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High N₂O variations induced by agricultural practices in integrated weed management systems

Anthony Vermue^{1,3} · Bernard Nicolardot^{1,4} · Catherine Hénault²

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Abstract Integrated weed management aims to decrease the dependence of cropping systems on herbicides by using a combination of several agricultural practices. Environmental impacts of individual practices under various conditions are already known. However, there is scarce knowledge on the impact of combining several practices. Therefore, we studied N₂O emissions of weed management cropping systems that use differing practices such as crop diversity, tillage, and herbicide pressure, during about 1 year. Data were compared with a conventionally managed cropping system. Results were also simulated using the NOE model. Results show a large variation of N₂O emissions, ranging from 177 g N₂O-N/ha for intensive tillage integrated weed management to 362 g N₂O-N/ha for the conventional cropping system, 777 g N₂O-N/ha for the no herbicide cropping system and 5226 g N₂O-N/ha for no-till integrated weed management. Most N₂O emissions occurred in spring, despite the absence of fertilizing related N₂O peaks. The lowest emissions occurred in autumn and winter. Emissions are explained by interactions between specific soil potential denitrification rate, soil bulk density,

temperature, soil water, and inorganic N contents. N₂O emissions are accurately predicted by the model NOE, with a global Nash-Sutcliffe coefficient of efficiency equal to 0.80.

Keywords Nitrous oxide · High-resolution measurements · Modeling · NOE model · Integrated weed management · Cropping systems · Tillage

1 Introduction

With the increase in food needs and environmental concerns, agriculture is currently facing new challenges (Foley et al. 2011) requiring the development of more sustainable cropping systems (Reganold et al. 2001).

Integrated weed management aims to lower reliance on herbicides in cropping systems (Chikowo et al. 2009) with limited economic and social impacts (Pardo et al. 2010), while introducing new combinations of agricultural practices in the development of these systems. These combinations may differ greatly from one system to another and include a large variety of practices, such as false seed beds, late sowing, mechanical weeding, reduced tillage, specific crop rotations that alternate spring and winter crops, the choice of crop varieties, and the use of pesticides with low ecotoxic impacts (Zoschke and Quadranti 2002). However, in the framework of developing more sustainable agriculture, the performance of systems is not only evaluated as a function of agronomical, economic, and social criteria but also as a function of environmental criteria. When several agricultural practices are implemented, they are likely to alter soil biogeochemical cycles (Chêneby et al. 2009) and different components of the greenhouse gas budget (balance between carbon sequestration and greenhouse gas emissions). Crop rotations are also reported to potentially affect N₂O emissions from soils (Vilain et al. 2010), as are the

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dates of application and levels of nitrogen fertilization inputs (Van Groenigen et al. 2010). N_2O emissions from soils are the result of complex interactions between the structure and functioning of microbial communities, as well as soil conditions affected by climatic conditions and land use (Weitz et al. 2001). The variety of pedoclimatic contexts and combinations of soil agricultural practices make their impacts on N_2O emissions difficult to study in croplands (Hénault et al. 2012). Soil management practices have been reported to affect N_2O emissions, especially no-till systems which are widely promoted for carbon sequestration purposes (Rochette 2008). Nevertheless, due to considerable N_2O emission during the first years following a no-till conversion, the benefits for mitigating global warming apparently occur only in the long term (Six et al. 2004).

In combination with the field measurement of N_2O fluxes and environmental parameters, models can provide better understanding of the mechanisms of N_2O emissions in soils and estimate the N_2O budgets of cropping systems. The comparison between measured and simulated fluxes can help to analyze the impacts of each soil variable and the effects of different types of agricultural management (Gabrielle et al. 2006). This approach requires using adequate statistical indicators to assess both the association of and the coincidence between simulated and observed N_2O emissions (Smith et al. 1996).

The environmental impacts of integrated weed management cropping systems were the subject of global studies using, for example, life cycle analysis (Deytieux et al. 2012), but specific impacts on soil, atmosphere, and water compartments were not assessed experimentally. Thus, our main objective was to contribute to the environmental evaluation of integrated weed management cropping systems by measuring the greenhouse gas emissions emitted from these systems. The specific objectives of this study are (i) to evaluate and compare the N_2O fluxes emitted from soil for 1 year for four cropping systems (i.e., three integrated weed management systems using different strategies to control weeds and a local conventional system as reference); (ii) to investigate the relationship between N_2O fluxes, soil parameters, and cropping systems; and (iii) to perform simulations using the NOE model in order to better explain the N_2O emission patterns observed.

2 Materials and methods

2.1 Experimental site

The study was conducted at the INRA experimental farm of Dijon-Epoisses, eastern France, (47° 20' N, 5° 2' E). The climate was semi-continental, with a mean annual temperature of 10.5 °C and an average rainfall of 770 mm year⁻¹. The experimental site was set up in 2000 to assess the performance of four cropping systems based on integrated weed

management compared to a reference standard system (Chikowo et al. 2009). The original design of the experiment integrated two geographically distant blocks including all the systems, but only one block is electrified at present and therefore used in this study. The surface areas of the four systems investigated were about 1.7 ha each, separated by a grass strip. These systems differed in terms of crop rotations, soil tillage, mechanical and chemical weeding, and crop management.

The first cropping system was the standard reference system (S1) designed to maximize financial returns, with emphasis placed on the use of chemical herbicides for weed control. Moldboard plowing was carried out each year during summer, and herbicides were chosen in accordance with the recommendations of extension services and pesticide producers that must conform to the rules of application defined by ANSES (French Agency of Food, Environmental, and Occupational Health and Safety). The second system (S2) was an integrated weed management system with no-tillage and direct seeding, with a herbicide treatment frequency reduced to 25 % in comparison with S1. The third system (S3) was an integrated weed management system that allows for plowing and other soil tillage operations when necessary, according to field observations, for weed seed bank management. Herbicide treatment frequency for system S3 was half that of the reference system S1. The last system studied (S5) was a total integrated weed management system with no herbicide treatment. However, in contrast with organic farming, tillage and other pesticide treatments were possible when required. All the systems were studied for 1 year, between March 2012 and March 2013. Crops and cultural operations during the experiment are detailed in Table 1 for each cropping system.

The soil of the experimental site was a Cambisol (Hypereutric) (IUSS Working Group WRB 2006) with a clayey surface layer developed on alluvial deposits. The main mean (\pm standard deviation) soil characteristics for the four cropping systems were as follows: clay 411 \pm 32 g kg⁻¹, silt 534 \pm 32 g kg⁻¹, sand 53 \pm 1 g kg⁻¹, pH 6.9 \pm 0.1, CaCO₃ <1 g kg⁻¹, organic C 19.1 \pm 2.4 mg C kg⁻¹, organic N 1.58 \pm 0.29 mg N kg⁻¹, cation exchange capacity 21.5 \pm 2.3 cmol(+) kg⁻¹, bulk density 1.49 \pm 0.02 g cm⁻³.

2.2 Soil physical and chemical parameters

The inorganic N contents, bulk densities, temperature, and water contents of the 0–30-cm soil layer were monitored over the experimental period in each plot.

2.2.1 Soil inorganic N content

Three soil samples per cropping system were collected randomly each month. Inorganic nitrogen, i.e., NH_4^+ -N and $(NO_3^- + NO_2^-)$ -N, was extracted by shaking 10 g of moist soil (with measured water content) with 50 mL

Table 1 Crops and agricultural operations for the four systems studied from autumn 2011 to spring 2013

Cropping systems	S1	S2	S3	S5
Seedling	12 Oct. 2011 (Winter wheat)		24 Oct. 2011 (winter wheat)	
Tillage	9 Sept. 2011			
Rolling	13 Oct. 2011	17 Jan. 2012		
Fertilization	28 Feb. 2012 (50 N) ^a	20 Feb. 2012 (108 Mg, 1083 Ca)	28 Feb. 2012 (50 N)	
Seedling		9 Mar. 2012 (Spring barley)		29 Mar. 2012 (Alfalfa)
Fertilization		9 Mar. 2012 (47 N, 6 P, 7 S)		
Rolling		12 Mar. 2012		
Herbicide ^b	22 Mar. 2012	16 Mar. 2012	22 Mar. 2012	
Fertilization	3 Apr. 2012 (85 N)	17 Apr. 2012 (84 N)	2 Apr. 2012 (90 N)	27 Apr. 2012 (42 S, 52.5 Mg)
Herbicide		7 May 2012		
Fungicide ^c	10 May 2012		10 May 2012	
Fertilization	15 May 2012 (30 N)		15 May 2012 (25 N)	
Insecticide ^d	27 May 2012		27 May 2012	
Fungicide ^c		29 May 2012		
Fertilization				14 Jun. 2012 (3 S, 2 Mg)
Harvest	18 Jul. 2012	25 Jul. 2012	24 Jul. 2012	14 Jun. 2012 31 Jul. 2012
Tillage	20 Jul. 2012 30 Jul. 2012		26 Jul. 2012 20 Jul. 2012	
Seedling		1 Aug. 2012 (winter oat)	1 Aug. 2012 (catch crop) ^f	30 Aug. 2012 (alfalfa)
Antislug ^e		7 Aug. 2012		
Fertilization	5 Sept. 2012 (43 P, 47 K)			
Tillage	14 Sep. 2012 1 Oct. 2012		5 Sep. 2012 22 Oct. 2012	
Seedling	5 Oct. 2012 (winter barley)			
Cover crushing		29 Oct. 2012	23 Oct. 2012	23 Oct. 2013
Tillage			30 Oct. 2012 6 Mar. 2013 8 Apr. 2013	14 Nov. 2012 6 Apr. 2013 29 Mar. 2013 23 Apr. 2013
Herbicide	25 Oct. 2012			
Antislug ^e	30 Oct. 2012			
Fertilizer	21 Feb. 2013 (50 N)	20 Feb. 2013 (108 Mg, 1083 Ca)		
Fertilizer	29 Mar. 2013 (90 N)			
Herbicide ^b	25 Apr. 2013	14 May 2013		
Fongicide ^c	6 May 2013			
Seedling		14 May 2013 (soybean)	13 May 2013 (soybean)	25 Apr. 2013 (grain maize)

^a Kilogram of N (P, K, S, Mg or Ca) per hectare

^b Glyphosate isopropyl amine salt, tribenuron-methyl, mefenpyr-diethyl or metsulfuron-methyl, isoproturon, pendimethaline, prohexadione, mepiquat-chloride, ester 1-methylheptyl, or florasulam

^c Cyproconazole, bixafen, azoxystrobin, prothioconazole, or fluoxastrobin

^d Deltamethrin

^e Metaldehyde

^f Spring oat, Linse, Niger, Phacelia, and Faba

of a 1-M KCL solution for 1 h. The slurries were then filtered using Whatman No. 42 filter paper (Whatman Group®), and the extract was collected and frozen (−20 °C). Ammonium and nitrate contents were deter-

mined by automated colorimetry. The soil gravimetric water content of each subsample was estimated by mass difference before and after an oven-drying period of 24 h at 105 °C.

2.2.2 Soil bulk density

The bulk density of the soil was measured each spring during the 2 years, 2011–2012 and 2012–2013, in each plot, using steel cylinders of known volumes (10 cm diameter, 5 cm height). Samples were taken between 0 and –30 cm depth, dried at 105 °C for 24 h, and then weighed. Bulk density (ρ) was calculated as the ratio between the dried soil mass and the volume of the cylinder. Average system bulk density was defined as the mean of three samples taken randomly in the system for each date. Moreover, the estimation of bulk density allowed the calculation of the soil total porosity (P) as per Formula (1):

$$P = 1 - \rho / \rho_s \quad (1)$$

where ρ is the bulk density of the soil and ρ_s is the density of the soil solid particles (2.6).

2.2.3 Soil temperature, volumetric moisture, and water-filled pore space

Soil temperature and volumetric moisture were monitored using four thermistor probes (Campbell Scientific® PT100) and four time-domain reflectometer (TDR) site-calibrated probes (Campbell Scientific® CS616), respectively, in each plot. Both TDR and temperature probes were set in pairs at two depths (–5 and –20 cm). Temperature and moisture were measured automatically for 5 min every 2 h, every day. The data recorded data were then stored in a CR-1000 (datalogger Campbell Scientific®). A daily average was then calculated as the mean response from the two probes in each layer and plot. The daily proportion of water-filled pore space (WFPS) was calculated afterwards using the following Formula (2):

$$\text{WFPS} = \theta / P \quad (2)$$

where θ is the daily average volumetric water content measured by the TDR probes and P is the estimated total porosity of the soils.

2.3 N₂O emission measurements

Nitrous oxide (N₂O) emissions were measured continuously using the automated chamber method described in Vermue et al. (2013). A specific set of 24 static chambers (length 70 cm, width 70 cm, height 30 cm) was set up (Fig. 1). The chambers were placed on each plot (six chambers for each system, i.e., S1, S2, S3, and S5) within a 25-m radius around the analyzing device located on the grass strip to take into account the spatial variability inside each plot as soon as possible. The chambers were present during the entire study period and removed for very short periods when tillage operations were performed. In the field, the chambers were

pressed 10 cm into the soil, giving a 98-L headspace volume. Vegetation was present inside the chambers for the early growth stages of the different crops. Nevertheless, when the crop height was higher than the top of the chamber, the plants were cut to allow measurements. Nitrous oxide (N₂O) concentrations in the headspaces were measured by a Megatec® IR analyzer 46i (Thermo Scientific), connected to each chamber using an automated screening system. The evolution of N₂O concentrations in the headspace of the chamber was recorded over a 20-min period, with a time step of 20 s, four times a day for each chamber. The chambers were automatically re-opened at the end of each kinetic, to ensure the normal fertilization and humidification of the soils by rain. Recorded data were then stored on a CR1000 datalogger (Campbell Scientific®) and collected once a week. N₂O fluxes were calculated as the slope of the linear regression between the N₂O concentrations with the 20-min measurement period expressed in seconds and converted into grams of N-N₂O ha⁻¹ day⁻¹. The sensitivity threshold, determined from the error on the slope measurements as per the equation defined by Laville et al. (2011), was 0.7 g N-N₂O ha⁻¹ day⁻¹. Daily N₂O fluxes were estimated as the means of the 4 × 6 replicates calculated each day in each system. Absences of measurements mostly corresponded to periods during which the chambers were removed for technical operations on the fields (e.g., sowing, tillage, harvest). To ensure comparison if measurements were missing in one system, the data in all the remaining systems were excluded. Cumulative N₂O emissions therefore corresponded to the sum of N₂O emissions observed over a total of 255 days without extrapolation and during which measurements were available for all the systems simultaneously.

2.4 N₂O emission simulations

N₂O emissions were simulated with the NOE model (Hénault et al. 2005). NOE is an algorithm which allows simulating N₂O fluxes at the cropland scale by cumulating both nitrification and denitrification emissions of N₂O (Formula 3) and taking N₂O reduction into account. Each subpart of the model relies on the combination between specific soil biological (D_p , z , and r_{max}) and environmental parameters (soil temperature, WFPS, nitrate and ammonium contents) which were measured in situ (Formulas 4 and 5).

$$N_2O = [N_2O]_{denit} + [N_2O]_{nit} \quad (3)$$

$$[N_2O]_{denit} = r_{max} \cdot D_p \cdot F_N \cdot F_w \cdot F_T \quad (4)$$

$$[N_2O]_{nit} = r_{max} \cdot z \cdot N_w \cdot N_{NH4} \cdot N_T \quad (5)$$

where $[N_2O]_{denit}$ is the N₂O produced by denitrification and $[N_2O]_{nit}$ is the N₂O produced by nitrification, both in



Fig. 1 Partial view of the experimental design equipped with automated static chambers, each of which has a 98-L headspace and equipped with an air temperature sensor and a fan to homogenize the atmosphere during the measurement. The chambers are connected to a waterproof cabinet containing the N₂O gas analyzer, the datalogger to record soil moisture and temperature, headspace temperature, and data from the analyzer. It also contains the automated screening system that controls the opening and closing of the chambers and starts/stops the fan inside each chamber

kilograms of N per hectare per day. r_{max} , revealing the capacity of the soil to reduce N₂O, is the maximum ratio of accumulated N₂O to denitrified nitrate under anaerobic conditions; D_p is the potential denitrification rate (also in kg N ha⁻¹ day⁻¹); F_N , F_w , and F_T are response functions for soil nitrate content, water-filled pore space, and temperature, respectively (Hénault and Germon 2000); z is the proportion of nitrified nitrogen emitted as N₂O; and N_w , N_{NH_4} , and N_T are response functions for soil water content, ammonium content, and temperature, respectively.

NOE has been applied in different areas and climatic contexts (Gabrielle et al. 2006; Hergoualc'h et al. 2009). NOE was used here under the conditions defined in the original publication (Hénault et al. 2005). For each system, the soil potential denitrification rate was measured using eight undisturbed soil cores according to Hénault and Germon (2000), in May 2011. The soil's capacity to reduce N₂O was estimated on sieved composites of soil according to Hénault et al. (2001), in January 2013. The soil nitrification parameters ($z = 0.0006$) used in the simulation were measured previously on a site close to the current experimental site (Hénault et al. 2005). Cumulative simulated N₂O emissions corresponded to the sum of N₂O emissions simulated over a total of 163 days without extrapolation. This period corresponded to the number of days during which all the data required to perform the simulation were available for all the systems (e.g., TDR measurements).

2.5 Statistical analysis

All analyses were conducted using R (R Core Team 2016). Most of the data acquired during this study did not follow a

normal law according to the Shapiro-Wilk test; therefore, only nonparametrical tests were used. All tests were performed with a statistical significance set at 0.05.

The Kruskal-Wallis test was used to highlight the treatment effect with the null hypothesis that tested that variables were not significantly different between the treatments. Regarding the measured data, Kruskal-Wallis tests were used to evaluate the effects of cropping system management (S1, S2, S3, S5) on WFPS, temperature, mineral nitrogen contents, and N₂O cumulative emissions. These tests were associated with Dunn paired groups to highlight any significant differences and similarities between the cropping systems. Similarly, the effect of cropping systems on simulated cumulative N₂O emissions; soil water, nitrogen, and temperature functions; and biological parameters were assessed using Kruskal-Wallis tests with Dunn paired groups.

Linear correlations between both measured and simulated N₂O emissions and soil WFPS and soil temperatures were assessed using Spearman's rank correlation coefficient. Nonlinear correlations were assessed using generalized additive mixed models. In order to avoid interdependences, smooth functions were employed to describe the relationships between observed N₂O emissions and predictors.

Association and coincidence criteria between simulated and observed data were used to determine the performance of the models (Smith et al. 1996). A high association meant that the model accurately captured the dynamic data while a high coincidence meant that the model accurately reproduced their range. Spearman's rank coefficient (ρ) was used to assess the association between simulated and measured data while the coincidence was assessed with the root mean square error (RMSE) (6), relative root mean square error (rRMSE) (7), and bias error (BE) (8), calculated from the following formulas:

$$\text{RMSE} = \sqrt{\frac{\sum_{i=1}^n (S_i - O_i)^2}{n}} \quad (6)$$

$$\text{rRMSE} = \sqrt{\frac{\sum_{i=1}^n (S_i - O_i)^2}{\sum_{i=1}^n (O_i)^2}} \quad (7)$$

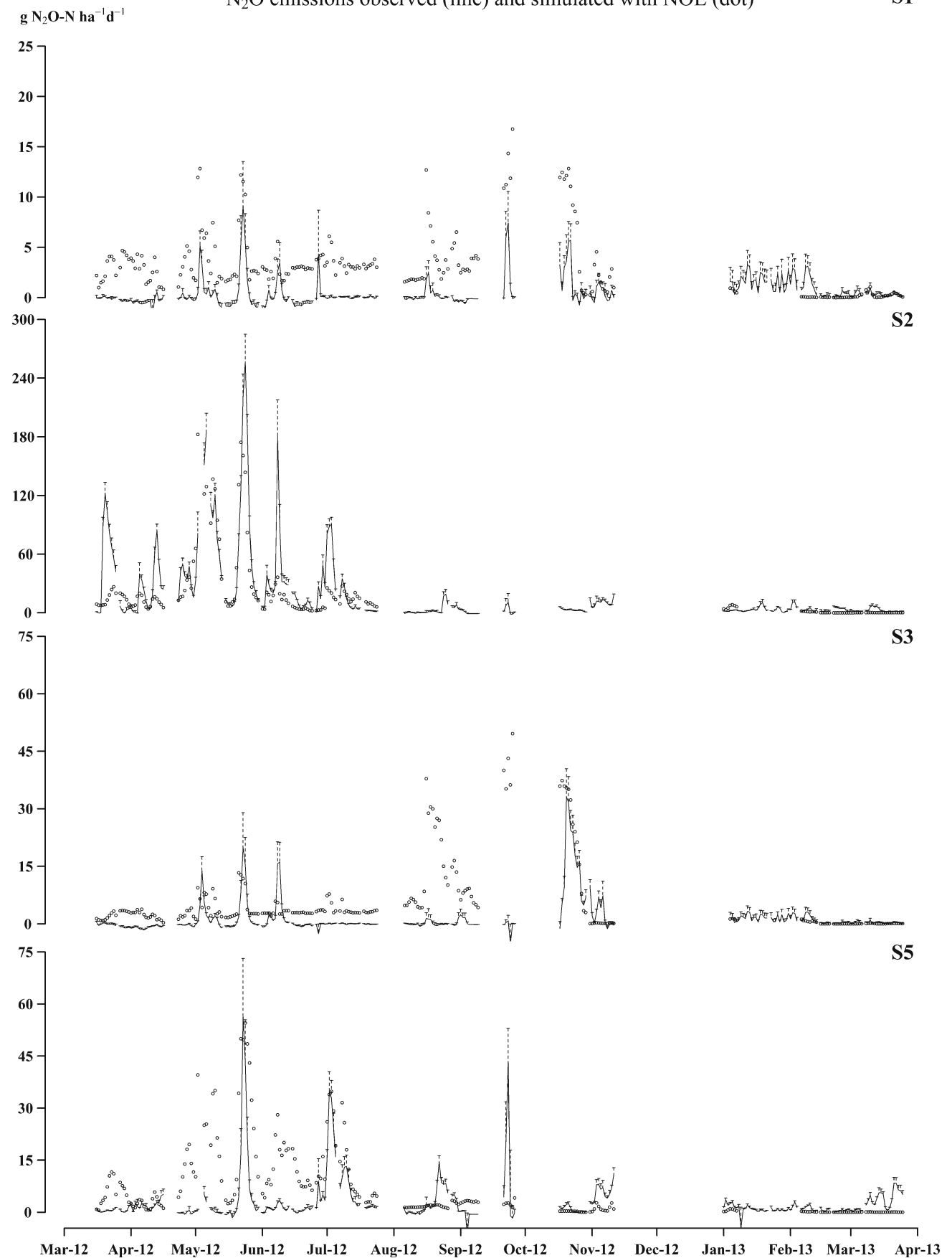
$$\text{BE} = \frac{\sum_{i=1}^n (O_i - S_i)}{n} \quad (8)$$

Additionally, the Nash-Sutcliffe model efficiency coefficient (NSE) was determined to confront cumulative observed N₂O emissions and N₂O emissions simulated with the NOE model (9).

$$\text{NSE} = 1 - \left[\frac{\sum_{i=1}^n (O_i - S_i)^2}{\sum_{i=1}^n (O_i - \bar{O}_i)^2} \right] \quad (9)$$

N₂O emissions observed (line) and simulated with NOE (dot)

S1



◀ **Fig. 2** Comparison between observed and simulated N₂O emissions (expressed in g N-N₂O ha⁻¹ year⁻¹) dynamics over 1 year in cropping systems S1 (control), S2 (no tillage), S3 (tillage, half herbicide use), and S5 (zero herbicide). Note that emissions were considerably influenced by cropping systems, with the highest emissions recorded for cropping system S2 (no tillage). Emissions also varied with time: the highest emissions were recorded during spring and after fertilizer applications whereas they were lower for the rest of the year

where O stands for observed fluxes, S for simulated fluxes, and n for the number of observations.

To ensure comparison, comparisons between measured and simulated N₂O fluxes were performed over the days when both were available, i.e., 163 days.

3 Results and discussion

3.1 Intensity and determinism of observed N₂O emissions

High-resolution measurements were performed on all the systems over 1 year, between March 2012 and March 2013, to estimate N₂O emissions as recommended in the literature (Yao et al. 2009). The specific design used here allowed capturing the high temporal and the spatial variability of the N₂O emissions inside each plot for all the systems. The limits required by the experimental design (i.e., absence of plants for some periods of measurements, possible modification of soil functioning induced by the presence of chambers) may have

modified N₂O emissions from the soil-plant system. However, the experimental conditions could be considered as comparable between cropping systems, thereby allowing us to compare their emissions; thus, seasonal patterns were identified for the different systems. The emission levels of systems S1, S3, and S5 were globally comparable to those observed in 2011 in the same area (Vermue et al. 2013), as well as to punctual measurements performed in the agricultural region of eastern France (Henault et al. 1998) for comparable pedoclimatic conditions.

Daily N₂O emissions from the reference system S1 remained relatively low, ranging from -2 to 9 g N-N₂O ha⁻¹ day⁻¹ (Fig. 2), with an average flux of 0.5 g N-N₂O ha⁻¹ day⁻¹. Cumulated emissions were 326 g N-N₂O ha⁻¹ over the experimental period for system S1. For system S3, N₂O emissions were significantly higher ($p < 0.05$). Daily N₂O fluxes ranged from -5 to 33 g N-N₂O ha⁻¹ day⁻¹ (Fig. 2), with an average daily flux of 1.5 g N-N₂O ha⁻¹ day⁻¹. The cumulated emissions were 177 g N-N₂O ha⁻¹ over the year. Despite the absence of N fertilization, N₂O emissions from system S5 were significantly higher than those observed for cropping systems S1 and S3 ($p < 0.05$), but nevertheless lower than those observed for the no-till system S2 ($p < 0.05$). For system S5, daily N₂O fluxes ranged between -6 and 56 g N-N₂O ha⁻¹ day⁻¹ with an average daily N₂O flux of 3 g N-N₂O ha⁻¹ day⁻¹ (Fig. 2) and cumulated N₂O emissions reached 777 g N-N₂O ha⁻¹. The emissions from system S2 were significantly higher ($p < 0.05$) than all the previous measurements performed in this area, ranging

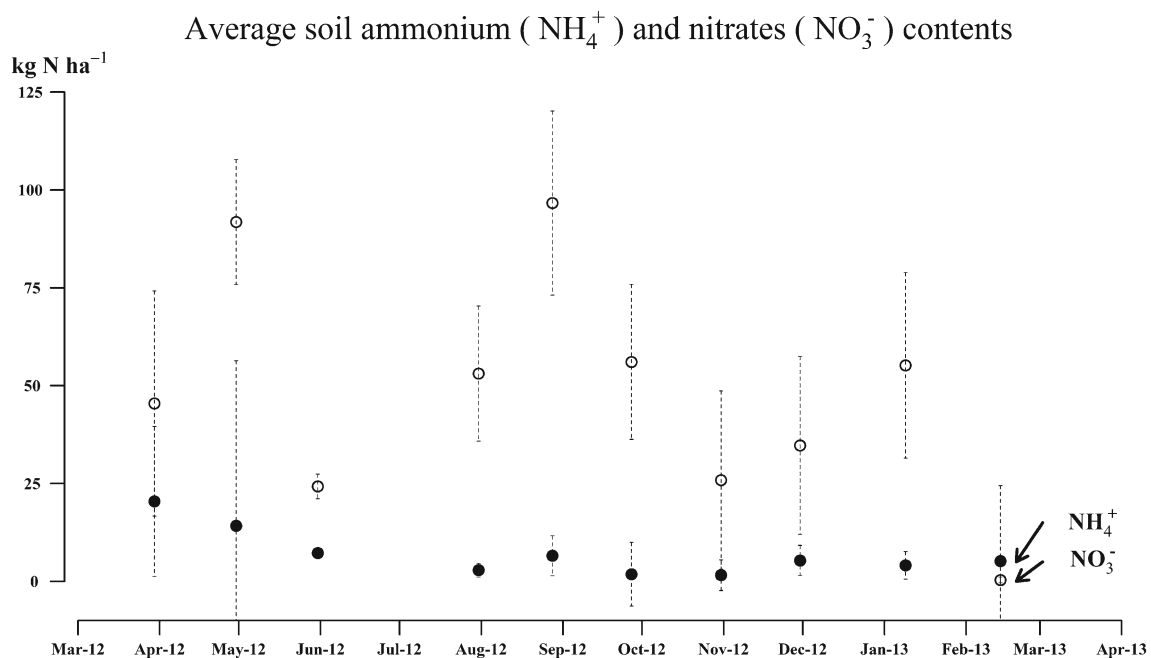


Fig. 3 The evolution of average ammonium and nitrate contents (expressed in kg N ha⁻¹) in the soils of the experimental site during the study period. Note that soil ammonium contents were very low compared

to soil nitrate contents. Elsewhere, ammonium and nitrate contents were not significantly different between the four systems on each date of measurement

Table 2 Spearman's coefficients (ρ) for the rank correlation between N_2O emissions simulated with NOE and functions recording temperature (F_T), nitrate (F_N), and water (F_W) effects, as well as average values observed for soil inorganic N, WFPS, and soil temperature during the study period

System	F_N	N- NO_3^- (kg ha $^{-1}$)	N- NH_4^+ (kg ha $^{-1}$)	F_W	WFPS (%)	F_T	Temperature ($^{\circ}\text{C}$)
S1	0.48	46 \pm 23	9 \pm 11	0.30	64 \pm 4	0.46	12.4
S2	0.66	64 \pm 65	41 \pm 1	0.22	68 \pm 3	0.32	12.6
S3	0.47	31 \pm 25	6 \pm 10	Not significant	66 \pm 7	0.42	12.4
S5	0.14	53 \pm 32	4 \pm 3	0.73	63 \pm 5	0.44	13.0

WFPS water-filled pore space

from -2 to 257 g N- N_2O ha $^{-1}$ day $^{-1}$ (Fig. 2) with an average flux of 20 g N- N_2O ha $^{-1}$ day $^{-1}$. The cumulated emission of system S2 was 5226 g N- N_2O ha $^{-1}$. Maximum emissions were observed during spring, while the soil emitted an average of 80 g N- N_2O ha $^{-1}$ day $^{-1}$ during May 2011, accounting for 40 % of the total N_2O emitted over the measurement period from this system.

The temporal pattern of N_2O emissions differed between the systems. N_2O peaks were observed at about equal levels in spring, summer, and autumn for each of the S1, S3, and S5 systems, while much higher peaks were observed during spring, especially for system S2 where 71 % of N_2O emissions were recorded before June 20, 2012. Finally, N_2O emissions from all the systems remained low most of the time whereas N_2O was mainly emitted during short peak periods over the year of measurement for all the systems. The lowest N_2O emissions were observed in winter in all the systems. This can be attributed to lower temperatures, as observed by Laville et al. (2011). Soil temperatures were the factors having the greatest impact on the N_2O emissions observed. Both non-linear and linear significant correlations were observed between soil temperature recorded at -5 cm and N_2O emissions in all the systems.

Fertilization periods associated with rainfalls often lead to high N_2O emissions (Dobbie et al. 1999). In our study, the application of fertilizers in April 2012 in systems S1, S2, and S3 did not immediately cause N_2O emission peaks, a phenomenon also reported by Zebarth et al. (2008). However, no significant correlation was established between soil N dynamics and the N_2O emissions observed later. Soil N dynamics were not significantly different between the systems, even including the nonfertilized system S5. High inorganic N contents were measured in all the systems after the harvesting period in August 2012 (Fig. 3), probably resulting from the mineralization of crop residues and soil organic matter (Velthof et al. 2002). This did not clearly induce N_2O emissions, whatever the system. Moreover, the emissions of systems S1, S3, and S5 were observed in autumn 2012 despite lower soil inorganic N contents during this period. Altogether, these observations suggest that soil mineral nitrogen contents were not the main limiting factor for the production of N_2O during this study.

The N_2O peaks observed often coincided with high WFPS values while the N_2O emissions observed were significantly nonlinearly correlated to WFPS ($p < 0.05$) in all the systems and linearly correlated ($p < 0.05$) to systems S1 ($\rho = 0.63$), S3 ($\rho = 0.60$), and S5 ($\rho = 0.27$). Among the soil processes resulting in N_2O emissions, nitrification and denitrification are often identified as main contributors to N_2O emissions under dry and wet soils, respectively, with denitrification producing more N_2O (Mathieu et al. 2006). This is consistent with the significantly higher WFPS and N_2O emissions measured for system S2 and lower soil WFPS and N_2O emissions measured in S1 and S3. This suggests that system S2 contributed most to the denitrification process while systems S1 and S3 contributed to the nitrification processes.

3.2 Simulation of N_2O emissions

The denitrification potentials used to parametrize NOE appeared to differ between systems, as they were significantly higher for S2 with 4036 g N- N_2O ha $^{-1}$ day $^{-1}$ while they were equal to 1820 , 1220 , and 1964 g N- N_2O ha $^{-1}$ day $^{-1}$ for systems S1, S3, and S5, respectively. Elsewhere, a soil capacity to reduce N_2O to N_2 with r_{max} values of 0.69 , 0.78 , 0.70 , and 1 was obtained for cropping systems S1, S2, S3, and S5, respectively. These latter values were used to parametrize NOE.

Simulated emissions ranged between 0 and 17 , 182 , 50 , and 55 g N- N_2O ha $^{-1}$ day $^{-1}$, for systems S1, S2, S3, and S5, respectively (Fig. 2). In terms of N_2O emission intensity, the NOE model significantly discriminated and ranked the cropping systems, in the same order as observed. Significantly higher emissions from system S2 were simulated, followed by, in order, systems S5, S3, and S1. According to the observations, nitrification appeared to be the main contributor to N_2O emissions for system S1 with 58 % of the emissions. In contrast, a contribution of 68 and 72 % of N_2O emissions by denitrification was simulated for systems S3 and S5. The highest N_2O emissions by denitrification, with a proportion of 89 %, were simulated for system S2, which is consistent with higher WFPS (Table 2) and higher measured N_2O emissions (Fig. 2). Seasonal patterns were accurately reproduced by the model, especially the high proportion of N_2O emissions during spring and the low N_2O emissions

during winter. As found with the generalized additive mixed modeling of observed N_2O emissions (detailed results not shown), NOE identified the soil temperature function F_t as the main factor regulating simulated N_2O emissions (Table 2). Despite the fact that no significant correlation was observed between measured N_2O emissions and soil inorganic N dynamics, as is usually observed in fertilized soils (Zebarth et al. 2008), simulated N_2O emissions from systems S1, S2, and S3 appeared to be highly correlated with the response factors for soil nitrate content obtained using the nonlinear F_n function (Table 2). N_2O emissions from system S5 simulated by NOE were determined more by the response factor for water contents (Table 2). No similar linear correlation was observed for systems S1, S2, and S3.

The statistical analysis of simulations versus observations revealed that the NOE model accurately reproduced N_2O emission dynamics and levels for all the cropping systems. The global Nash-Sutcliffe coefficient of efficiency, considering the four cropping systems, was $\text{NSE} = 0.80$, suggesting better predictions from the model than the estimations from the mean of the N_2O emissions. Both association and coincidence criteria suggested that the simulation of N_2O emissions was very accurate. Indeed, simulated and observed N_2O fluxes were significantly correlated ($p < 0.0001$) for all four systems with rho values of 0.34, 0.76, 0.43, and 0.61 for cropping systems S1, S2, S3, and S5, respectively. The RMSE between the observed N_2O emissions and those simulated with NOE averaged at $13.6 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$, which was consistent with previously reported errors. Hergoualc'h et al. (2009) and Gabrielle et al. (2006) reported a maximum RMSE of 9.4 and $38.4 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$, respectively, with NOE. While NOE could tend to underestimate denitrifying activity related to high WFPS (Gabrielle et al. 2006), we observed an underestimation of the intensity of N_2O peaks, especially for system S2, where a maximum BE and RMSE of 9 and $30.4 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$ were observed. However, system S2 presented the lowest rRMSE, with $0.00 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$, while the average rRMSE was $0.10 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$. Similarly, the highest Nash-Sutcliffe efficiency coefficient ($\text{NSE} = 0.57$) was obtained for system S2. In contrast, our results suggest that NOE tended to overestimate the background nitrification rates for all the cropping systems. From March to December 2012, the model almost continuously simulated low N_2O emissions ($<5 \text{ g N}_2\text{O-N ha}^{-1}$) from nitrifying activity. However, while soil temperatures and water and ammonium contents were not limiting for nitrification, significantly lower N_2O emissions were observed in all the systems. The overestimation was particularly visible for systems S1, S3, and S5 which contributed the most nitrification to N_2O emissions and where BEs of -3 , -5 , and $-4 \text{ g N-N}_2\text{O ha}^{-1} \text{ day}^{-1}$ were observed, respectively. The Nash-Sutcliffe efficiency coefficients were therefore low ($\text{NSE} < 0$) for these systems.

As NOE accurately captured the temporal variability of the emissions, it probably allows accurate characterization of the interactions between the main factors involved in regulating N_2O emissions. The simulations obtained also discriminated the systems studied well, especially the system with the highest emission. Indeed, NOE identified the no-tillage system S2 as the most N_2O -emitting system, with a total emission of $3209 \text{ g N-N}_2\text{O ha}^{-1}$ emitted in comparison with the three conventionally tilled systems (S1, S3, and S5). This discrimination was achieved by the system-specific measurement of both the biological and environmental systems, revealing a clearly higher potential denitrification rate of the soil of the S2 system combined with a lightly higher bulk density and thus WFPS.

No-till systems are commonly reported to enhance N_2O emissions in comparison to conventional systems, especially during the initial years of introduction (Six et al. 2004; Oorts et al. 2007). NOE simulated an average additional emission of $2 \text{ kg N-N}_2\text{O ha}^{-1}$ while an average of $1.95 \text{ kg N-N}_2\text{O ha}^{-1}$ is usually observed over 1 year in temperate soils (Six et al. 2002). Our study therefore confirms the potential for using NOE to estimate the impact of different agricultural practices on N_2O emissions (Gabrielle et al. 2006). It also reveals that the use of NOE also helps in understanding contrasting levels of N_2O emissions in different agronomic systems including innovative cropping systems based on integrated weed management.

4 Conclusions

Our study contributes to the environmental evaluation of cropping systems that incorporate integrated weed management principles. Thus, our main objective was to compare N_2O emissions for four cropping systems based on different weed management strategies. We observed a very strong effect of integrated weed management on N_2O fluxes over a 1-year period of investigation in different integrated weed management cropping systems using high-resolution measurements. N_2O emissions from integrated weed management with no tillage since 2008 ($>5000 \text{ g N-N}_2\text{O ha}^{-1} \text{ year}^{-1}$) were clearly higher than the emissions measured on the other cropping systems with integrated weed management. The NOE model successfully reconstituted patterns of N_2O emissions, indicating that N_2O emissions were driven by the interactions between specific potential denitrification rates; bulk densities; and temperature, water, and nitrogen dynamics in soils. Since one of the challenges now facing agriculture is that of mitigating its global warming potential, we have taken a step in this direction by highlighting the reliability of the NOE model as a tool for estimating and understanding N_2O emissions in the framework of cropping system development and evaluation.

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