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Can alternative cropping systems mitigate nitrogen losses and improve GHG balance? Results from a 19-yr experiment in Northern France

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Abstract

Alternative cropping systems are promoted to reduce nitrogen (N) losses in the environment and mitigate greenhouse gas (GHG) emissions. However, these supposed benefits are not fully known, rarely studied together and on the long-term. Here, we studied the N inputs, N exports, soil organic N (SON) storage, N leaching, gaseous N emissions and GHG balance in a 19-yr field experiment comparing four arable cropping systems without manure fertilization, under conventional (CON), low-input (LI), conservation agriculture (CA) and organic (ORG) managements. The N surplus, *i.e.* the difference between total N inputs and exports, was lowest in LI (43 kg ha^{-1} yr^{-1}), intermediary for CON and ORG with 63 kg ha^{-1} yr^{-1} and highest in CA (163 kg ha^{-1} yr^{-1}). CA and ORG received high amounts of N derived from biological fixation from alfalfa. The annual SON storage rates markedly differed between CA (55 kg ha^{-1} yr^{-1}) and both CON and LI (13 and 6 kg ha^{-1} yr^{-1}), with intermediary value in ORG (30 kg ha^{-1} yr^{-1}). N leaching, calculated using soil mineral N measurements, reached an average of 21 kg ha^{-1} yr^{-1} and did not significantly differ between treatments. The gaseous N emissions (volatilization + denitrification), calculated as the difference between N surplus, SON storage and N leaching, ranged from 12 kg ha^{-1} yr^{-1} in ORG to 83 kg ha^{-1} yr^{-1} in CA. N_2O emissions were continuously monitored with automatic chambers during 40 months. They varied from 1.20 kg ha^{-1} yr^{-1} in LI to 4.09 kg ha^{-1} yr^{-1} in CA system and were highly correlated with calculated gaseous N emissions. The GHG balance, calculated using SOC and N_2O measurements, varied widely between systems: it was highest in CON and LI, with 2198 and 1763 kg CO_{2eq} ha^{-1} yr^{-1} respectively. In CA, the GHG balance was much more favourable (306 kg CO_{2eq} ha^{-1} yr^{-1}), despite important N_2O losses which partly offset the benefit of SOC storage. ORG was the system with the smallest GHG balance (–65 kg CO_{2eq} ha^{-1} yr^{-1}), acting as a CO_2 sink in the long-term. Similar trends were observed when GHG was expressed per unit of N input or N exported. The N surplus alone was not a good indicator of the N fate in the four agricultural

systems. Complementary predictors of N losses and GHG balance are required to obtain a true overview of the C and N environmental impacts of cropping systems. On an operational point of view, these results should lead to investigate the variability of the GHG emissions within each cropping system.

1. Introduction

The objective of increasing crop yields to meet the worldwide increasing food demand has been putting pressure on the use of nitrogen (N) fertilizers for sixty years, the amount of synthetic N fertilizer applied being multiplied by 7 between 1960 and 1995 (Tilman et al., 2002). However, the N use efficiency (NUE), *i.e.* the ratio between crop N export and N inputs, is often less than 50% (Tilman et al., 2002). The unrecovered N in the crop can be stored in the soil or released in the environment throughout the "nitrogen cascade" (Galloway et al., 2003) as dinitrogen (N_2), nitrogen oxides (NO_x), nitrous oxide (N_2O), ammonia (NH_3) and nitrate (NO_3). These N losses cause environmental damages, with eutrophication of rivers, algal blooms in estuaries or slimes under forest (Sutton et al., 2011) and for human health (Habermeyer et al., 2015; WHO, 2013). The other important environmental challenge of sustainable agriculture is to improve the greenhouse gas (GHG) balance, with two aspects: reduce the CO2 emissions or increase the soil organic C (SOC) stocks and decrease the emissions of N_2O, with regard to its high global warming potential (296 times greater than CO_2 over a 100-year time span).

The awareness of this situation led to the implementation of regulations or initiatives such as the Nitrate Directive in Europe (91/676/CEE), which aims at reducing N losses from agriculture to the groundwater, or the "4 per 1000" initiative launched by the COP21 in 2015 (http://4p1000.org) in order to increase SOC stocks by 4‰ every year. To meet these objectives, alternative farming practices have been promoted, such as the reduction of mineral N fertilization, the establishment of catch crops, the cultivation of legume crops as organic N fertilizer, the introduction of perennial crops in arable systems, the reduction of tillage. An ideal cropping system combining these practices would result in high N exportations, high NUE, low N surplus (Eurostat, 2016) and high C and N storage in soil.

Most studies focused on evaluating separately the environmental impacts of each cropping practice with regard to N losses. For example, straw incorporation, catch crops and reduced N fertilization practices were shown to reduce N leaching (e.g. Beaudoin *et al.*, 2005; Constantin *et al.*, 2010; Hansen *et al.*, 2015), while gaseous N emissions can be decreased by reduced fertilization or increased by no-till (Constantin et al., 2010). These improved practices are often clustered in alternative farming systems, such as low-input farming, organic farming or conservation agriculture. These systems are generally assumed to be more environmental-friendly than conventional farming systems, because of their smaller nutrient inputs, improved biodiversity and/or lower soil disturbance. However, the combination of various practices may complicate the effects of alternative cropping systems on the N flows, and the biochemical processes involved in N fate still need to be clarified. While the positive effect of long-term reduced fertilization on N leaching is commonly accepted, the effect of organic farming and conservation agriculture is still a matter of controversy. Conservation agriculture is characterized by no-tillage and permanent living mulch growing under the main crop (Soane et al., 2012) and its impact on N flows is unclear. For example, it may either result in a reduced N leaching by immobilization of N in the topsoil layer during crop residues decomposition or increase it because of preferential water flows occurring under no-till (Kay et al., 2009). Palm *et al.* (2014) report inconsistent results concerning N_2O emissions which could be increased with a higher moisture at soil surface in CA or decreased in the long-term. In organic farming systems, N input to soil is mainly ensured by organic fertilizers and crop residues derived from N_2 fixing legumes, but this N needs to be mineralized before becoming available (Thorup-Kristensen and Dresbøll, 2010). The N inputs in stockless organic systems mainly derive from the second source and are usually lower than in the corresponding conventional systems. A greater N sequestration seems to occur with these inputs compared to mineral N inputs (Kramer et al., 2002). Gaseous N emissions from organic cropping systems are expected to be smaller since soil mineral N is scarcer than in highly fertilized conventional systems (Skinner *et al.*, 2014).

At the field scale, the N surplus of a cropping system is calculated as the difference between N inputs and N exported by crop harvests. N surplus is often considered as an indicator of the negative impact of agricultural practices on the environment (OECD, 2001). N surplus must be calculated over long enough period, in order to characterize properly the long-term effect of a cropping system on the N fate in the soil-plant system.

Few studies have simultaneously estimated the components of the N balance (N surplus, N leaching, SON storage, gaseous N emissions), the SOC storage and the GHG balance of alternative cropping systems in the long-term. This was the general objective of this paper, which analyses the long-term experiment of "La Cage", comparing conventional (CON), low input (LI), conservation agriculture (CA) and organic (ORG) cropping systems. Autret *et al.* (2016) previously determined the changes in soil organic carbon (SOC) stocks in these four cropping systems. In this paper, we i) compare changes in SON stocks between systems from 1998 to 2014; ii) quantify the N losses through leaching and N_2O emission, iii) determine the N surplus and iv) calculate the GHG balance of each cropping system on the long-term (1998-2017). The SON stocks measured in 1998 and 2014 were used to estimate the SON sequestration rates. N leaching was calculated using measurements of soil mineral N and LIXIM model (Mary et al., 1999). The direct N_2O emissions were continuously measured during 40 months (2014-2017). The N surplus was calculated using records of N inputs and exportations. With regard to the previous observations concerning SOC changes, we hypothesized that the GHG balance could vary widely among the four cropping systems.

2. Material and methods

2.1 Experimental site

The study was conducted at the experimental site of "La Cage" at Versailles, France (48°48' N, 2°08' E) as described by Autret *et al.* (2016). The experiment was established in 1998 in order to assess the agronomic, economic and environmental performances of three alternative systems compared to a prevalent conventional cropping system of Northern France. Before the experimental establishment, the site was evenly maintained under a conventional management. The climate is oceanic temperate with mean annual temperature of 11.3°C and mean annual precipitation of 673 mm. The soil is an artificially drained deep Luvisol (IUSS Working Group WRB, 2006). Its clay content measured in 1998 is 167 g kg^{-1} over the whole field and the pH in water is 7.38 (Appendix A).

2.2 Cropping systems and management

Four cropping systems are compared: a conventional (CON), low input (LI), conservation agriculture (CA) and organic farming (ORG) system. They differ in soil tillage, crop succession and protection, fertilization and crop residues management. Soil was ploughed every year in CON and ORG, and only one out of two years in LI. The CA system, consisting in direct seeding with a permanent plant cover, is conducted in no-till since 1998. The crop rotation of CON and LI was the following: rapeseed *(Brassica napus* L.), winter wheat (*Triticum aestivum* L.), spring pea (*Pisum sativum* L.) and winter wheat. It slightly differed in CA where maize (*Zea mays* L.) was grown two years instead of rapeseed. The main difference occurred in the ORG system in which alfalfa (*Medicago sativa* L.) was introduced thereafter to replace pea and rapeseed. All crop residues were left on the soil surface at harvest. Some of the alfalfa cuts were exported while the others were returned to soil: the proportion of cuts return was 50% and 67% in CA and ORG systems respectively. Catch crops were grown only in the CA system, being composed of oat (*Avena sativa* L.), vetch (*Vicia sativa* L.), white mustard (*Sinapis alba* L.) or fodder radish (*Raphanus sativus* L.). A permanent cover crop, composed of festuca (*Festuca rubra* L.) in the first years and alfalfa thereafter, was maintained under the main crop in CA. It was chemically destroyed at least once every 4 years, before seeding pea crops. Pesticides were used in the CON, LI and CA systems for weed, pest and disease control; their application rate was lower in LI and CA compared to CON system. The ORG system did not receive any pesticide and received authorized P and K fertilizers, according to the European specifications for organic farming. Phosphate fertilizer was applied at a mean rate of 14, 14, 9 and 3 kg P ha^{-1} yr^{-1} and potassium at a rate of 26, 26, 18 and 6 kg K ha^{-1} yr^{-1} in CON, LI, CA and ORG systems respectively. The rate was proportional to the crop yield. Mineral N fertilizers were applied to non-legume crops of CON, LI and CA systems. The N fertilizer rate of each crop was calculated in mid-winter according to a balance-sheet method and split in 2 or 3 applications. The N fertilization rate was highest in CON system and was reduced on average by 22% in LI and 29% in CA systems. In the ORG system, organic N fertilizers (feather meal and guano) were applied solely on wheat during the first years and in 1999 and 2005 for rapeseed, at a low rate (6% of the CON system). No other manure or organic fertilizer was applied.

2.3 Measurements

The timeline of N-related measurements made at La Cage between 1998 and 2017 is summarized in Figure 1.

2.3.1 Crop yields and N uptake

Grain yield was determined based on the quantity of grain collected by the combine harvester and average values of dry matter content of grain. Alfalfa yield was calculated as the sum of the biomass content of exported cuts which was measured directly, and the aboveground biomass returned to soil (not exported) which was estimated using regional references (Autret *et al.*, 2016). When not available, N content in harvested grains and aboveground biomass was assumed to be equal to national reference values (Appendix B.1). They were used to estimate N export by main crops (Table 1). Crop residues, cover crops and catch crops were returned to the soil and therefore not accounted for in the N outputs.

2.3.2 *Soil water and mineral N contents*

Soil mineral N (SMN) and soil water content (SWC) were measured three times per year: after harvest (average date July 30), in autumn (average November 2) and winter (average February 11) between November 2012 and February 2017. Three soil samples were collected in each subplot from soil surface down to 90 cm depth, corresponding to the maximum rooting depth. Soil cores were divided into three layers of 30 cm and frozen until mineral N extraction. The gravimetric water content was determined after 48h drying at 105°C. Mineral N was extracted from 100 g of soil shaken in 200 mL of a potassium chloride solution (1M). Nitrate (NO_3) and ammonium (NH_4) concentrations of the solution were measured by continuous-flow colorimetry (Khan et al., 2007). Bulk densities used to calculate SMN stocks were taken from previous measurements made in 2014 (Autret et al., 2016).

2.3.3 *SOC and SON stocks*

The SOC and SON concentrations and bulk densities were measured at different dates between 1998 and 2014, in order to calculate SON stocks on equivalent soil mass (ESM) basis (Ellert and Bettany, 1995) over a depth at least equal to the deepest tillage event. Twenty samples were taken in each subplot down to 30 cm depth in 1998, whereas six samples were picked up in each subplot at 60 cm depth in 2014 with a hydraulic gauge of 6 cm diameter (Autret *et al.*, 2016). Soil samples were coarsely crushed, oven dried for 48h at 35°C and sieved (2 mm). Soil subsamples were finely ground in a ball mill (PM 400, Retsch, Germany) before measurement of the total N content by Dumas (dry combustion) with an elemental analyser (EURO EA, Eurovector, Italy). Bulk density was measured in 1998 for the layers 0-10, 10-20 and 20-30 cm and in 2014 on the top soil layer 0-5 cm, using a steel cylinder of 98 cm^3. Soil was weighed after drying during 48 h at 105 °C. A gamma-densitometer (LPC-INRA, Angers, France) was used in 2014 to measure the bulk density every 5 cm in the layers from 5 to 40 cm depth.

2.3.4 N_2O emissions

Direct N_2O emissions were monitored continuously between April 2014 and July 2017, using automatic chambers as described by Bessou *et al.* (2010) and Peyrard *et al.* (2016, 2017). On April 8, 2014, twelve automatic, large size chambers (0.70 m x 0.70 m) were installed in four plots of the block 1, representing the four cropping systems with three replicates. Three chambers were installed at a mean distance of 10 m each other in each plot of 0.56 ha. They were maintained in the plot throughout the growing season and removed only for the main operations (sowing, mechanical weeding, harvest and disk ploughing), receiving the same management than the rest of the plot. Chambers were connected to two infra-red gas analysers, one for CO_2 (LiCor 820, LiCor Biosciences, USA) and the other for N_2O (Thermo 46c, Thermo Fisher Scientific, USA) installed in an enclosure left *in situ*. The automatic chambers were closed sequentially four times a day during 20 minutes (starting at 0, 6, 12 and 18 hours GMT). The CO_2 and N_2O concentrations were measured every 10 seconds during the closure. Additional corrections (besides those provided by the manufacturer) were made to account for the (small) interferences between N_2O and CO_2 concentrations. N_2O concentrations were converted into fluxes by fitting the concentration kinetics to a linear or an exponential model. The smaller measurable flux using this method is lower than 1 g N_2O-N ha^{-1} day^{-1}. Cumulative N_2O emissions were obtained by summing up the fluxes measured during 20 minutes every 6 hours.

2.4 Calculations

2.4.1 Biological N fixation

The biological nitrogen fixation (BNF) of legume crops was estimated, either for the main crops, cover crops undersown in the main crop, or catch crops preceding the main crop. The total amount of fixed N was calculated as the sum of the contributions of each category of grown legumes. BNF was calculated using the empirical relation established by Anglade *et al.*

(2015) between the amount of N derived from atmosphere (*Ndfa*) and the N yield of the legume crop:

$$Ndfa = \alpha.Ny + \beta \qquad (1)$$

where α and β are crop specific parameters (Appendix B.2) and *Ny* is the N yield, defined as the total N accumulated in the aboveground biomass (kg N ha^{-1}), calculated as follows:

$$Ny = Y.Nc/NHI \qquad (2)$$

where *Y* is the harvested crop yield (t DM ha^{-1}), *Nc* is the nitrogen content of the harvested material (g kg^{-1}), and *NHI* is the nitrogen harvest index (ratio of N in the harvested material to N in total aboveground biomass). *Ny* was determined using the following variables: i) the measured grain yield for pulses (fababean, lupin, pea and soybean), ii) the estimates of aboveground biomass for the other legumes (alfalfa, vetch and clover), iii) an average value of measured N content for pea, and iv) standard values of N content for the other species (CORPEN, 1988; Parr *et al.*, 2011; Anglade *et al.*, 2015). The *Ndfa* was finally corrected by a multiplicative factor (BGN-F) accounting for belowground contributions which varied between species, in order to estimate the total BNF (Anglade *et al.*, 2015).

2.4.2 N surplus

The N surplus (N_{sur}) relative to the soil-plant system was calculated for each of the four cropping systems according to OECD (2001) by subtracting the total N exportation from the total N inputs. A positive N surplus indicates that N losses occur in the environment and/or that SON stock increases, whereas a negative surplus means a soil impoverishment. The annual surplus (kg ha^{-1} yr^{-1}) was calculated as:

$$N_{sur} = N_{fert} + N_{fix} + N_{atm} - N_{exp} \qquad (3)$$

where N_{fert} is the annual N fertilization, N_{fix} the N derived from symbiotic fixation, N_{atm} the atmospheric N deposition and N_{exp} the N exported from the field by harvests, all values expressed in kg ha^{-1} yr^{-1}. The values of N_{fert} and N_{exp} were recorded each year whereas N_{fix} was calculated as indicated previously. N_{atm} was estimated based on the European Monitoring and Evaluation Program (http://www.emep.int/), providing an annual deposition of 12.9 kg ha^{-1} yr^{-1} at the regional scale in France over the 2000-2015 period.

2.4.3 N leaching

Water drainage, net N mineralization and N leaching were calculated using both measurements of SWC and SMN contents and the LIXIM model (Mary et al., 1999). LIXIM simulates simultaneous N mineralization and water and nitrate transfer between soil layers at a daily time-step, fitting simulated water and mineral N stocks to observed values. We used the adaptation made by Beaudoin *et al.* (2005) to account for the concomitant crop N uptake which can occur when soil is not completely bare fallow during winter. Results showed that leaching predictions were only slightly impacted by crop N uptake.

Input data for LIXIM model were the mean daily temperature and precipitation, potential evapotranspiration, soil bulk density, soil water contents at permanent wilting point and field capacity of each layer, SWC and SMN contents in autumn and winter. When soil was covered by a crop (catch crop, cover crop, main crop or rapeseed regrowth), the date of seeding, depth of rooting, root growth rate, base temperature and N absorption rate were also provided. The depth of rooting, root growth rate and base temperature of each crop were set to values given in Appendix C. The actual to potential evapotranspiration ratio (AET / PET) was set at 0.5 for a bare soil and 0.6 or fitted by the model when a crop was present in winter. The net N mineralization rate was optimized, as well as the nitrogen absorption rate, by fitting the simulated values of water and SMN contents to the observed values.

Drainage and leaching were estimated differently according to the time period. First, they were calculated for the five drainage seasons (2012 to 2017) during which water and mineral N contents were measured both in autumn and winter. During this period, the amounts of drained water (*D*, mm yr^{-1}) and N leached (*L,* kg N ha^{-1} yr^{-1}) were estimated using SWC and SMN measurements together with LIXIM model. This calculation was not possible during the other period (1998-2011), because SWC and SMN contents were measured only in winter. Therefore, drainage was calculated using a simulator based on evapotranspiration, precipitation and soil water capacity (Leenhardt, 1991) and leaching was determined using a statistical relationship between N leached, drained water and SMN measured in February, as follows:

$$L = D^2\left(a_1 S_1 + a_2 S_2 + a_3 S_3\right) \quad (4)$$

where S_1, S_2 and S_3 are the amounts of soil nitrate-N measured in mid-February in layers 0-30, 30-60 and 60-90 cm, respectively. This relationship, established with the data of the second period (2012-2017), explained satisfactorily the leaching simulated by LIXIM: $r^2 = 0.85$, n=20. The fitted value of coefficients a_1, a_2 and a_3 was 0.98, -0.89 and 0.74 respectively.

2.4.4 Gaseous N losses

Based on N surplus, N leaching and SON storage estimated between 1998 and 2016, we calculated the total gaseous N losses (G, in kg ha^{-1} yr^{-1}) with the following equation (Mary *et al.,* 2002; Constantin *et al.*, 2010):

$$G = N_{sur} - (N_{stored} + L) \quad (5)$$

where N_{stored} is the SON sequestration rate which was measured between 1998 and 2014. This equation assumes that the variation in soil mineral N between the beginning and end of the period is negligible, which is true. G includes all N emissions through denitrification (N_2+N_2O), nitrification (N_2O+NOx) and volatilization (NH_3), without detail of their respective proportions.

2.4.5 GHG balance

The total emissions of greenhouse gases at La Cage experiment result from total equivalent CO_2 losses, deriving from the soil and crop management of each cropping. We calculated the annual GHG balance (GHG_b, in kg CO_2eq ha^{-1} yr^{-1}) as follows:

$$GHG_b = F + M + 296.\frac{44}{28}(direct\ N_2Oe + indirect\ N_2Oe) - \frac{44}{12}SOC_{storage} \quad (6)$$

where F is the amount of CO_2 emitted during the fertilizer synthesis (in kg CO_2 ha^{-1} yr^{-1}), M the amount of CO_2 emitted during the plant and soil management (in kg CO_2 ha^{-1} yr^{-1}), *direct N_2Oe* the amount of N_2O emitted by the soil (kg N_2O-N ha^{-1} yr^{-1}), *indirect N_2Oe* the amount of N_2O emitted throughout the N cascade (kg N_2O-N ha^{-1} yr^{-1}) and $SOC_{storage}$ the amount of carbon yearly stored in the soil (kg C ha^{-1} yr^{-1}). F was calculated as the product of the amount of fertilizer applied per hectare and the corresponding emission factors which were 6.17 kg CO_2eq kg^{-1} of N for ammonium nitrate, 1.30 kg CO_2eq kg^{-1} of phosphate (P) and 0.54 kg CO_2eq kg^{-1} of potassium (K) for binary P-K fertilizers (Gac et al., 2010). M was obtained by multiplying the amount of fuel consumed per hectare for soil and crop management (Appendix D) by the emission factor of 0.81 kg CO_2eq per liter of fuel consumed. The direct N_2Oe emissions were measured for the 2014-2017 period. They were used to calculate the impact factors relative to mineral fertilizers and organic N residues for each system during three years. These factors were then used to estimate the *direct N_2O* emissions from 1998 to 2013. The *indirect N_2Oe* emissions were estimated with the emission factors defined in the IPCC Guidelines for National GHG Inventories (De Klein et al., 2006), namely 1% of the N volatilized derived from synthetic fertilizer being transformed into N_2O and 0.75% of the leached N being transformed into N_2O all along the N cascade in groundwater, rivers and estuaries. Finally, the $SOC_{storage}$ was taken from by Autret *et al.* (2016) who measured it during the 1998-2014 period, with 78, 22, 625 and 277 kg C ha^{-1} yr^{-1} for CON, LI, CA and ORG respectively.

2.5 Statistical analysis

Statistical analyses were performed using the R software (R Core Team, 2017). The weakness of the experiment was its design with only two randomized blocks, previously underlined by Henneron *et al.* (2014), Pelosi *et al.* (2015) and Autret *et al.* (2016). These authors considered each of the two sided subplots of a given cropping system as replicate, thus producing four pseudo replicates. Their choice was supported by different arguments and particularly the fact that the intra-plot variability (between subplots) was as important as the inter-plot variability (within blocks), as indicated by the comparison of variances (F=1.83, p<0.05) made by Autret *et al.* (2016). In the present study, we made a more rigorous analysis as recommended by Webster and Lark (2018). We did not consider 16 randomized plots (4 treatments x 2 blocks x 2 subplots) such as the previous authors but 8 randomized plots (4 treatments x 2 blocks) with 2 replicates within plots (in the sided subplots), applying a "nested" analysis of variance. This ANOVA was performed to test the effect of cropping system on 1) SON stocks in 1998 and 2014 and their variation; 2) N inputs, N exportations and N surplus from 1998 to 2016; 3) N leaching between 1998 and 2017; 4) N_2O emissions between April 2014 and July 2017 and 5) GHG balance. The assumptions of ANOVA were checked by using the Shapiro-Wilk and Levene's tests. The existence of significant effects (p<0.05) was followed by a post-hoc comparison test of means with the CLD from the *emmeans* package (Lenth, 2018).

3. Results

3.1 N inputs and export by main and cover crops

The mean input of fertilizer-N over the 19 years experiment was 199, 144 and 166 kg N ha^{-1} yr^{-1} for wheat crops and 189, 169 and 98 kg N ha^{-1} yr^{-1} for rapeseed in the CON, LI and CA cropping systems, respectively (Table 1). It was much lower in the ORG system, with an average of 12 kg N ha^{-1} yr^{-1} (wheat) and 44 kg N ha^{-1} yr^{-1} (rapeseed), both applied as organic fertilizer. The N input derived from symbiotic fixation, related to the legume crop N yield, varied between species. It was estimated at 166, 168, 191 and 105 kg N ha^{-1} yr^{-1} for pea crop in the CON, LI, CA and ORG systems, respectively. Alfalfa, whether grown as a main crop in CA and ORG or as a cover crop mixed with a cereal in CA, provided the highest N inputs, ranging from 154 to 350 kg N ha^{-1} yr^{-1}. BNF represented 20%, 23%, 56% and 84% of total N inputs for CON, LI, CA and ORG systems, respectively. After 2008, organic N fertilizers were no longer applied to the ORG system, which then relied exclusively on legume crops to inject reactive N. The part of BNF in total N inputs also increased with time in the CA system, due to the more frequent use of legumes as cover crop and green manure, from 30% until 2008 to an average of 77% since 2009.

N outputs (export) are directly linked to the grain yields and aerial biomass of alfalfa. Average wheat yields varied between cropping systems, respectively 8.1, 7.4, 6.2 and 4.6 t DM ha^{-1} in CON, LI, CA and ORG. They were correlated to the N fertilizer rate (r=0.51, n=152, p<0.001). The yields of spring pea and winter rapeseed were also lower in the ORG than in the other cropping systems. The mean N export through wheat grains was 149, 137, 123 and 76 kg N ha^{-1} yr^{-1} in CON, LI, CA and ORG respectively. It represented 49 to 59% of the total N export.

3.2 N surplus

The average N surplus calculated between 1998 and 2016 was clearly positive for all systems (Table 2). It varied from 43 to 163 kg N ha^{-1} yr^{-1} and ranked as follows: LI < CON = ORG < CA. The smaller N surplus observed in LI compared to CON results from lower N inputs (169 kg N ha^{-1} yr^{-1}) and a similar symbiotic N fixation. The N surplus of the ORG system did not differ from the CON system, despite its very low fertilizer input but the presence of alfalfa generated high N input (101 kg N ha^{-1} yr^{-1}), three times greater than in the CON system. The very high N surplus observed in CA results from the important amounts of symbiotic N fixation, especially from alfalfa which accounts for 107 kg N ha^{-1} yr^{-1}, corresponding to 65% of the total BNF inputs. The N surplus was positive in all cropping systems, indicating that N had been stored in the soil and/or lost through leaching and gaseous emissions in all systems.

3.3 SON storage

The SON stocks calculated at ESM in the 0-30 cm layer (old ploughed layer) in 1998 and 2014 are presented in Table 3. Small differences, but not significant, were found between treatments in SON stocks in 1998 (p<0.05), the average SON value being 4.25 t N ha^{-1}. In 2014, SON stocks were much higher in CA than in the three other systems, which did not differ significantly each other. SON stocks in the layer 30-60 cm did not differ between systems (results not shown), indicating that most SON variations occurred in the upper layer. The change in SON stocks during the 16 years varied between 0.10 and 0.89 t N ha^{-1}. It was significantly different from 0 in the CA system but not in the other systems. The average rates of N sequestration were 13, 6, 55 and 30 kg N ha^{-1} yr^{-1} in CON, LI, CA and ORG respectively.

3.4 SMN stocks, drainage and N leaching

The measured SMN stocks and the calculated drainage, N leaching and NO_3 concentrations in drained water are presented in Table 4. The SMN stocks in autumn, measured from 2012 to 2017, varied little between systems: they were on average 62, 58, 43 and 58 kg N ha^{-1} in the CON, LI, CA and ORG systems respectively. The SMN stocks found in winter during the same years were lower, respectively 29, 33, 35 and 42 kg N ha^{-1}. The winter SMN during the rest of experiment (1998-2011) were slightly higher but with the same ranking between systems. Although no significant difference was found, the ORG system which received no mineral N fertilizer tended to have the highest SMN stocks in February, varying between 18 and 86 kg N ha^{-1}.

The calculated drainage did not vary significantly between cropping systems but varied markedly between years, from 11 to 334 mm yr^{-1}. The average drainage over the whole period (1998-2017) was 145 mm yr^{-1}. Similarly, the amounts of N leached did not differ significantly between systems but widely among years. The mean amounts of leached N over the whole period were 18, 22, 24 and 20 kg N ha^{-1} yr^{-1} for CON, LI, CA and ORG respectively. The nitrate concentration in drained water did not differ significantly between systems with an average value of 56 (period 2012-2017) and 65 mg NO_3 L^{-1} (whole period), *i.e.* slightly greater than the maximum content of 50 mg NO_3 L^{-1} set by the Nitrate Directive (91/676/CEE). The concentrations calculated during the last five years differed more between systems (37 mg NO_3 L^{-1} in CA *vs* 63 mg NO_3 L^{-1} in CON), suggesting that the improvement in the management of the non-conventional systems might reduce nitrate leaching losses in the near future.

3.5 N surplus and gaseous N losses

The greatest source of uncertainty in the calculation of the N surplus lies in the BNF input from alfalfa in the CA and ORG systems, with two components: 1) the estimate of aboveground biomass and 2) the belowground N derived from BNF. We conducted a sensitivity analysis to determine the change in the N surplus in response to a variation in aboveground production of alfalfa and in the BGN-F factor (ratio of total to aboveground N fixed) (Table 5). Biomass production varied between 90 and 110% of the nominal value (range determined using the measurements made in the last years) and three values of the BGN factor were tested: 1.4, 1.7 and 2.1 corresponding to its minimum, average and maximum value reported by Anglade *et al.* (2015). The N surplus thus calculated varied widely, from 134 to 196 kg N ha^{-1} yr^{-1} in CA and 43 to 95 kg N ha^{-1} yr^{-1} in ORG system. The average values were close to those previously calculated, 161 ± 20 kg N ha^{-1} yr^{-1} for CA and 65 ± 17 kg N ha^{-1} yr^{-1} for ORG. It is noticeable that the difference between the two cropping systems was much more stable, 95 ± 3 kg N ha^{-1} yr^{-1}. This indicates that despite the rather large uncertainty in the estimates of N surplus, the CA system is characterised by a much higher surplus than the ORG system.

The unrecovered N, *i.e.* the difference between the N surplus and the sum of N stored in soil and N leached, corresponds to the gaseous N losses (denitrification + volatilization). Figure 2 displays the N surplus and its partitioning into its three components over the 1998-2017 period.

N surplus was mainly correlated with gaseous losses (r=0.97, p<0.001), moderately with SON storage (r=0.93, p<0.01) and not correlated with N leaching (r=0.60, ns). The gaseous losses differed widely among cropping systems: they were much greater in the CA (83 ±22 kg N ha^{-1} yr^{-1}) than in the CON system (32 ±11 kg N ha^{-1} yr^{-1}) and smallest in the LI (15 ±24 kg N ha^{-1} yr^{-1}) and ORG system (12 ±14 kg N ha^{-1} yr^{-1}). Similar differences were obtained when calculations were made only during the period 2014-2017 (results not shown).

3.6 N_2O emissions

Emissions varied widely between systems and throughout time (Table 6). Over the whole monitoring period (April 2014 – July 2017) the N_2O emissions were highest in the CA system (11.96 kg N_2O-N ha^{-1}), intermediate in the CON system (6.85 kg N_2O-N ha^{-1}) and lowest in the LI and ORG system (3.51 and 4.39 kg N_2O-N ha^{-1}). Emissions occurred mainly but not exclusively after fertiliser application, in response to rainfall events, indicating that denitrification was the main source of N_2O production. The ORG system produced very small emissions except during the last period (November 2016 – July 2017) under wheat established after the destruction of alfalfa. Conversely, the three other systems had small emission rates during this period; this is likely due to a low denitrification itself linked with a low rainfall during this period.

The N_2O emission factor relative to the total N inputs (mineral + organic N) during the whole monitoring period was 1.19%, 0.92%, 1.49% and 1.42% for the CON, LI, CA and ORG systems, respectively. These values are close to IPCC references. In addition, N_2O emissions were highly correlated with total gaseous losses (r=0.97, *p*<0.001). This result suggests that a large part of the gaseous losses originated from denitrification.

3.7 Global GHG balance

The GHG balance calculated for each cropping system is presented in Table 7. The upper part of the table shows the GHG balance during the 2014-2017 period, during which N_2O fluxes were continuously measured, whereas the lower part concerns the 19-yr period (1998-2017). Over the 19 yr-period, emissions deriving from fertilizer synthesis were high in CON (927 kg CO_2eq ha^{-1} yr^{-1}) and low in ORG (44 kg CO_2eq ha^{-1} yr^{-1}) which received small amounts of organic fertilizers in the early years of the trial. The CO_2 emissions related to agricultural operations varied less between systems, from 161 kg CO_2eq ha^{-1} yr^{-1} in CA to 272 kg CO_2eq ha^{-1} yr^{-1} in CON. The differences were mainly due to the absence of soil tillage in CA, ploughing being particularly fuel consuming. The indirect N_2O emissions occurring during the N cascade contributed very little (average 118 kg CO_2eq ha^{-1} yr^{-1}) to the GHG balance. The direct N_2O emissions and the direct CO_2 emissions (estimated by the SOC variation) were the main sources of variability. Taking into account the annual SOC sequestration, the net GHG balance was estimated for each cropping system on the 1998-2017 period and ranked as follows: CON (+2198 kg CO_2eq ha^{-1} yr^{-1}) > LI (+1763) > CA (+306) > ORG (-65). A similar ranking and amplitude of variation was obtained for the period 2014-2017, showing the robustness of the calculations. The ranking was conserved when the GHG balance was expressed per unit of N input or unit of exported N. Intensive mineral N fertilization and mechanization, associated with a poor SOC sequestration rate lead to high GHG emissions in the CON, as well as in LI even with reduced farming intensity. The CA system had a much better GHG balance due to its very high SOC storage rate. However, its high N_2O emissions offset this beneficial effect, resulting in a slightly positive GHG balance. Finally, the ORG system was the only cropping system leading to a negative GHG balance.

4. Discussion

We have studied the long-term impacts of alternative cropping systems on the N fate, including N uptake by crops, SON storage, N leaching and gaseous N emissions, and the GHG balance, including direct and indirect emissions. The results are summarized in Figure 3: each variable of each cropping system was scored between 0 (adverse impact) and 1 (beneficial impact), compared to two reference values (low and high) of each variable, given in the legend.

3.8 N use efficiency

The NUE of each cropping system can be compared to the reference value of 0.66 obtained on average for France during the 2000-2014 period (Eurostat, 2016). A nominal comparison of the NUE of cropping systems requires precaution because NUE is linked to the N fertilizer rate (Mary *et al.*, 2002). Thus, the NUE variability within each system would deserve to be investigated. However, both the N input rate and origin is structurally determined by the nature of each cropping system. During the 19 years of the experiment, LI system yielded the highest NUE, 0.84, CON and ORG obtained respectively 0.69 and 0.55, and the lowest was found in CA system, 0.39. This latter value was explained by the high organic N inputs derived from legume cover crop residues which were preferentially stored in the soil rather than used for plant uptake. Given that straw was systematically returned to soil during the experiment in all cropping systems, the differences in NUE refer mainly to the efficiency of conversion of the supplied N, either by inorganic fertilizers or through BNF, into crop uptake and grain N. Aronsson *et al*. (2007) reported similar NUE values in Sweden: 0.54 for an organic system relying on green manure, 0.68 for a conventional system, but higher efficiency in an organic system with animal manure (0.70). In Italy, higher NUE were found for an organic system (average 0.74 over 16 years) than for a conventional (average 0.51) (Migliorini et al., 2014).

Lin *et al*. (2016) also reported higher NUE for organic systems in a 20-yr experiment in Germany.

3.9 N surplus, an ambiguous indicator

The N surplus is often presented as an indicator of the N losses in arable fields (OECD, 2001), useful to compare management practices at annual time step. However, a positive N surplus may also reflect a SON storage. In our experiment, the N surplus was three times higher in CA than in the other systems (157 *vs* 50 kg N ha^{-1} yr^{-1}), in spite of a smaller addition of mineral N fertilizer compared to the CON system. The important amounts of symbiotic fixed N combined with smaller N exportations are responsible of the very high N surplus in CA. The introduction of legume cover crops in conservation agriculture is used to provide N for the subsequent crop after crop residues mineralization and allow to reduce mineral N fertilization (Scopel et al., 2012). Blesh and Drinkwater (2013) made contrasted observations in fields from the Mississippi River Basin, where legumes and complex crop rotations including annual and perennial species were grown. They found smaller surpluses in these cropping systems compared to mineral N based systems (<10 and 35 kg N ha^{-1} yr^{-1} respectively). The difference between their results and ours can be explained by the large proportion of alfalfa cuts returned to soil in La Cage experiment: 90% in the CA system and 75% in ORG. The N surplus in the LI and CON systems is equal or slightly greater than the mean value (47 kg N ha^{-1} yr^{-1}) reported by Poisvert *et al.* (2017) in the same region over the period 2000-2010. The ORG system is characterized by an unusual high N surplus (63 kg N ha^{-1} yr^{-1}) compared to other published references. This results from the long duration of legumes in the rotation (42%), particularly alfalfa (29%) whose cuts were often returned to soil. The asynchrony between the crop N uptake and the release of mineral N from decomposing residues, described by Crews and Peoples (2005), may have limited N uptake and increased the N surplus.

3.10 N and C storage

The SON measurements realized in 1998 and 2014 indicated that N storage occurred in all treatments but was significant only in the CA system which had a mean rate of sequestration of 55 kg N ha^{-1} yr^{-1}. This result is consistent with Autret *et al.* (2016) who found significant SOC sequestration in the CA system. The authors attributed the SOC increase to important crop residues and root inputs, deriving from supplementary catch crops and cover crops. SOC sequestration rates can be compared to the yearly increase of 4 ‰ targeted by the "4 per 1000" initiative to mitigate CO_2 emissions. They were 1.8, 0.5, 14.1 and 7.0 ‰ yr^{-1} for CON, LI, CA and ORG systems respectively (Autret *et al.*, 2016). SON sequestration rates were close: 3.2, 1.3, 12.5 and 7.6 ‰ yr^{-1} respectively; the conservation and organic systems thus demonstrated a high potential of C and N storage in soil over almost a two-decade period. If the important storage could be expected in the CA system (González-Sánchez et al., 2012), the storage in the ORG system is more surprising. The positive impact of organic systems on SOC/SON storage claimed by some studies (Gattinger et al., 2012; Lin et al., 2016) is debated (Leifeld et al., 2013) and/or attributed to higher application of organic fertilizer in organic farming experiments (Leifeld and Fuhrer, 2010). In our experiment, very low amounts of organic fertilizer were applied in the ORG system. We hypothesize that the storage in ORG (and also CA) mainly result from the important amount of legume residues, particularly alfalfa, which provided both C and N substrate needed for C and N sequestration in soil: large C inputs were reported by Autret *et al.* (2016) and large N surpluses were shown in this study. This hypothesis is consistent with Van Groenigen *et al.* (2017) who pointed out the importance of N required for SOC sequestration.

3.11 N leaching

The N leaching mitigation (Figure 3) was assessed by comparing the NO_3 concentrations in drained water to a reference threshold of 50 mg L^{-1} defined by the Nitrate Directive. The average leaching loss over the 1998-2017 period was 16 kg N ha^{-1} yr^{-1}, without any difference between cropping systems. This value was lower than the average N leaching reported by Benoit *et al.* (2014) in the same region for conventional cropping systems (32-77 kg N ha^{-1} yr^{-1}) but closer to those reported for organic farms (13-37 kg N ha^{-1} yr^{-1}). N leaching was not related with N surplus. The correlation between N leaching and N surplus, reported by Billen *et al.* (2013) for conventional systems, does no longer apply when comparing complex cropping systems involving diversified cropping practices and/or rotation. Other studies also confirmed this poor or absence of correlation in arable cropping systems (Sieling and Kage, 2006; Pugesgaard *et al.*, 2017).

A moderate reduction in mineral fertilizer N has been shown to have small effects on N leaching if the reference system is not over-fertilized (*e.g.* Constantin *et al.*, 2010). This was the case of LI compared to CON. In the case of CA system, opposite effects occurred: the presence of permanent cover crop favoured catching of mineral N whereas alfalfa destruction and decomposition during autumn and winter increased SMN. The absence of tillage in this system did not seem to reduce N leaching, in agreement with previous results (Oorts *et al.*, 2007; Hansen *et al.*, 2015; Daryanto *et al.*, 2017). The similar N leaching observed in the ORG system is more surprising with regard to the literature. Organic cropping systems generally lead to a decrease of N leaching because of a lower level N fertilizer applied, as highlighted by Tuomisto *et al.* (2012) in their meta-analysis. Syswerda *et al.* (2012) also found smaller nitrate losses in an organic system using legume catch crops compared to a mineral N fertilizer-based conventional system; the average nitrate concentration in drained water over 11 years was decreased to 40 instead of 84 mg NO_3 L^{-1}, respectively. In our study, the favourable effect of low fertilizer rate was probably offset by the risky phase of alfalfa destruction. Several authors have mentioned that the poor synchrony between mineral N availability derived from alfalfa residues and the subsequent crop uptake could increase N leaching (Crews and Peoples, 2005; Aronsson *et al.*, 2007).

3.12 Gaseous N emissions

The N_2O emissions monitored continuously during 40 months are assumed to be representative of the four systems, at least during the last years. The emission factor (ratio of cumulative N_2O-N emissions to total N inputs) calculated for this period vary from 0.7% (LI) to 1.5% (CA), *i.e.* close to the 1% value of IPCC. The CA system was characterized by high rates of N_2O emissions after fertilization events, much higher than those observed in the LI system, although both systems received almost the same amount of fertilizer-N. Such difference may result from the absence of tillage or the presence of the living mulch or both effects. The influence of no tillage on N_2O emissions is not clear yet, since opposite results have been reported. In their meta-analysis, van Kessel *et al.* (2013) did not point out differences between tilled or no-tilled systems during the first ten years, and found lower emissions in no-till systems after 10 years under dry climates. Therefore, tillage does not seem to be the main reason for the difference between systems at La Cage. The mulch formed by dead crop residues falling on the top soil layer together with the more humid micro-climate created by the living cover crop may have stimulated the emissions in the CA system. Shan and Yan (2013) have shown in their meta-analysis that the presence of mulch of plant residues stimulates N_2O emissions compared to their incorporation in soil. Crop residues, left as a mulch on the top soil layer, have been shown to increase soil moisture content of soil surface and exacerbate the N_2O emissions under annual or perennial crops (Peyrard et al., 2017, 2016). Moreover, residues deriving from alfalfa probably increased N_2O emissions in the CA system, since legumes residues are known to induce higher N losses than non-legume (Basche et al., 2014).

The total gaseous N emissions, assessed by the N mass balance, correspond to the sum of NH_3 volatilization, production of N_2 and N_2O by denitrification and NOx and N_2O by nitrification, all fluxes being stimulated by mineral N fertilization (De Klein et al., 2006). In our study these losses were not correlated with the fertilizer rate, but with the N surplus ($r = 0.97$, $p < 0.001$). They were also well correlated with N_2O emissions ($r = 0.98$, $p < 0.001$), which suggests that denitrification was the major source of emission to the atmosphere. The CA system exhibited the highest N losses (average 79 kg N ha^{-1} yr^{-1}) and ORG the smallest ones (10 kg N ha^{-1} yr^{-1}). If we assume that most gaseous losses are due to denitrification, then the $N_2O/(N_2+N_2O)$ molar ratio would vary from 5% (CA) to 15% (ORG), *i.e.* in the lower range of reported values (*e.g.* Wang *et al.*, 2011).

3.13 The GHG balance, an ultimate environmental indicator

Our GHG balance accounts for the main sources of CO_2 emissions, including mineral fertilizer synthesis, fuel combustion due to crop and soil management, direct and indirect N_2O emissions from soil, groundwater, rivers and estuaries and net CO_2 emissions from soil (assessed using SOC change rate). The average GHG balance was 2198 kg CO_{2eq} ha^{-1} yr^{-1} in the conventional system and 20% smaller in the LI system. Due to its very high C sequestration rate, the CA system had a much more favourable balance, emitting only 306 kg CO_{2eq} ha^{-1} yr^{-1} to the atmosphere. The best situation was found in the ORG system which was a sink for the atmosphere (-65 kg CO_{2eq} ha^{-1} yr^{-1}). The relative differences between systems are maintained when the GHG is expressed per unit of N input or N exported instead of area unit.

Very few studies have quantified the GHG balance of such arable alternative cropping systems without manure application. Most studies focused on gross GHG emissions, without considering SOC storage and/or N_2O emissions. Six *et al.* (2004) estimated a negative GHG balance in no-till systems compared to conventional tillage, but they mentioned the large uncertainty of their estimation, related to the variability of N_2O emissions. Mary *et al.* (2014)

also compared tilled versus no-tilled systems in a long-term experiment at Boigneville (France) in which SOC stocks had been monitored for 41 years and N_2O emissions for three years. They found high values of the GHG balance both for no-till (3350 kg CO_{2eq} ha^{-1} yr^{-1}) and ploughed systems (4030 kg CO_{2eq} ha^{-1} yr^{-1}). Aguilera *et al*. (2015) estimated the carbon footprint of rainfed crops under conventional and organic management in 8 Spanish farms. They estimated higher net GHG emissions from conventional than organic systems when expressed per unit of area (1024 *vs* 361 kg CO_{2eq} ha^{-1} yr^{-1}) and also per unit of production (315 *vs* 182 g CO_{2eq} kg^{-1}). The authors attributed the low carbon footprint of organic management to reduced CO_2 emissions deriving from synthetic fertilizers use and lower direct N_2O emissions since SOC sequestration rates were small and similar in conventional and organic systems.

The GHG balance can be presented as an ultimate indicator giving a wider evaluation of cropping systems, in the context of the global climate change mitigation with alternative cropping managements. In our study, the ranking of GHG balance among cropping systems differed completely from the ranking of N surplus. The N_2O losses accounted for a small share (5-15%) of the total N losses in the four cropping systems, whereas they represented on average 49% of total GHG emissions. In the CA system, N_2O emissions offset 66% of the SOC sequestration, even though SOC change rate reached the top range achievable in arable cropping systems. The importance of assessing N_2O fluxes on the global warming potential was previously pointed out by Six *et al.* (2004). Using a simulation model, Li *et al.* (2005) predicted that C sequestration often goes along with increased N_2O emissions in alternative cropping systems. Our results confirm the idea that no-till management, which is often considered as the main attribute of conservation agriculture, should not be seen as an ultimate solution to mitigate global warming (VandenBygaart, 2016). Hence, the need for a complete evaluation of the GHG balance in designing alternative cropping systems is highly critical, yet being an arduous task (Skinner *et al.*, 2014).

5. Conclusion

The N surplus, N fate and the GHG balance were compared in the conventional and three alternative arable cropping systems over 19 years in La Cage experiment (Northern France). Our purpose was not to conclude about the performance of cropping systems in a nominal approach but to understand how their characteristics could impact their N balance and GHG emissions. The four systems had contrasted impacts on C and N cycles: they had similar effects on nitrate leaching, but very different ability to sequester C and N in soil, and contrasted gaseous N emissions, including denitrification and N_2O losses. Their GHG mitigation potential was not reflected by their N surplus. The alternative systems all improved the GHG balance, slightly for the low input system, markedly for the conservation agriculture system systems and even more in the organic cropping system, which led to a negative GHG balance. In conservation agriculture, the high N_2O emissions partially offset the very high carbon sequestration rate.

Agricultural policies targeting a single environmental objective, such as the “4 per 1000” initiative (Minasny et al., 2017), are welcome but must be considered cautiously since they may potentially overestimate the CO_2 sequestration potential (*e.g.* White *et al.*, 2017). Our results clearly demonstrate that the full GHG balance has to be considered when comparing the potential of new management practices. Hence, an appropriate assessment of the environmental impact of a cropping system should be based on a global evaluation, considering both C and N fluxes modified by the farming practices and not be limited to a patchy indicator. Our study confirms the interest of long-term monitoring to accurately evaluate the impact of alternative systems. Moreover, there is a need to investigate other CA or ORG systems varying in the nature and importance of legume residue returns, which constitute the main alternative to mineral N fertilizer for injecting reactive nitrogen into soils. This investigation can be done using experiments, modelling or both.

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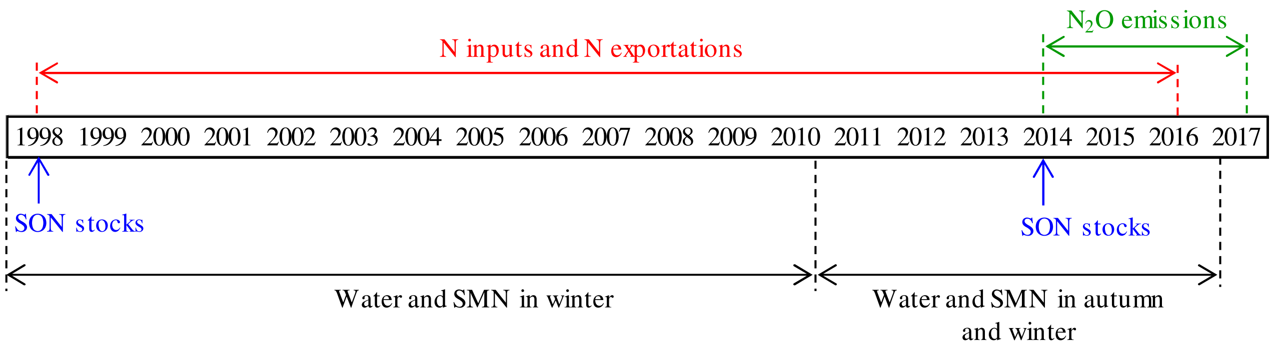

Figure 1.

Sequences of N measurements made between 1998 and 2017, used in this paper. SON = soil organic nitrogen; SMN = soil mineral nitrogen

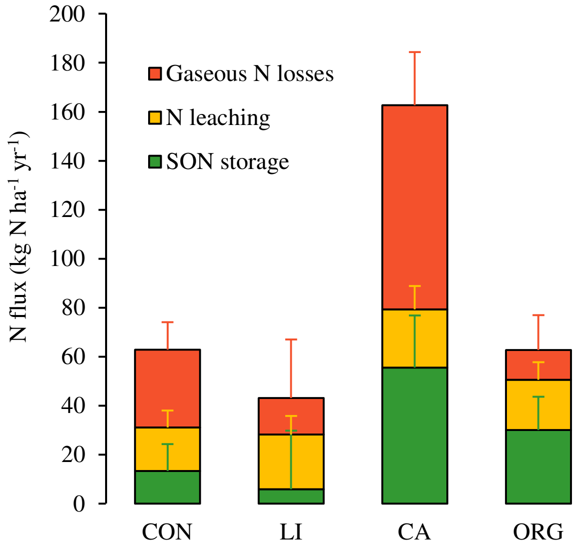

Figure 2.

Partitioning of the N surplus between SON storage, N leaching and gaseous N emissions (see eq. 5). Gaseous N losses were calculated as the difference between the N surplus and the sum of SON storage and N leaching. Error bars represent the standard deviations.

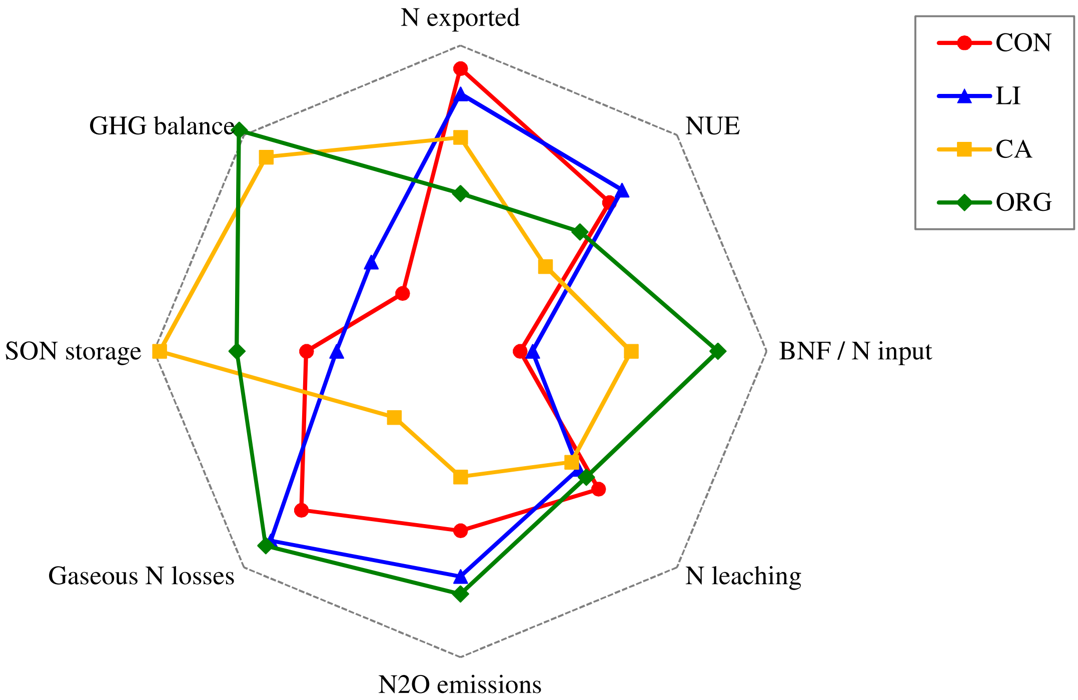

Figure 3.

Diagram summarizing the environmental C and N impacts of the four cropping systems. Each dimension was scored between 0 (adverse impact, center of the radar) and 1 (beneficial impact, periphery). These scores correspond to the following values (respectively): N exported: 0 and 150 kg N ha^{-1}; NUE: 0 and 1; BNF/N input: 0 and 1; N leaching: 150 and 0 mg NO_3 l-1; N2O emissions: 6 and 0 kg N ha^{-1} yr^{-1}; gaseous N losses: 120 and 0 kg N ha^{-1} yr^{-1}; SON storage: -8 and +13 ‰ yr^{-1}; GHG balance: 0 and 3000 kg CO_2eq ha^{-1} yr^{-1}

Table 1.

Crop frequency, mean N inputs from fertilizers and BNF (biological N fixation), mean N offtake by crops and mean grain yield per crop and cropping system during the whole period (1998-2016). Values in brackets are standard deviations between years.

cropping system	main crop	cover crop	crop frequency	N fertilizer		BNF		N offtake		Grain yield		
				kg N ha^{-1} yr^{-1}						t DM ha^{-1}		
CON	wheat		50%	199	(25)			149	(27)	8.1		(1.5)
	rapeseed		26%	189	(31)			123	(24)	3.9		(0.8)
	pea		24%			166	(58)	135	(53)	3.7		(1.3)
LI	wheat		50%	144	(27)			137	(21)	7.4		(1.2)
	rapeseed		26%	169	(15)			99	(26)	3.1		(0.8)
	pea		24%			168	(62)	134	(55)	3.7		(1.4)
CA	wheat	fescue	26%	154	(19)			116	(12)	6.0		(0.9)
	wheat	alfalfa	21%	181	(36)	204	(44)	128	(26)	6.3		(1.5)
	wheat	clover	3%	166		112		149	(0)	7.3		
	rapeseed	alfalfa	5%	98	(4)	154	(7)	91	(57)	2.9		(1.8)
	pea		16%	8		191	(79)	121	(52)	3.3		(1.4)
	oat	alfalfa	3%			237		42	(0)	2.6		
	maize	clover	11%	154	(36)	82	(99)	74	(30)	5.2		(2.1)
	alfalfa		16%			317	(25)	67	(5)	9.8	*	(0.7)
ORG	wheat		50%	12	(23)			76	(27)	4.6		(1.6)
	rapeseed		8%	44	(40)			20	(18)	0.6		(0.6)
	pea		5%			105	(27)	94	(26)	2.6		(0.7)
	barley-pea		3%			112		67	(0)	2.9		
	lupin		3%			190		142	(0)	3.2		
	soyabean		3%			117		84	(0)	1.5		
	alfalfa		29%			350	(30)	87	(37)	10.7	*	(0.9)

** total aerial biomass produced for alfalfa as a main crop*

Table 2.

Mean annual values of N input, N exported and N surplus (period 1998-2016). Total N input is the sum of N fertilization, BNF and atmospheric N deposition. Values in brackets are standard deviations between replicates. Different letters indicate significant differences between cropping systems (p<0.05).

Cropping system	N fertilization			BNF			atmospheric deposition	total N input			total N exported			N surplus		
	kg N ha^{-1} yr^{-1}															
CON	149	(9)	a	39	(13)	b	13	202	(5)	b	139	(10)	a	63	(6)	b
LI	117	(8)	b	40	(17)	b	13	169	(10)	c	126	(11)	a	43	(3)	c
CA	106	(4)	b	149	(9)	a	13	268	(7)	a	105	(7)	b	163	(11)	a
ORG	9	(1)	c	118	(6)	a	13	140	(7)	d	77	(6)	c	63	(2)	b

Table 3.

SON stocks at equivalent soil mass in 1998 and 2014 and SON storage rate between 1998 and 2014. Values in brackets are standard deviations. Small letters indicate significant differences between cropping systems, capital letters indicate differences between years (p<0.05).

Cropping system	SON stock 1998			SON stock 2014				SON storage rate (1998-2014)		
	t N ha^{-1}			t N ha^{-1}				kg N ha^{-1} yr^{-1}		
CON	4.13	(0.33)	*a*	4.34	(0.31)	*b*	*A*	13	(11)	*b*
LI	4.47	(0.74)	*a*	4.57	(0.43)	*b*	*A*	6	(24)	*b*
CA	4.44	(0.64)	*a*	5.33	(0.58)	*a*	*B*	55	(21)	*a*
ORG	3.95	(0.42)	*a*	4.44	(0.27)	*b*	*A*	30	(14)	*ab*

Table 4.

Mean annual drainage, SMN stocks in autumn and winter, N leached and NO_3 concentration in drained water for each cropping system. NO_3 concentrations are weighted by water drainage. N leached and NO_3 concentrations were estimated with LIXIM model between 2013 and 2017 and calculated during the remaining years using eq. 4 and 5.

Drainage season	Drained water	CON				LI				CA				ORG			
		SMN autumn	SMN winter	N leached	$[NO_3^-]$	SMN autumn	SMN winter	N leached	$[NO_3^-]$	SMN autumn	SMN winter	N leached	$[NO_3^-]$	SMN autumn	SMN winter	N leached	$[NO_3^-]$
	mm yr^{-1}	kg ha^{-1}		kg ha^{-1} yr^{-1}	mg L^{-1}	kg ha^{-1}		kg ha^{-1} yr^{-1}	mg L^{-1}	kg ha^{-1}		kg ha^{-1} yr^{-1}	mg L^{-1}	kg ha^{-1}		kg ha^{-1} yr^{-1}	mg L^{-1}
1998-1999	269		9	23	37		21	44	25		24	67	35		18	15	28
1999-2000	286		15	60	93		18	40	29		14	27	23		21	62	27
2000-2001	334		8	39	52		14	64	19		44	142	61		22	68	30
2001-2002	191		32	28	66		46	35	64		20	14	28		41	8	48
2004-2005	99		49	7	31		47	8	64		25	6	35		86	0	94
2005-2006	37		49	0	5		80	0	120		32	1	49		25	1	37
2006-2007	171		12	20	53		42	50	59		44	45	64		63	62	67
2007-2008	60		19	2	16		16	1	16		18	2	25		22	2	24
2008-2009	11		49	0	10		36	0	49		11	0	17		55	0	71
2009-2010	56		25	3	24		29	2	39		17	2	24		33	5	37
2011-2012	150		8	8	24		19	14	10		42	20	61		21	23	29
1998-2012	152		25	17	51		33	23	69		27	30	87		37	22	65
2012-2013	183	113	30	58	140	92	38	56	136	40	28	17	40	63	29	37	89
2013-2014	192	35	19	18	42	45	27	21	48	42	38	24	55	55	43	28	65
2014-2015	139	58	21	10	31	65	29	11	35	38	17	8	26	38	38	7	22
2015-2016	110	29	3	6	23	27	8	6	25	25	9	5	20	58	27	11	45
2016-2017	26	74	70	1	11	60	61	5	78	70	81	1	19	73	71	1	16
2012-2017	130	62	29	18	63	58	33	20	67	43	35	11	37	58	42	17	57
1998-2017	145		26	18	54		33	22	68		29	24	73		38	20	63

Table 5.

Sensitivity analysis of N surplus (kg N ha^{-1} yr^{-1}) to the aboveground biomass and BGN factor of alfalfa for the conservation agriculture (CA) and organic (ORG) cropping systems. Values in brackets are standard deviations. The N surplus used in the study is shown in bold.

Cropping system	BGN factor	Relative variation of AG biomass				
		90%	100%	110%	mean	sd
CA	2.1	176	190	204		
	1.7	152	**163**	173	167	(21)
	1.4	139	148	157		
ORG	2.1	76	86	97		
	1.7	55	**63**	71	67	(17)
	1.4	44	51	58		
Difference CA-ORG	2.1	100	103	107		
	1.7	97	100	103	100	(4)
	1.4	94	97	99		

Table 6.

Cumulative N_2O-N fluxes (kg N ha-1) measured continuously during 6 periods (4 crops and 40 months). Values in brackets are standard deviations. Letters indicate significant differences between cropping systems at each period (Student test, p<0.05).

Measurement period		Duration days	Crop		Cropping system CON			LI			CA			ORG		
08/04/2014	20/07/2014	104	wheat		1.50	(0.55)	b	0.93	(0.19)	b	2.98	(0.90)	a	0.22	(0.07)	c
02/09/2014	08/07/2015	310	rapeseed	*	1.44	(0.14)	b	1.30	(0.15)	b	2.51	(0.39)	a	0.51	(0.04)	c
10/07/2015	19/10/2015	102	fallow	*	0.17	(0.05)	a	0.17	(0.10)	a	0.09	(0.05)	a	0.23	(0.02)	a
10/11/2015	26/07/2016	260	wheat		2.31	(0.54)	b	0.86	(0.16)	c	5.21	(1.39)	a	0.54	(0.13)	d
31/08/2016	19/10/2016	50	fallow	*	0.38	(0.08)	b	0.15	(0.03)	c	0.55	(0.09)	a	0.57	(0.18)	a
16/11/2016	16/07/2017	243	wheat		1.04	(0.10)	b	0.11	(0.02)	d	0.62	(0.14)	c	2.32	(0.56)	a
08/04/2014	16/07/2017	1069	all		6.85	(1.48)	b	3.51	(0.88)	c	11.96	(3.29)	a	4.39	(0.95)	c

** alfalfa for the ORG system*

Table 7.

GHG balance and components at La Cage estimated during the 2014-2017 period (with N_2O monitoring) and the whole experimental period (1998-2017).

Period	Cropping system	*F*		*M*		*direct* N_2Oe		*indirect* N_2Oe	SOC storage rate		GHG balance kg CO_{2eq} ha^{-1} yr^{-1}		GHG balance kg CO_{2eq} kg^{-1} N input		GHG balance kg CO_{2eq} kg^{-1} N exported	
		kg CO_{2eq} ha^{-1} yr^{-1}														
2014-2017	CON	861	*a*	281	*a*	1088	*b*	94	286	**	2039	*a*	10.1	*a*	14.7	*a*
	LI	644	*b*	251	*a*	558	*c*	85	81	**	1457	*b*	8.6	*b*	11.5	*b*
	CA	606	*b*	131	*b*	1900	*a*	79	2292	**	425	*c*	1.6	*c*	4.0	*c*
	ORG	0	*c*	243	*a*	698	*c*	41	1016	**	-33	*d*	-0.2	*d*	-0.4	*d*
1998-2017	CON	927	*a*	272	*a*	1153	*	131	286	*a*	2198	*a*	10.9	*a*	15.8	*a*
	LI	728	*b*	249	*b*	736	*	132	81	*a*	1763	*b*	10.4	*b*	14.0	*b*
	CA	659	*b*	161	*c*	1645	*	132	2292	*c*	306	*c*	1.1	*c*	2.9	*c*
	ORG	44	*c*	255	*b*	576	*	76	1016	*b*	-65	*d*	-0.5	*d*	-0.8	*d*