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Greenhouse gas emissions from the grassy outdoor run of organic broilers

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Abstract. Nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) fluxes over the grassy outdoor run of organically grown broilers were monitored using static chambers over two production batches in contrasted seasons. Measured N_2O and CH_4 fluxes were extremely variable in time and space for both batches, with fluxes ranging from a small uptake by soil to large emissions peaks, the latter of which always occurred in the chambers located closest to the broiler house. In general, fluxes decreased with increasing distance to the broiler house, demonstrating that the foraging of broilers and the amount of excreted nutrients (carbon, nitrogen) largely control the spatial variability of emissions. Spatial integration by kriging methods was carried out to provide representative fluxes on the outdoor run for each measurement day. Mechanistic relationships between plot-scale estimates and environmental conditions (soil temperature and water content) were calibrated in order to fill gaps between measurement days. Flux integration over the year 2010 showed that around $3 \pm 1\,\mathrm{kg}\,N_2O\text{-N}\,\mathrm{ha}^{-1}$ were emitted on the outdoor run, equivalent to 0.9 % of outdoor N excretion and substantially lower than the IPCC default emission factor of 2 %. By contrast, the outdoor run was found to be a net CH_4 sink of about $-0.56\,\mathrm{kg}\,CH_4\text{-C}\,\mathrm{ha}^{-1}$, though this sink compensated less than 1.5 % (in CO_2 equivalents) of N_2O emissions. The net greenhouse gas (GHG) budget of the outdoor run is explored, based on measured GHG fluxes and short-term (1.5 yr) variations in soil organic carbon.

1 Introduction

Carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) are among the most important greenhouse gases (GHG) responsible for climate change (IPCC, 2007b), and N_2O also contributes to stratospheric ozone depletion (Ravishankara et al., 2009). Climate change is one of the most challenging environmental issues of the 21st century (UNFCCC, 2009) and solutions must be found in order to mitigate GHG emissions on a global scale. Agriculture contributes 13.5 % of global anthropogenic GHG emissions (expressed in CO_2 equivalents, CO_2 eq.) (IPCC, 2007a), and livestock production is known to be a major emitter of GHG with 9 % of total CO_2 eq. emissions (FAO, 2006).

In poultry production, GHG are mainly emitted by manure in houses, in storage facilities or during and after field application of manure. In comparison with cattle production, which accounts for 23 % of total N_2O emissions and more than 15 % of total CH_4 emissions in France, poultry production is responsible for relatively little GHG emission, i.e. about 1.5 % of total N_2O emissions and 2.2 % of total CH_4 emissions (Gac et al., 2007). Nevertheless, few data concerning GHG emissions from poultry rearing systems are available and uncertainties on emission factors are high (Meda et al., 2011).

This lack of knowledge is even larger for alternative, less intensive rearing systems, which provide an outdoor access to the animals (free-range and/or organic farming). In these systems, a fraction of droppings is excreted onto an outdoor run (called "run" hereafter), leading to gaseous emissions on the run as well as in the house, due to microbial processes occurring in the soil and in the litter. Estimates of GHG and ammonia emissions from organic broiler houses were proposed by Meda (2011) but outdoor emissions also need to be quantified. To our knowledge, there is no study concerning outdoor GHG emissions from poultry in the literature. Nonetheless, the general mechanisms regulating emissions are known. N_2O is widely believed to be emitted as a by-product of the oxidation of ammonium (NH_4^+) to nitrite (NO_2^-) and to nitrate (NO_3^-) (nitrification), and of the reduction of NO_3^- ultimately to N_2 (denitrification), the latter becoming dominant with increasing soil moisture (Davidson, 1991; Smith et al., 2003), and thus N_2O can be released on the run under favourable conditions. Droppings excreted on the run indeed add biodegradable carbon (C), available nitrogen (ammonium, uric acid) and moisture to the soil, thereby enhancing nitrification and/or denitrification processes and thus N_2O emissions (Smith et al., 2003). CH_4 is produced by the microbial degradation in anaerobic conditions of organic matter found in droppings and soil (Le Mer and Roger, 2001; Smith et al., 2003) but under well aerated conditions CH_4 is also often oxidized (consumed) by agricultural soils (Smith et al., 2000; Le Mer and Roger, 2001; Castaldi et al., 2007). The interplay of all these different processes makes the prediction of net N_2O and CH_4 fluxes a difficult exercise.

In this paper, we present the results of GHG flux measurements (N_2O, CH_4 and CO_2) over a period of 1 yr over a sown grassland, which served as the run for organic broilers, as part of the French AlterAviBio project. The study was conducted on two broiler batches, reared in winter/spring 2009–2010 and in summer/autumn 2010, providing a range of climatic and environmental conditions known to control GHG emissions. The study also focuses on broiler behaviour as a factor controlling the spatial and temporal variability of GHG fluxes. The measurements are used to provide annual emission fluxes, which for N_2O are also expressed as emission factors (EF) as the percentage of emission relative to the N input to the run (Pain and Menzi, 2011), and to estimate the net GHG exchange (NGHGE) of the run.

2 Materials and methods

2.1 Study site characteristics

The study took place on the experimental facility of the French National Institute for Agricultural Research (INRA) at Le Magneraud (Long. 00°41′25″ W, Lat. 46°09′04″ N) at an altitude of 60 m above mean sea level. The climate in the region is temperate oceanic with an interannual mean air temperature of around 12 °C and mean annual precipitation of around 780 mm. The soil is a clay loam and moderately alkaline (pH about 8.2). It is morphologically simple, consisting of an A horizon (25–35 cm thick) with about 15 % of limestone coarse elements (from a few millimetres to a few centimetres) above a C horizon (rendzicleptosol (calcaric)). The upper layer (0–15 cm) is rich in organic carbon (2.45 %), with a C/N ratio of about 9.7. Before 2008, the field was used for crop production (maize, winter wheat, sunflower and winter pea) before being converted into a grassland during spring 2008. The sown grassland mix consisted of 41 % tall fescue, 19 % lolium, 11 % Kentucky bluegrass, 11 % birdsfoot trefoil, 11 % alsike clover, 7 % white clover. Before the start of the first broiler batch (March 2009), the canopy was cut in order to offer a short canopy to the broilers. The field has a gentle slope down to the broiler house (Fig. 1).

2.2 Organic broiler production system

Five broiler batches were reared consecutively on the study site between March 2009 and November 2010. The GHG flux measurements presented in this paper were carried out during two batches only (batches 3 and 5). Batch 3 was studied from December 2009 to May 2010 (winter and early spring, denoted "WS" hereafter) while batch 5 was studied from August 2011 to December 2010 (late summer and autumn, or "SA").

The main characteristics of the two studied batches (number of animals, important dates, weight at slaughter) are listed in Table 1. During each batch, between 750 and 800 slow-growing strain broilers were reared in a broiler house of 75 m^2 ($l = 12.5$, $L = 6$ m) with straw bedding. After 35 days (batch WS) or 28 days (batch SA), two pop-holes of 2 m were continuously open and allowed the animals an unlimited access to the outdoor run of 2350 m^2 surface area ($l = 50$, $L = 47$ m).

2.3 Flux measurement technique

Greenhouse gas fluxes were measured using static chambers (Smith et al., 1995; Conen and Smith, 1998). Square steel chambers ($L = 50$, $l = 50$, $H = 30$ cm) were operated manually and fluxes were measured at (a maximum of) 16 locations on the run, and also at 3 locations just outside the fence to measure background fluxes (Fig. 1), using steel frames which were permanently inserted 10 cm into the soil. Among these frames, 9 were located within the first 15 m of the chicken house, as previous behavioural observations had shown that broilers are preferentially present in this zone (Zeltner and Hirt, 2003, 2008; Hegelund et al., 2005), and we therefore expected higher emissions due to a higher amount of nutrients (carbon, nitrogen) excreted.

Measurements were carried out routinely at regular time intervals after broilers gained first access to the run. In addition, measurements were also carried out after the last access
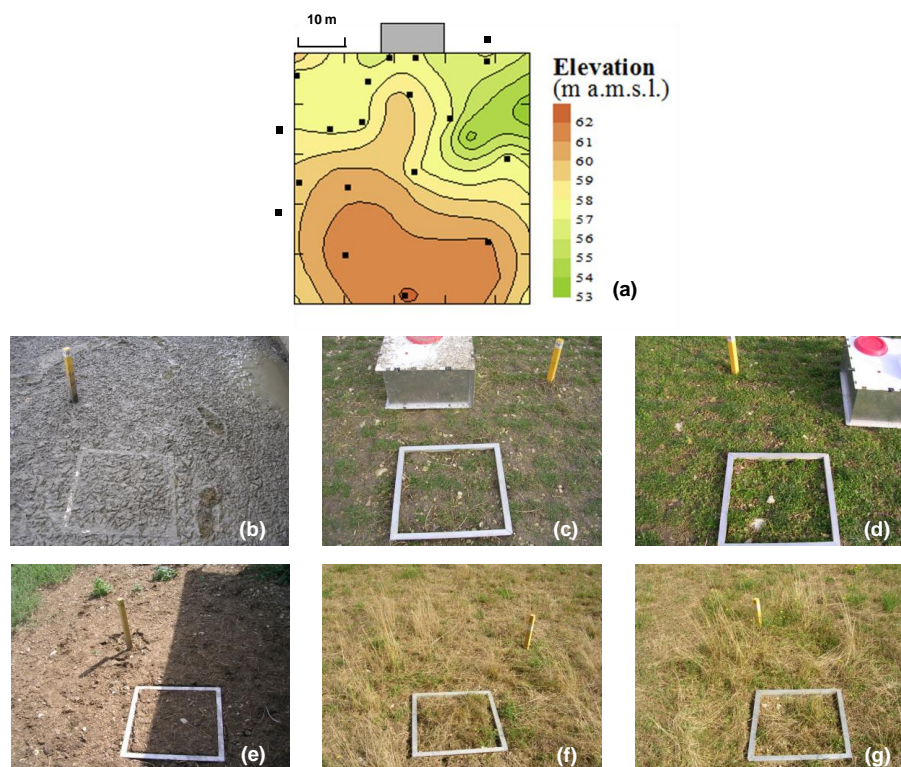
Fig. 1. (a) Site map showing the approximately square-shaped fenced-in outdoor run (50 × 47 m), the broiler house (grey rectangle at the top), the topography of the outdoor run, and the distribution of the static chambers (black square). **(b–g)** Photographic records of the state of soil and vegetation, showing differences in soil moisture, vegetation density and soil compaction for chamber frames located at 5 m **(b, e)**, 24 m **(c, f)** and 41 m **(d, g)** from the broiler house. Pictures were taken on 2 March 2010 **(b–d)** and on 27 July 2010 **(e–g)**, a.m.s.l. = above mean sea level.

Table 1. Main characteristics of the two broiler batches during which GHG fluxes were measured.

Batch	Number of animals		Weight at slaughter	Dates		
	at start	at slaughter	(kg/broiler)	Chicks arrival	First outdoor access	Slaughter
WS	758	678	2.21	8 December 2009	13 January 2010	8 March 2010
SA	788	711	2.23	3 August 2010	1 September 2010	10 November 2010

(slaughter) to the run. On each day of flux measurements, chamber sampling was done in the early to mid-afternoon i.e. between 14:00 to 16:00. At the start of each flux measurement, the chambers were placed on top of the steel frames, their lid being equipped with a septum for gas sampling using a 50 ml syringe. Four air samples of 15 ml were taken at $t = 0$, $t = 10$, $t = 20$ and $t = 30$ min and injected into 5-ml evacuated vials closed with leak free septa before being analysed using gas chromatography (Agilent 6890N, Agilent Technologies, Santa Clara, USA). The trace gas (CO_2, N_2O and CH_4) exchange flux between soil/vegetation/atmosphere is proportional to the temporal change in head-space gas concentration (C):

$$F(t) = \frac{V}{A} \frac{\partial C}{\partial t} = H \frac{\partial C}{\partial t} \qquad (1)$$

with V the head-space volume, A the surface area and H the chamber height. In practice, the slope of the linear regression of gas concentration vs. time was used. It should be recognised that in theory the temporal evolution of concentration inside the closed chamber is curvilinear, owing to a gradual reduction over time of the soil/headspace concentration gradient and hence of the flux. However we did not fit non-linear functions because only four concentration points (one sample every 10 min during a half-hour) were available to constrain non-linear models, which, accounting for the uncertainty in individual concentration measurements, may lead to unstable

solutions (e.g. Levy et al., 2011). The linear regression slope was deemed a more robust option, and the GHG flux data presented hereafter should be viewed as conservative estimates.

The lower detectable fluxes (LDF) for CO_2, N_2O and CH_4 were determined on the basis of the precision of GHG concentrations measured by GC. Values of LDF were calculated following Flechard et al. (2005) as $LDF = H \cdot \Delta C_{noise}/\Delta t_{chb}$, where ΔC_{noise} is the noise or lower detectable concentration difference, and Δt_{chb} is chamber closure duration (30 min). The noise levels for the three gases were defined as twice the standard deviation of successive concentration measurements of gas standards during calibrations, and determined in our GC system as $\Delta C_{noise}(CO_2) = 11$ ppm at 600 ppm (precision $\sim 2\%$), $\Delta C_{noise}(N_2O) = 21$ ppb at 475 ppb (precision $\sim 4\%$), and $\Delta C_{noise}(CH_4) = 0.06$ ppm at 2.05 ppm (precision $\sim 3\%$). The resulting LDF values (using $H = 0.3$ m) were $0.1\,\mu\text{mol}\,CO_2\,\text{m}^{-2}\,\text{s}^{-1}$, $6.7\,\text{ng}\,N_2O\,\text{m}^{-2}\,\text{s}^{-1}$ and $7.1\,\text{ng}\,CH_4\,\text{m}^{-2}\,\text{s}^{-1}$.

2.4 Soil and meteorological measurements

The site was equipped with a meteorological station and datalogger (CR10X, Campbell Scientific Ltd., Loughborough, UK) providing half-hourly means of air temperature, global radiation, wind speed and rainfall. Soil temperature was measured at -5 cm, while volumetric soil water content (SWC) at -5 cm was measured using time domain reflectometry (TDR) probes (CS616, Campbell Scientific Ltd., Loughborough, UK). These measurements were made on site but outside the run, at a distance of around 150 m from the broiler house. To fill occasional SWC data gaps mechanistically, a simple one-dimensional soil water budget model was used. The soil water balance of the uppermost 10 cm of topsoil was computed every half-hour using the following equation:

$$\frac{d\Phi}{dt} = \frac{1}{Z_s}(P - I - D - E) \tag{2}$$

with Φ the mean soil water content ($\text{m}^3\,\text{m}^{-3}$) of the considered topsoil layer, $Z_s = 0.1$ the depth of considered topsoil layer (m), P the precipitation rate ($\text{m}^3\,\text{m}^{-2}\,30\,\text{min}^{-1}$), I the rate of rainfall interception by the canopy ($\text{m}^3\,\text{m}^{-2}\,30\,\text{min}^{-1}$), D the gravitational drainage rate from the top 10-cm layer down to the deeper layers ($\text{m}^3\,\text{m}^{-2}\,30\,\text{min}^{-1}$), and E the evapotranspiration flux ($\text{m}^3\,\text{m}^{-2}\,30\,\text{min}^{-1}$). D was parameterised as a function of Φ based on rainless nighttime observations of the rate of SWC change (when also both $E = 0$ and $P = 0$). I was modelled as a function of precipitation rate and leaf area index (LAI) according to Norman and Campbell (1983). E was modelled using the Penman-Monteith equation (Monteith, 1965), with stomatal resistance of the grass being adjusted on a monthly basis for the modelled SWC to fit the observations by TDR.

2.5 Spatial integration of GHG fluxes

For each measurement day (Table 2), the spatial integration of GHG fluxes was carried out using geostatistical methods with the SURFER software (Golden Software, Golden, USA). A grid map was generated with its origin at the top-left corner of the map. On each measurement day, the GHG flux values on the run were used to generate a map of fluxes with a 1×1 m resolution; the mean of the control chambers was used as background flux around the fringes of the run. As datasets were small (about 20 observations per measurement day), kriging and radial basis function interpolation methods were selected. Gridded fluxes were summed to provide spatial integrals for each GHG and each measurement day. The average of both interpolation methods was used in the rest of the study.

2.6 Temporal gap-filling and integration of GHG fluxes

To fill gaps between two measurement days, and to compute time-integrated fluxes over a batch or a year, three methods were used and compared. First, the temporal mean of daily spatial integrals was calculated and multiplied by the duration of the batch. Second, a simple linear dot-to-dot interpolation was carried out between daily spatial integrals obtained from Sect. 2.5. Note that the first and second methods would be equivalent if consecutive measurement days were all equidistant, which was not the case here. Third, to provide more mechanistically-based estimates of GHG fluxes, relationships between measured fluxes and assumed control variables, such as soil temperature and SWC, were investigated. These functions are detailed in Sect. 3.3. To derive estimates of batch-scale fluxes, temporal integrals were calculated by summing actual measurement-based spatial flux integrals for each measurement day and gap-filled fluxes inbetween. The batch-scale integrals included data from first outdoor access of the N-th batch to the first outdoor access of the following $(N + 1)$-th batch, since we considered that emissions before first outdoor access of batch $N + 1$ should be attributed to batch N.

2.7 Outdoor nitrogen excretion

In order to estimate the amount of N excreted on the run during batches WS and SA, mass balances of N, P and K were computed at the house + run scale. These mass balances are presented in another study (Meda, 2011) in which more methodological details are given. Total ingestion was calculated as the product of feed consumption and nutrient content of the diet, ignoring in a first approach any ingestion of nonfeed material from the outdoor run. Body retention for N, P, and K was calculated according to CORPEN (2006). Total excretion was calculated by difference between total ingestion and body retention. Straw and litter were weighed, sampled and analysed for chemical composition in N, P and K,

Table 2. Summary of N_2O, CO_2 and CH_4 chamber flux statistics for each measurement date.

		WS batch								Intermediate		SA batch								
N_2O fluxes (ng m^{-2} s^{-1})		11.12.09	19.01.10	28.01.10	09.02.10	19.02.10	02.03.10	09.03.10	06.04.10	04.05.10	29.07.10	27.08.10	02.09.10	16.09.10	27.09.10	07.10.10	18.10.10	04.11.10	24.11.10	13.12.10
Outdoor run (within fence)	N	16	8	6	5	15	14	10	12	13	16	16	8	8	8	16	16	16	16	16
	Mean	−4	9	20	31	25	19	56	166	1	12	63	12	42	841	49	70	48	15	123
	Median	−5	5	14	39	13	13	11	15	0	3	16	7	14	94	30	10	45	12	6
	Min	−32	1	6	5	−6	6	−18	2	−5	−4	4	1	−3	16	0	−12	14	0	−11
	Max	21	28	45	60	80	63	323	1026	8	162	476	39	219	5910	159	597	96	53	1871
Control (outside fence)	N	3	1	1	2	2	2	0	2	3	3	3	2	2	2	3	2	3	2	2
	Mean	10	10	15	0	10	11		2	3	3	13	6	5	5	5	4	11	4	−1
	Median	15	10	15	0	10	11		2	2	2	15	6	5	5	9	4	1	4	−1
	Min	−10	10	15	−16	4	10		−2	1	1	5	3	−2	4	−4	0	−3	1	−7
	Max	26	10	15	15	16	11		6	5	4	20	9	12	6	9	8	34	8	5
R_{eco} CO_2 fluxes (µmol m^{-2} s^{-1})		11.12.09	19.01.10	28.01.10	09.02.10	19.02.10	02.03.10	09.03.10	06.04.10	04.05.10	29.07.10	27.08.10	02.09.10	16.09.10	27.09.10	07.10.10	18.10.10	04.11.10	24.11.10	13.12.10
Outdoor run (within fence)	N	16	7	7	5	14	16	6	16	14	16	16	8	8	8	16	15	16	16	16
	Mean	0.9	1.3	1.6	1.8	1.3	2.3	1.3	2.8	1.0	1.3	3.0	1.5	1.6	6.3	3.2	1.1	4.0	1.5	0.9
	Median	0.9	1.1	1.6	1.6	1.0	2.1	0.7	2.5	1.0	1.1	2.7	1.5	1.5	3.7	2.8	0.8	3.6	1.4	0.8
	Min	0.2	0.7	0.6	1.0	0.3	0.9	0.2	0.9	0.2	0.3	1.7	0.5	0.7	2.8	1.2	0.4	1.5	1.0	0.2
	Max	1.9	2.5	2.5	2.7	3.9	5.0	3.1	5.3	2.1	2.5	5.7	2.6	3.1	24.2	5.8	3.4	10.6	2.2	3.7
Control (outside fence)	N	3	1	1	1	3	3	1	3	2	3	3	2	2	2	3	2	3	2	2
	Mean	0.9	0.7	0.9	0.4	0.7	1.5	0.9	4.0	1.3	1.3	2.8	1.2	1.2	2.0	2.4	1.2	2.2	1.0	0.6
	Median	1.1	0.7	0.9	0.4	0.7	1.2	0.9	3.0	1.3	1.4	2.3	1.2	1.2	2.0	2.6	1.2	2.0	1.0	0.6
	Min	0.2	0.7	0.9	0.4	0.5	0.9	0.9	2.6	1.1	0.9	2.1	0.5	1.0	1.8	1.3	0.5	0.3	1.0	0.1
	Max	1.4	0.7	0.9	0.4	0.9	2.5	0.9	6.5	1.4	1.7	4.0	1.9	1.5	2.1	3.4	2.0	4.2	1.0	1.1
CH_4 fluxes (ng m^{-2} s^{-1})		11.12.09	19.01.10	28.01.10	09.02.10	19.02.10	02.03.10	09.03.10	06.04.10	04.05.10	29.07.10	27.08.10	02.09.10	16.09.10	27.09.10	07.10.10	18.10.10	04.11.10	24.11.10	13.12.10
Outdoor run (within fence)	N	16	7	7	5	10	8	9	6	16	16	16	8	8	8	16	15	16	16	13
	Mean	−10	−8	3	−3	1	114	−1	−6	−2	−6	−4	−4	−3	13	12	0	48	5	−6
	Median	−6	−8	0	−2	−5	17	9	−17	−1	−8	−4	−3	−4	2	−3	−2	19	1	−3
	Min	−47	−18	−50	−23	−39	−13	−27	−30	−22	−27	−11	−9	−17	−6	−42	−18	−2	−5	−29
	Max	17	6	41	28	44	566	22	56	14	17	9	−1	15	86	115	27	169	35	9
Control (outside fence)	N	3	2	2	2	1	3	0	1	2	3	3	2	2	2	3	2	3	2	2
	Mean	−3	−7	4	−12	−15	2		−9	−2	0	−2	−8	−6	−4	−9	−1	−2	−9	−6
	Median	2	−7	4	−12	−15	−5		−9	−2	0	−2	−8	−6	−4	−21	−1	−1	−9	−6
	Min	−12	−9	−8	−21	−15	−29		−9	−4	−3	−8	−8	−7	−11	−44	−4	−5	−10	−11
	Max	3	−5	16	−2	−15	40		−9	0	4	3	−7	−6	3	36	1	1	−7	0

respectively at the beginning of the batch, and at the end of the batch. For non-volatile compounds (P and K), outdoor excretion was computed according to Eq. (3). The average outdoor/total excretion ratio was then calculated for P and K, and then applied to N to estimate the outdoor N excretion.

$$X_{\text{Outdoor}_{\text{excretion}}} = X_{\text{Total}_{\text{excretion}}} - (X_{\text{Litter}} - X_{\text{Straw}}) \qquad (3)$$

where X is the mass of either P or K.

2.8 Temporal and spatial variations in soil nutrient (P, N, C) stocks

Changes in soil organic carbon (SOC), total nitrogen (N) and total phosphorus (P) stocks in the soil of the run were estimated in order to assess the temporal and spatial impacts of broiler foraging on soil biogeochemistry. Soil samples were taken before the first broilers had access to the run (March 2009) and after the end of the last batch (December 2010). The run was divided into 25 grid squares of equal surface area. In each grid square, 5 to 6 auger samples were dug up to characterize the 0–15 cm horizon. Bulk density (g cm^{-3} soil) was measured by the cylinder method for each sample and a correction for the stone content estimated by 2 mm mesh sieving was applied. These samples were then mixed to create a composite sample for each grid square for chemical analysis. Total contents in C and N (g kg^{-1} soil) were measured by dry combustion (ISO, 1995, 1998) while P content was measured with spectrometry (ISO, 1994). For each grid square, stocks were finally calculated according to Eq. (4).

$$S_X = A \cdot d \cdot \theta_X \cdot h \qquad (4)$$

with X the considered nutrient (C, N or P); S_X the stock of X in the soil (kg); A the area of the grid square (m^2); d the bulk density (kg m^{-3} soil); θ_X the content in X of the soil (in kg kg^{-1} soil) and h the horizon thickness (m).

Stock variations for each grid square of the run were then calculated as the difference between initial and final stocks. Data were interpolated spatially by kriging in order to provide a spatially integrated stock variation for the whole run.

2.9 Outdoor spatial distribution of broilers

The spatial distribution of broilers on the run was obtained by a series of behavioural observations, which were performed on days 35, 49 and 63 of each rearing period (5 and 7 scans per day for batches WS and SA, respectively), using the method described by Germain et al. (2010, 2011). About 100 broilers were tagged with a numbered poncho placed through the back of the neck. During each observation period, the locations of the tagged broilers on the run were recorded according to 16 predefined zones. To that purpose, 4 equal stripes of 11.75 m wide and perpendicular to the broiler house were divided into 4 zones of respectively 5, 10, 15 and 20 m wide parallel to the broiler house. Data were interpolated spatially by kriging in order to derive a spatially explicit "foraging index", which represents a percentage of total outdoor time spent in each square meter of the run.
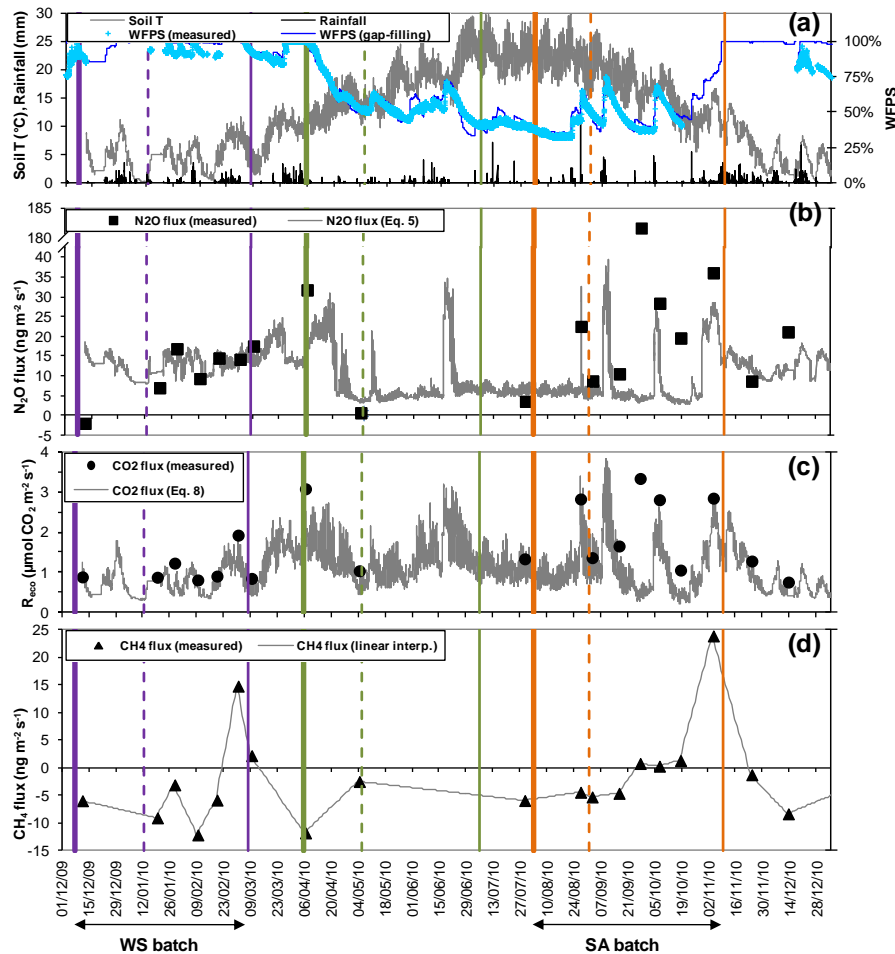
Fig. 2. Overview of soil conditions, meteorology, measured (black symbols) and gap-filled (grey lines) GHG fluxes at the outdoor run scale. **(a)** Soil temperature at -5 cm depth (Soil T), rainfall and water-filled pore space (WFPS). **(b)** N_2O fluxes. **(c)** Ecosystem respiration (R_{eco}). **(d)** CH_4 fluxes. GHG fluxes for each measurement day are spatial integrals computed from individual measurements using geostatistical methods. Vertical lines indicate batch start (thick, continuous), broilers first outdoor access (dotted) and slaughter (thin, continuous).

3 Results

3.1 Overview of measured chamber fluxes

Measured N_2O and CH_4 fluxes were extremely variable in time and space for both batches, with temporal variations shown in Table 2 and Fig. 2 and spatial variability shown in Figs. 3, 4 and 5 for the three GHG. N_2O fluxes ranged -32 from to $+1026$ ng N_2O m^{-2} s^{-1} and from -12 to $+5910$ ng N_2O m^{-2} s^{-1} during batches WS and SA, respectively, with the usual atmospheric science sign convention of plus for emissions and minus for deposition. The emission peaks observed during the study mostly occurred in chambers located closest to the house and the two pop-holes for outdoor access (Fig. 3). Uptake fluxes on the run represented about 21 % and 14 % of all individual chamber measurements during batches WS and SA, respectively.

CH_4 exchange was largely dominated by small uptake fluxes, observed in 64 % and 61 % of all individual flux data for batches WS and SA, respectively. Nevertheless, fluxes ranged from -50 to $+566$ ng CH_4 m^{-2} s^{-1} and from -42 to $+169$ ng CH_4 m^{-2} s^{-1} during batches WS and SA, respectively, suggesting that the net sink activity on the run was offset by rare emission peaks (Table 2). Once again, these peaks were observed in the chambers located closest to the broiler house (Fig. 4).

Concerning CO_2, emission fluxes generated by the (heterotrophic and autotrophic) respiration of the soil + grass ecosystem (R_{eco}) ranged from $+0.2$ to $+5$ µmol CO_2 m^{-2} s^{-1} and from $+0.2$ to $+24$ µmol CO_2 m^{-2} s^{-1} for batches WS and SA, respectively.
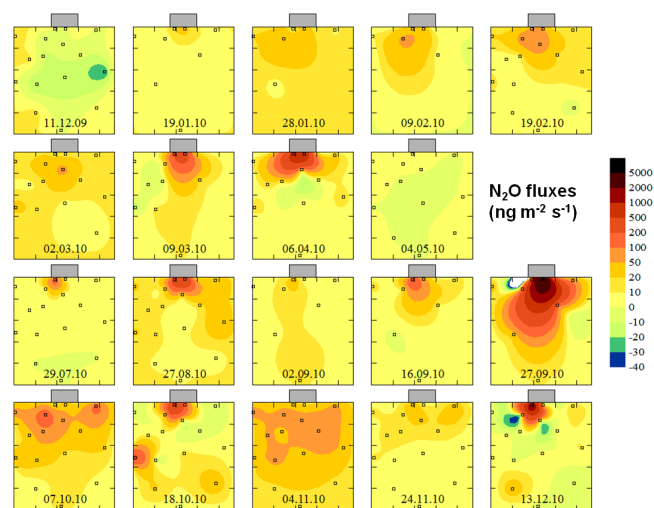
Fig. 3. Spatially interpolated (kriging method) maps of N_2O fluxes. For each measurement day, the kriging was done on the basis of the available chamber measurements as shown by the number of chambers displayed in each map.
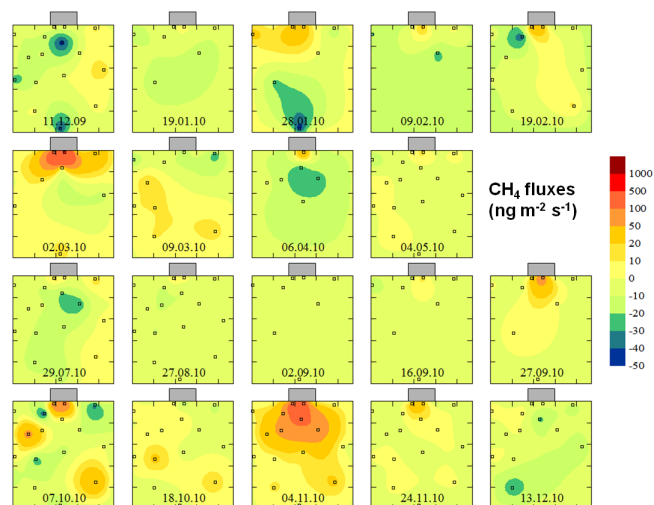

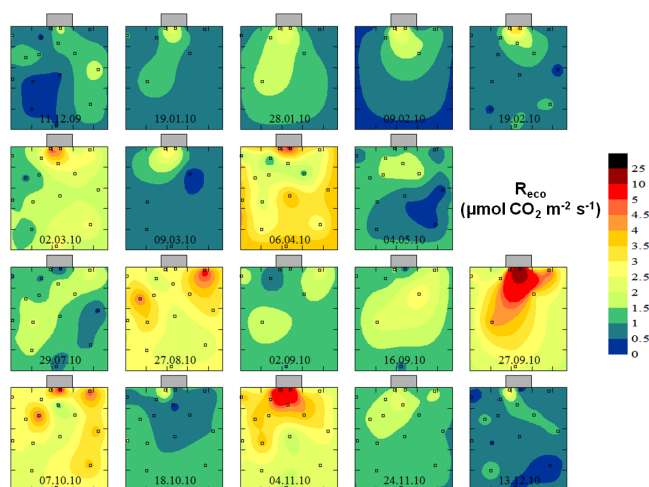**Fig. 4.** Spatially interpolated (kriging method) maps of CH_4 fluxes. Comments as in Fig. 3.

Fig. 5. Spatially interpolated (kriging method) maps of ecosystem respiration R_{eco} (not including broiler respiration). Comments as in Fig. 3.

3.2 Spatially integrated GHG fluxes

Figures 3, 4 and 5 present the spatial distributions of N_2O, CH_4 and CO_2 fluxes, respectively, for each measurement day and based on kriging. Table 3 summarizes the spatial N_2O, CH_4 and CO_2 flux integrals estimated by kriging and radial basis function. The discrepancy between the two interpolation methods was generally below 10 % and on average −4 %, −0.9 % and −1.3 % for N_2O, CH_4 and CO_2 fluxes, respectively, showing that the spatial integrals are not very sensitive to the interpolation method.

Plot-scale N_2O fluxes ranged from −2.1 to +31.8 ng N_2O $m^{-2} s^{-1}$ and from +3.5 to +181.7 ng N_2O $m^{-2} s^{-1}$, respectively for all measurement days of batches WS and SA. Plot-scale CH_4 fluxes were frequently negative, confirming a sink activity at the scale of the run, with values for different measurement days ranging from −12.2 to +15.1 and from −8.3 to +23.7 ng CH_4 $m^{-2} s^{-1}$, respectively, for batches WS and SA. The range of plot-scale integrals of respiration of the ecosystem R_{eco}, from +0.8 to +3.1 µmol CO_2 $m^{-2} s^{-1}$ and from +0.7 to +3.3 µmol CO_2 $m^{-2} s^{-1}$ respectively for batches WS and SA (Table 3), indicates somewhat weaker spatial patterns of R_{eco} than of N_2O, even if large CO_2 fluxes were also clearly associated with the area just outside the broiler house (Fig. 5).

3.3 Temporal gap-filling functions

To derive annual-scale estimates of N_2O fluxes based on a discrete time series of measurements, one possible method consists in an investigation of the relationships of N_2O emission fluxes to environmental variables known to control N_2O production and consumption in soils, with meteorology and soil moisture key controlling variables (Davidson, 1991; Granli and Bøckman, 1994). While the effect of soil temperature on N_2O emissions is predictably positive, the effect of water-filled pore space (WFPS, expressed as the percentage ratio of actual SWC to saturation SWC), is unlikely to be monotonous from 0 to 100 % WFPS, since in wet saturated soils total denitrification is expected to proceed all the way to N_2, with less or no potential for N_2O to evolve. At lower WFPS, nitrification is expected to be the dominant mechanism for N_2O emission (Davidson, 1991). As a result, the overall effect of WFPS on the N_2O emission potential can be thought of as bell-shaped, with maximum likelihood of

Table 3. Summary of spatially integrated GHG fluxes for each measurement day, computed using (an average of) kriging and radial basis functions to interpolate between all available individual chamber measurements. Soil temperature and water-filled pore space (WFPS) are values observed at the same time as GHG fluxes, i.e. generally around mid-afternoon.

| | Soil T 5 cm (°C) | WFPS 5 cm (%) | N_2O flux (ng N_2O m^{-2} s^{-1}) | | | | CH_4 flux (ng CH_4 m^{-2} s^{-1}) | | | | R_{eco} CO_2 (µmol m^{-2} s^{-1}) | | | |
			Kriging	Radial basis function	%diff.*	Average	Kriging	Radial basis function	%diff.*	Average	Kriging	Radial basis function	%diff.*	Average
WS batch														
11 Dec 2009	8.6	87 %	−2.2	−1.9	−13.5 %	−2.1	−5.9	−6.0	0.3 %	−5.9	0.9	0.9	0.0 %	0.9
19 Jan 2010	5.1	100 %	6.9	7.0	0.7 %	7.0	−9.1	−9.0	−0.3 %	−9.1	0.9	0.9	−1.2 %	0.9
28 Jan 2010	4.6	94 %	16.8	16.9	0.4 %	16.9	−3.0	−3.1	2.0 %	−3.1	1.2	1.2	−1.1 %	1.2
9 Feb 2010	4.8	93 %	9.6	9.0	−6.9 %	9.3	−12.1	−12.2	1.4 %	−12.1	0.8	0.8	−3.9 %	0.8
19 Feb 2010	5.6	91 %	14.5	14.5	−0.1 %	14.5	−5.3	−6.4	19.5 %	−5.8	0.9	0.9	−0.7 %	0.9
2 Mar 2010	11.4	99 %	14.2	14.2	0.0 %	14.2	14.5	15.1	3.9 %	14.8	1.9	1.9	−1.1 %	1.9
9 Mar 2010	4.6	91 %	18.2	16.9	−7.4 %	17.5	2.0	2.5	23.1 %	2.2	0.9	0.8	−4.1 %	0.8
6 Apr 2010	13.9	100 %	33.2	30.3	−8.8 %	31.8	−11.8	−11.7	−0.7 %	−11.8	3.1	3.1	0.8 %	3.1
Intermediate measurements														
4 May 2010	14.5	53 %	0.6	0.7	13.1 %	0.6	−2.5	−2.5	0.8 %	−2.5	1.0	1.0	1.3 %	1.0
29 Jul 2010	25.3	41 %	3.6	3.4	−5.5 %	3.5	−5.9	−5.9	0.8 %	−5.9	1.3	1.3	−0.1 %	1.3
SA batch														
27 Aug 2010	23.6	64 %	22.8	22.2	−2.4 %	22.5	−4.4	−4.4	−0.6 %	−4.4	2.8	2.8	−0.5 %	2.8
2 Sep 2010	25.7	50 %	8.8	8.5	−3.5 %	8.7	−5.2	−5.3	3.1 %	−5.3	1.4	1.4	−0.8 %	1.4
16 Sep 2010	20.5	50 %	11.1	9.9	−10.2 %	10.5	−4.6	−4.6	1.9 %	−4.6	1.7	1.6	−0.8 %	1.7
27 Sep 2010	16.5	38 %	186.9	176.6	−5.5 %	181.7	0.9	0.7	−28.0 %	0.8	3.4	3.3	−2.5 %	3.3
7 Oct 2010	20.7	64 %	28.9	27.8	−3.9 %	28.4	0.3	0.3	−19.2 %	0.3	2.8	2.8	−0.9 %	2.8
18 Oct 2010	13.6	42 %	19.9	19.2	−3.7 %	19.6	1.3	1.3	0.6 %	1.3	1.1	1.1	0.2 %	1.1
4 Nov 2010	16.6	77 %	36.5	35.8	−1.9 %	36.1	24.1	23.7	−1.9 %	23.9	2.9	2.8	−0.8 %	2.8
24 Nov 2010	7.9	99 %	8.8	8.5	−3.5 %	8.6	−1.1	−1.4	20.0 %	−1.2	1.3	1.3	−0.7 %	1.3
13 Dec 2010	2.1	98 %	22.6	19.7	−12.7 %	21.1	−8.4	−8.3	−1.5 %	−8.3	0.8	0.8	−0.8 %	0.8

* Between kriging and radial basis function.

emission somewhere in the range 60–90 % WFPS, and minimum flux in either very dry or waterlogged conditions. The optimum range depends on the soil texture, pH, substrate C/N ratio and the composition of microbial communities.

To interpret the temporal variations in measured N_2O fluxes, and to derive a mechanistically grounded gap-filling algorithm, we therefore adapted the empirical model used by Flechard et al. (2007a), in which a bell membership function (BMF, ranging from 0 to 1), characterizing the optimum WFPS range for N_2O emission, was parameterized on the basis of plot-scale N_2O fluxes (Table 3). The model responds exponentially to soil temperature, and the WFPS and temperature effects are multiplicative. Datasets for soil temperature and WFPS are also given in Table 3. The N_2O flux was modelled as:

$$F_{N_2O} = e^{(c_0 + c_1 T_s + c_2 B(\text{WFPS}))} \qquad (5)$$

with F_{N_2O} expressed in ng N_2O m^{-2} s^{-1}, T_s the soil temperature at −5 cm (°C), WFPS at −5 cm expressed as a percentage of pore saturation (0–100), and c_0, c_1 and c_2 the parameters of a multiple linear regression between $\ln(F_{N_2O})$ and the independent variables T_s and $B(\text{WFPS})$. The bell membership function was given as:

$$B(\text{WFPS}) = \frac{1}{1 + (\frac{\text{WFPS}-c}{a})^{(2b)}} \qquad (6)$$

The three parameters of the bell function (a, width; b, taper coefficient; c, centre) were adjusted to maximize the goodness of fit between the observed data and the fluxes predicted by Eq. (5). With the parameters $a = 20.1$ (% WFPS), $b = 6$ and $c = 82.1$ (% WFPS), the resulting regression parameters were $c_0 = 0.57 \pm 0.77$, $c_1 = 0.05 \pm 0.03$ and $c_2 = 1.87 \pm 0.61$, and the multiple R^2 was 0.41. The observation-based model thus predicts a maximum likelihood of N_2O emission at 82 % WFPS, suggesting that denitrification was the dominant mechanism of N_2O production at this site.

Note that two measurement days excluded from the data set used to derive these equations. First, 11 December 2009 (before batch WS) was removed because N_2O uptake fluxes were considered to be representative of background conditions. Second, 27 September 2010 was removed because the low apparent value of WFPS (38 %) measured at −5 cm depth could not be considered representative of soil moisture at the surface (1–2 cm depth), which was certainly influenced by rainfall of 24 September after a long dry period and triggered the highest measured N_2O flux (Table 3).

Results of temporal N_2O gap-filling using Eq. (5) are presented in Fig. 2b. Measured fluxes indicate a steady increase over the course of batch WS (cool, wet), which is reasonably well simulated by Eq. (5). By contrast, measured emissions during batch SA (warm and dry with wet episodes) were much more heterogeneous, being much more dependent on

recent rainfall. Here, the gap-filling algorithm does reproduce some of the small-scale temporal variability, but the peak on 27 September is not simulated since the model is driven by SWC at -5 cm depth, which did not respond much to the rainfall of 24 September. Thus, for the special case of this peak, we used a linear interpolation between 24 September up to 27 September, and again from 27 September down to the next measurement day on 7 October.

For the temporal gap-filling of the (spatially integrated) R_{eco} (CO_2) fluxes, an empirical algorithm based on functional relationships to T_s and WFPS was also derived. The temperature response of R_{eco} for non water-limiting conditions (set arbitrarily by WFPS > 75 %) was determined by fitting the parameters of the Lloyd and Taylor (1994) Arrhenius-type function:

$$R_{eco}(\text{opt}) = R_{ref} \cdot e^{\left(E_0\left(\frac{1}{T_{ref}-T_0} - \frac{1}{T_s-T_0}\right)\right)} \tag{7}$$

with R_{eco} in μmol CO_2 m^{-2} s^{-1}, temperatures in K, R_{ref} the ecosystem respiration at the reference temperature T_{ref} ($=283.15$ K), T_0 the growth characteristic of the exponential function, and E_0 an activation energy analogue representing the ecosystem respiration sensitivity to temperature (set to 309 K). For the subset of CO_2 fluxes measured at WFPS > 75 %, the fitted parameters were $R_{ref} = 1.60$ μmol CO_2 m^{-2} s^{-1} and $T_0 = 234$ K, and the R^2 was 0.87. For drier periods, however, ecosystem respiration was assumed to be limited by the lack of moisture, and R_{eco} no longer responds to the high summer temperatures as predicted by the Lloyd and Taylor model (e.g. Balogh et al., 2011; Longdoz et al., 2008). We therefore introduced a soil moisture multiplicative term (a sigmoidal function S(WFPS), ranging from 0 to 1) to modulate the temperature response:

$$R_{eco} = R_{eco}(\text{opt}) \cdot S(\text{WFPS}) \tag{8}$$

$$\text{with } S(\text{WFPS}) = S_2 + \frac{(S_1 - S_2)}{1 + e^{\left(\frac{\text{WFPS}-S_3}{d}\right)}} \tag{9}$$

Here, S_2 was set to 1 (maximum value for S, i.e. no water limitation for R_{eco}), and the parameters S_1, S_3 and d were adjusted to maximise the goodness of fit between the observed data and the fluxes predicted by Eq. (8). The resulting values were $S_1 = 0.125$, $S_3 = 65$ (% WFPS) and $d = 9$ (% WFPS), with an R^2 of 0.86. These numbers suggest 50 % and 80 % reductions of R_{eco} at WFPS levels of 62 % and 44 %, respectively, compared to non water-limiting conditions (Eq. 7).

Note that for the same reason as for N_2O (see above), the 27th September 2010 datapoint was removed from the data set used to fit parameters of the CO_2 algorithm. Also, due to a lack of data points measured in or near waterlogged conditions (100 % WFPS), we could not identify whether a reduction in R_{eco} would actually occur at saturation, due to limited soil CO_2 diffusion (Flechard et al., 2007b). However, since waterlogged conditions tend to occur predominantly in the winter half year at this site, when temperatures are low

and therefore R_{eco} is small, the overall (annual) error would likely be small.

Results of temporal CO_2 gap-filling using Eq. (8) are presented in Fig. 2c. The semi-empirical model of R_{eco} reproduces satisfactorily the short-term as well as the seasonal fluctuations observed in system respiration, with summer values severely limited by the lack of moisture, despite the higher temperatures.

Concerning CH_4 emissions, no significant relationship between fluxes and environmental conditions was found. Therefore, temporal gap-filling was carried out either with a simple linear interpolation between measurement days (Fig. 2d), or with an average flux multiplied by the duration of the batch as presented in Sect. 2.6.

3.4 Batch-scale and annual-scale GHG fluxes

Results of temporal integration for batches WS and SA obtained from the different gap-filling methods described previously are given in Table 4. Emission estimates for N_2O ranged from 0.86 to 1.01 kg N_2O-N ha^{-1} and from 1.45 to 2.44 kg N_2O-N ha^{-1} for batches WS and SA, respectively. The run was found to be a net CH_4 sink during batch WS (-0.39 to -0.34 kg CH_4-C ha^{-1}) but the flux was probably not significantly different from zero during batch SA ($+0.03$ to $+0.07$ kg CH_4-C ha^{-1}). Cumulative estimates of R_{eco} ranged from $+1284$ to $+1819$ kg CO_2-C ha^{-1} during batch WS and from $+1560$ to $+2312$ kg CO_2-C ha^{-1} during batch SA, depending on the temporal integration method. For both N_2O and CO_2, the estimates based on the upscaled mean flux and from linear interpolation were systematically higher than estimates using the T- and WFPS-based functions. This is logical since, (i) the distributions of fluxes were not normal but skewed or log-normal, with few elevated (peak) values leading to a bias in average-based estimates, and (ii) fluxes were always measured in the afternoon, when soil temperatures and microbial activity were highest.

Although flux measurements focused on batches WS and SA, and only two flux datapoints are available for batch 4 (Fig. 2), we present tentative estimates of GHG fluxes for batch 4 from 6 May to 31 August 2010 (Table 4), based either on the mean of these two fluxes, or on functional relationships derived from batches WS and SA. Although these values are highly uncertain, we speculate that N_2O fluxes were rather low due to dry soil conditions and low rainfall during these summer months. These estimates are also needed to provide annual-scale GHG flux estimates, corresponding approximately to the sum of the three batches, and ranging from 2.7 to 4.8 kg N_2O-N ha^{-1} yr^{-1}, -0.5 to -0.6 kg CH_4-C ha^{-1} yr^{-1}, and 4.4 to 6.1 t CO_2-C ha^{-1} yr^{-1} (R_{eco} only).

Table 4. GHG fluxes at the batch and annual time scales, using alternative temporal integration methods.

Integration period	Time interval	Temporal gap-filling method	kg N_2O-N ha^{-1}	kg CH_4-C ha^{-1}	kg CO_2-C ha^{-1} R_{eco}
Batch 3 (measured and gap-filled)	13.01.2010 to 05.05.2010	Mean Flux · N days[a]	0.87	−0.39	1569
		Linear (dot to dot)	1.01	−0.34	1819
		Flux = $f(T_s,$ WFPS)[b]	0.86	na	1284
Batch 4 (simulation)	06.05.2010 to 31.08.2010	Mean Flux · N days[a,c]	0.84	−0.61	2546
		Flux = $f(T_s,$ WFPS)[b]	0.44	na	1469
Batch 5 (measured and gap-filled)	01.09.2010 to 05.01.2011	Mean Flux · N days[a]	2.44	0.03	2312
		Linear (dot to dot)	2.18	0.07	2210
		Flux = $f(T_s,$ WFPS)[b,d]	1.45	na	1560
Year (measured and gap-filled)	01.01.2010 to 31.12.2010	Mean Flux · N days[e]	4.76	−0.46	6147
		Linear (dot to dot)	3.56	−0.65	5883
		Flux = $f(T_s,$ WFPS)[b]	2.71	na	4351

R_{eco} = Respiration of the (grassland + soil) ecosystem, [a] Calculated from the average of all daily fluxes obtained from measurements between the date of first outdoor access of animals (and that of the next batch), [b] Measured fluxes + mechanistic gap-filling function (Eq. 5 for N_2O-N and Eq. 8 for CO_2-C; not available for CH_4-C), [c] Based on only two days measurements made after the slaughter of Batch 4., [d] For N_2O and CO_2, emission peaks measured on 27.09.10 added on top of base emission (see text for details), [e] Calculated from the average of all daily fluxes obtained from measurements from year start to year end.

3.5 Magnitude and significance of temporal soil nutrient changes

Soil nutrient stocks for C, N and P at the paddock scale all increased significantly from start to end of the experiment, as shown in Table S1 (Supplement). The stocks were determined for the 0–15 cm horizon only, by means of stratified sampling for the 0–5 cm and 5–15 cm layers separately, and the data in Table S1 are therefore lower bound estimates of stock changes for the whole soil depth. The paddock-scale stock increments for the 0–15 cm horizon were +377 g C m^{-2} yr^{-1} (+11 % yr^{-1}), +25 g N m^{-2} yr^{-1} (+7 % yr^{-1}) and +9 g P m^{-2} yr^{-1} (+20 % yr^{-1}). Paired-sample t-test statistics on the 25 paired composite samples (initial vs. final) were used since the five soil samples used for each composite sample were extracted from the very same locations (±10 cm) at the start and end of the experiment. The temporal stock increments were thus statistically highly significant ($P < 0.01$) for all three nutrients.

The main source of uncertainty in the calculation of soil carbon stocks was the evaluation of the fraction of coarse stony elements at the paddock scale, which was of the order of 15 % in the 0–15 cm layer. We tested the error propagation from the coarse element fraction to the calculated soil elemental stock, and found that hypothetical random errors for the coarse fraction of 5 %, 10 % and 30 % would translate into an error (of opposite sign) of 1 %, 2 % and 5 %, approximately, for the soil stock. Given the magnitude of observed changes in C, P and N stocks (~10–20 % per year), and given that the relevant error is that of the stock change (or difference between two measurements, not that of the absolute stock value at a given time), this study concludes that

very significant C, N and P stock changes took place during the experiment.

4 Discussion

4.1 Emission factors for the outdoor run

There are no measurement-based GHG emission factors (EF) for poultry with outdoor access in the literature. Our flux measurements indicate substantial annual N_2O emissions and a weak net CH_4 sink on the run. For N_2O, the calculation of an EF requires the knowledge of total N inputs to the grassland, as well as background N_2O emissions outside the run, which were provided in our case by three control chambers (Table 2). Note that in this paper, the EF we calculate is based on the total N input (as in Tier 1 methodology of IPCC), not on a "net" N input as defined by total input minus NH_3 volatilisation (as in Tier 2 methodology of IPCC), which would obviously result in much higher EF values (~factor 2, if one considers a 50 % NH_3 volatilisation/abatement rate).

The only way to quantify N inputs to the run by broiler droppings is by a calculation of the mass balance of chemical elements in animal feed through the production system (ingestion, retention, excretion), whereby the fraction of outdoor excretion is estimated as the difference between total and indoor excretion. Based on P and K data, Meda (2011) showed that the ratios of outdoor/total excretion were 6 % and 47 % for batches WS and SA, respectively, and that excretion of N on the run was estimated to 5.4 and 42.2 kg N (equivalent to 23 and 180 kg N ha^{-1}) for batches WS and SA, respectively. For batch 4, a similar calculation yielded an outdoor excretion rate of 33.2 kg N

(equivalent to $141\,\text{kg}\,\text{N}\,\text{ha}^{-1}$), and the annual N input by outdoor excretion (3 batches) to the run was thus estimated at $344\,\text{kg}\,\text{N}\,\text{ha}^{-1}\,\text{yr}^{-1}$. This value is high, but according to CORPEN French references (2006), a large part of this N input is lost through volatilization (60 %), certainly albeit with considerable variability and uncertainty (Sintermann et al., 2011).

Our estimates of annual N_2O emission range from $+2.71$ to $+4.76\,\text{kg}\,N_2\text{O-N}\,\text{ha}^{-1}\,\text{yr}^{-1}$ for the run (Table 4). The background annual emission for the experimental site, outside the run, calculated as the mean of all control chamber fluxes, was $+1.25\,\text{kg}\,N_2\text{O-N}\,\text{ha}^{-1}\,\text{yr}^{-1}$. This background emission is relatively high but can possibly be explained by a high atmospheric N input (NH_3 dry deposition from the surrounding experimental farm buildings (poultry, pigs)). The EF for N_2O is calculated as the run emission minus the background flux, and divided by the excretion N input. This yields a range of EF of 0.4 % to 1.0 % for outdoor N excretion. These values are substantially lower than the IPCC default value of 2 % (IPCC, 2006b). However, our N input estimate of $344\,\text{kg}\,\text{N}\,\text{ha}^{-1}\,\text{yr}^{-1}$ obtained by mass balance calculations was rather high and is also subject to considerable uncertainty.

For CH_4, no study on outdoor emissions by free range poultry was found in the literature. However, based on IPCC references (IPCC, 2006a) and on our estimation of droppings partitioning between the house and the run, we calculated a theoretical CH_4 EF of $0.03\,\text{kg}\,CH_4\,\text{ha}^{-1}\,\text{yr}^{-1}$ due to the degradation of organic matter in droppings on the run. This value does not take into account the potential CH_4 consumption by the soil microorganisms as frequently observed during our study, with an annual flux of the order of $-0.5\,\text{kg}\,CH_4\text{-C}\,\text{ha}^{-1}\,\text{yr}^{-1}$ showing a net sink activity of the grassland. This was at the low end of the range of values reported in the literature (from -1 to $-13\,\text{kg}\,CH_4\,\text{ha}^{-1}\,\text{yr}^{-1}$) for non-grazed grasslands (Smith et al., 2000; Castaldi et al., 2007; Klemedtsson et al., 2009), but consistent with those given by Saggar et al. (2008) for New-Zealand pastures ($<1\,\text{kg}\,CH_4\,\text{ha}^{-1}\,\text{yr}^{-1}$). Methane oxidation rates are influenced by many factors, many of them related to oxygen availability such as soil water content or porosity. Methane can indeed only be oxidised in specific ranges of moisture when oxygen is available and can diffuse into the soil (Smith et al., 2000; Le Mer and Roger, 2001) though we did not found any significant relationship between average CH_4 fluxes and soil moisture (WFPS).

In this study, respiration of the grassland ecosystem (R_{eco}) was estimated at $+4351\,\text{kg}\,CO_2\text{-C}\,\text{ha}^{-1}\,\text{yr}^{-1}$, and is lower than values reported in the literature for cattle-grazed pastures, ranging from 6400 to more than $15\,000\,\text{kg}\,CO_2$-$\text{C}\,\text{ha}^{-1}\,\text{yr}^{-1}$ (Allard et al., 2007; Gilmanov et al., 2007). Respiration is known to be influenced by many factors (Jones et al., 2006) such as grassland composition (i.e. species), grassland management (mowing, grazing, fertilization), clim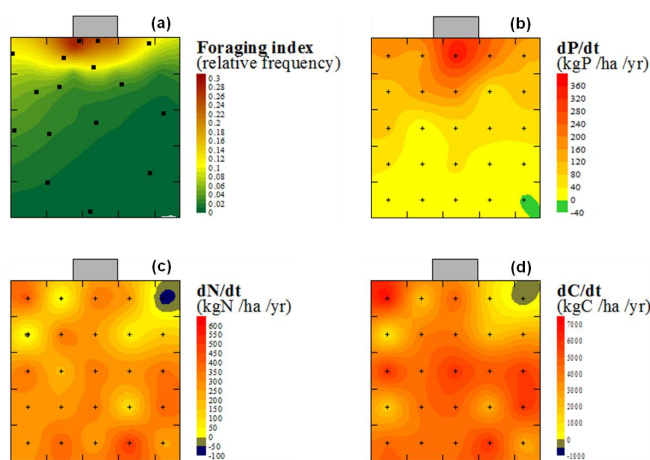atic conditions (temperature, rainfall), soil conditions (water content, temperature) and gross photosynthetic assimilation rates. The low value of R_{eco} cannot be explained by limiting conditions in organic matter availability (C and N) due to the amount brought by droppings. However, soil water was limiting throughout the summer and certainly affected ecosystem respiration as reflected by the need to modulate the temperature response of R_{eco} by a SWC factor (Eqs. 7 to 9). The mechanism certainly involved a reduction of both autotrophic and heterotrophic components in dry conditions, with limited plant CO_2 assimilation rates reducing the supply of root exudates to soil microorganisms.

Fig. 6. Kriged maps of broiler foraging index (collective probability of broilers being present on a surface area of $1\,\text{m}^2$ at a given location) **(a)** and annual variations of phosphorus **(b)**, total nitrogen **(c)** and total carbon **(d)** soil stocks. Black squares **(a)** indicate locations for chamber measurements, while crosses **(b, c, d)** indicate soil sampling sites.

4.2 Spatial and temporal controls of GHG fluxes

The initial distribution of the static chambers on the run was determined under the assumption that broilers would spend more time in the first half of the run, close to the broiler house, where emissions should be higher than in the rest of the run due to a larger amount of nutrients (carbon, nitrogen) excreted. Behavioural observations (Fig. 6a) confirmed indeed that, when they were outdoors, broilers spent most of their time in the first 15 m of the run (75 % of the total time spent outdoors), which led to a higher soil accumulation of phosphorus here (65 % of the total stock increment for the whole run) (Fig. 6b). Similar nutrient accumulation was also reported by Aarnink et al. (2006) and Kratz et al. (2004) who concluded that the phosphorus load in the first meters close to the broiler house was several times higher than Dutch and German standards, respectively, and indicates an enhanced risk of P run-off, especially in a zone where the plant cover has been suppressed by broilers. Variations in phosphorus soil stock were positively correlated (spatially)
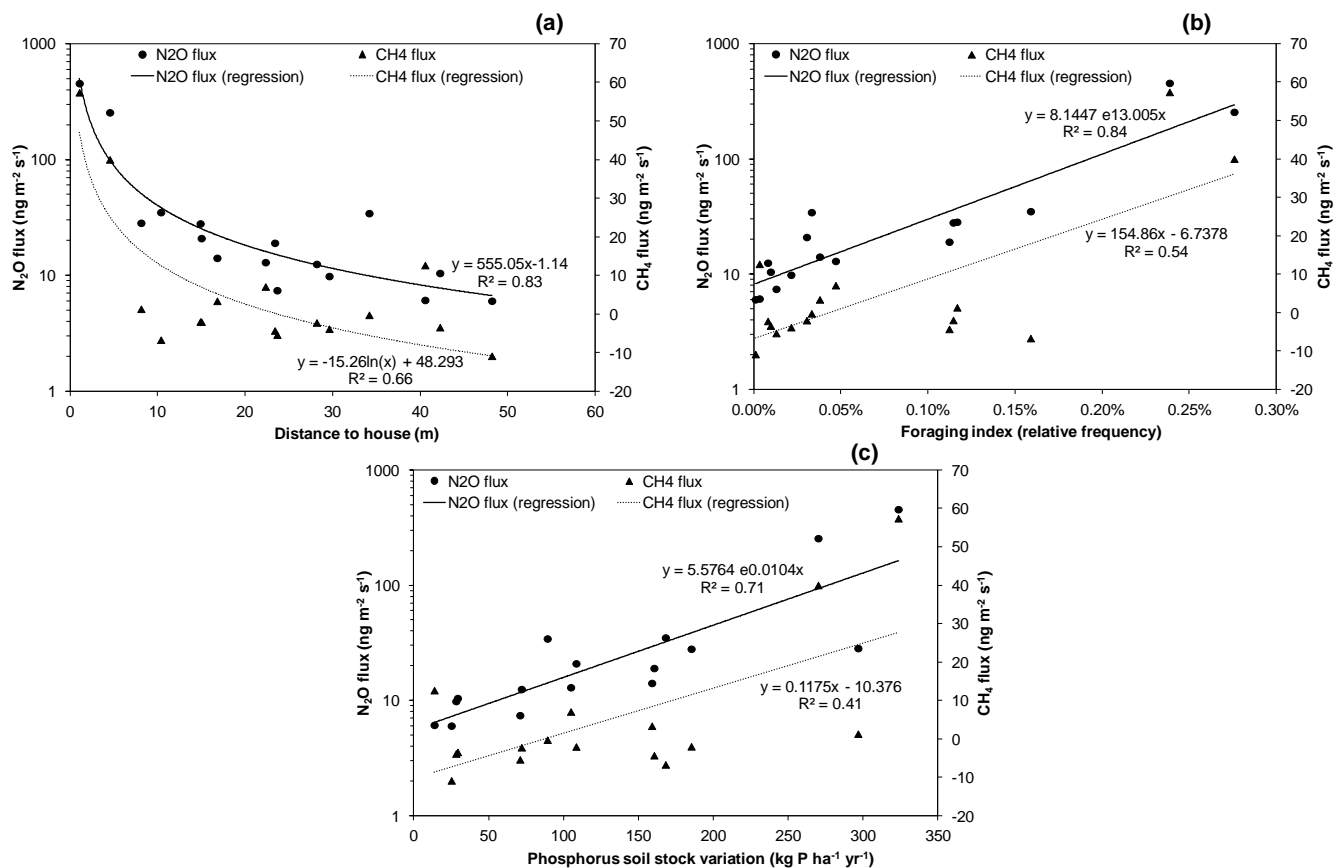
Fig. 7. Relationships between average measured N_2O and CH_4 chamber fluxes and control variables: **(a)** distance to broiler house. **(b)** Foraging index. **(c)** Annual variation of phosphorus soil stock. Each datapoint represents the mean measured flux at one chamber location on the plot.

with the foraging index ($R^2 = 0.74$), but no relationship between foraging index and variations in total nitrogen or total carbon soil stocks was found ($R^2 < 0.1$). This is due to a faster turnover of carbon and nitrogen than for phosphorus, partly due to the uptake by plants, soil fauna and microorganisms, but more importantly here by mineralisation and gaseous emissions (NH_3, NO, N_2O, N_2, CO_2) in the soil, and possibly leaching (NO_3^-, dissolved inorganic or organic C and N). Further, we may hypothesize that variations in C and N soil stocks were of the same magnitude close to the house and at the back of the run because of two distinct phenomena. In the first part of the run, the vegetation was very sparse and the increase in C and N in soil was explained by the concentration of nutrients excreted in this zone, whereas at the back of the run, the increase in C and N was mostly due to C sequestration and N fixation by the soil and the vegetation.

The spatial interpolation of N_2O, CH_4 and CO_2 fluxes, summarized in Figs. 3 to 5, respectively, confirmed the dominance of near-house emissions. The highest emission speaks were generally measured in the two chambers placed only one meter away from the pop-holes. Relationships between average measured N_2O and CH_4 fluxes and variations in phosphorus stock soil, foraging index and distance to the broiler house, for all chamber sampling points on the field, are given in Fig. 7. As expected, N_2O and CH_4 fluxes were positively correlated with the animals presence and their cumulative droppings (as shown by the increase in soil P), and negatively correlated with distance to the broiler house. Good correlation between CO_2 fluxes and these variables (not shown in this paper) was also found ($R^2 > 0.50$).

The spatial variability of fluxes and the higher emissions measured in the first 15 m of the run could be explained by the amount of carbon and nitrogen available for soil microorganisms. We may assume that the high amount of nutrients excreted close to the house generated a non-limiting situation for soil microbial activity (Flechard et al., 2005), which, coupled with the disappearance of the plant cover in the first meters in front of the broiler house, because of broiler consumption, stalling and compaction, led to the absence of competition for resources for soil microorganisms. This indicates that R_{eco} was dominated by soil heterotrophic respiration. Furthermore, the surface compaction of the soil in the first meters of the run (Fig. 1), coupled with high input of reactive

C, may have generated local anaerobic conditions and thus enhanced the production of N_2O through denitrification and CH_4 production. By the same token, this compaction certainly reduced oxygen availability in the soil and thus limited methane oxidation. The large amounts of N dropped in this zone could also have limited the potential for CH_4 oxidation and increased N_2O emission by microorganisms through the increased oxidation of ammonium into nitrite (nitrification) by the same enzyme (methane monooxygenase) as reported in the literature (Le Mer and Roger, 2001; Jassal et al., 2011).

N_2O emission is known to respond to soil pH, but the effect is complex. The rates of both nitrification and denitrification increase with pH from acidic conditions (pH 3 to 5) to slightly alkaline soils. However, for N_2O emissions conflicting results have been reported, with a positive impact of H^+ on N_2O emissions in acidic soils, if denitrification is the main source, and a negative impact of H^+ on emissions in near-neutral to slightly alkaline conditions, if nitrification is the main source (Granli and Bøckman, 1994). The slightly alkaline soil pH at this site (8.1) might suggest a dominant role of nitrification. On the other hand, the change in pH from start to end of this experiment was only of the order of -0.1 pH units, and no significant spatial variability in pH was observed in response to the differential loads of organic matter and N across the paddock, due to the strong stability and buffer provided by the high calcium carbonate content (20 %).

The effect of run characteristics and season on broiler behaviour has been studied in the literature. It is commonly accepted that the foraging of free-range poultry increases with better climatic conditions (warmer temperature for instance) and with the presence on the run of "vertical structural elements" (trees, bushes), because they provide shade and shelters from predators and snow or rain (Haneklaus et al., 2000; Cornetto and Estevez, 2001; Lubac et al., 2003; Zeltner and Hirt, 2003, 2008; Hegelund et al., 2005; Germain et al., 2010, 2011). The presence on the run of a number of vertical elements such as trees could therefore lead to a better spatial distribution of droppings, reducing the occurrence of emission (gaseous, leaching) hotspots, limit the risks of vegetation degradation close to the house and improve the general functioning of the ecosystem (e.g. Fig. 1), which would thus absorb a greater fraction of the excess nutrients deposited. In these conditions, we suggest that plot-scale levels of emission should be lower in runs planted with trees and bushes than those we measured in this study. Further experiments are required to confirm this hypothesis.

The temporal variability of fluxes was mainly due to seasonal changes in soil moisture (WFPS) and temperature. Since fluxes were measured in a large range of WFPS (30–100 %) and temperature (0–30 °C), we may reasonably think that the uncertainty of our annual estimates of N_2O and CO_2 fluxes based on gap-filling functions is moderate. A part of the uncertainty in our estimates stems from the fact that environmental variables were measured outside the run and that the same value of WFPS and temperature were used for the whole field, while measurements and observations all indicate large heterogeneity in density of the plant cover (leaf area index) and soil characteristics. The slope of the run (Fig. 1) probably also induced a higher SWC close to the broiler house (Fig. 1) than upslope, due to surface run-off, especially during high intensity precipitation events. Therefore, local predictions of T and SWC would reduce uncertainty of annual averages of GHG fluxes from the run.

4.3 Towards a net GHG budget of the outdoor run

Emission factors are needed from an inventory viewpoint (IPCC), but to assess the whole climate change impact of a production system, a total net GHG exchange (NGHGE), taking into account all global warming potential (GWP) contributions by CO_2, N_2O and CH_4 fluxes is required (Soussana et al., 2007). In this study we determined the net N_2O and CH_4 fluxes, but for CO_2 only ecosystem respiration (R_{eco}) was measured; for a full GHG balance at the paddock scale, the net ecosystem exchange (NEE, measurable by eddy covariance, see e.g. Ammann et al., 2007), defined as gross primary production (GPP) minus R_{eco}, would be required. In addition, CO_2 losses by animal respiration ($R_{broilers}$) would add to emissions by plants and soil, as would losses by soil dissolved organic and inorganic carbon leaching, as well as exports by animals as they consume worms and other forms of terrestrial organic materials. C imports by animal excreta on the other hand contribute very significantly to an increase in soil organic matter in this system.

In the absence of measurements for many of these terms, we derived a tentative net GHG budget by inferring NEE from the observed and significant change in soil organic carbon ($+377 \, g \, C \, m^{-2} \, s^{-1}$, see Sect. 3.5) and by making various assumptions concerning C imports, C exports and C leaching. The details of the methodology and reference values from the literature used for this calculation are provided in the Supplement. The NEE estimate for the grassland + broiler system (NEE_{G+B}) thus obtained was $-1163 \, kg \, C \, ha^{-1} \, yr^{-1}$, implying that the whole ecosystem (soil + vegetation + animals) would have been a net C sink, which was offset by roughly one third by N_2O emissions with a negligible impact by CH_4 oxidation. Overall, the NGHGE of $-2561 \, kg \, CO_2 \, eq. \, ha^{-1} \, yr^{-1}$ would suggest that the outdoor part of the rearing system behaves as a net sink of greenhouse gases, but it must be stressed that uncertainties are very large and plot-scale measurements of all component terms are required to provide a more reliable and robust estimate. In addition, much of the short-term ($\sim$2-yr) increase in soil organic carbon was likely contributed by deposited bird excreta, the longer term fate of which is uncertain, and which therefore does not necessarily represent a potential for C sequestration.

5 Conclusions

Annual GHG flux estimates suggest that the run of organic, free-range broilers emitted between 2.7 and 4.7 kg N_2O ha^{-1} yr^{-1} and acted as a very weak sink of CH_4, which compensated less than 1.5 % of the N_2O forcing effect. Spatial variability of fluxes was high. As expected, GHG fluxes were higher close to the broiler house where large amounts of organic matter are excreted by the broilers. The temporal variability of fluxes is related to soil moisture and temperature, which regulate soil microbial processes. The uncertainty associated with our estimates can be reduced by an improved understanding and modelling of processes controlling GHG fluxes. This study provides the first measurement-based assessment of outdoor N_2O emissions factors for "grazing" poultry. However, this value is site-specific and more measurements are needed in different climatic conditions and over different soil and vegetation types, in order to derive a comprehensive and representative set of values to be used in emission inventories and in Life Cycle Assessment approaches. Based on measured GHG fluxes and on the change of soil organic carbon during the time frame of the experiment, we calculated a short-term net GHG budget for the run, equivalent to a net sink of -2600 kg CO_2 equivalent ha^{-1} yr^{-1}. However, more data over the lifetime of a production system (e.g. 10 yr) would be required to assess the long term soil carbon sequestration potential. This study focused on GHG fluxes, and outdoor ammonia emissions should be also quantified, given the potential impacts associated to ammonia deposition downwind from the source (eutrophication, acidification, secondary N_2O emissions).

Supplementary material related to this article is available online at: http://www.biogeosciences.net/9/1493/2012/bg-9-1493-2012-supplement.pdf.

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