Bergstrom, L.: Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden, Agron. J., 98(3), 603–615 [online] Available from: isi:000237867600023, 2006.

Tosti, G., Benincasa, P., Farneselli, M., Tei, F., and Guiducci, M.: Barley–hairy vetch mixture as cover crop for green manuring and the mitigation of N leaching risk, Eur. J. Agron., 54, 34–39, doi:10.1016/J.EJA.2013.11.012, 2014.

Tosti, G., Farneselli, M., Benincasa, P., and Guiducci, M.: Nitrogen fertilization strategies for organic wheat production: Crop yield and nitrate leaching, Agron. J., 108(2), 770–781, doi:10.2134/agronj2015.0464, 2016.

Tuomisto, H. L., Hodge, I. D., Riordan, P., and Macdonald, D. W.: Does organic farming reduce environmental impacts? - A meta-analysis of European research, J. Environ. Manage., 112(834), 309–320, doi:10.1016/j.jenvman.2012.08.018, 2012.

Tyson, K. C., Scholefield, D., Jarvis, S. C., and Stone, A. C.: A comparison of animal output and nitrogen leaching losses recorded from drained fertilized grass and grass/clover pasture, J. Agric. Sci., 129(3), 315-323, 1997.

Ussiri, D., and Lal, R.: Global Sources of Nitrous Oxide, in Soil Emission of Nitrous Oxide and its Mitigation, pp. 131–175, Springer Netherlands, Dordrecht., 2013.

Valkama, E., Lemola, R., Känkänen, H., and Turtola, E.: Meta-analysis of the effects of undersown catch crops on nitrogen leaching loss and grain yields in the Nordic countries, Agr. Ecosyst. Environ., 203, 93–101, doi:10.1016/j.agee.2015.01.023, 2015.

Van Groenigen, J. W., Velthof, G. L., Oenema, O., Van Groenigen, K. J., and Van Kessel, C.: Towards an agronomic assessment of N2O emissions: A case study for arable crops, Eur. J. Soil Sci., 61(6), 903–913, doi:10.1111/j.1365-2389.2009.01217.x, 2010.

Wagner-Riddle, C., Congreves, K.A., Abalos, D., Berg, A.A., Brown, S.E., Ambadan, J.T., Gao, X., Tenuta, M.: Globally important nitrous oxide emissions from croplands induced by freeze-thaw cycles, Nat. Geosci. 10, 279–283. https://doi.org/10.1038/ngeo2907, 2017.

Watson, C.A., Fowler, S.M., and Wilman, D.: Soil inorganic-N and nitrate leaching on organic farms, J. Agric. Sci., 120(3), 361-369, 1993.

Watson, C.A., Atkinson, D., Gosling, P., Jackson, L.R., and Rayns, F.W.: Managing soil fertility in organic farming systems, Soil Use Manage., 18, 239-247, 2002.

Weier, K.L., Doran, W.L., Power, J.F., and Walters, D.T: Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. Soil Sci. Soc. Am. J. 57, 66-72, 1992.

Westphal, M., Tenuta, M., and Entz, M. H.: Nitrous oxide emissions with organic crop production depends on fall soil moisture, Agr. Ecosyst. Environ., 254, 41–49, doi:10.1016/j.agee.2017.11.005, 2018.

Willer, H., Schaack, D., and Lernoud, J.: Organic farming and market development in Europe and the European Union. In: Willer, H., and Lernoud, J. (Eds.): The World of Organic Agriculture. Statistics and Emerging Trends 2018. Research Institute of Organic Agriculture (FiBL), Frick, and IFOAM – Organics International, Bonn, p 218 – 255, 2018.

Willer, H. and Lernoud, J. (Eds.): The World of Organic Agriculture. Statistics and Emerging Trends 2018. Research Institute of Organic Agriculture (FiBL), Frick, and IFOAM – Organics International, Bonn, 2018.

Xia, L., Lam, S.K., Wolf, B., Kiese, R., Chen, D., Butterbach-Bahl, K.: Trade-offs between soil carbon sequestration and reactive nitrogen losses under straw return in global agroecosystems, Glob. Chang. Biol., 24, 5919–5932. https://doi.org/10.1111/gcb.14466, 2018.

Captions supplementary material

S1. Soil mineral nitrogen (SMN) contents in soil profiles in organic field trials in Norway and Poland as influenced by crop rotation, soil tillage and N fertilization.

S2. N_2O emission and the five highest daily N_2O flux rates in the given measurement periods for organic field trials in Switzerland, Denmark, Scotland and Norway. WFPS, soil temperature and soil mineral-N at 0-20 cm depth are given for the day with highest flux rate. Abbreviations: CS = Cattle Slurry, CCM = composted cattle manure, PS = Pig slurry, P=ploughing, H = Harrowing, CCinc= Cover crop incorporated, CCs= Cover Crop under sown, CCh = cover crop harvested

S3. Annual N leaching (total N or nitrate N) reported from organic field trials in Europe, as influenced by crops, soil type and N applied as organic fertilizers.

S4. Statistical analyses Eq. (1).

S5. Statistical analyses Eq. (2).