CO₂ uptake is offset by CH₄ and N₂O emissions in a poplar short-rotation coppice

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Abstract

The need for renewable energy sources will lead to a considerable expansion in the planting of dedicated fast-growing biomass crops across Europe. These are commonly cultivated as short-rotation coppice (SRC), and currently poplar (*Populus* spp.) is the most widely planted. In this study, we report the greenhouse gas (GHG) fluxes of carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) measured using eddy covariance technique in an SRC plantation for bioenergy production. Measurements were made during the period 2010–2013, that is, during the first two rotations of the SRC. The overall GHG balance of the 4 years of the study was an emission of 1.90 (±1.37) Mg CO₂eq ha⁻¹; this indicated that soil trace gas emissions offset the CO₂ uptake by the plantation. CH₄ and N₂O contributed almost equally to offset the CO₂ uptake of -5.28 (±0.67) Mg CO₂eq ha⁻¹ with an overall emission of 3.56 (±0.35) Mg CO₂eq ha⁻¹ of N₂O and of 3.53 (±0.85) Mg CO₂eq ha⁻¹ of CH₄. N₂O emissions mostly occurred during one single peak a few months after the site was converted to SRC; this peak comprised 44% of the total N₂O loss during the two rotations. Accurately capturing emission events proved to be critical for deriving correct estimates of the GHG balance. The nitrogen (N) content of the soil and the water table depth were the two drivers that best explained the variability in N₂O and CH₄, respectively. This study underlines the importance of the 'non-CO₂ GHGs' on the overall balance. Further long-term investigations of soil trace gas emissions should monitor the N content and the mineralization rate of the soil, as well as the microbial community, as drivers of the trace gas emissions.

Keywords: bioenergy crops, greenhouse gas balance, self-organizing map, soil nitrogen

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Introduction

The Renewable Energy Directive (EU, 2013) of the European Commission calls for 20% of the EU's total energy consumption to be supplied from renewable sources by 2020. One of these sources is biomass from agricultural and forestry waste, but an increase in the cultivation of bioenergy feedstock crops is necessary to achieve the proposed displacement of fossil fuels (Ashworth *et al.*, 2013). Dedicated bioenergy crops grown as short-rotation coppice (SRC) have been receiving increased attention over the past decades (Ceulemans & Deraedt, 1999; Dallemand *et al.*, 2008; Rowe *et al.*, 2009; Rosso *et al.*, 2013). Poplar (*Populus*), willow (*Salix*), red alder (*Alnus rubra*) and black locust (*Robinia pseudoacacia*) are the most significant species commercially cultivated as SRC in Europe, and they cover an area of about 50 kha (Don *et al.*, 2011). Compared to food crops, SRC requires low

inputs of fertilizers and herbicides, and SRC cultures grow well on lands that are less suitable for agriculture (Dallemand *et al.*, 2008).

Nevertheless, there are some environmental concerns. The large expansion of SRC and the consequent direct and indirect land-use change (LUC/ILUC) may have considerable impacts on climate mitigation in terms of greenhouse gas (GHG) emissions (Searchinger *et al.*, 2008; Fargione *et al.*, 2008). While LUC from perennial pasture into annual biofuel crops induces large CO₂ emissions into the atmosphere (Zenone *et al.*, 2011), land conversion into perennial SRC may have a beneficial effect on the soil organic carbon (SOC) balance. Several authors report SOC initially declines, but then levels off after 7–15 years followed by SOC sequestration (Hansen, 1993; Grigal & Berguson, 1998; Jug *et al.*, 1999; Arevalo *et al.*, 2011; Verlinden *et al.*, 2013). On the other hand, Walter *et al.* (2014) indicated that there is no evidence of any general variation in the soil organic stock when cropland is converted into SRC. The identification of the factors determining whether carbon may be

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sequestered in the soil under SRC remains a major challenge. While SOC plays an important role in the overall carbon balance of SRC, other GHGs may have a significant effect on the global warming impact of a cropping system.

N₂O emission from soils is governed by two microbial processes: nitrification and denitrification (Robertson & Groffman, 2015). Management factors, as N fertilization as well as soil environmental conditions (soil temperature, moisture, pH and oxygen), control soil microbial processes and, thus, soil N₂O production (Firestone & Davidson, 1989; Robertson & Groffman, 2015). N₂O can also be consumed by soil microbes under specific environmental conditions, although this process is still under debate (Chapuis-Lardy *et al.*, 2007; Schlesinger, 2013). Methane is produced by soil microbial fauna at anaerobic microsites in water-saturated zones by reduction of CO₂ with hydrogen (Conrad, 1996). In contrast to N₂O, the consumption of CH₄ in soils is well documented and aerobic soils can act as a strong sink for CH₄ (Conrad, 1996; Schimel, 2004; Whalen, 2005). Variables controlling CH₄ production and consumption include soil temperature, available C pool, soil oxygen concentration, and the presence or absence of alternative electron acceptors as iron, sulfate or nitrate (NO₃⁻). Soil physical properties [i.e., texture, porosity, bulk density (BD), water-filled pore space (WFPS)] and hydrology (i.e., soil moisture, water table depth and lateral flow rate) also play an important role as drivers of CH₄ production and consumption (Brinson, 1993; Whiting & Chanton, 1993; Holden, 2005; Whalen, 2005; Von Fischer & Hedin, 2007). Recently, fast-response N₂O and CH₄ analyzers based on spectroscopic techniques such as tunable diode laser and quantum cascade laser spectrometers have become commercially available. Using these new technologies, an increasing number of sites have been equipped for eddy covariance measurements. They measure single or multiple GHG fluxes at a high temporal resolution and they cover annual timescales, providing the essential datasets for the whole GHG balance (McManus *et al.*, 2010; Merbold *et al.*, 2014).

In this study, we report fluxes of all major GHGs (N₂O, CH₄ and CO₂) measured in a poplar SRC plantation for four consecutive years after establishment. Our objectives were as follows: (i) to quantify the seasonal dynamics of CO₂, N₂O and CH₄ fluxes and their cumulative fluxes; (ii) to identify, through the use of the neural network technique of self-organizing maps (SOM), the environmental drivers of the N₂O and CH₄ production; and (iii) to quantify the whole GHG balance for the first two (2-year) rotations of the plantation.

Materials and methods

Site description

The research site was an operational SRC of poplar (*Populus*) located in Lochristi, East Flanders (Belgium; 51°06'44" N, 3°51'02" E) at an elevation of 6.25 m above sea level. The 18.4 ha site was a former agricultural area (62% of the total) with cultivations of ryegrass, wheat, potatoes, beet, and most recently corn monoculture regularly fertilized with N at a rate of 200–300 kg ha⁻¹ year⁻¹ as liquid animal manure and chemical fertilizer. The remaining part (38% of the total area) was an extensively grazed pasture. An area of 14.5 ha (excluding the headlands that remained unplanted) was planted on April 7–10, 2010, with 12 selected and commercially available poplar and three willow genotypes. No fertilizers and no irrigation were applied during the 4 years of the study (2010–2013). The soil was characterized as a sandy texture with a clay-enriched deeper soil layer and poor natural drainage. The C and N mass fractions of the two previous land-use types were significantly ($P = 0.000$) different: In the upper 0–15 cm soil layer, they were lower in cropland ($1.48\% \pm 0.32$ and $0.12\% \pm 0.03$, respectively) as compared to previous pasture ($1.95\% \pm 0.36$ and $0.18\% \pm 0.03$, respectively). Between the two different land-use types, the contents of C and N were not significantly different in the upper 90 cm (Broeckx *et al.*, 2012).

The long-term (30-year) average annual air temperature at the site was 9.5 °C and the average total annual precipitation was 726 mm (collected by the nearby station of the Royal Meteorological Institute of Belgium; www.meteo.be). Rainfall was evenly distributed across the year. Total N deposition in Flanders (Belgium) ranged from 20 to 35 kg N ha⁻¹ y⁻¹. (Flemish Environment Agency; www.vmm.be). The aboveground biomass was harvested on February 2–3, 2012 (end of the first rotation), and on February 16–17, 2014 (end of the second rotation). The aboveground biomass productivity was 4 Mg ha⁻¹ y⁻¹ during the first rotation and 12.2 Mg ha⁻¹ y⁻¹ during the second rotation (Verlinden *et al.*, 2015). Mean height growth rate raised from 2.08 m y⁻¹ during the first rotation to 2.99 m y⁻¹ during the second rotation (Verlinden *et al.*, 2015).

Eddy covariance measurements

The eddy covariance (EC) system used to measure gas and energy fluxes between the SRC plantation and the atmosphere included the following: a three-dimensional sonic anemometer (model CSAT3, Campbell Scientific, Logan, UT, USA) to measure the wind speed components, a closed-path differential infrared gas analyzer (LI-7000, LI-COR, Lincoln, NE, USA) for the measurements of the CO₂ and H₂O mole fractions in the air, a closed-path fast-response N₂O/CO analyzer (model 908-0014, Los Gatos Research, Mountain View, CA, USA) and a closed-path fast-response CH₄ analyzer (model DLT-100, Los Gatos Research, Mountain View, CA, USA). Measurements were made continuously with all the instruments sampling at a frequency of 10 Hz. The sonic anemometer and the inlets of the sampling lines were positioned 5.8 m above the soil surface during a first period (from June 1, 2010 to August 30, 2011) and were raised to 6.6 m thereafter (from August 31, 2011 to

December 31, 2013) to match the rapid height growth of the plantation. A vacuum pump, positioned at the outlet of the LI-7000 gas analyzer, generated a flow rate of about $20\text{--}22\text{ L min}^{-1}$. Two scroll pumps (model XDS-35i, Edwards, MA, USA) were used to draw air through the N_2O and the CH_4 analyzers, which resulted in a flow rate of $\sim 25\text{ L min}^{-1}$ for the N_2O analyzer and $\sim 45\text{ L min}^{-1}$ for the CH_4 analyzer. This flow rate maintained a turbulent flow regime in the sampling line – as required by standard EC methodology – to avoid concentration dilution in the air samples between the sampling line inlet and the gas analyzer, and the consequent, high-frequency fluctuation dampening (Munger *et al.*, 2012). To eliminate fluctuations caused by the pump, two buffer tanks (each of 0.5 L) were positioned between the pump and the outlet of the analyzer. The sampling tubes were heated and thermally insulated from the environment to prevent condensation and deliquescence of the air samples in the inlet tubes; this avoids potential latent heat (LE) flux underestimation (Fratini *et al.*, 2012).

The CH_4 analyzer was calibrated every 1–6 months using 4 ppm (CH_4 mixture balanced with air with 5% accuracy). The N_2O analyzer was calibrated at the Los Gatos Research Company on July 6, 2010, using a NOAA primary standard at 322.24 ppbv N_2O in air (uncertainty <0.1 ppbv). The linearity of the analyzer was then tested by diluting a higher concentration bottle (440 ppbv N_2O) by known amounts and measuring the response of the analyzer. More detailed information has been previously provided by Zona *et al.* (2013a,b).

The half-hourly average rate of change in CO_2 storage was calculated according to Aubinet *et al.* (2001). For this calculation, air temperature and $[\text{CO}_2]$ profiles were measured at the meteorological mast (12 m from the eddy covariance mast) at different heights that reflected the height of the growing canopy. At each height, $[\text{CO}_2]$ and air temperature were continuously recorded with the CR 1000 data logger (Campbell Scientific, Logan, UT, USA). The $[\text{CO}_2]$ was measured with a LI-840 analyzer (LI-COR, Lincoln, NE, USA) that switched among the different sample heights. As the storage term represented a minor percentage of the total fluxes and as concentration profiles of CH_4 and N_2O were not measured, no storage term was added to the final calculation of the CH_4 and N_2O fluxes.

Postprocessing and gap filling

Fluxes of CO_2 , N_2O and CH_4 were calculated using the EDIRE software (R. Clement, University of Edinburgh, UK; www.geos.ed.ac.uk/abs/research/micromet/EdiRe/) from the high-frequency data series divided into half-hourly averaging periods. A two-component rotation was applied to set mean lateral and vertical wind velocity components to zero, while the time delay between scalar and vertical wind velocity fluctuations was determined by cross-correlation optimization (Foken *et al.*, 2012). The frequency response correction was applied to the EC fluxes following Horst (1997). The Webb–Pearman–Leuning (WPL) H_2O vapor correction term was applied to all the data collected (cfr. Webb *et al.*, 1980). A filter was applied to the data using the following rejection criteria: (i) more than ten

standard deviations from the 30-min mean for CO_2 , and for the wind velocity components, u , v and w ; (ii) more than eight standard deviations from the 30-min mean for N_2O , and more than three standard deviations for CH_4 ; (iii) when CH_4 and N_2O minimum concentration was <0 ; and (iv) for quality flags 9 (data that should be removed) as suggested by Foken & Wichura (1996) and Foken *et al.* (2004). To maximize the data coming from the plantation, (the mast was located in the north-eastern corner of the site), only data with wind direction between 50° and 250° were used. To avoid the possible underestimation of fluxes during stable (nonturbulent) conditions at night, data with friction velocity (u^*) below a threshold were treated separately using the procedure described by Reichstein *et al.* (2005). The u^* thresholds adopted for each year were 0.15 m s^{-1} , 0.13 m s^{-1} , 0.18 m s^{-1} and 0.20 m s^{-1} for the datasets of 2010, 2011, 2012 and 2013, respectively. A storage term – about 3.4%, 3.8%, 3.1% and 3.7% of the cumulative net ecosystem exchange (NEE) in 2010, 2011, 2012 and 2013, respectively – was added to the CO_2 flux calculation before setting the u^* thresholds. The storage term accounts for a minor percentage of the total fluxes. As no environmental variable consistently explained either CH_4 or N_2O fluxes at a 30-min time step, we used the linear extrapolation (Mishurov & Kiely, 2011) of the CH_4 and N_2O fluxes divided into different periods with similar emission rates. Methane fluxes were averaged over monthly intervals; these were then summed to give the annual total emission. The N_2O fluxes were gap filled in the same way, except for the emission peak in 2010 where the N_2O fluxes were extrapolated into periods with similar emission rates (Zona *et al.*, 2013a,b). Minimum detectable fluxes for the half-hourly averaged fluxes were as follows: $0.21\text{ nmol m}^{-2}\text{ s}^{-1}$ for N_2O (Kroon *et al.*, 2007) and $3.8\text{ nmol m}^{-2}\text{ s}^{-1}$ for CH_4 (Detto *et al.*, 2011). An uncertainty analysis was performed following the methodology described by Kroon *et al.* (2010). The fluxes of CO_2 were gap filled to calculate the cumulative net ecosystem production (NEP) and the ecosystem respiration (R_{eco}) using the marginal distribution sampling (MDS) method (Reichstein *et al.*, 2005) implemented in www.bgc-jena.mpg.de/~MDIwork/eddyproc/ (2013 version) with the error propagated from the half-hourly averaged data into the annual uncertainty. The quality of the EC system performance was assessed by the energy balance closure $(H + LE)/(R_n - G)$, where H is sensible heat flux, LE is latent heat flux, R_n is net radiation, and G is average soil heat flux. Gap-filled CH_4 , N_2O and CO_2 fluxes were then converted into CO_2 equivalents (CO_2eq) using GWPs of 25 and 298 for CH_4 and N_2O , respectively (Stocker *et al.*, 2013) to assess the GWP of the SRC plantation.

Environmental and meteorological parameters

Several environmental variables were continuously recorded. Volumetric soil water content (SWC) was measured vertically in the soil layers of 0–30 cm, 0–20 cm, 0–10 cm, and horizontally at specific depths of 1 m, 60 cm, 40 cm, 30 cm and 20 cm below the surface, using soil moisture probes (TDR model CS616, Campbell Scientific, Logan, UT, USA). Soil water content was then converted to water-filled pore space (WFPS) according to Wu *et al.* (2010). The water table depth was

measured using a pressure transducer (model PDCR1830, Campbell Scientific, Logan, UT, USA) installed in a perforated pipe inserted into the ground to a depth of 1.97 m. Precipitation was measured using a tipping bucket rain gauge (model 3665R; Spectrum Technologies Inc., Plainfield, IL, USA). Soil temperature (T_{soil}) was measured as the average temperature of a soil layer of 8 cm depth (model TCAV-L averaging thermocouples, Campbell Scientific, Logan, UT, USA). The G was measured by eight heat flux plates (HFT3, REBS Inc., Seattle, WA, USA) installed at 6–8 cm soil depth. Air temperature (T_a) and relative humidity (RH) were recorded on the EC mast using a Vaisala probe (model HMP45C, Vaisala, Helsinki, Finland). The net radiation was measured with a four-component radiometer (model CNR1, Kipp & Zonen, Delft, The Netherlands). Incoming photosynthetically active radiation (PAR; 400–700 nm) was recorded above the canopy using quantum sensors (LI-190, LI-COR, Lincoln, NE, USA). The CO_2 efflux from the soil (R_{soil}) was monitored by an automated soil CO_2 flux system (LI-8100, LI-COR, Lincoln, NE, USA). Sixteen chambers operating as closed systems were connected to an infrared gas analyzer through a multiplexer (LI-8150, LI-COR, Lincoln, NE, USA). The 16 chambers were spatially distributed in the plantation and covered both former land-use types. A more detailed description of the setup of the soil CO_2 flux system and the most important results were previously published (Verlinden *et al.*, 2013).

Self-organizing maps

The self-organizing map (SOM) is a popular neural network that is based on nonsupervised learning. A SOM consists of neurons which are organized on a low-dimensional grid. The number of neurons may vary from a few dozen up to several thousands, and each neuron is connected to adjacent neuron by a neighborhood relation, which dictates the topology of the map. The use of a neighborhood function differentiates the SOM from other neural networks because in this way they preserve the topological properties of the input vector (Kohonen, 2001; Oja *et al.*, 2003). In this study, the SOM was used to cluster half-hourly N_2O and CH_4 fluxes, observed T_{soil} , SWC, soil respiration (R_{soil}) and R_{eco} . Each cell on the two-dimensional grid has a d-dimensional prototype vector m , where d is the dimension of the data vectors x . In this study, $x = [\text{N}_2\text{O}, \text{CH}_4, T_{\text{soil}}, \text{SWC}, R_{\text{eco}}, R_{\text{soil}}]$ and each half-hourly flux in the time series received a single location on the two-dimensional grid which resembles a topographic map; each half hour in the time series receives a location on the map, and similar half hours in terms of flux and meteorological conditions received a nearby position on the grid. Each layer has been plotted on the same underlying grid. These plots are called component planes. The values of the fluxes and meteorological measurements on each of the component planes are represented by a color scale. The relation between the GHG fluxes and the meteorological observations can thus be studied by comparing the respective values of a certain grid cell on the different component planes. In this way, the SOM can be used to visualize the often complex relationships between several time series of greenhouse gas fluxes and meteorological variables. The different component planes

of the SOM shared the same underlying grid and, therefore, an individual half hour had the same location on each component plane. In each component plane, the Z-axis (given by the colors) represented a single variable describing the GHG fluxes or climatic conditions. The combination of the values of all component planes for a given location described the fluxes as well as the associated environmental conditions that were allocated to that location for a particular half hour. The data processing was conducted applying the SOM toolbox version 2 (<http://www.cis.hut.fi/projects/somtoolbox/>) implemented in Matlab R2014a (The MathWorks Inc., Natick, MA, USA). A more detailed description of the SOM technique has been given by Luyssaert *et al.* (2007).

Groundwater nitrogen measurements

Samples for total N concentrations (i.e., all inorganic and organic N forms) in the groundwater were measured from six water access tubes installed in July 2011 at different locations across the plantation, covering the two former land-use types. The 5-cm-diameter access tubes were made of PVC, with openings along the whole of their 2 m length. From August 2011 to July 2013, groundwater samples were taken from each access tube approximately once per month, with the exception of October 2011, November 2011 and January 2012 when no readings were made. Water samples were collected by applying a vacuum of 60 kPa to a 2-m plastic tube connected to a glass bottle. Each 200 mL sample of groundwater was divided into a 100 mL subsample for nitrate (NO_3^-) and nitrite (NO_2^-) analysis, and a 100 mL subsample for N measurements. After collection, the samples were stored at 4 °C and sent to the laboratory within the next 24 h for analysis. Nitrate and nitrite concentrations were determined using a spectrophotometric method (Aquakem 250, discrete analyzer, Labmedics, Manchester, UK). Other N compound concentrations such as ammonia (NH_3), ammonium (NH_4^+) and organic N were determined following the Kjeldahl method (SKALAR autosampler segmented flow analyzer, Skalar, Breda, The Netherlands).

Results

Environmental conditions and soil nitrogen

The mean daily SWC followed the variations in water table depth throughout the entire duration of the study (Fig. 1a). Total precipitation was 556 mm in 2010 (from June to December only), 718 mm in 2011, 857 mm in 2012 and 947 mm in 2013 (entire years). The daily precipitation values are reported in Fig. 1b. The annual precipitation was in each year close to the long-term average. The average daily T_a (Fig. 1b) during the period June–September ranged between 18 °C and 22 °C in all four growing seasons. Seasonal dynamics of the daily average T_{soil} measured in the topsoil (0–8 cm) followed the dynamics of T_a in all 4 years (Fig. 1c).

The N concentration in the groundwater (Table 1) was low: The total N concentration during the

Fig. 1 Evolution of the environmental parameters during the course of the four-year study. (a) Temporal trend of the daily average water table depth and soil water content (SWC as measured with time domain refractometers); (b) mean daily air temperature (T_a) and precipitation; (c) daily average soil temperature (T_{soil}) measured at 8 cm depth.

measuring period showed an average value of $10 \text{ mg of N L}^{-1}$, with most of it in the form of NO_3^- . Nitrite concentration was very low or below the detection limit in most of the samples. Finally, NH_4^+ , NH_3 and organic N forms also showed concentrations close to the detection limit. The highest NO_3^- (close to 37 mg L^{-1}) and NO_2 concentrations were measured on August 31, 2011; NO_3^- stabilized at around 7 mg L^{-1} thereafter. Other N forms (NH_4^+ , NH_3) as well as organic nitrogen concentrations remained relatively constant during the two rotations and were around 2 mg L^{-1} on average.

Temporal dynamics of CO_2 , N_2O and CH_4 fluxes

The energy balance closure was 87% and 93% in 2010 and 2011 (first rotation), while it was 70% and 82% in

2012 and 2013 (second rotation). The percentage of data coverage after quality control ranged from 32 to 56% (Table 2). For the entire eddy flux dataset, the main reasons for data removal were (i) wind direction outside of the desired footprint (removal of 27% of all data in 2010, of 34% in 2011, 38% in 2012 and 33% in 2013), (ii) followed by the u^* threshold, (iii) concentrations outside the reasonable range and (iv) instrument maintenance. For the CO_2 data, we removed all data flagged '9' (bad data) as suggested by Foken *et al.* (2004).

The daily NEE (Fig. 2) showed a considerable variation among the 4 years: The plantation was characterized by a different length of the growing season (i.e., days with a net CO_2 uptake) within each of the two rotations. During the first rotation, the growing season ranged from August to September 2010 (establishment

Table 1 Concentrations of NO_3^- , NO_2^- , NH_4^+ , NH_3^- (mg L^{-1}) in the groundwater (and corresponding water table depth (cm below the surface) on different sampling dates during the course of the two rotations of the plantation

	NO_3	NO_2	NH_4 and NH_3 and other organic N	Water table (cm)
18/08/2011	27.4 ± 17.0	0.052 ± 0.054	3.9 ± 3.5	-134.8 ± 21.9
31/08/2011	36.9 ± 38.0	$0.022 \pm .015$	1.5 ± 0.1	-129.5 ± 22.5
19/09/2011	31.2 ± 29.1	0.032 ± 0.028	1.3 ± 0.3	-121.4 ± 23.9
19/12/2011	1.8 ± 2.9	0.035 ± 0.019	1.9 ± 1.0	-11.2 ± 13.8
13/02/2012	2.4 ± 4.7	0.014 ± 0.008	2.2 ± 0.4	-113.6 ± 28.5
06/03/2012	3.7 ± 5.9	<0.01	2.1 ± 0.7	-54 ± 25.9
06/04/2012	2.4 ± 4.5	<0.01	2.5 ± 1.0	-109 ± 20.2
10/05/2012	6.7 ± 8.6	<0.01	3.3 ± 2.1	-42.4 ± 25.3
05/06/2012	3.3 ± 5.5	0.012 ± 0.005	2.3 ± 0.5	-115.4 ± 21.7
02/07/2012	2.7 ± 4.4	<0.01	3.5 ± 2.9	-117.3 ± 21
26/07/2012	2.5 ± 6.4	0.015 ± 0.013	2.5 ± 0.9	-80.4 ± 16.2
29/08/2012	1.9 ± 4.9	<0.01	4.8 ± 2.4	-140 ± 20.6
16/10/2012	1.5 ± 3.7	<0.01	3 ± 2.5	-95.9 ± 22.1
20/11/2012	5.2 ± 12.7	<0.01	2 ± 0.9	-76.3 ± 19.2
17/12/2012	13.7 ± 26.6	<0.01	2.7 ± 1.0	-38.6 ± 20.9
31/01/2013	4.8 ± 8.6	<0.01	1.6 ± 0.6	-15.4 ± 16.1
04/03/2013	4.7 ± 10.3	<0.01	1.9 ± 0.4	-88 ± 21.2
28/03/2013	4 ± 7.1	<0.01	1.9 ± 0.5	-76.6 ± 19.3
25/04/2013	4.4 ± 9.6	0.013 ± 0.007	2.3 ± 1	-109.9 ± 18
03/06/2013	3.4 ± 7.4	<0.01	2 ± 0.5	-104.1 ± 19
27/06/2013	3.4 ± 8.2	<0.01	2.1 ± 1.4	-136.6 ± 20.4
29/07/2013	3.7 ± 9.4	0.01 ± 0.001	2.3 ± 0.7	-141.9 ± 23.4
13/02/2014	8.3 ± 10.7	<0.01	1.7 ± 0.6	-25.3 ± 22.4
05/03/2014	8.9 ± 10.5	0.013 ± 0.007	1.6 ± 0.4	-63 ± 21.6

For the chemical analyses, average $\pm$ standard deviation is shown ($n = 7$ for sampling dates between 04/2013 and 07/2013 and $n = 6$ for sampling dates before 04/2012 and after 07/2013).

Table 2 Percentage of data coverage and values below the detection limit for CH_4 and N_2O flux measurements over the 4 years of the study

	2010		2011		2012		2013	
	CH_4	N_2O	CH_4	N_2O	CH_4	N_2O	CH_4	N_2O
Data coverage (%)	32	35	39	39	43	44	36	56
Data below detection limit (%)	28	31	25	38	27	35	25	26

All values are expressed in %.

year) and from April to September 2011 (Fig. 2a–b). During the second rotation, the growing season lasted from June to August 2012 and from April to October 2013 (Fig. 2c–d). The vegetation started later in the first year of each rotation because the crop had to become established either from the woody cuttings (first rotation) or from the harvested stumps (second rotation). Using the sign convention that negative fluxes represent

an uptake by the SRC plantation, the net carbon balance was $75.2 (\pm 4.4) \text{ g C m}^{-2}$ in 2010 (from June to December) and $-95.6 (\pm 5.9) \text{ g C m}^{-2}$ in 2011. After the coppice, the net carbon balance was $151.0 (\pm 10.5) \text{ g C m}^{-2}$ in 2012 and $-274.6 (\pm 18.8) \text{ g C m}^{-2}$ in 2013. Considering the 4 years of the two rotations together, the SRC plantation was a net sink of carbon with an uptake of $-144.0 (\pm 22.8) \text{ g C m}^{-2}$. The average rate of carbon uptake during the growing season (i.e., months with net carbon uptake) was $-39.6 (\pm 16.4) \text{ g C m}^{-2}$ per month during the first rotation and $-59.6 (\pm 42.5) \text{ g C m}^{-2}$ per month during the second rotation.

N_2O fluxes in the establishment year were characterized by a remarkable peak of emissions from the second half of August until the middle of September 2010 (Fig. 3a). This peak was extensively described and analyzed previously (Zona *et al.*, 2011, 2013a,b) and is included in the present GHG balance. From mid-September to December 2010, daily average N_2O emissions ranged from $-0.22 (\pm 0.4)$ to $0.95 (\pm 0.2) \text{ nmol m}^{-2} \text{ s}^{-1}$. Negative fluxes of N_2O (i.e., uptake) were sporadically measured during 2010 (for a total number of 11 days), particularly in December 2010. From September to November 2011, a fairly pronounced diurnal pattern in

Fig. 2 Daily net ecosystem exchange (NEE; $\text{g C m}^{-2} \text{ day}^{-1}$) of CO_2 measured during the 4 years of the study. First rotation: 2010 (panel a) and 2011 (panel b); second rotation: 2012 (panel c) and 2013 (panel d). Gap-filled data are presented with error bars propagated from the half-hourly data into the daily uncertainties.

N_2O fluxes were observed, with daytime N_2O uptake (between September and October 2011, data not shown). In 2011, N_2O fluxes (Fig. 4b) were similar to 2010: The maximum daily N_2O emission was $1.23 (\pm 1.6) \text{ nmol m}^{-2} \text{ s}^{-1}$, while for most of the time the fluxes were lower and ranged between -0.5 and $0.5 \text{ nmol m}^{-2} \text{ s}^{-1}$. Some negative (uptake) fluxes of N_2O were sporadically measured during 2011; they ranged from $-1.0 (\pm 0.5) \text{ nmol m}^{-2} \text{ s}^{-1}$ on day of the year (DOY) 9 in January to $-1.8 (\pm 0.13) \text{ nmol m}^{-2} \text{ s}^{-1}$ on

DOY 280 in October 2011. In late summer and autumn, we identified two periods (from 1 to 28 September and from 13 to 19 October 2011) with a net N_2O uptake. In September 2011, the N_2O uptake showed a clear daily trend (data not shown) with a net uptake during daytime and an emission during the night. On the other hand, after coppice in January 2012, no diurnal or seasonal patterns of N_2O emissions were measured (Fig. 3c). After coppice, a slight increase of the emissions in February 2012 was detected with a daily

Fig. 3 Daily averaged N_2O exchange ($\text{nmol N}_2\text{O m}^{-2} \text{s}^{-1}$; $\pm$ standard deviation). Negative fluxes indicate a net uptake of N_2O , and positive values indicate a net release of N_2O . Panel (a) has a different scale to highlight the contribution of the peak emission in the first year (2010). Data from two rotations are shown. First rotation: 2010 (panel a) and 2011 (panel b); second rotation: 2012 (panel c) and 2013 (panel d). The plantation was coppiced on February 2–3, 2012.

average of about $1.1 \text{ nmol m}^{-2} \text{s}^{-1}$, and a similar trend was observed in May 2012. Overall N_2O uptake was observed in 2012 for a total of 25 days. The magnitude and the temporal dynamics observed in the second year after coppice (2013) (Fig. 3d) were similar to other years. They were, however, more variable in 2013 as compared to 2010–2012 and characterized by a larger number of days (124) with a net N_2O uptake. The negative fluxes of N_2O were especially pronounced toward the end of

the 2013 growing season with values that reached $-4.2 (\pm 3.8) \text{ nmol m}^{-2} \text{s}^{-1}$.

A multiple regression between daily average N_2O fluxes and NO_3^- content in the groundwater table, WFPS and T_{soil} (Fig. 4a) showed a correlation coefficient (r^2) of 0.41 ($P = 0.021$). The assumption of homoscedasticity was met (Fig. 4b), and the predictors did not show multicollinearity [variance inflation factor (VIF) < 10 and tolerance statistic $(1/\text{VIF}) > 0.2$ (Bowerman and

Fig. 4 Multiple linear regression (panel a) analysis for the prediction of the N_2O daily average variability from the concentrations of NO_3 in the groundwater, the water-filled pore space (WFPS) and soil temperature (T_{soil}) measured at 8 cm depth. The equation for the regression line is: $Y = 0.032 - 2.8\text{E-}04 (\text{NO}_3) - 3.1\text{E-}04 (\text{WFPS}) - 2.8\text{E-}04 (\text{Soil T})$. Panel b: regression between the standardized predicted values and the standardized residual. This pattern is indicative of a situation in which the assumptions of linearity and homoscedasticity have been met.

O'Connell, 1990)]. In contrast to the N_2O fluxes, CH_4 fluxes (Fig. 5) showed a more consistent pattern during the period of the measurements with high emission rates observed during autumn and winter. High emissions were observed between November 2011 and February 2012 with values between 40 and 62 $\text{nmol m}^{-2} \text{s}^{-1}$ and during December 2013 with daily average emissions $>40 \text{ nmol m}^{-2} \text{s}^{-1}$. Periods with high emission were also characterized by large uncertainties in the fluxes. During the summer, the values were considerably lower and ranged between 10 and 30 $\text{nmol m}^{-2} \text{s}^{-1}$. Some sporadic days with CH_4 uptake (i.e., CH_4 oxidation) were observed in all 4 years. There were no consistent diurnal trends in the CH_4 emission during the entire period of the study. There was also no significant correlation between the CH_4 fluxes and the

principal environmental variables at 30-min time steps. The high emission rates were observed when the water table was shallow (in autumn and winter); the variation of the water table depth explained 26% of the average monthly CH_4 emission fluxes ($P = 0.001$; Fig. 6).

Controlling environmental drivers of CH_4 , N_2O and CO_2 fluxes

The component planes (Fig. 7) showed a clear link between R_{eco} and soil temperature, where both component planes show a very similar patterns. More specifically, highest R_{eco} is occurring at the same location on the component plane as the highest soil temperature values. A strong relation is also observed between R_{soil} and T_{soil} . In contrast, no such relationships were found between CH_4 and N_2O fluxes, and the meteorological observations. High N_2O fluxes occurred under high as well as under low T_{soil} and SWC. This indicated that N_2O fluxes were not driven by either of these environmental variables at the half-hourly time step. Also for CH_4 no clear link between T_{soil} and SWC was observed. Moreover, the N_2O and CH_4 planes (Fig. 7) showed a clear horizontal gradient in contrast to the environmental drivers of T_{soil} and SWC, which showed a vertical gradient. Subsequently, we were unable to find a link between the N_2O and CH_4 fluxes and the climatic/environmental variables.

Greenhouse gas budget

The whole GHG balance of the 4 years of the study was $1.90 (\pm 1.37) \text{ Mg CO}_2\text{eq ha}^{-1}$. The cumulative monthly GHG emissions for CH_4 , N_2O and CO_2 are given in Table 3, while the temporal dynamics of the cumulative GHG balance are shown in Fig. 8. In 2010 (the establishment year), the plantation was a net source of all three gases: The CO_2 emission was $2.76 \pm (0.16) \text{ Mg CO}_2\text{eq ha}^{-1}$, the CH_4 emission was $0.63 \pm 0.05 \text{ Mg CO}_2\text{eq ha}^{-1}$, and the N_2O was $1.97 \pm 0.49 \text{ Mg CO}_2\text{eq ha}^{-1}$ (from June to December). In 2011 (the second year of the first rotation), the plantation became a CO_2 sink with $-3.51 (\pm 0.56) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$ and a small source of CH_4 and N_2O to the atmosphere with an emission of $0.49 (\pm 0.05) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$ and $0.42 (\pm 0.17) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$, respectively. After the (first) coppice of the aboveground biomass, on February 2–3, 2012 (i.e., the first year of the second rotation), the plantation became a net source of all three gases: CO_2 emission was $5.54 (\pm 0.39) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$, while the CH_4 emission was $1.0 (\pm 0.05) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$ and the N_2O emission $0.96 (\pm 0.14) \text{ Mg CO}_2\text{eq ha}^{-1} \text{y}^{-1}$. In 2013 (i.e., the second year of the second rotation), the plantation was a net sink of CO_2 with $-10.07 (\pm 0.43)$

Fig. 5 Daily averaged CH_4 exchange ($\text{mg CH}_4 \text{ m}^{-2} \text{ s}^{-1}$; $\pm$ standard deviation) over the 4 years (two rotations) of the study. Negative fluxes indicate oxidation of CH_4 , and positive values indicate a net release into the atmosphere. First rotation: 2010 (panel a) and 2011 (panel b); second rotation: 2012 (panel c) and 2013 (panel d).

$\text{Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$, while the CH_4 emission was $1.30 (\pm 0.92) \text{ Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$ and the N_2O emission was $0.31 (\pm 0.25) \text{ Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$. When the three GHGs were integrated over a year, the SRC plantation was a net GHG source in 2010 with $5.36 (\pm 0.52) \text{ Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$ emitted, and a sink in 2011 with $-2.60 (\pm 0.59) \text{ Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$. During the second rotation, the GHG balance was $7.59 (\pm 0.42)$ and $-8.45 (\pm 1.0) \text{ Mg CO}_2\text{eq ha}^{-1} \text{ y}^{-1}$ in 2012 and 2013, respectively. The cumulative emissions of N_2O and CH_4 offset the CO_2

uptake by the SRC plantation and reversed the overall GHGs balance of the SRC from a sink to a source.

Discussion

The temporal variability of the N_2O fluxes can be explained by the large variation in soil properties as water content, N availability and decomposable organic matter as previously reported by Ambus & Christensen (1995), Pihlatie *et al.* (2005) and Silver *et al.* (2005). The

Fig. 6 Relationship between CH_4 fluxes and water table depth. Monthly averages and standard errors are shown for the water table (cm below the surface); monthly averages and propagated errors are shown for the CH_4 fluxes.

N concentration and mineralization rate in the soil influence the N_2O production/consumption, while N in groundwater influences the indirect N_2O emissions from the leached water (Reay *et al.*, 2009). Our plantation was not fertilized, and at the same time, N was continuously removed from the soil due to plant uptake and woody biomass harvest. This process probably led to a progressive N limitation. Most of the N was present in the leaves [(2.7% of leaf fresh weight, N content of 0.5% of the fresh weight of the woody biomass removed from the site) unpublished data from S. Vanbeveren] and therefore not removed from the site. The N dynamics of decaying litter is a complex process characterized by different phases: Litter is first leached, then N is immobilized and subsequently released from the leaves, after the maximum amount of N has been accumulated (Berg & McClaugherty, 1989; Berg & Laskowski, 2005). Litter decay is a slow process: During 8 months of field incubation at an Italian field site, poplar leaf litter only lost between 15% and 18% of its original mass (Cotrufo *et al.*, 2005). Low decay rates are characteristic for most woodlands, both for needle and for deciduous litter (Moro & Domingo, 2000). Additionally, the recent land-use change that characterized our site may have affected the capacity of the soil biological community to degrade a new substrate as poplar litter, slowing down the decomposition process and resulting in a rapid accumulation of litter to form a distinct litter layer on the top of the soil (Hoosbeek *et al.*, 2004; Cotrufo *et al.*, 2005). This process does not affect biomass production, and therefore, N fertilization is generally not recommended for SRC plantations. However, such conditions can lead to a temporary N limitation in the soil promoting sporadic N_2O negative fluxes. This is underlined by

Fig. 7 Component planes of the SOM. Left column: component planes for the N_2O fluxes ($\text{g N}_2\text{O m}^{-2} \text{s}^{-1}$), soil temperature (T_{soil} ; $^{\circ}\text{C}$) and ecosystem respiration (Reco $\mu\text{mol m}^{-2} \text{s}^{-1}$). Right column: component planes for CH_4 fluxes ($\text{mg CH}_4 \text{m}^{-2} \text{s}^{-1}$), soil water content (SWC ; %) and soil respiration ($\text{CO}_2 \mu\text{mol m}^{-2} \text{s}^{-1}$). Each cell on the self-organizing map corresponds to the same cohort of half-hourly observations.

the fact that negative fluxes are often confined to N-limited ecosystems (Chapuis-Lardy *et al.*, 2007; Saggard *et al.*, 2013; Schlesinger, 2013). Soil N_2O uptake was observed in our study, with increasingly negative fluxes after the second harvest, when ecosystem N availability would be the lowest (Fig. 3d). N_2O uptake has been observed in managed grasslands (Flechard *et al.*, 2005; Neftel *et al.*, 2010; Jones *et al.*, 2011). Overall, N_2O emissions here presented can be considered as background fluxes (i.e., $<5 \text{ nmol m}^{-2} \text{s}^{-1}$ as reported by Merbold *et al.*, 2014) due to the low management intensity of our plantation. Nevertheless, even background emissions of N_2O significantly contributed to offsetting the CO_2 uptake of the plantation and overturned the ecosystem GWI. They therefore need to be included in estimates of

Table 3 Temporal dynamics of the monthly GHG emissions: CH₄ (kg CH₄-C ha⁻¹); N₂O (kg N₂O-N ha⁻¹); CO₂ (g C m⁻²); ± standard deviation

	First rotation						Second rotation					
	2010			2011			2012			2013		
	CH ₄	N ₂ O	CO ₂	CH ₄	N ₂ O	CO ₂	CH ₄	N ₂ O	CO ₂	CH ₄	N ₂ O	CO ₂
Jan	n.a.	n.a.	n.a.	2.2 (±0.3)	0.03 (±0.07)	22.3 (±1.0)	1.82 (±0.5)	0.06 (±0.02)	30.4 (±1.2)	3.0 (±0.5)	0.11 (±0.07)	39.7 (±2.9)
Feb	n.a.	n.a.	n.a.	1.2 (±0.3)	0.06 (±0.01)	19.7 (±0.7)	1.4 (±0.4)	0.20 (±0.04)	34.6 (±2.0)	0.55 (±0.6)	0.04 (±0.04)	17.5 (±1.6)
Mar	n.a.	n.a.	n.a.	1.1 (±0.5)	0.08 (±0.08)	1.6 (±1.4)	0.76 (±0.6)	0.11 (±0.03)	46.5 (±1.33)	2.7 (±1.3)	0.002 (±0.03)	12.9 (±2.0)
Apr	n.a.	n.a.	n.a.	0.8 (±0.5)	0.123 (±0.12)	-41.4 (±1.8)	3.3 (±0.3)	0.24 (±0.04)	44.1 (±3.5)	2.5 (±0.5)	0.04 (±0.04)	-6.7 (±2.0)
May	n.a.	n.a.	n.a.	0.6 (±0.6)	0.15 (±0.04)	-74.2 (±1.9)	3.3 (±0.5)	0.33 (±0.06)	5.3 (±2.3)	3.6 (±0.9)	0.009 (±0.09)	-101.1 (±4.8)
Jun	2.8	n.a.	58.6 (±2.4)	0.4 (±0.3)	0.08 (±0.03)	-41.3 (±1.7)	2.2 (±0.4)	0.17 (±0.04)	-49.6 (±3.0)	1.6 (±0.8)	0.19 (±0.05)	-62.7 (±5.8)
Jul	2.4	n.a.	19.1 (±1.9)	1.2 (±0.3)	0.08 (±0.07)	-39.4 (±1.9)	3.1 (±0.6)	0.11 (±0.04)	-123.6 (±3.2)	4.5 (±4.7)	0.11 (±0.11)	-111.5 (±6.3)
Aug	1.2 (±0.4)	3.11 (±0.00)	-25.0 (±2.0)	0.7 (±0.2)	0.03 (±0.05)	-27.9 (±2.4)	4.0 (±0.6)	0.14 (±0.10)	-64.6 (±4.1)	2.9 (±1.5)	0.22 (±0.09)	-53.6 (±2.6)
Sep	2.5 (±0.5)	0.48 (±0.07)	-22.2 (±1.5)	0.8 (±0.2)	-0.03 (±0.04)	-45.2 (±2.1)	3.1 (±0.7)	0.22 (±0.08)	51.4 (±5.0)	1.2 (±1.7)	0.18 (±0.13)	-19.7 (±2.7)
Oct	3.2 (±0.4)	0.13 (±0.05)	4.9 (±1.2)	0.9 (±0.2)	0.02 (±0.05)	33.7 (±1.7)	2.8 (±0.6)	0.13 (±0.05)	71.2 (±4.4)	2.2 (±0.5)	-0.07 (±0.07)	-3.2 (±1.1)
Nov	3.3 (±0.4)	0.08 (±0.02)	19.6 (±1.0)	2.8 (±0.4)	0.143 (±0.05)	59.1 (±1.7)	2.8 (±0.4)	0.11 (±0.03)	65.1 (±1.2)	6.6 (±0.7)	-0.39 (±0.19)	2.4 (±1.0)
Dec	3.3 (±0.5)	0.03 (±0.03)	20.0 (±0.9)	1.3 (±0.7)	0.09 (±0.03)	37.3 (±1.0)	3.8 (±0.5)	0.15 (±0.03)	40.0 (±1.3)	7.3 (±0.4)	0.19 (±0.07)	11.2 (±1.1)

n.a. = not assessed.

the GHG balance of intensively managed terrestrial ecosystems.

The methane fluxes reported can also be considered as background fluxes, although the net CH₄ emission was surprising: Soils usually act as a net CH₄ sink (Robertson & Groffman, 2015). We attribute the CH₄ emission measured here to the relatively shallow water table depth. This is evidenced by the relationship between monthly CH₄ emission and water table depth (Fig. 7), by the associated anoxic niches in the soil as well as by the high diffusion rates. A second possible explanation for the CH₄ emissions could be the land-use history of the site. Prior to the SRC plantation, part of the site was intensively managed agricultural cropland, and this land use could cause a reduction in the CH₄ oxidation capacity of the soil. Soils with a history of intensive management lose their CH₄ oxidation ability and have very slow oxidation capacity recovery rates (Robertson *et al.*, 2000; Levine *et al.*, 2011).

Although SRC plantations are managed in a similar way as agricultural crops (e.g., soil preparation, weed control, irrigation, fertilization), the magnitude and the temporal dynamics of their GPP/R_{eco} are comparable to those of deciduous forests (Zenone *et al.*, 2015). Systematic changes in the length of the growing season (Randerson *et al.*, 1999; Keyser *et al.*, 2000) indicate an extension of the favorable period for assimilation. In an intensively managed crop – as the SRC of this study – the impact of management on the length of the growing season appears to be more important, as highlighted by the change in the length of the growing season after coppice.

The SOM is a well-established artificial neural network for the visualization of multi-dimensional data that has been applied to clustering problems and data exploration in several sectors as industry, finance, natural sciences and linguistics (Kohonen, 2013). A SOM projects nonlinear statistical relationships between multidimensional data into simple geometric relationships on a regular two-dimensional grid (Kohonen, 2001). These characteristics make the SOM an interesting alternative to some conventional statistical techniques because it is able to provide a visual representation of the multiple correlations among the variables investigated. This technique has been used to visualize the relationships between daily anomalies in NEE and environmental variables, underlining how annual anomalies are dominated by anomalies in gross primary production, which in turn were correlated with incident radiation and vapor pressure deficit (Luyssaert *et al.*, 2007). To the best of our knowledge, no previous studies have applied the SOM technique to investigate the relationships between soil trace gases and their environmental drivers. The lack of a significant relationship between the CH₄-N₂O emission fluxes and the environmental

Fig. 8 Monthly cumulative greenhouse gas (GHG) fluxes over the two rotations of the short-rotation coppice culture (2010–2013). The CO_2 is represented by the black line and the white circles; N_2O fluxes by the dashed line and white triangles; CH_4 fluxes by the dotted line and the black circles; cumulative GHGs are represented by the black solid line. Positive values denote a loss from (or flux out of) the ecosystem, and negative values denote an uptake (or flux toward the ecosystem). GHGs: greenhouse gases, N_2O : nitrous oxide, CH_4 : methane, CO_2 : carbon dioxide.

drivers (at 30-min time step) underlines the difficulties in creating reasonable gap-filling strategies for agroforestry ecosystems, such as SRC, which are characterized by low emission rates of soil trace gases and low management inputs. The best way to predict N_2O emissions appears to be a multiple regression that includes the level of N in the soil and/or the mineralization rate among the predictors. Nitrogen mineralization and nitrification rates combined with an indicator of soil anaerobicity (gas diffusivity or water content) explained 73–77% of the variation in N_2O fluxes of beech forests (Eickenscheidt & Brumme, 2013). In intensively managed agricultural sites, N fertilization rate represented the best single predictor of N_2O emissions (Scherbak *et al.*, 2014). After fertilizer application, N_2O emissions can be related to plant productivity (Merbold *et al.*, 2014). The prediction of the emissions at half-hourly time steps in unmanaged ecosystems remains challenging. At our SRC site, N_2O fluxes were characterized by an emission peak due to an extreme rain event (80 mm rainfall in 48 h) after a long dry period (Zona *et al.*, 2011) after the establishment of the plantation, followed by long periods of low and highly variable fluxes. Emission bursts of short duration, typically occurred after fertilizer application (Jones *et al.*, 2007; Scherbak *et al.*, 2014), associated soil tillage (Ruan & Robertson, 2013), thawing and rain events (Kroon *et al.*, 2007; Pihlatie *et al.*, 2010). Nitrous oxide was an important component of the balance of other biofuel production systems: Soil N_2O emissions contributed ~20% to the total GHG budget of a poplar SRC in southwest Michigan, USA (Gelfand *et al.*, 2013). In contrast to N_2O , methane usually

only makes a minor contribution to the balance of biofuel feedstocks (Gelfand *et al.*, 2013). In our SRC plantation, however, CH_4 contributed as much as N_2O to the net GHG balance due to the considerable emissions from the soil. The net ecosystem exchange of N_2O and CH_4 presented in this study is one of the very few cases where non- CO_2 GHG gases have been measured above an SRC plantation. Several studies showed a large uncertainty and temporal variability in N_2O and CH_4 fluxes measured by EC (e.g., Christensen *et al.*, 1996; Laville *et al.*, 1997; Scanlon & Kiely, 2003; Neftel *et al.*, 2007; Kroon *et al.*, 2007; Mammarella *et al.*, 2010). This variability was explained either by biogeochemical soil processes and/or by several systematic and random error sources in the EC measurements. In conclusion, the soil trace gas emissions of CH_4 and N_2O offset the CO_2 uptake by the poplar SRC and made the total GHG budget of the plantation (excluding the end use of the harvested biomass) positive. Ecosystem-level fluxes of N_2O were highly variable with a non-negligible proportion of N_2O uptake. On the other hand, ecosystem-level CH_4 fluxes were mostly positive with only a sporadic uptake by the soil. Further long-term investigations of soil trace gas fluxes should include continuous monitoring of the soil nitrogen content and the mineralization rate of the soil as well as the microbial community of the soil as primary drivers of the emissions.

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