Impact of extreme precipitation and water table change on $\text{N}_2\text{O}$ fluxes in a bio-energy poplar plantation

D. Zona$^1$, I. A. Janssens$^1$, M. S. Verlinden$^1$, L. S. Broeckx$^1$, J. Cools$^1$, B. Gioli$^2$, A. Zaldei$^2$, and R. Ceulemans$^1$

$^1$Department of Biology, Research Group of Plant and Vegetation Ecology, University of Antwerp, Belgium
$^2$National Research Council (CNR), Institute of Biometeorology, Firenze, Italy

Received: 18 February 2011 – Accepted: 22 February 2011 – Published: 2 March 2011

Correspondence to: D. Zona (donatella.zona@ua.ac.be)

Published by Copernicus Publications on behalf of the European Geosciences Union.
Abstract

A large fraction of the West European landscape is used for intensive agriculture. Several of these countries have very high nitrous oxide (N$_2$O) emissions, because of substantial use of fertilizers and high rates of atmospheric nitrogen deposition. N$_2$O production in soils is controlled by water-filled pore space (WFPS) and substrate availability (NO$_3$). Here we show that extreme precipitation (~80 mm rainfall in 48 h) after a long dry period, led to a week-long peak in N$_2$O emissions (up to about 2200 µg N$_2$O-N m$^{-2}$ h$^{-1}$). In the first four of these peak emission days, N$_2$O fluxes showed a pronounced diurnal pattern correlated to daytime increase in temperature and wind speed. It is possible that N$_2$O was transported through the transpiration stream of the poplar trees and emitted through the stomates. However, during the following three high emission days, N$_2$O emission was fairly stable with no pronounced diurnal trend, and was correlated with wind speed and WFPS (at 20 and 40 cm depth) but no longer with soil temperature. We hypothesized that wind speed facilitated N$_2$O emission from the soil to the atmosphere through a significant pressure-pumping. Successive rainfall events and similar WFPS after this first intense precipitation did not lead to N$_2$O emissions of the same magnitude. These findings suggest that climate change-induced modification in precipitation patterns may lead to high N$_2$O emission pulses from soil, such that sparser and more extreme rainfall events after longer dry periods could lead to peak N$_2$O emissions. The cumulative effects of more variable climate on annual N$_2$O emission are still largely uncertain and need further investigation.

1 Introduction

Nitrous oxide (N$_2$O) is one of the major greenhouse gases, with a global warming potential ~300 times higher than CO$_2$, and that plays also a role in the destruction of stratospheric ozone (Cicerone, 1989). Approximately two thirds of the atmospheric N$_2$O originates from the biogenic processes of nitrification and denitrification.
Davidson, 2010). As previously suggested (Kramer et al., 1999; Boeckx and Van Cleemput, 2001; Kroon et al., 2010a) N₂O emissions from agricultural soils are very important in European countries with a long history of fertilizer use. In addition to agricultural soils, forest soils in Western Europe could also present very high N₂O emissions (Pilegaard et al., 2006; Wu et al., 2010), caused by the high nitrogen deposition rates (Kristensen et al., 2004). The total emission of N₂O from forest soils in Germany (expressed as CO₂ equivalents) could reach up to 53% of the total emission of greenhouse gases (CO₂, CH₄, and N₂O) (Jungkunst et al., 2008). However, there are still large uncertainties on the estimates of N₂O fluxes from the biosphere (Houghton et al., 1996; Kroeze et al., 1999; Schindlbacher et al., 2004; Neftel et al., 2007).

The mechanisms responsible for N₂O emission and the environmental controls over its production are complex. For instance, increases in soil water content initially have a positive effect, but further increases have a negative effect on N₂O emissions; how soil water controls N₂O fluxes is not yet fully understood (Davidson et al., 2000; Jungkunst et al., 2008; Castellano et al., 2010). Dry, well-aerated soils favor the oxidative process of nitrification (with transformation of NH₄⁺ into NO₃⁻ and NO emission), wet soils favor NO₃⁻ and NO reduction, and N₂O emission, and finally extremely wet soils favor the complete N₂O reduction to N₂ by denitrifiers (Davidson, 1991; Davidson et al., 2000). A water-filled pore space (WFPS) of ~60% corresponds to the maximum N₂O emission (Davidson, 1991). However, N₂O emission is also strongly dependent on soil compaction, soil texture and bulk density (Ruser et al., 2006; Ball et al., 2008; Castellano et al., 2010). Water is a transport medium for NO₃⁻ and NH₄⁺, both substrates for the nitrifying and denitrifying microbes (Davidson et al., 2000). Soil aeration also affects gas diffusivity in the soil and increases N₂O release into the atmosphere (Davidson et al., 2000; Smith et al., 2003).

Previous studies highlighted the complexity in predicting N₂O fluxes from environmental variables, where similar soil water content could lead to very different emission rates (Schindlbacher et al., 2004; Castellano et al., 2010; Wu et al., 2010). Even long-term measurements (Hellebrand et al., 2003; Wagner-Riddle et al., 2007; Wu et
al., 2010) were not able to unequivocally explain the processes responsible for \( \text{N}_2\text{O} \) release. Moreover, the processes leading to \( \text{N}_2\text{O} \) consumption within the soil are still largely unknown (Chapuis-Lardy et al., 2007). Nitrous oxide uptake has been observed in different ecosystems, such as grasslands (Glatzel and Stahr, 2001; Neftel et al., 2007) and forests (Cavigelli and Robertson, 2001; Butterbach-Bahl et al., 2002), and it has been connected to anaerobic microbial denitrification (Zumft, 1997).

\( \text{N}_2\text{O} \) release mostly occurs in short peak emissions connected to fertilization and precipitation events (Wagner-Riddle et al., 2007; Eugster et al., 2007; Jungkunst et al., 2008; Neftel et al., 2010). The short-term nature of the \( \text{N}_2\text{O} \) release and the difficulties in modeling \( \text{N}_2\text{O} \) emission generate the need for continuous monitoring to estimate annual \( \text{N}_2\text{O} \) emission from ecosystems. Unfortunately most studies have been of discontinuous nature (from weekly to monthly) (Kavdir et al., 2007; Neftel et al., 2007; 2010; Mammarella et al., 2010) and/or involved the use of soil chambers (Hellebrand et al., 2003; Kroon et al., 2010b; Wu et al., 2010), with associated spatial scaling issues, and with resulting uncertainties in annual estimates of more than 50% (Flechard et al., 2007). Small spatial and discontinuous temporal resolution of chamber measurements prevent the accurate capture of some of these peak events-based \( \text{N}_2\text{O} \) release. Thus far, few studies have been performed at ecosystem scale with eddy covariance (Neftel et al., 2007; 2010; Eugster et al., 2007; Mammarella et al., 2010; Kroon et al., 2010a) or gradient techniques (Wager-Riddle et al., 2007). Neftel et al. (2007) reported that \( \text{N}_2\text{O} \) emission measured with eddy covariance exceeded that obtained by the chamber technique threefold.

Understanding the impact of climate and soil hydrology on \( \text{N}_2\text{O} \) emissions is particularly important, because the frequency and magnitude of drought and precipitation events are expected to increase with climate change (Kunkel et al., 2008). Future changes in rainfall patterns are predicted to increase \( \text{N}_2\text{O} \) emission by 45% even with reduced fertilizer application (Hsieh et al., 2005). As soil water content is believed to be among the most important controls on \( \text{N}_2\text{O} \) emission, altered precipitation patterns could significantly affect the emission of this greenhouse gas (Davidson et al., 1991;
The main objective of this study was to investigate the impact of soil hydrological changes (e.g. WFPS and water table change) on \( N_2O \) emission in a high-density bioenergy poplar plantation, recently converted from cropland and pasture. We hypothesized that increases in water table and WFPS connected to rain events lead to increases in \( N_2O \) emissions. We also hypothesized that increases in soil temperature stimulate \( N_2O \) production and thus increase \( N_2O \) emissions if adequate water is available in the soil.

2 Materials and methods

2.1 Site description

The research site is located in Lochristi, Belgium (51°06′44″ N, 3°51′02″ E), 11 km from the city of Ghent at an altitude of 6.25 m above sea level (Fig. 1). The long-term average annual temperature is 9.5 °C and the average total annual precipitation is 726 mm (Royal Meteorological Institute of Belgium). The soil has a sandy texture with a clay-enriched deeper soil layer. The soil C:N ratio (measured in February–March 2010) in the first 90 cm of the soils was on average 13.3 ± 1.4 (\( n = 110 \)) and the bulk density was \( \sim 1.482 ± 0.075 \) g cm\(^{-3} \). The soil pH was on average 5.51 ± 0.66 (\( n = 42 \)). Nitrogen deposition in northern Belgium (Flanders) is \( \sim 30–40 \) kg N ha\(^{-1} \) y\(^{-1} \) (Official report of the Flemish Environment Agency, Environmental Assessment Report).

A total of 18.4 ha were planted on 7–10 April 2010, with different poplar clones (belonging to the species \( Populus deltoides \), \( P. maximowiczii \), \( P. nigra \), and \( P. trichocarpa \) and interspecific hybrids) in a double-row planting scheme (with 0.75 m and 1.5 m in between rows; 1.1 m within the rows, and a planting density of 8000 plants ha\(^{-1} \)). The canopy height of the plantation (measured in front of the eddy covariance mast) increased from \( \sim 1.3 \) m on 2 August to 2.1 m on 29 September 2010. Ditches of \( \sim 80 \) cm depth were established by the previous land users around the field to drain excess...
water. These ditches were draining water into deeper canals (1.5 m depth) at the outer edges of the field site. As a consequence the soil surface was mostly dry and drainage of standing water was fairly rapid.

The previous land uses were pasture and cropland (ryegrass, wheat, potatoes, beets, and most recently monoculture corn with regular fertilization, 200–300 kg N ha\(^{-1}\) y\(^{-1}\) liquid animal manure and chemical fertilizers). Before establishment of the plantation the agricultural land was ploughed in March 2010, to 40–70 cm depth. Several herbicide treatments were applied between the end of March and the beginning of April 2010, and between 25 June and 7 July 2010. Mechanical weeding was conducted, both manually and by tilling, from June until the end of August 2010. No fertilization or irrigation were applied during this experiment.

The site is surrounded by intensively managed croplands (mostly monoculture corn, and potatoes) with regular fertilizer application (170–250 kg N ha\(^{-1}\) y\(^{-1}\); K. Mouton, personal communication, 2010; S. Overloop, Flemish Environment Agency, personal communication, 2010). The N concentration in the water in the ditch around the field (measured on 29 October 2010) was on average 2.37 ± 0.005 mg N l\(^{-1}\) (the sum of NO\(_3^-\)-N and NO\(_2^-\)-N) and 0.31 ± 0.0416 mg N l\(^{-1}\) (NH\(_4^+\)-N).

### 2.2 Environmental variables

A complete set of meteorological variables were recorded continuously from the beginning of June 2010 to the present day. Soil water content was measured at different depths (0–30 cm, 0–20 cm, 0–10 cm in different locations), and across a vertical transect (at 1 m, 60 cm, 40 cm, 30 cm, and 20 cm) in the proximity of the eddy covariance mast using 8 Time Domain Reflectometry (TDR, model CS616 Campbell Scientific, Logan, UT, USA) moisture probes. Soil water content was then converted to water-filled pore space (WFPS) according to Wu et al. (2010). Soil temperature was recorded by temperature probes which provided the average temperature of a soil layer of 8 cm
depth (model TCAV-L averaging thermocouples, Campbell Scientific, Logan, UT, USA). These probes parallel four type-E thermocouples together into one, 24-gauge wire, and were inserted in proximity of each of the soil water content sensors. Surface temperature was recorded using an Apogee infrared sensor (Apogee Instruments, Inc., Logan UT, USA) pointing into the footprint of the eddy covariance mast at an angle of \( \sim 45^\circ \) with the ground (with field of view \( \sim 22^\circ \), height above the surface of 4.8 m, corresponding to a footprint of \( \sim 7.7 \text{ m}^2 \)). Air temperature and relative humidity was recorded both on the eddy covariance mast using Vaisala probe (model HMP45C, Vaisala, Helsinki, Finland) at a height of 5.4 m above the ground surface. The air temperature profile was also measured at a meteorological tower at three different heights (50 cm, 1 m, and 2 m above the ground) with type-T thermocouples. Air pressure was measured with an electronic barometer (model PTB 101B, Vaisala, Helsinki, Finland). Incoming photosynthetically active radiation (PAR) (400–700 nm) was recorded above the canopy using quantum sensors (Li-190, Li-COR, NE, USA). Net radiation (0.2 to 100 \( \mu \text{m} \)) was recorded using a net radiometer (NR Lite Kipp & Zonen, Delft, The Netherlands). The incoming and reflected shortwave solar (0.3 to 3 \( \mu \text{m} \)) and longwave (far infrared 4.5 to 42 \( \mu \text{m} \)) radiation were collected using two pyranometers and two pyrgeometers (model CNR1, Kipp & Zonen, Delft, The Netherlands). Diffuse radiation was monitored with a shadow-band pyranometer (model LP PYRA 02, Delta Ohm, Padova, Italy). The PAR sensor, the net radiometer, the pyranometers and pyrgeometers, and the diffuse radiation sensors were all mounted on the meteorological tower (where also sensors for measuring air temperature profile were installed) at 2 m above the surface and at \( \sim 10 \text{ m} \) distance from the eddy covariance mast. Eight heat flux plates (HFT3, REBS Inc., Seattle, WA, USA) were installed in the soil at 6–8 cm depth. Precipitation was recorded using a tipping bucket rain gauge (model 3665R, Spectrum Technologies Inc., Plainfield, IL, USA) installed on top of the cabin where the gas analyzers were installed. Water table was recorded with a pressure transducer (model PDCR1830, Campbell Scientific, Logan, UT, USA) installed in a pipe inserted into the ground to a 1.85 m depth. All instruments were connected to two data loggers (model CR5000...
and model CR1000, Campbell Scientific, Logan, UT, USA) and each environmental variable was read once every 0.1–10 s and the 30 min averages are output to a PC.

2.3 Eddy covariance measurements

An eddy covariance mast was installed at the beginning of June 2010 and it was continuously been operated to the present day. The eddy covariance mast was positioned in the northeast part of the plantation (Fig. 1) including areas with both previous land use types (cropland and pasture). The eddy covariance mast included a sonic anemometer for the measurement of the three-dimensional wind components, wind speed, wind direction, and the energy fluxes (Model CSAT3, Campbell Scientific, Logan, UT, USA), and several fast-response analyzers, among them a closed-path Los Gatos N\textsubscript{2}O/CO\textsubscript{2} analyzer (model 908-0014, Los Gatos Research, Mountain View, LGR, CA, USA) and a closed path CO\textsubscript{2}/H\textsubscript{2}O infrared analyzer (LI-7000, LI-COR, Lincoln, NE, USA). The sonic anemometer and the inlet of the sampling lines were positioned at 5.8 m above the surface. The mast location was chosen according to the prevalent wind direction (from southeast, Fig. 1), to maximize the footprint of the tower. The sonic anemometer was oriented to 175\degree from true north. The large majority of wind directions were between 198\degree and 252\degree from north (Fig. 1).

The Los Gatos N\textsubscript{2}O analyzer employs a cavity enhanced laser absorption technique in which an optical cavity is used as the measurement cell. This allows for a longer optical pathlength (400 ± 10 m) compared to conventional laser absorption techniques, resulting in increased sensitivity. The analyzer utilizes a room temperature mid-infrared quantum cascade laser and detector at a specific narrow band (4.6 \textmu m). The internal pressure of the optical cell is fixed at 10 kPa. The analyzer has a 1s 1\sigma precision of 0.3 ppbv for both N\textsubscript{2}O and CO. A scroll pump (model XDS-35i, Edwards, MA, USA) was used to draw air through the N\textsubscript{2}O analyzer. A two-meter long vacuum tubing was used to dampen the air flow and pressure in the air stream. The flow rate in the sampling line of the N\textsubscript{2}O analyzer was \sim25 \text{l min}^{-1}.
The N\textsubscript{2}O analyzer was calibrated at the LGR Company on 6 July 2010 using a NOAA primary standard at 322.24 ppbv N\textsubscript{2}O in air (uncertainty less than 0.1 ppbv). The linearity of the analyzer was then tested by diluting a higher concentration bottle (440 ppbv N\textsubscript{2}O) by known amounts and measuring the analyzer response. This dilution test proved that the accuracy of the instrument was better than 1% over the range of 40–440 ppbv N\textsubscript{2}O (R. Provencal, Los Gatos Research, personal communication, 2010). We calibrated the N\textsubscript{2}O analyzer again on 31 August 2010 with 733 ppbv (ultra high purity ≥99.997 vol% with 10% accuracy, limited by dilution system).

The LI-7000 (LI-COR, Lincoln, NE, USA) was used to measure CO\textsubscript{2} and H\textsubscript{2}O fluxes. A vacuum pump was positioned at the outlet of the LI-7000 analyzer, generating a flow of \(\sim 22 \text{l min}^{-1}\). Two buffer volumes of 0.5 l each respectively were positioned between the pump and the outlet of the analyzer to dump the fluctuations of the pump. Here we describe only calibration procedure for the H\textsubscript{2}O fluxes as they were used to correct the N\textsubscript{2}O fluxes (see following sections). CO\textsubscript{2} fluxes are presented and discussed elsewhere. The H\textsubscript{2}O vapor was calibrated every week using ultra-high purity nitrogen for the zero, and a dew point generator (LI-610, LI-COR, Lincoln, NE, USA) to produce an air stream with a known water vapor dew point (typically 7°C lower than the ambient air temperature) for the H\textsubscript{2}O span.

Fluxes of H\textsubscript{2}O, N\textsubscript{2}O, and momentum were measured using eddy covariance, a micrometeorological method that quantifies the net exchange of a scalar between the biosphere and the atmosphere (Swinbank, 1951; Desjardins and Lemon, 1974; Baldocchi, 2003).

Teflon tubing (~15 m long and 8 mm inner diameter) was used for two separate sampling lines for the LI-7000 and for the N\textsubscript{2}O analyzer. The two inlets were positioned 10 cm from the center of the sonic anemometer. A 1 µm teflon filter (Gelman) was used at the inlet of the sampling line of the LI-7000 analyzer. A stainless steel Swagelok™ filter (60 µm pore size SS-4FW4-60) was positioned at the inlet to protect the sampling line of the N\textsubscript{2}O analyzer. Another stainless steel Swagelok™ filter (2 µm pore size, SS-4FW4-2) was also present at the input of the sampling line to prevent dust from
entering the sample cell.

The H$_2$O, N$_2$O fluxes, and sonic wind components were recorded at 10 Hz using a data logger (model CR 5000, Campbell Scientific, Logan, Utah, USA). All the analyzers, the data loggers, and the PC were positioned inside a wooden cabin maintained at a stable temperature (21°C).

### 2.4 Post-processing of the eddy covariance data

Fluxes of N$_2$O, H$_2$O, sensible heat, and momentum were calculated using the EdiRe software (version 1.4.3.1169, R. Clement, University of Edinburgh, UK; http://www.geos.ed.ac.uk/abs/research/micromet/EdiRe/) and averaged over 30 min.

A two-components rotation was applied to set mean vertical ($\bar{w}$) and lateral ($\bar{v}$) velocity components to zero. Time delays (on average 1.6 s for N$_2$O and 1.8 s for H$_2$O) were calculated using a cross-correlation function of the scalar fluctuation and the vertical wind velocity. A frequency response correction was applied to the eddy covariance fluxes following Moore (1986) and using theoretical attenuation functions and Kaimal model spectra to account for high frequency and low frequency fluctuations in signal losses (Kaimal et al., 1972). We also applied a correction for density change (WPL) according to Webb et al. (1980). We only applied the water vapor term of the WPL correction as we assume the long tube attenuated the temperature fluctuation. It should be noted that N$_2$O and H$_2$O fluxes were measured by different instruments (N$_2$O fluxes with the Los Gatos analyzer and H$_2$O fluxes with the Li-7000) with separate lines but similar flow rates.

Data quality was assessed by analysis of energy budget closure and by comparison of co-spectra of $\bar{w}'T's'$, $\bar{w}'H_2O'$, $\bar{w}'N_2O'$ (Kaimal et al., 1972). Obvious data outliers were removed, which were values more than 30 standard deviations from the 30 min mean for H$_2$O vapor, N$_2$O and for the wind velocity components, $u$, $v$, and $w$. The remaining N$_2$O fluxes were filtered according to the following procedures: when N$_2$O concentration was < 0 (due to instrumental failure), in correspondence of error of the
sonic anemometer (reported by the diagnostics of the CSAT-3D), when failing the sta-
tionarity test with a threshold of 30% as suggested by Foken and Wichura (1996). A
footprint model was applied to the data (Klijun et al., 2004) indicating that the 90% of
the fluxes were coming from the first ∼200 m upwind of the eddy mast. Data with wind
direction between 285° and 135° (from the north, back, and from the right of the tower)
were removed.

Only N₂O fluxes are being presented in this manuscript, but CO₂ fluxes were used
to derive \( u^* \) (defined as \( \sqrt{\langle u'w' \rangle} \)) threshold, then applied to the N₂O fluxes. CO₂ fluxes
for a solar radiation < 10 Wm⁻² were regressed with \( u^* \) and a \( u^* \) threshold was set to
0.15 m s⁻¹ (data not shown). H₂O fluxes were used for the WPL correction of the N₂O
fluxes. As no standard procedure exists for gap-filling of the N₂O fluxes, we did not gap-
fill them. The cumulative N₂O emission during 19–25 August 2010, was estimated from
the daily average of the available data. The temperature sensitivity of N₂O emission
(\( R \)) was investigated by fitting the following equation to the data:

\[
R = BR \times Q_{10}^{((\text{soil } T - 10)/10)}
\]

Where \( BR \) is the basal respiration and \( Q_{10} \) describes the response of respiration to
temperature (soil \( T \)) increase.

2.5 Statistical analyses

General linear modeling (GLM) was used to identify the most important predictors of
N₂O fluxes (Systat version 13, Systat Software Inc., 2002, Chicago, IL, USA)). A single
variable and a forward stepwise multiple regression approach were used to discrimi-
nate among and rank the most important variables (surface temperature, soil tempera-
ture at 0–8, 20, 30, 40, and 60 cm, WFPS at 0–10, 20, 30, 40, and 60 cm, water table
depth, wind speed, \( u^* \)) in explaining the variability in N₂O fluxes. Models were applied
to the instantaneous N₂O fluxes during the peak emission days (for 19–25 August, and
separately for 19–22 August and 23–25 August 2010).
3 Results

3.1 Environmental conditions

Atmospheric conditions at the experimental field site during August 2010 were exceptional. From 11–14 August 2010, a depression over the North Sea brought maritime air over the European continent. From 15 to 17 August 2010, the depression moved toward Germany causing very active rain zones over Belgium (Royal Meteorological Institute of Belgium). At the field site the total measured rainfall on 16–17 August was 81 mm. The normal monthly total precipitation is 75 mm (at Ukkel, ∼50 km from Lochristi, Royal Meteorological Institute of Belgium), less than what we measured in only 48 h. During the second decade of August the total precipitation reported for Ukkel was 111 mm (the long-term average for that decade is 25 mm), which was the highest value since 1901 (the second largest occurred in 1951 and was 71 mm).

Over the entire month of August 2010 the total precipitation at our site was 185 mm (187 mm at Ukkel, classified as an “exceptional event” by the Royal Meteorological Institute of Belgium, a denomination used for events that occur once every 30 yr). The record high total monthly precipitation at Ukkel was measured in 1996 (231 mm).

This extreme precipitation event led to a steep increase in water table and WFPS (Fig. 2). Prior to the precipitation event the water table was at ∼136 cm below the surface and it was below 80 cm for the entire summer season (Fig. 2). The heavy rain on 16–17 August caused flooding of the field site (in several locations there was standing water) and overflowing of the ditches. The weekly total precipitation from 20 June to 16 August 2010 was on average $13 ± 11$ mm while from 16 August to 3 October it was on average $37 ± 27$ mm, not allowing the shallower soil layers (0–10 cm) to become drier than ∼60% WFPS after 17 August (Fig. 2).
3.2 N$_2$O fluxes

The spectral analysis showed that the co-spectra of $w'Ts'$ and of $w'N_2O'$ presented a reasonable comparison, demonstrating the good performance of the instruments (Fig. 3). The co-spectra of $w'N_2O'$ showed a slight loss at the high frequencies (typical for closed path analyzers) (Fig. 3). The energy budget closure for the presented data averaged 85%.

During the days immediately following the large rainfall event and the steep increase in water table and WFPS on 16–17 August, a steep increase in N$_2$O emission from the plantation was observed (Fig. 4). This large N$_2$O emission started on 19 August when the water table and WFPS progressively decreased (Fig. 4 and Fig. 5). From 19 to 22 August the N$_2$O emission presented a pronounced diurnal trend following the daytime increase in soil temperature (Fig. 5), and wind speed (or $u^*$) (Fig. 6). From 23 to 25 August when the wind speed was generally $> 2 \text{ m s}^{-1}$ (and the $u^*$ was mostly $> 0.3 \text{ m s}^{-1}$) N$_2$O emissions did not present a diurnal pattern any more (Fig. 6).

To identify the most important environmental variables controlling N$_2$O emission during these peak release days, a general linear model was applied to the half hourly averaged N$_2$O fluxes. The results of the single variable model for 19–25 August, and for 19–22 and 23–25 August separately, are shown in Table 1–3. If the entire peak N$_2$O emission period was modeled together (19–25 August), the best single variable model included surface temperature that explained 48% of the variability in N$_2$O fluxes (Table 1). A multi-variable model that included surface $T$, $u^*$, and WFPS (at 60 cm) presented a slightly higher explanatory power of the N$_2$O fluxes ($R^2 = 51\%$, F-ratio 74, $p < 0.001$).

Similar results were found for the N$_2$O fluxes from 19 to 22 August (Table 2). During this period surface temperature and the shallow soil temperature (0–8 cm) explained 56% and 54% of the N$_2$O fluxes, respectively (Table 2). At this time wind speed and $u^*$ were also important but presented lower explanatory power (29% and 33% respectively) than temperature (Table 2). The relation between N$_2$O emission (from
19–22 August) and soil temperature (0–8 cm) was exponential and exhibited a $Q_{10}$ of 3 (Fig. 7). The multi-variable model that presented the highest explanatory power of the N$_2$O fluxes from 19–22 August included surface temperature, soil $T$ (60 cm depth), and wind speed, and it was able to explain 68% of the variability in N$_2$O emissions (F-ratio 67, $p < 0.001$).

N$_2$O emissions between 23 and 25 August did not present a diurnal trend (Fig. 5). During 23–25 August, soil and surface temperature were no longer significant predictor of N$_2$O fluxes (Table 3). During these days, wind speed and $u^*$ were the variables with the highest explanatory power of N$_2$O fluxes and they explained 38% and 42% of the variability in N$_2$O fluxes, respectively, (Table 3). A multi-variable model that included $u^*$, WFPS (20 cm), and WFPS (40 cm) was able to explain 79% of the variability in N$_2$O fluxes (F-ratio 148, $p < 0.001$). Interestingly, we noticed that the WFPS at intermediate depth in the soil profile (20 cm) was sometimes lower than in the shallower and deeper layers. The water content in the shallower layers increased due to the mist and light rainfall (a clear example is shown on 7 July, Fig. 2 when WFPS at 0–10 cm increased right after a small rainfall, even while WFPS at 20 cm increased later only after a larger rainfall event).

The low turbulence at night ($u^* < 0.1$ m s$^{-1}$) and moderately turbulent conditions during daytime ($u^* \sim 0.5$ m s$^{-1}$), during the first four days (19–22 August), led to N$_2$O concentration increases at night, ranging from $\sim$325 ppb to $\sim$340 ppb over a few hours period (Fig. 6). The last three days (23–25 August) exhibited higher turbulent conditions, with $u^*$ spanning from $\sim$0.3 m s$^{-1}$ at night to $\sim$0.8 m s$^{-1}$ during daytime, and presented a lower variability of N$_2$O concentration with no marked diurnal cycle (Fig. 6).

The daily total N$_2$O-N emission from 19–22 August was fairly stable, on average $0.26 \pm 0.01$ (SD) kg N$_2$O-N ha$^{-1}$, while the total daily emission from 23–25 August was also fairly stable (e.g. on average $0.13 \pm 0.014$ kg N$_2$O-N ha$^{-1}$). We also estimated the approximate N present in the soil water using the average sum of NO$_3^-$-N, NO$_2^-$-N (their sum was $\sim$90% of the total inorganic nitrogen) and NH$_4^+$-N in the water from the ditch around the site and the average soil water content in the different soil layers on 19–
25 August 2010. According to this calculation the total nitrogen present in the soil water was 9.7 kg N ha$^{-1}$. The total N emitted as N$_2$O from 19 to 25 August ($\sim$1.44 kg N$_2$O-N ha$^{-1}$) thus represented 15% of this soil water N content.

Rain events that occurred after 25 August 2010 led to similar increases and decreases in water table (and WFPS), but did not lead to N$_2$O emissions of the same magnitude of the one observed on 19–25 August (Fig. 4). Overall, N$_2$O fluxes before and after the peak emissions of 19–25 August, were mostly close to zero.

4 Discussion

The emission of N$_2$O differed dramatically between the week following the first extreme rain event and the rest of the study period. The low N$_2$O emission observed before the large rainfall on 16–17 August, could be related to the fact that under normal conditions well aerated sandy-loam soils are unlikely to develop the large number of anaerobic micro-sites necessary for N$_2$O production by denitrification (Skiba et al., 1993).

In contrast, a first extreme rain event induced production and release of substantial amount of N$_2$O. The maximum N$_2$O emission observed after the large rain fall in this study was several orders of magnitude higher than what is usually observed (Pilegaard et al., 2006; Davidson et al., 2000; Schaufler et al., 2010). Our maximum emission of about 2200 µg N$_2$O-N m$^{-2}$ h$^{-1}$ was comparable to reported peak emissions (Jungkunst et al., 2008; Kroon et al., 2009; Kroon et al., 2010a; Wu et al., 2010). Some of the highest N$_2$O peak emissions reported were recorded in a managed fen meadow (with a maximum of about 3200 µg N$_2$O-N m$^{-2}$ h$^{-1}$ in The Netherlands, Kroon et al., 2009), in spruce forests in Germany (a maximum of about 800 µg N$_2$O-N m$^{-2}$ h$^{-1}$, Wu et al., 2010, and almost 3000 µg N$_2$O-N m$^{-2}$ h$^{-1}$, Jungkunst et al., 2008), and in a bio-energy poplar plantation (in Germany, up to 900 µg N$_2$O-N m$^{-2}$ h$^{-1}$, Hellebrand et al., 2003). The average daily emissions during 19–25 August resulted in a cumulative N$_2$O-N
loss of 1.44 kg N$_2$O-N ha$^{-1}$ ($\sim$10% of the emission from agricultural soils in European countries, estimated to be $\sim$15 kg N$_2$O-N ha$^{-1}$ y$^{-1}$; Boeckx and Van Cleemput, 2001).

Peak N$_2$O emission with re-wetting of dry soils has been observed in several ecosystems (Sexstone et al., 1985; Wagner-Riddle et al., 1996; Hsieh et al., 2005; Wagner-Riddle et al., 2007). The large release in N$_2$O emissions observed on 19–25 August may have been connected to multiple mechanisms. The flooding of the land could have transported NH$_4^+$ and NO$_3^-$ from the ditches or from surrounding agricultural fields to the plantation at a rate that exceeded the uptake of plants and microorganisms, leading to significant rates of denitrification and N$_2$O emission. It is likely that the extreme rain event probably also caused the reactivation of water-stressed bacteria following the dry period, which decomposed and mineralized the labile organic matter fraction, suddenly available in the soil (Birch, 1964). Additionally, the prolonged drier conditions before 16 August could have led to death of the microbial population in the shallower soil layers and the release of nitrogen in the soil, emitted as N$_2$O once the intense rain event suddenly increased moisture availability.

The observed lag between the rain event (16–17 August) and N$_2$O emission (19–25 August) was probably related to the rate of water infiltration through the soil profile (Fig. 5). The sustained high N$_2$O emission that we observed for a week was accompanied by the drop in the water table from the surface until about 60 cm below the surface from 16 to 23 August 2010 (Fig. 4). We observed the highest N$_2$O emission when the soil profile became less anoxic (e.g. WFPS 0–10 cm between 60–72%, Fig. 5) preventing the complete reduction of N$_2$O into N$_2$ (Davidson, 1991), but leaving sufficient anaerobic micro-sites available for denitrification (Rolston et al., 1982; Sexstone et al., 1985). Deeper soil layers presented a stable and higher WFPS (80%, Fig. 5) where probably N$_2$ production was dominant instead (Davidson, 1991; Davidson et al., 2000).

While N$_2$O fluxes from 19–22 August presented a pronounced diurnal trend with increased emission during daytime, the following days (23–25 August) presented stable emissions and no diurnal trend. This could be related to the interaction of different processes responsible for the N$_2$O emission. The observed diurnal pattern in N$_2$O
emission between 19–22 August could be related to the N\textsubscript{2}O release through poplar leaves. N\textsubscript{2}O could be transported by the transpiration stream and emitted to the atmosphere (Chang et al., 1998; McBain et al., 2004). Poplar seedlings emit N\textsubscript{2}O when extremely high soil N\textsubscript{2}O concentration was applied to the root zone (Chang et al., 1998; McBain et al., 2004). The dependence of N\textsubscript{2}O emission on temperature on the 19–22 August could suggest the dependence of N\textsubscript{2}O release from daytime increase in stomatal conductance and transpiration (Chang et al., 1998; McBain et al., 2004). The increase in transpiration with increasing wind speed, with abundant water in the soil and high stomatal conductance (Campbell and Norman, 1998), could explain the dependence of N\textsubscript{2}O fluxes from wind speed at this time.

As N\textsubscript{2}O emission from poplar leaves has been observed only under extremely high soil N\textsubscript{2}O concentration (McBain et al., 2004), it was probably connected to the observed nighttime decrease in wind speed (and $u^*$) and increase in N\textsubscript{2}O concentration (Fig. 6). This decrease in turbulence probably led to high N\textsubscript{2}O concentration in the soil during 19–22 August. On the other hand, the high diffusion rates (Chang et al., 1998) and pressure pumping between 23–25 August prevented an N\textsubscript{2}O concentration increase (Fig. 6), and probably an increase in concentration in the soil profile, thus reducing the importance of N\textsubscript{2}O emission through leaves.

From 23–25 August, soil temperature lost its importance in explaining N\textsubscript{2}O fluxes and the main environmental variables controlling N\textsubscript{2}O release were wind speed (or $u^*$) in combination with moisture content in deeper soil layers (WFPS at 20 and 40 cm depth). At this time the main mechanism of N\textsubscript{2}O emission was probably mass flow through the soil layers, not transpiration through the poplar leaves anymore. The wind pumping effect (Gu et al., 2005) probably pushed N\textsubscript{2}O from deeper soil layers (where temperature was more stable, not presenting a diurnal trend) into the atmosphere, thus reducing the residence and the travelling time of N\textsubscript{2}O in and from deeper soil profiles, and preventing its reduction to N\textsubscript{2}. The occurrence of a more aerobic layer between 20 and 40 cm depth into the soil may have been either the site of production or storage of N\textsubscript{2}O, that was released once the wind speed (or $u^*$) increased (Fig. 5).
The $Q_{10}$ values reported for $\text{N}_2\text{O}$ emission in laboratory incubations of sandy-loamy soils span a very wide range, 1.9–8.9 (Maag and Vinther, 1996), 1.4–5.2 (Vicca et al., 2009), 12.4 (Vinther, 1992), and up to 23 (Christensen, 1983). Possible mechanisms behind these very high $Q_{10}$ values include the increase in size of the existing anaerobic micro-sites and the generation of new ones (Dowdell and Smith, 1974), connected to the increased respiratory oxygen consumption with increasing temperature (Tiedje et al., 1984). On the other hand, high $Q_{10}$ could be related to the confounding effect of changes in microbial population size and/or substrate availability (Davidson et al., 2006). As the $R^2$ of the $Q_{10}$ function in this experiment from 19–22 August was only 51% and could not explain the release between 23–25 August (see Fig. 7) we believe that several complex mechanisms were responsible for the $\text{N}_2\text{O}$ emission. The stomatal transport would provide a possible mechanism for this release in the first peak emission days while increased mass flow with higher wind speed probably explained the high emission on 23–25 August.

Successive rain events after 25 August and associated fluctuation of water table and WFPS which were of same magnitude as those on 16–25 August, did not lead to peak $\text{N}_2\text{O}$ emissions (Fig. 4). $\text{N}_2\text{O}$ fluxes after the 25 August were very low, and did not respond to temperature increase, water table, or WFPS fluctuations. The lack of large $\text{N}_2\text{O}$ emission events after the ones of 19–25 August would indicate that the large soil nitrogen pool was probably completely used (either emitted as $\text{N}_2\text{O}$, immobilized by the recovering microbial population, or taken up by the vegetation) or leached to some other location. Moreover, surrounding agricultural fields were no longer fertilized after mid-August. This observed lack of response after successive rainfall events was previously explained as nitrate or carbon limitation (Sexstone et al., 1985; Wagner-Riddle et al., 1996), as a very specific combination of water content and nutrient availability is necessary to produce denitrification peak fluxes (Grundmann et al., 1988). Not only the observed extreme rainfall event is important but the rainfall pattern over extended period is extremely important in influencing $\text{N}_2\text{O}$ emissions. This result is confirmed by the higher $\text{N}_2\text{O}$ emission with the same total rainfall, but a longer dry period observed...
in previous studies (Rolston et al., 1982; Smith and Patrick, 1983). The intense precipitation event after a long dry period could also have been responsible for higher NO$_3^-$ leaching than the one occurring under lower and more frequent precipitation events (Rolston et al., 1982).

5 Conclusions

Intense precipitation events after extended dry periods could have a large impact on N$_2$O emission; weekly or monthly monitoring schemes of N$_2$O fluxes could largely underestimate these emissions. In this study we showed that water table and soil water content could affect N$_2$O fluxes, but that an increase in soil water content does not necessarily lead to peak N$_2$O emissions, leaving large uncertainties on the controls of N$_2$O fluxes. Notably, the use of eddy covariance allowed capturing the effect of pressure pumping and increase in turbulence on N$_2$O emissions. Overall the results presented here explore multiple important mechanisms responsible for peak N$_2$O emission. The first peak emission days presented a diurnal increase in N$_2$O emission which suggested that N$_2$O was transported through the transpiration stream of the poplar trees and emitted through their stomates. However, during the last days of high emission, N$_2$O loss was fairly stable with no pronounced diurnal trend. Overall, wind speed, and increasing gas flow through the soil, played a major role on the N$_2$O emission. These results confirm the complexity in modeling N$_2$O emission and the need for continuous larger-scale studies.

Acknowledgements. The research leading to these results has received funding from a Marie Curie Reintegration grant (PIRG07-GA-2010-268257) and from the European Research Council ERC grant agreement nr. 233366 (POPFULL) under the European Community’s Seventh Framework Programme (FP7/2007-2013). We would like to thank Robert Provencal and Douglas Bear from Los Gatos Research for assistance with the N$_2$O analyzer. We thank Kristof Mouton for logistic support at and management of the field site, Franco Miglietta and Piero Toscano for field assistance, Robert Clements, George Burba and Gerardo Fratini for help with...
the eddy covariance data analysis and insight in the data processing, the Royal Meteorological Institute of Belgium for providing climate data, Frans Fierens and the ECMWF (www.ecmwf.int) for the boundary layer data, Ann Cools and Tom Van der Spiet for the water sample analysis, Toon De Groote for help with the meteorological data analysis, and Sara Vicca for help with the $Q_{10}$ analysis, John King for the revision of the manuscript.

References

Arah, J. R. M. and Vinten, A. J. A.: Simplified models of anoxia and denitrification in aggregated and simple-structured soils, Eur. J. Soil Sci., 46, 507–517, 1995.

Baldocchi, D.: Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future, Glob. Change Biol., 9, 479–492, 2003.

Ball, B. C., Crichton, I., and Horgan, G. W.: Dynamics of upward and downward $N_2O$ and $CO_2$ fluxes in ploughed or no-tilled soils in relation to water-filled pore space, compaction and crop presence, Soil Till. Res., 101, 20–30, 2008.

Birch, H. F.: Mineralisation of plant nitrogen following alternate wet and dry conditions, Plant Soil, 20, 43–49, 1964.

Boeckx, P. and Van Cleemput, O.: Estimates of $N_2O$ and $CH_4$ fluxes from agricultural land in various regions of Europe, Nutr. Cycl. Agroecosys., 60, 35–47, 2001.

Birch, H. F.: Mineralisation of plant nitrogen following alternate wet and dry conditions, Plant Soil, 20, 43–49, 1964.

Butterbach-Bahl, K., Breuer, L., Gasche, R., Willibald, G., and Papen, H.: Exchange of trace gases between soils and the atmosphere in Scots pine forest ecosystems of the northeastern German lowlands: 1. Fluxes of $N_2O$, $NO/NO_2$ and $CH_4$ at forest sites with different N-deposition, Forest Ecol. Manag., 167, 123–134, 2002.

Campbell, G. S. and Norman, J. M.: An Introduction to Environmental Biophysics, Springer-Verlag, New York, 1998.

Castellano, M. J., Schmidt, J. P., Kaye, J. P., Walker, C., Graham, C. B., Lin, H., and Dell, C. J.: Hydrological and biogeochemical controls on the timing and magnitude of nitrous oxide flux across an agricultural landscape, Glob. Change Biol., 16, 2711–2720, 2010.

Cavigelli, M. A. and Robertson, G. P.: Role of denitrifier diversity in rates of nitrous oxide consumption in a terrestrial ecosystem, Soil Biol. Biochem., 33, 297–310, 2001.
Chang, C., Janzen, H. H., Cho, C. M., and Nakonechny, E. M.: Nitrous oxide emission through plants, Soil Sci. Soc. Am. J., 62, 35–38, 1998.

Chapuis-Lardy, L., Wrage, N., Metay, A., Chotte, J. L., and Bernoux, M.: Soils, a sink for N₂O?, a review, Glob. Change Biol., 13, 1–17, 2007.

Christensen, S.: Nitrous oxide emission from a soil under permanent grass: Seasonal and diurnal fluctuations as influenced by manuring and fertilization, Soil Biol. Biochem., 15, 531–536, 1983.

Davidson, E. A.: Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems, in: Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halomethanes, edited by: Rogers, J. E. and Whitman, W. B., American Society for Microbiology, Washington, DC, 219–236, 1991.

Davidson, E. A., Keller, M., Erickson, H. E., Verchot, L.V., and Veldkamp, E.: Testing a conceptual model of soil emissions of nitrous and nitric oxides, BioScience, 50, 667–680, 2000.

Davidson, E. A., Janssens, I. A., and Luo, Y. Q.: On the variability of respiration in terrestrial ecosystems: moving beyond Q_{10}, Glob. Change Biol., 12, 154–164, 2006.

Desjardins, R. L. and Lemon, E. R.: Limitations of an eddy-correlation technique for the determination of the carbon dioxide and sensible heat fluxes, Bound.-Lay. Meteorol., 5, 475–488, 1974.

Dowdell, R. J. and Smith K. A.: Field studies of the soil atmosphere ii. occurrence of nitrous oxide, J. Soil Sci., 25, 231–238, 1974.

Eugster, W., Zeyer, K., Zeeman, M., Michna, P., Zingg, A., Buchmann, N., and Emmenegger, L.: Methodical study of nitrous oxide eddy covariance measurements using quantum cascade laser spectrometry over a Swiss forest, Biogeosciences, 4, 927–939, doi:10.5194/bg-4-927-2007, 2007.

Flechard, C. R., Ambus, P., Skiba, U., Rees, R. M., Hensen, A., Van Amstel, A., Van den Polvan Dasselaar, A., Soussana, J.-F., Jones, M., Clifton-Brown, J., Raschi, A., Horvath, L., Neftel, A., Jocher, M., Ammann, C., Leifeld, J., Fuhrer, J., Calanca, P., Thalman, E., Pilegaard, K., Di Marco, C., Campbell, C., Nemitz, E., Hargreaves, K. J., Levy, P.E., Ball, B. C., Jones, S. K., Van de Bulk, W. C. M., Groot, T., Blom, M., Domingues, R., Kasper, G., Allard, V., Ceschia, E., Cellier, P., Laville, P., Henault, C., Bizouard, F., Abdalla, M., Williams, M., Baronti, S., Berretti, F., and Grosz, B.: Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe, Agr. Ecosyst. Environ., 121, 135–152, 2007.
Foken, T. and Wichura B.: Tools for quality assessment of surface-based flux measurements, Agric. Forest Meteorol., 78, 83–105, 1996.

Glatzel, S. and Stahr, K.: Methane and nitrous oxide exchange in differently fertilized grassland in southern Germany, Plant Soil, 231, 21–35, 2001.

Groffman, P., Butterbach-Bahl, K., Fulweiler, R., Gold, A., Morse, J., Stander, E., Tague, C., Tonitto, C., and Vidon, P.: Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models, Biogeochemistry, 93, 49–77, 2009.

Grundmann, G. L., Rolston, D. E., and Kachanoski R. G.: Field soil properties influencing the variability of denitrification gas fluxes, Soil Sci. Soc. Am. J., 52, 1351–1355, 1988.

Gu, L. H., Falge, E. M., Boden, T. A., Baldocchi, D. D., Black, T. A., Saleska, S. R., Suni, T., Verma, S. B., Vesala, T., Wofsy, S. C., and Xu, L. K.: Objective threshold determination for nighttime eddy flux filtering, Agric. Forest Meteorol., 128, 179–197, 2005.

Hellebrand, H. J., Kern, J., and Scholz, V.: Long-term studies on greenhouse gas fluxes during cultivation of energy crops on sandy soils, Atmos. Environ., 37(12), 1635–1644, 2003.

Houghton, J. T., Meira Filho, L. G., Callander, B. A., Harris, N., Kattenberg, A., and Maskell, K.: Climate Change 1995; The Science of Climate Change, Cambridge University Press, New York, 1996.

Hsieh, C. I., Leahy, P., Kiely, G., and Li, C.: The effect of future climate perturbations on N₂O emissions from a fertilized humid grassland, Nutr. Cycl. Agroecosys., 73, 15–23, doi:10.1007/s10705-005-7129-4, 2005.

Cicerone, R.: Analysis of sources and sinks of atmospheric nitrous oxide, J. Geophys. Res., 94, 18265–18271, 1989.

Jungkunst, H. F., Flessa, H., Scherber, C., and Fiedler, S.: Groundwater level controls CO₂, N₂O and CH₄ fluxes of three different hydromorphic soil types of a temperate forest ecosystem, Soil Biol. Biochem., 40, 2047–2054, 2008.

Kaimal, J. C., Wyngard, J. C., Izumi, Y., and Cote, O. R.: Spectral characteristics of surface-layer turbulence, Q. J. Roy. Meteor. Soc., 98, 563–589, 1972.

Kavdir, Y., Hellebrand, H. J., and Kern, J.: Seasonal variations of nitrous oxide emission in relation to nitrogen fertilization and energy crop types in sandy soil, Soil Till. Res., 98, 175–186, 2008.

Kljun, N., Calanca, P., Rotach, M. W., and Schmid, H. P.: A simple parameterisation for flux footprint predictions, Bound.-Lay. Meteorol., 112, 503–523, 2004.
Kroene, C., Mosier, A., and Bouwman, L.: Closing the global N\textsubscript{2}O budget: A retrospective analysis 1500–1994, Global Biogeochem. Cy., 13, 1–8, 1999.

Kroon, P., Hensen, A., van den Bulk, W. C. M., Jongejan, P., and Vermeulen, A.: The importance of reducing the systematic error due to non-linearity in N\textsubscript{2}O flux measurements by static chambers, Nutr. Cycl. Agroecosyst., 83, 97–98, 2009.

Kroon, P. S., Schrier-Uijl, A. P., Hensen, A., Veenendaal, E. M., and Jonker, H. J. J.: Annual balances of CH\textsubscript{4} and N\textsubscript{2}O from a managed fen meadow using eddy covariance flux measurements, Eur. J. Soil Sci., 61, 773–784, 2010a.

Kroon, P. S., Hensen, A., Jonker, H. J. J., Ouwersloot, H. G., Vermeulen, A. T., and Bosveld, F. C.: Uncertainties in eddy covariance flux measurements assessed from CH\textsubscript{4} and N\textsubscript{2}O observations, Agric. Forest Meteorol., 150, 806–816, 2010b.

Kunkel, K. E., Bromirski, P., Brooks, H., Cavazos, T., Douglas, A. V., Easterling, D. R., Emanuel, K. A., Holland, G. J., Knutson, T. R., Kossin, J. P., Komar, P. D., Levinson, D. H., and Smith, R. L.: Observed changes in weather and climate extremes, in: Weather and Climate Extremes in a Changing Climate, Regions of Focus: North America, Hawaii, Caribbean, and US Pacific Islands, edited by: Karl, T. R., Meehl G. A., Hassol S. J., Murray W. L., and Waple A. M., A Report by the US Climate Change Science Program and Subcommittee on Global Change Research, Washington, DC, 2008.

Mammarella, I., Werle, P., Pihlatie, M., Eugster, W., Haapanala, S., Kiese, R., Markkanen, T., Rannik, Ü., and Vesala, T.: A case study of eddy covariance flux of N\textsubscript{2}O measured within forest ecosystems: quality control and flux error analysis, Biogeosciences, 7, 427–440, doi:10.5194/bg-7-427-2010, 2010.

Maag, M. and Vinther, F. P.: Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil water content contents and temperatures, Appl. Soil Ecol., 4, 5–14, 1996.

McBain, M. C., Warland, J. S., McBride, R. A., and Wagner-Riddle, C.: Laboratory-scale measurements of N\textsubscript{2}O and CH\textsubscript{4} emissions from hybrid poplars (Populus deltoides X Populus nigra), Waste Manag. Res., 22, 454–465, 2004.
Impact of extreme precipitation and water table change on N$_2$O fluxes

D. Zona et al.

McClain, M. E., Boyer, E. W., Dent, C. L. Gergel, S. E., Grimm, N. B., Groffman, P. M., Hart, S. C., Harvey, J. W., Johnston, C. A., Mayorga, E., McDowell, W. H., and Pinay, G.: Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems, Ecosystems, 6, 301–312, 2003.

Moore, C. J.: Frequency response corrections for eddy correlation system, Bound.-Lay. Meteorol., 37, 17–35, 1986.

Neftel, A., Flechard, C., Ammann, C., Conen, F., Emmenegger, L., and Zeyer, K.: Experimental assessment of N$_2$O background fluxes in grassland systems, Tellus B, 59, 470–482, 2007.

Neftel, A., Ammann, C., Fischer, C., Spirig, C., Conen, F., Emmenegger, L., Tuzson, B., and Wahlen, S.: N$_2$O exchange over managed grassland: Application of a quantum cascade laser spectrometer for micrometeorological flux measurements, Agric. Forest Meteorol., 150, 775–785, 2010.

Pilegaard, K., Skiba, U., Ambus, P., Beier, C., Brüggemann, N., Butterbach-Bahl, K., Dick, J., Dorsey, J., Duyzer, J., Gallagher, M., Gasche, R., Horvath, L., Kitzler, B., Leip, A., Pihlatie, M. K., Rosenkranz, P., Seufert, G., Vesala, T., Westrate, H., and Zechmeister-Boltenstern, S.: Factors controlling regional differences in forest soil emission of nitrogen oxides (NO and N$_2$O), Biogeosciences, 3, 651–661, doi:10.5194/bg-3-651-2006, 2006.

Rappoldt, C. and Crawford, J. W.: The distribution of anoxic volume in a fractal model of soil, Geoderma, 88, 329–347, 1999.

Rolston, D. E., Sharples, A. N., Toy, D. W., and Broadbent, F. E.: Field Measurement of denitrification: III. Rates during irrigation cycles, Soil Sci. Soc. Am. J., 46, 289–296, 1982.

Ruser, R., Flessa, H., Russow, R, Schmidt, G., Buegger, F., and Munch, J. C.: Emission of N$_2$O, N$_2$ and CO$_2$ from soil fertilized with nitrate: effect of compaction, soil water content and rewetting, Soil Biol. Biochem., 38, 263–274, 2006.

Schauffler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton, M. A., and Zechmeister-Boltenstern, S.: Greenhouse gas emissions from European soils under different land use: effects of soil water content and temperature, Eur. J. Soil Sci., 61, 683–696, doi:10.1111/j.1365-2389.2010.01277.x, 2010.

Schindlbacher, A., Zechmeister-Boltenstern, S., and Butterbach-Bahl, K.: Effects of soil water content and temperature on NO, NO$_2$, and N$_2$O emissions from European forest soils, J. Geophys. Res., 109, D17302, doi:10.1029/2004JD004590, 2004.

Sexstone, A. J., Parkin, T. B., and Tiedje, J. M.: Temporal response of soil denitrification rates to rainfall and irrigation, Soil Sci. Soc. Am. J., 49, 99–103, 1985.
Skiba, U., Smith, K. A., and Fowler, D.: Nitrification and denitrification as sources of nitric oxide and nitrous oxide in a sandy loam soil, Soil Biol. Biochem., 25, 1527–1536, 1993.

Smith, C. J. and Patrick Jr., W. H.: Nitrous oxide emission as affected by alternate anaerobic and aerobic conditions from soil suspensions enriched with ammonium sulfate, Soil Biol. Biochem., 15, 693–697, 1983.

Smith, K. A, Ball, T., Conen, F., Dobbie, K. E., Massheder, J., and Ray, A.: Exchange of greenhouse gases between soil and atmosphere: Interactions of soil physical factors and biological processes, European J. Soil Sci., 54, 779–791, 2003.

Swinbank, W. C.: The measurement of vertical transfer of heat and water vapor by eddies in the lower atmosphere, J. Meteorol., 72, 135–132, 1951.

Tiedje, J. M., Sexstone, A. J., Parkin, T. B., Revsbech, N. P., and Shelton, D. R.: Anaerobic processes in soil, Plant Soil, 76, 197–212, 1984.

van Groenigen, J. W., Zwart, K. B., Harris, D., and van Kessel, C.: Vertical gradients of $\delta^{15}N$ and $\delta^{18}O$ in soil atmospheric $N_2O$ – temporal dynamics in a sandy soil, Rapid Commun. Mass Sp., 19, 1289–1295, 2005.

Verchot, L. V., Davidson, E. A., Cattânio, J. H., Ackerman, I. L., Erickson, H. E., and Keller, M.: Land-use change and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia, Global Biogeochem. Cy., 13, 31–46, 1999.

Vicca, S., Janssens, I. A., Flessa, H., Fiedler, S., and Jungkunst, H. F.: Temperature dependence of greenhouse gas emissions from three hydromorphic soils at different groundwater levels, Geobiology, 7, 465–476, 2009.

Vinther, F. P.: Measured and simulated denitrification activity in a cropped sandy and loamy soil, Biol. Fert. Soils, 14, 43–48, 1992.

Wagner-Riddle, C., Thurtell, G. W., King, K. M., Kidd, G. E., and Beauchamp, E. G.: Nitrous oxide and carbon dioxide fluxes from a bare soil using a micrometeorological approach, J. Environ. Qual., 25, 898–907, 1996.

Wagner-Riddle, C., Furon, A., McLaughlin, N., Lee, I., Barbeau, J., Jayasundara, S., Parkin, G., Von Bertoldi, P., and Warland, J.: Intensive measurements of nitrous oxide emissions from a corn-soybean-wheat rotation under two contrasting management systems over 5 years, Glob. Chang Biol., 13, 1722–1736, 2007.

Webb, E. K., Pearman, G. I., and Leuning, R.: Correction of flux measurements for density effects due to heat and water-vapor transfer, Q. J. Roy. Meteor. Soc., 106, 85–100, 1980.

Wu, X., Brüggemann, N., Gasche, R., Shen, Z., Wolf, B., and Butterbach-Bahl, K.: Environ-
mental controls over soil-atmosphere exchange of N$_2$O, NO, and CO$_2$ in a temperate Norway spruce forest, Global Biogeochem. Cy., 24, GB2012, doi:10.1029/2009GB003616, 2010. Zumft, W. G.: Cell biology and molecular basis of denitrification, Microbiology and Molecular Biology Review, 61, 533–616, 1997.
Table 1. Single variable model (GLM) for the \( \text{N}_2\text{O} \) fluxes (19–25 August 2010), displayed are \( R^2 \), F-ratio and p-values; in bold the single environmental variable with the highest explanatory power of the \( \text{N}_2\text{O} \) fluxes. Notice that during this period surface temperature explains most of the \( \text{N}_2\text{O} \) fluxes.

| variable          | \( R^2 \) | F-ratio | p-value |
|-------------------|-----------|---------|---------|
| \( \text{N}_2\text{O} \) fluxes |           |         |         |
| surface \( T \)   | **0.48**  | **178** | < **0.001** |
| soil \( T \) (0–8 cm) | 0.37     | 127     | < 0.001 |
| soil \( T \) (20 cm) | 0.27     | 76      | < 0.001 |
| soil \( T \) (30 cm) | 0.06     | 14      | < 0.001 |
| soil \( T \) (40 cm) | 0        | 1       | 0.395   |
| soil \( T \) (60 cm) | 0.025    | 6       | 0.019   |
| WFPS (0–10 cm)    | 0         | 0.58    | 0.45    |
| WFPS (20 cm)      | 0         | 0       | 0.96    |
| WFPS (30 cm)      | 0         | 0.02    | 0.88    |
| WFPS (40 cm)      | 0         | 1       | 0.31    |
| WFPS (60 cm)      | 0.27      | 83      | < 0.001 |
| water table       | 0         | 0.023   | 0.88    |
| wind speed        | 0         | 0.3     | 0.62    |
| \( u^* \)         | 0         | 0.7     | 0.42    |
Table 2. Single variable model (GLM) for the N₂O fluxes (19–22 August 2010), displayed are $R^2$, F-ratio and p-values; in bold the single environmental variable with the highest explanatory power of the N₂O fluxes. Notice that during this period surface temperature explains most of the N₂O fluxes.

| variable      | $R^2$ | F-ratio | p-value |
|---------------|-------|---------|---------|
| N₂O fluxes    |       |         |         |
| surface $T$   | 0.56  | 113     | < 0.001 |
| soil $T$ (0–8 cm) | 0.54  | 113     | < 0.001 |
| soil $T$ (20 cm) | 0.45  | 80      | < 0.001 |
| soil $T$ (30 cm) | 0.26  | 33      | < 0.001 |
| soil $T$ (40 cm) | 0.17  | 19      | < 0.001 |
| soil $T$ (60 cm) | 0.11  | 12      | < 0.001 |
| WFPS (0–10 cm) | 0.31  | 42      | < 0.001 |
| WFPS (20 cm)  | 0.31  | 45      | < 0.001 |
| WFPS (30 cm)  | 0.3   | 41      | < 0.001 |
| WFPS (40 cm)  | 0.21  | 18      | < 0.001 |
| WFPS (60 cm)  | 0.14  | 15      | < 0.001 |
| water table   | 0.29  | 41      | < 0.001 |
| wind speed    | 0.29  | 40      | < 0.001 |
| $u^*$         | 0.33  | 48      | < 0.001 |
Table 3. Single variable model (GLM) for the N$_2$O fluxes (23–25 August 2010), displayed are $R^2$, F-ratio and p-values; in bold the single environmental variable with the highest explanatory power of the N$_2$O fluxes. Notice * during this period $u^*$ explains most of the N$_2$O fluxes.

| variable         | $R^2$ | F-ratio | p-value |
|------------------|-------|---------|---------|
| N$_2$O fluxes    |       |         |         |
| surface $T$      | 0.0   | 0       | 0.93    |
| soil $T$ (0–8 cm)| 0.02  | 2       | 0.13    |
| soil $T$ (20 cm) | 0.0   | 1       | 0.3     |
| soil $T$ (30 cm) | 0.02  | 3       | 0.086   |
| soil $T$ (40 cm) | 0.0   | 2       | 0.16    |
| soil $T$ (60 cm) | 0.035 | 4       | 0.039   |
| WFPS (0–10 cm)   | 0.054 | 7       | 0.011   |
| WFPS (20 cm)     | 0.026 | 3       | 0.078   |
| WFPS (30 cm)     | 0.0   | 0       | 0.89    |
| WFPS (40 cm)     | 0.12  | 16      | < 0.001 |
| WFPS (60 cm)     | 0.0   | 2       | 0.21    |
| water table      | 0.38  | 73      | < 0.001 |
| wind speed       |       |         |         |
| $u^*$            | 0.42  | 86      | < 0.001 |
Fig. 1. Topographic map of the study site (Lochristi, East of Flanders, Belgium) and wind rose for the entire period of measurements (4 August–30 September 2010). The position of the eddy covariance mast is also indicated.
Fig. 2. Soil temperature at 1 m and 0–8 cm depth (a), total daily precipitation (b), water table (c), water-filled pore space (WFPS) at 0–10 cm, 20 cm, 40 cm, and 60 cm depth in the soil during the period 1 June to 1 October 2010. Precipitation data were only available from 18 June 2010 onward (notice dashed line in panel (b)).
Fig. 3. Cospectra normalized by frequency (n) and covariance (cov) of the fluctuation in vertical wind velocity and the fluctuation in sonic temperature \(w' T'\) and \(N_2O \ (w' N_2O')\) averaged for 19–25 August 2010 (in logarithmic scale); \(f\) is the frequency in Hz, \(z\) is the measurement height (5.8 m) and \(\bar{u}\) is the mean wind speed in each corresponding half hour (m s\(^{-1}\)).
Fig. 4. Height of the water table (a) and N$_2$O fluxes (b) for the entire measurement period. Notice the large N$_2$O emission from 19 to 25 August 2010 after the water table increased to the surface and then decreased below 20 cm.
Fig. 5. \( \text{N}_2\text{O} \) fluxes superimposed to WFPS (at 0–10 cm, 20 cm, and 60 cm) (a), and to soil temperature (0–8 cm and 60 cm) (b), right before and during the peak emission period (16–25 August 2010).
Fig. 6. Wind speed and $u^*$ (a), N$_2$O concentration (b), and N$_2$O fluxes (c), during the peak emission days (19–25 August 2010).
Fig. 7. \(Q_{10}\) dependence of \(N_2O\) fluxes for the different peak emission periods (19–22 August and 23–25 August 2010). The fitted curve, \(R^2\) and p-value displayed refer to the period 19–22 August, as \(N_2O\) fluxes from 23–25 August 2010 were not correlated with soil temperature. BR indicates basal respiration.