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Impact of 40 years poplar cultivation on soil carbon stocks and greenhouse gas fluxes

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Abstract

Within the JRC Kyoto Experiment in the Regional Park and UN-Biosphere Reserve "Parco Ticino" (North-Italy, near Pavia), the soil carbon stocks and fluxes of CO_2, N_2O, and CH_4 were measured in a poplar plantation in comparison with a natural meso-hygrophilous deciduous forest nearby, which represents the pristine land cover of the area. Soil fluxes were measured using the static and dynamic closed chamber techniques for CH_4, N_2O, and CO_2, respectively. We made further a pedological study to relate the spatial variability found with soil parameters.

Annual emission fluxes of N_2O and CO_2 and deposition fluxes of CH_4 were calculated for the year 2003 for the poplar plantation and compared to those measured at the natural forest site. N_2O emissions at the poplar plantation were 0.15±0.1 g N_2O m^{-2} y^{-1} and the difference to the emissions at the natural forest of 0.07±0.06 g N_2O m^{-2} y^{-1} are partly due to a period of high emissions after the flooding of the site at the end of 2002. CH_4 consumption at the natural forest was twice as large as at the poplar plantation. In comparison to the relict forest, carbon stocks in the soil under the poplar plantation were depleted by 61% of surface (10 cm) carbon and by 25% down the profile under tillage (45 cm). Soil respiration rates were not significant different at both sites with 1608±1053 and 2200±791 g CO_2 m^{-2} y^{-1} at the poplar plantation and natural forest, respectively, indicating that soil organic carbon is much more stable in the natural forest. In terms of the greenhouse gas budget, the non-CO_2 gases contributed minor to the overall soil balance with only 0.9% (N_2O) and −0.3% (CH_4) of CO_2-eq emissions in the natural forest, and 2.7% (N_2O) and −0.2% of CO_2-eq. emissions in the poplar plantation.

The very high spatial variability of soil fluxes within the two sites was related to the morphology of the floodplain area, which was formed by the historic course of the Ticino river and led to a small-scale (tenth of meters) variability in soil texture and to small-scale differences in elevation. Differences of site conditions are reflected by differences of inundation patterns, ecosystem productivity, CO_2 and N_2O emission rates, and soil

contents of carbon and nitrogen. Additional variability was observed during a flooding event and after fertilisation at the poplar site. Despite of this variability, the two sites are comparable as both originate from alluvial deposits.

The study shows that changes in soil carbon stocks and related fertility are the most visible phenomena after 40 years of land use change from a pristine forest to a fast growing poplar plantation. Therefore, the conservation and careful management of existing carbon stocks deserves highest priority in the context of the Kyoto Protocol.

1. Introduction

Efforts to combat Climate Change has high priority on the agenda of the European Union, and synergies with other policies, such as the sustainable development strategy are among the main preconditions of achieving an environmentally integer development path at minimum cost (Commission of the European Communities, 2005). Minimizing terrestrial carbon losses will be an important pillar in this context. Stimulating measures to preserve or enhance terrestrial C stocks would retard the increase in atmospheric CO_2 and may be therefore gain some time during which other measures can take effect (Janssens, 2003). Trade-off's between the sinks and sources of biogenic greenhouse gases from different land use types has become a major issue in the context of the Kyoto Protocol and the Marrakech Accords.

According to a recent paper of Janssens et al. (2005), forests and grasslands are a net sink, whereas croplands are a major source of carbon in all European countries. Hence, the overall terrestrial carbon balance of a country is mainly controlled by its mix of land uses; land use changes and changes in management of lands can easily change the terrestrial carbon balance of a country to turn from a sink into a source and vice versa. In a comparison of annual CO_2 exchanges in different forest and agricultural land uses in Germany, Anthoni et al. (2004) observed large sinks for a managed and a unmanaged beech forests, a young spruce plantation was near carbon neutral due to high respiration; the agricultural area, when taking into account harvest,

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was a small or large carbon source, depending on crop type and the management of crop residues and organic amendments. For agricultural soils, various management options to increase carbon stocks have been proposed (Freibauer et al., 2004; Smith, 2004) including the use of no-till systems, increased applications of animal manure, and the incorporation of crop residues into the soil. However, a modelling exercise comparing the implementation of these techniques in the United States, Germany, and China with baseline simulations has shown that for all options a significant reduction or even inversion of the gains in terms of CO_2-equivalents might be caused by increased nitrous oxide (N_2O) emissions (Li et al., 2005). Likewise, studies in grasslands soils showed those carbon uptake rates are partly offset by emissions of CH_4 and N_2O (Soussana et al., 2004). Only few studies, however, has yet been undertaken on forest or agroforestry sites (Albrecht and Kandji, 2003).

Most efforts for the contribution of forest ecosystems to greenhouse gas emission reductions are focusing on the enhancement of their carbon sink strength; the effects of typical land-use changes on the overall balance of GHG are not yet properly considered. The JRC Kyoto experiment aims at filling this gap for a series of the most relevant land uses in the Lombardy Region in Italy. Starting from a pristine floodplain forest, it compares different land uses (e.g. high stand poplar plantation, short rotation poplar plantation for bioenergy use, rice and maize cultivation) with regard to the stocks of biomass and carbon, and with regard to fluxes of CO_2, N_2O, CH_4. We present here results of the measurements of CO_2, N_2O and CH_4 fluxes from soils performed in 2002/2003 in a poplar plantation in comparison with the natural mesohygrophilous deciduous forest nearby, which represents the pristine land cover of the area. Based on a pedological characterization, we will highlight on a few aspects that are relevant to understand the complexity of processes leading to high variability of the greenhouse gas fluxes, both spatially and temporally. Finally, we will highlight some clear trends showing the impact of the transformation of a pristine forest into an agroforestry land-use that occurred some 40 years ago.

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2. Materials and methods

2.1. Study site

The study area is located about 10 km north-west of the city of Pavia within the "Parco Regionale del Ticino", Italy. It is represented by an intensive poplar plantation and a relict of a pristine floodplain forest, the "Bosco Siro Negri" at about 2 km distance (see Fig. 1). The poplar site is part of the Carboeurope network with the flux tower situated in the middle of a 120 ha even-aged plantation (Clone I-214 Polulus x Euroamerican Populus hybrida). It is part of a larger area of poplar plantations "Bosco di Mezzanone" that was covered by forests of the Bosco Siro Negri type until 40 years ago. Most part of this area is now under 3–4 generations of poplar with turnover times of 9–12 years. After each logging of poplar, the residues and stumps are removed to allow ploughing 40–50 cm deep and insertion of 4–5 m long shoots for 150–200 cm into the ground. Poplar management includes regular fertilisation and removal of ground vegetation, irrigation is normally applied on demand.

Both natural forest and poplar plantation (in the following referred to as NF and PP site, respectively) are within the boundary of a river arm and its embankment that protects the area outside (mainly rice fields) from inundations by the Ticino river (see Fig. 1). The 14 years old poplar stand was harvested in spring 2005. The relict of pristine forest is about 200 years old and was not managed for at least 70 years. It is made up of some 15 tree species with dominant Common Oak and co-dominant Elm, Black Alder, White Poplar, White Willow and Locust. The canopy of 40 m height is not suited for eddy flux measurements due to its heterogeneity and limited extension of 12 ha. The climate of the site is characterised by the long term meteorological station at Pavia as temperate continental with average rainfall of 984 mm and temperature of 12.7°C. Precipitation in 2003 was significantly lower (545 mm) than the long-term average, with strong precipitation events in November (165 mm) and December (96 mm). During 2002, instead, precipitation was over average (1020 mm) with peak event of

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135 mm in May and 228 mm in November.

2.2. Flux measurements

At both sites, measurements of soil respiration were performed along two transects (see Fig. 1). In the poplar stand, the transect pointed to the NW and SW directions starting from the eddy flux tower base. Based on detailed mapping of soils and of tree height and diameter at breast height (DBH), the transects were selected to follow the gradient of growth and site conditions from the area around the tower (trees up to 28 m high, BHD up to 40 cm, dominant sandy texture) to the end points of the transects with trees not higher than 15 m, BHD 25 cm, dominant coarse texture. Each transect was made by 5 measuring points (collars) spaced at 20 m one from each other. In the natural forest, the two perpendicular transects, forming an L-shape, were installed in order to sample the large variability of the forest with a total of 15 measuring points. N_2O and CH_4 flux measurements were done at the end points of the two transects (four chambers) and using only one of the transects in the natural forest as a preliminary study showed that it represented the whole range of conditions encountered. The analysis presented in this paper is limited to those CO_2 respiration plots which were used also for N_2O and CH_4 flux measurements, and where the pedological study was undertaken. As the numbering of the plots is done sequentially we therefore present results from plot #1 and #5 (endpoint of the SW pointing transect with plot #1 being close to the tower) and #6 and #10 (endpoint of the NW pointing transect with plot #6 being close to the tower) in the poplar plantation and with plots #19, #16, #22 and #24 laying along the second transect in the natural forest.

2.2.1. N_2O and CH_4 soil fluxes

For the measurement of N_2O and CH_4 soil fluxes we used cylindric chambers of 40 cm diameter which were covered by a Perspex lid and laterally by Teflon foil to ensure realistic light conditions within the chambers. Even though the microbial processes, which are responsible for production and consumption of N_2O, are not directly affected by light (e.g. Horrigan et al., 1981), this was regarded important in view of the effect of light on the respiratory activity of soil microorganism, which may have secondary effects on N_2O and CH_4 fluxes. Mixing of the air within the chambers was ensured by large slowly rotating impellers. Turbulent flow conditions were established in the chambers with four baffles inside the chamber walls.

For sampling, the chambers were tightly set on permanently installed stainless steel frames. As the flux rates observed were in most cases low, we left the chambers closed for 1.5 h taking a gas sample immediately after closing of the chamber, and after 30, 60, and 90 min. Unless we observed a zero flux rate, only the measurements with linearity of R^2 better than 0.65 were used. With this criterion, we allowed a higher degree of non-linearity than used for example by Czepiel et al. (1996), Zongliang et al. (1994) or Hellmann (1993) under usually conditions of higher flux rates.

Sampling was done by using previously flushed and evacuated glass vials connected to the chamber air with the help of a needle inserted through a rubber stopper in the chamber lid. The quality of the vacuum of the vials is the most critical analytical step and care has to be taken to ensure reproducible quality of the vacuum (Loftfield et al., 1997). We used a automated procedure based on the system developed by Leip (2000). Leip (2000) carried out detailed tests on the uncertainty introduced by the quality of the vacuum as a function of evacuation procedure and quality of the rubber stopper (CHROMATOCOL) and found an upper limit of 3%, being in most cases ≤1.5%. In this study, we used a conservative estimate of 3%.

Analysis of the gas samples was done with a Shimadzu GC 14B gas chromatograph provided with a 6-port valve to interface with the injection system and a Porapak Q column (2 m×1/8″×2 mm SS, 80/100 mesh, CHROMPACK). Gases were detected with an ECD (N_2O) and a FID (CH_4) (both Shimadzu), mounted in series. The gas chromatographic peaks were evaluated with the software Chrom-Card (CE Instruments, UK). Calibration was done by means of a standard mixture of N_2O and CH_4 in nitrogen, which has been calibrated with a standard mixture (SCOTT SPECIALITY GASES) using tunable diode laser absorption spectroscopy (Restivo, 1999).

For injection of the gas samples a self-built automatic system for automatic injection of up to 163 gas samples, temperature and pressure close to the sample loop was measured (HONEYWELL) prior to transfer of the sample on the gas chromatographic column. As the system is tight, this information was used to calculate the mass of gas sampled in the vial. During the closure time of a flux chamber, changes in temperature or changes of air humidity lead to an additional, “artificial”, change in the mixing ratio of the gas; measured changes in the observed mixing ratio must be corrected for this effect (e.g. Schmidt et al., 1988). By ensuring that equilibrium between the conditions in the chamber and vial were established during field sampling, we were able to correct for this effect using the information obtained during the analyses (for details see Leip, 2000).

2.2.2. CO_2 soil fluxes

CO_2 soil fluxes were measured with a portable infrared gas analyser equipped with a chamber based on the principle of closed dynamic systems. The EGM-3 (PP-Systems, Hitchin, UK) is a portable infrared gas analyser connected to the SRC-1 CO_2 soil fluxes chamber and to the STP-1 sensor for soil temperature (both from PP-Systems, Hitchin, UK). The chamber is cylindrical with a diameter of 10 cm, base area of 78.5 cm^2, and a volume of 1170 cm^3. A small fan in the chamber ensures air mixing; negative effects to the soil-to-air boundary layer are prevented by a flow-breaking stainless-steel grid (cell size approx. 4 mm^2) installed below the fan. The permanently installed collars were custom-made from stainless-steel (inner diameter 10 cm; height 6.5 cm) with four rods of 20 cm length and 3 mm diameter helping to fix the collar tidily into the soil. The factory-made chamber was slightly modified and an o-ring was added to the base in order to improve its sealing with the collars.

The collars were installed at the beginning of the experiment and measurements started 15 to 30 days after collars installation. After the insertion, the collars remained in place for all the duration of the experiment. CO_2 soil fluxes were measured the same day at both stands every 15 to 30 days, depending on the time of the year. At each sampling point, a single measurement was performed and, for each stand, the measurements from all collars were averaged to give soil flux rate. Time by time, variability of CO_2 soil flux was checked by performing three subsequent measurements over single collars resulting in very limited changes (within 3 to 5% of the average value).

2.3. Soil characterization

2.3.1. Soil profile

A total of eight soil profiles of about 60 cm depth were opened near the plots for N_2O flux measurements for soil description and sampling. Soil bulk density was determined on dry (105°C, 24 h) undisturbed core samples of 100 cm^3. Soil moisture content was determined gravimetrically for each plot on a bulk sample of three soil cores of 5 cm depth and converted to water filled pore space (WFPS).

2.3.2. Soil chemical analysis

For chemical analyses, soil samples were air-dried and sieved (2 mm mesh). Soil pH was determined potentiometrically in a soil-to-solution (water and KCl 1N) ratio of 1:2.5 (Jackson, 1958). Organic carbon was measured using the method of Walkley and Black (1934) and total N was determined colorimetrically following Kjeldhal digestion (Kjeldahl, 1883). Further we measured soil texture (method of sedimentation), cation exchange capacity (Cecconi and Polesello, 1956) and base saturation. On organic samples (Ol and Of horizons) we measured pH (water), organic carbon, total nitrogen and C/N.

2.4. Ancillary measurements

Concurrently with soil fluxes, soil temperature and soil water content were measured at all sites. Soil temperature (Ts) was determined with the built-in temperature probes of the analysers, usually at a depth of 5 cm, while soil water content was measured with a portable TDR system (IMKO Micromodultechnik, GmbH, Ettlingen, Germany) using a 12 cm long trifurcated probe. At both sites, together with other parameters, a weather station measured continuously (1 min, 30 min averages) soil heat flux, soil temperature profiles (5, 25, 45 cm depths) and soil water content in the first 20 cm.

2.5. Statistical analyses

Statistical analyses were performed using Statistica (version 6). Data were interpreted using principal component analyses (PCA). Annual emission (N_2O and CO_2) and consumption (CH_4) rates were determined by integrating the curve obtained by interpolation of the flux rates (shape-preserving interpolation, Matlab version 6.5).

3. Results and discussion

3.1. Overview of soil fluxes

Throughout the study period all plots at both the Natural Forest site and at the Poplar Plantation site were a source for nitrous oxide and carbon dioxide and a sink for methane. PP site is indicated by a dot.

Figure 2 shows an overview of the monthly averaged fluxes measured from April 2002 through December 2003. Due to problems with the gas chromatographic analyzer for methane in the first months, statistical analyses will be done on the measurements carried out in 2003.

With the exception of some N_2O emission events observed at the PP site during spring/early summer, the fluxes measured at the PP and the NF site were not significantly different, as shown in Fig. 2a. However, within each site the fluxes were characterized by a large spatial variability as discussed in the following chapter. The distribution of the flux rates measured at the two sites was highly skewed with a ratio of approximately 40:20:10:5 for flux rates in the range of 0–2.5, 2.5–5, 5–7.5, and 7.5–10 μg N_2O m^{-2} h^{-1}, respectively, at both sites. This is a common feature in studies on N_2O fluxes (e.g. Flessa et al., 1995; Smith et al., 1994). At the PP site, 9% of the flux rates, were 70 μg N_2O m^{-2} h^{-1} or higher, whereby at the NF site such flux rates were measured only at a few occasions in spring 2002 and early 2003. The highest fluxes have been measured in May 2002 at both sites. This is not surprising for the PP site, as on 10 May, the only application of fertilizer (300 kg ha^{-1} urea) occurred causing a pulse of N_2O emissions that lasted several months with flux rates between 65±35 and 398±113 μg N_2O m^{-2} h^{-1} observed at the PP site on 16 May 2002, which decreased during the following months to 41±7 to 108±8 μg N_2O m^{-2} h^{-1} on 15 July 2002. The highest flux rate registered in July 2003 amounted to 10.8±5.6 μg N_2O m^{-2} h^{-1}. But emission rates were also high at the NF site in spring 2002. Here, the explanation must rely on environmental factors. Heavy precipitation events of 75 mm rain in mid April and of 120 mm rain in the week before the measurements started, combined with already warm soil temperature of about 15°C caused ideal conditions for springtime N_2O emissions.

CH_4 consumption rates varied between values of less than −10 μg CH_4 m^{-2} h^{-1} to high CH_4 oxidation rates of more than −100 μg CH_4 m^{-2} h^{-1} at the PP and NF sites. Generally somewhat higher consumption rates occurred a the NF site: the highest CH_4 consumption at this site was −156±24 μg CH_4 m^{-2} h^{-1}, while at the PP site no CH_4 oxidation higher than 67±28 μg CH_4 m^{-2} h^{-1} was measured. Very high standard deviations for the NF site are reported in Fig. 2b. These stem largely from the CH_4 oxidation rates measured at a single plot, which were significantly higher than those measured at the other plots. This will be further discussed below. Consumption of CH_4 in aerobic soils is a common process and can be performed by methanotrophic bacteria which can grow on CH_4 as sole energy source and ammonium oxidizing nitrifiers wich can co-oxidize CH_4 (e.g. Bender and Conrad, 1992; Knowles, 1993). Due to competition between ammonium and methane, inhibition of CH_4 oxidation rates at high nitrogen input levels have been observed (see Mosier and Schimel, 1991; Nesbit and Breitenbeck, 1992). However, other studies report of no effect of ammonium on the rate with which methane is being consumed by the soil suggesting an inhibitory effect of nitrate (Reay and Nedwell, 2004). The reduced rates of CH_4 oxidation measured at the PP site compared to the NF site, however, cannot unequivocally be attributed to the higher nitrogen load by fertilization as other differences as the higher soil compaction (1.08 ± 0.24 and $0.84\pm0.23\,cm^3\,cm^{-3}$ for the first 10 cm of mineral soil at the PP and NF sites, respectively) are also known to reduce CH_4 consumption rates (e.g. Teepe et al., 2004).

Some authors report of strong correlations between soil temperature and CH_4 oxidation rates with subsequently pronounced seasonal variations (e.g. Reay et al., 2005; Steinkamp et al., 2001). No consistent effect of soil temperature has been observed in our study. CH_4 oxidation is characterized by a relatively low activation energy which buffers the effect of temperature (Mosier et al., 2004) suggesting that the consumption of CH_4 in forests soils is limited by diffusion rather than by enzymatic processes as is the case for CO_2 respiration fluxes (King and Adamsen, 1992). Across the field sites of the NOFRETETE project, the type of litter layers found in deciduous forests has been suggested to be one of the factors explaining the lower importance of nitrification and NO emissions in deciduous compared to the coniferous forests.

A compilation by Bradford et al. (2001) gives mean CH_4 oxidation rates for different forest soils of 77 ± 14 or 63 ± 13 g CH_4 m^{-2} h^{-1} for a sampling period $\leq$364 days or >364 days, respectively. The CH_4 consumption rates measured in our study are at the lower end of this range.

In contrast to CH_4 and N_2O fluxes, the variability in soil CO_2 fluxes in 2002 was explained by 58% by soil temperature at 5 cm (see Fig. 2c) in consistency with other studies (e.g. Fang and Moncrieff, 2001; Longdoz et al., 2000; Reichstein et al., 2002). This is indicated in Fig. 2d, showing mean soil CO_2 flux rates and soil temperature.

However, during summer 2003, soil temperature explained much less of the variability of soil respiration rates (29%). Also the direct effect of soil moisture, which is believed to affect heterotrophic respiratory activity at very low values only (Fang and Moncrieff, 2001; Longdoz et al., 2000; Smith et al., 2003), which was the case in summer 2003, cannot fully explain the difference. Schindlbacher et al. (2004) compared the controlling factors for nitrogen oxides emissions from different European forest soils studied in the frame of the NOFRETETE project; they found the lowest optimum for NO fluxes (15% WFPS) for the soil taken at the NF site and conclude that this soil is well adapted to drought stress.

Other studies report a close relationship between forest productivity and respiratory activity in the soil (Janssens et al., 2001; Valentini et al., 2000) due to the coupling of autotrophic root respiration to photosynthetic activity of leaves. For the poplar site it could be observed (Meroni, 2005) that net ecosystem exchange (NEE) of CO_2 as measured by the flux tower was 37% lower in summer 2003 compared to 2002. Thus, the decline in respiratory activity must indirectly be linked to the drought situation, even though the relative contribution of autotrophic and heterotrophic respiration could not be quantified.

Anyhow, in comparison of summer 2002 and 2003, the slight reduction of CO_2 emission from soil in the range of 10% was more than compensated by the substantial reduction of CO_2 uptake by the canopy, resulting in the overall difference in NEE of 37%.

3.2. Spatial variability of site conditions and fluxes

The results of the pedological study are shown in Table 1. The soils at two investigated sites were classified as Regosols according to the nomenclature of the World Reference Base for Soil Resources (FAO et al., 1998). These soils show a low pedogenic development, have a prevalently silty-sandy texture, and belong to the soil subunits Humic, Arenic, Dystric and Eutric.

High spatial variability is commonly found in N_2O flux studies and is usually higher in soils where nitrous oxide is produced predominantly by denitrifiers (e.g. Ambus and Christensen, 1994; Parkin, 1990). Denitrification is a form of anaerobic respiration in bacteria which couples the stepwise reduction of nitrate oxides to electron transport with the intermediate products nitrite, nitric oxide, and nitrous oxide (Firestone and Davidson, 1989). It needs therefore the presence of anaerobic microsites, which are formed if the supply of oxygen is inadequate to meet the respiratory demand, either by the presence of water in the soil pore system (due to the low diffusivity of oxygen in water) or by high oxygen consuming process rates being faster than the delivery of oxygen, or both (e.g. Arah and Smith, 1989; Granli and Bøckman, 1994). Nitrification, the biological oxidation of ammonium to nitrite and nitrate is an aerobic process and has also been described as a possible source for nitrous oxide (e.g. Bremner and Blackmer, 1981; Goodroad and Keeney, 1984; Klemedtsson et al., 1988). Both processes are linked by the nitrate pool of the soil (Russow et al., 2000).

In the present study, spatial variability is caused not only by a random distribution of soil moisture and organic substrate, but also by larger scale (tens of meters) variability in soil conditions; these are caused by the pedologic history of the site with old arms of the River Ticino giving rise to very different conditions of some plots and for a certain period, at both the NF and the PP site. This will be discussed more in detail below.

3.2.1. Spatial variability at the poplar plantation

At the PP site, we observed large differences in soil characteristics with the soil at plot #10 on one hand being characterised by a coarser texture and lower organic carbon and nitrogen content than the soil at plots #6 and #1 on the other hand. The soil at plot #5 showed intermediate characteristics. The moisture content measured in the period 2002–2003 was low for the sandy soil at plot 10 and ranged between 3 and 40% WFPS, while higher moisture content was obtained for soils at plots #1 and #6, ranging between 16 and 57%.

The flux rates of the two soil types are shown in Fig. 3. Emissions were greatest in the finer textured soils at plots #1 and #6, where between May 2002 and December

2003 the measured flux rates of N_2O and CO_2 were 0.52 g N_2O m^{-2} and 6.2 kg CO_2 m^{-2}, respectively. During the same period the flux rates at plot #10 were 0.27 g N_2O m^{-2} and 2.5 kg $CO_2 m^{-2}$, respectively.

The higher flux rates of N_2O and CO_2 were particularly evident at two occasions, i.e. after the input of fertilizer in April 2002 and after the flooding event, which occurred in November/December 2002 and flooded the whole PP site by about 1 m for several weeks (Furlanetto, 2003). In both cases, significant ($p<0.05$) higher emissions were found at the finer textured soils. For example, maximum flux rates at plots #1 and #6 after fertilizer input were 280±89 μg N_2O m^{-2} h^{-1} at moisture content of 51% WFPS with corresponding maximum flux rates of 89±34 μg N_2O m^{-2} h^{-1} at plot #10 with only 37% WFPS. The flooding event enhanced flux rates clearly at plots #1 and #6, while no effect was visible for the Dystri-Arenic Regosol at plot #10. Drying/wetting cycles are known to stimulate nitrogen mineralization and to enhance N_2O emissions by reducing the competition between plant and microbe for N, and thereby potentially increase emissions of N_2O (e.g. Mamilov and Dilly, 2002; Skiba and Smith, 2000). While in most cases the emission pulses are observed when dried soils is being rewetted, in our case the wet period acted as stressor.

We assign the difference in the reaction between the two investigated soil types to the largely differing productivity (Janssens et al., 2001; Raich et al., 2002; Valentini et al., 2000). Smith et al. (2003), which is mirrored at the site by substantial differences in height and BHD of the poplar clone (Meroni, 2005) and is also reflected in the lower content of organic carbon. Another major impact of the different soil textures on N_2O and CO_2 fluxes might be exerted via the effect on soil water retention (McTaggart et al., 2002) and soil moisture. The importance of soil moisture in determining N_2O, CO_2 and NO emissions was reported in several studies (Dilustro et al., 2005; Dobbie and Smith, 2001; Linn and Doran, 1984; Reichstein, 2003; Schindlbacher et al., 2004; Weitz et al., 2001).

3.2.2. Spatial variability at the natural forest site

Irregular morphology of natural forest is the consequence of both the historical changes of the course of the river and periodical flood events. As for the PP site, the flooding event at the end of 2002 gave the opportunity to study in detail variability of parameters relevant for gas exchange processes. In contrast to the PP site, only one plot (plot #19) was affected by the flooding event of November 2002. As these events are occurring with irregular time steps (years to decades) in the Ticino Park (Furlanetto, 2003), we can assume that plot #19, located at lower elevation, is stronger influenced by these events than the other investigated field plots.

As a consequence, the soil at plot #19 shows higher organic carbon and total nitrogen contents, a finer soil texture and a higher surface pH than the other plots (see Table 1). Also, following 2002 flooding event, this plot exhibited significant higher N_2O flux rates (Fig. 4a) than the other plots at the NF site. Enhanced N_2O flux rates were maintained also during spring 2003. At the same time, we observed higher CH_4 consumption rates at the plot #19 (Fig. 4b).

We attribute both effects to an increased bioactivity which affected soil aeration. This led to a more rapid incorporation in the upper mineral soil of the organic material accumulated during previous autumn and in consequence to higher N_2O emissions compared with the other soils (Papen and Butterbach-Bahl, 1999). For CH_4 uptake, the primary control is usually considered to be the rate of gas diffusion within the topsoil (Born et al., 1990; Dörr et al., 1993) which is controlled by both the physical structure and moisture content of the soil. The presence of macropores, such as worm channels, increased gas diffusivity.

Soil respiration fluxes were much less affected by the flooding event; emissions measured at plot 19 were only slightly enhanced between February and May 2003. This was reversed on 16 April 2003, when exceptionally high CO_2 fluxes were measured – for yet unclear reasons – on two plots which have not been affected by the flooding even (Fig. 4c).

3.3. Land use change

A principal component analysis (PCA, Jolliffe, 2002) was carried out comparing the investigated soil plots at both sites respective to their soil properties and flux rates. Each principal component (PC) is generated as a linear combination of the original variables and combines variables that are correlated internally but mostly independent of other variables. The PCs successively are able to explain the majority of the total variation in the dataset. As shown in Fig. 5, 79% of the total variance was explained by the first two factors. The results suggest that positive soil respiration rates were affected by soil texture. Clay and silt contents correlated positively with CO_2 emissions while sandy fractions were inversely correlated with soil respiration. The PCA confirmed also the positive interaction of CH_4 oxidation with water filled pore space (Del Grosso et al., 2000) and a negative correlation with bulk density. As a consequence, higher CH_4 consumption was observed at NF sites compared with PP site and highest N_2O fluxes at plots #1 and #6 of the PP site.

With the exception of plot #10, which was recognized as an outlier in the analysis due to its high sand content, the soils at the PP site showed higher N_2O emissions and lower nitrogen and organic carbon contents than the soils at the NF site. Despite the fact that surface carbon stocks in the natural forest (4.0 ± 0.6 kg m^{-2}) are much higher than at the poplar plantation (1.7 ± 0.4 kg m^{-2}), no significant difference in respiration rates at $p<0.05$ was found between the two sites. This means that specific respiration is higher at the PP site (see Fig. 5a), which can be attributed to the difference in the C:N ratio (14.7 ± 1.7 and 12.5 at the NF and PP sites, respectively) as a result of carbon depletion and management (Tisdall and Oades, 1982). The PCA resulted in positive correlation between specific soil respiration rate and N_2O fluxes. Highest C:N ratio occurred at plot #24 (NF site) and is indicative of a low turnover rate of organic material. At this site, the lowest N_2O fluxes have been measured.

A humus layer was found only at the NF site. The litter was predominantly composed by leaves of *Quercus robur* and, to a smaller extent by *Acer campestre, Corylus avel-*

lana and *Robinia pseudoacacia*. The humus layer was characterized by a C:N ratio of 23, which decreased during litter decomposition and incorporation of the material into the mineral soil stabilizing on characteristic values (Cortez et al., 1996). At the poplar plantation we observed the loss of the humus layer and the depletion of organic carbon in the first 10 cm of the mineral soil (Fig. 6). Higher soil organic carbon content, however, was observed in deeper soil layers (20–45 cm) resulting in a difference of total organic carbon content (0–45 cm of mineral soil, which is the lower boundary of soil tillage at the PP site) of 1.14 kg m^{-2}. A soil stratification ratio, expressed as a ratio of soil C or N content at a deeper and an upper soil layer, respectively, has been proposed as an indicator for soil quality. Higher ratios correspond to higher soil quality and ratios >2 would be uncommon under degraded conditions (Franzluebbers, 2002). Indeed, from organic carbon at 0–10 cm divided by that at 10–20 cm we calculated soil stratification values of 1.6 at the PP site and 3.1 at the NF site.

Considering the carbon content of the humus layer at natural forest site (0.19±0.06 kg m^{-2}), we observed a decrease of 61% of surface carbon (up to 10 cm depth) and a decrease of 25% of carbon content down the profile to 45 cm. This is in line with several studies (Guo and Gifford, 2002; Johnson, 1992; Tiessen et al., 1994; Turner and Lambert, 2000) which have demonstrated that land use changes like the one from forest to crop land lead to soil disturbance and consequently to loss of organic carbon in soil.

3.4. Overall GHG budget

Both N_2O and CH_4 soil fluxes can be considered as negligible in terms of emissions of greenhouse gas equivalents. Table 2 shows that at the NF site, emissions of CH_4 and N_2O contributed to global warming emissions (defined here as the annual emission flux times the global warming potential on a 100 years time horizon as defined in Houghton et al., 2003) of only 0.9% at the soil surface and CH_4 fluxes contributed with –0.3% leaving 99.4% of the fluxes of CO_2 equivalents to CO_2 itself. The picture changes slightly when looking at the PP site: here, the contributions of N_2O, CH_4 and CO_2 were

2.7%, −0.2% and 97.6%, respectively. Meroni (2005) measured a net ecosystem exchange of −1918 g CO_2 m^{-2} y^{-1} at the PP site, thus detecting a significant carbon sink. Looking at the greenhouse gas balance at canopy, the impact of non-CO_2 greenhouse gases at the PP site would become therefore −2.3% and 0.2% of the net ecosystem exchange for N_2O and CH_4, respectively.

4. Conclusions

We studied two different forms of land use at a site which up to 40 years ago was covered by a natural forest as now represented by one of our field sites. The other has been converted into a poplar plantation. We observed a significant degradation of the soil under the poplar trees with a decrease of 61% of surface (10 cm) carbon and a decrease of 25% of carbon down the profile (45 cm). Spatial variability was very high at both sites and could be assigned to a large degree to the morphological situation formed by historical course of the Ticino River. Some events occurring during the study period have evidenced the impact of this spatial variability, i. e., the application of fertiliser at the Poplar plantation and a flooding period lasting for several weeks. They showed that differences in soil texture, via the impact on water / nutrient availability and ecosystem productivity, largely affect the exchange of greenhouse gases between the soil and the atmosphere. It is to expect that a significant part of nitrogen fertiliser applied at the sandy plot was lost to the hydrologic system. N_2O emissions were strongly correlated to the specific respiration rate of the soil, which was enhanced at the Poplar site respective to the conditions in the natural forest.

Still, in terms of greenhouse gas budget, the non-CO_2 gases contributed only minor to the overall balance with only 0.9% and −0.3% of CO_2-eq. emissions as N_2O and CH_4, respectively, at the soil surface of the natural forest, and 2.7% and −0.2% of CO_2-eq. emissions at the soil surface in the poplar plantation. This again points towards the degradation of the soil under tillage for poplar cultivation. We expect that, considering the whole cultivation cycle, the decrease in carbon stocks at this site will be aggravated by the clear-cut, removal of stumps and deep ploughing before replanting in 2005. In this context it is important to mention that the 14 years old poplar stand represents a conservative case study due to its reduced management. Under similar site conditions of the neighbouring Bosco Mezzanone, a comparable stock of poplar biomass is normally obtained within 8–10 years through regular fertilisation, tillage of ground vegetation and, most important, regular irrigation during dry periods. Short rotation cycles are known to decline soil carbon contents (e.g. Harrison et al., 1995; Turner and Lambert, 2000).

Nevertheless, our study provides another proof for the "slow in – fast out" concept of asymmetry between carbon gains and losses due to land use changes (Koerner, 2003; e.g. Thuille et al., 2000). Additionally, enhanced denitrification rates will increase N_2O emission fluxes (e.g. Skiba and Smith, 2000). When planting fast growing trees with the purpose of carbon sequestration, the overall impact on the full budget of greenhouse gases depends very much on the preceding land cover, e.g. if one starts from a forest or a maize field, on the type of management, and on the full life cycle of the plantation. In any case, the conservation and careful management of existing carbon stocks deserves highest priority.

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Table 1. Soil parameters at the studied plots at the Poplar plantation (PP) and the Natural Forest (NF) sites. PP: Poplar Plantation; NF: Natural Forest "Bosco Negri"; Humus: Humus layer. [1] RG: Regosol; dy: Dystric; eu: Eutric; hu: Humic; ar: Arenic (FAO et al., 1998). [2] t-uoEnt: Typic Ustorthent coarse-loamy, mixed, mesic; t-upEnt: Typic Ustipsammment mixed, mesic (Soil Survey Staff, 2003). [3] Jabiol et al. (1995).

	Horizon	pH H_2O	pH KCl	C_{org} ($g\cdot kg^{-1}$)	Ntot ($g\cdot kg^{-1}$)	C/N	Bulk density ($g\cdot cm^{-3}$)	Texture sand (%)	silt (%)	clay (%)	CEC (meq/100 g)	BS (%)
a) PP												
plot 1	Ap1 (0–12 cm)	6.3	5.0	14	1.3	11	1.0	51	38	11	7.54	50
[1]dy-huRG	Ap2 (12–44 cm)	5.8	4.3	9	0.8	11	1.3	22	63	15	9.89	27
[2]t-uoEnt	C (44–60 cm)	5.6	4.1	9	0.6	15	n.d	31	57	12	8.06	42
plot 6	Ap1 (0–11 cm)	6.5	5.2	15	1.2	13	0.9	33	56	11	7.91	53
[1]dyRG	Ap2 (11–45 cm)	6.2	4.5	6	0.7	9	1.3	29	58	13	6.58	55
[2]t-uoEnt	Ap3 (45–55 cm)	6.3	4.6	6	0.5	12	n.d.	17	69	14	7.96	52
	C (55–62 cm)	6.3	4.5	4	0.5	8	n.d.	6	80	14	7.89	47
plot 5	Ap1 (0–18 cm)	6.3	4.9	11	1.0	11	1.0	54	35	11	7.08	40
[1]dyRG	Ap2 (18–45 cm)	6.1	4.5	4	0.3	13	1.3	77	16	7	5.36	34
[2]t-upEnt	C1 (45–65 cm)	6.8	5.5	1	tr	-	n.d.	93	4	3	2.19	80
plot 10	Ap1 (0–12 cm)	7.0	5.6	6	0.4	15	1.4	86	10	4	2.07	95
[1]dy-arRG	Ap2 (12–47 cm)	6.4	4.6	2	0.2	10	1.4	83	14	3	1.85	95
[2]t-upEnt	C (47–65 cm)	7.0	5.5	tr	0.4	-	n.d.	98	2	0	2.61	43
b) NF												
plot 24	A1 (0–8 cm)	5.5	4.6	48	2.9	17	0.5	53	36	11	17.09	32
[1]dy-huRG	A2 (8–13 cm)	4.6	3.7	11	0.8	14	1.0	67	24	9	6.03	24
[2]t-upEnt	CA (13–32 cm)	4.7	3.8	5	0.3	18	1.2	78	16	6	3.61	34
	C (32–62 cm)	5.9	4.1	1	0.1	12	n.d.	88	8	4	3.31	60
plot 22	A1 (0–8 cm)	5.6	4.4	35	2.6	13	0.97	47	42	11	11.59	42
[1]dy-huRG	A2 (8–25 cm)	4.7	3.6	13	1.1	12	1.1	55	35	10	5.11	37
[2]t-upEnt	C (25–60 cm)	6.3	4.5	1	0.1	11	1.22	93	4	3	2.78	56
plot 16	A1 (0–10 cm)	5.8	4.5	35	2.6	13	0.89	22	61	17	14.13	33
[1]euRG	A2 (10–20 cm)	5.5	4.1	7	0.7	10	1.04	55	36	9	2.87	96
[2]t-upEnt	C (20–60 cm)	6.5	4.6	1	0.1	11	1.18	91	6	3	2.13	84
plot 19	A1 (0–6 cm)	6.2	5.1	50	3.2	16	0.84	26	60	14	15.02	46
[1]eu-huRG	A2 (6–17 cm)	5.5	4.2	25	2	12	1	31	56	13	10.09	37
[2]t-uoEnt	CA (17–41 cm)	5.8	4.3	8	0.7	11	1.13	39	49	12	3.95	74
	C (41–60 cm)	6.5	4.6	2	0.2	8	1.23	91	6	3	3.34	70
c) Humus												
plot 24;22;16;	[3]Mesomull	5.5		40.7%	1.8%	23						
plot19	[3]Eumull	5.8		31.7%	1.4%	23						

Table 2. Greenhouse gas budget on the soil surface for the a) PP site and b) NF site in 2003.

	Fluxes of greenhouse gases ($g\ m^{-2}\ y^{-1}$ of gas)	($g\ CO_{2-eq.}\ m^{-2}\ y^{-1}$)
a) Poplar Plantation		
Soil N_2O emissions	0.15±0.1	44±31
Soil CH_4 consumption	−0.15±0.02	−3±0
Soil CO_2 emissions	1608±1053	1608±1053
b) Natural Forest		
Soil N_2O emissions	0.07±0.06	20±18
Soil CH_4 consumption	−0.31±0.14	−7±3
Soil CO_2 emissions	2200±791	2200±791

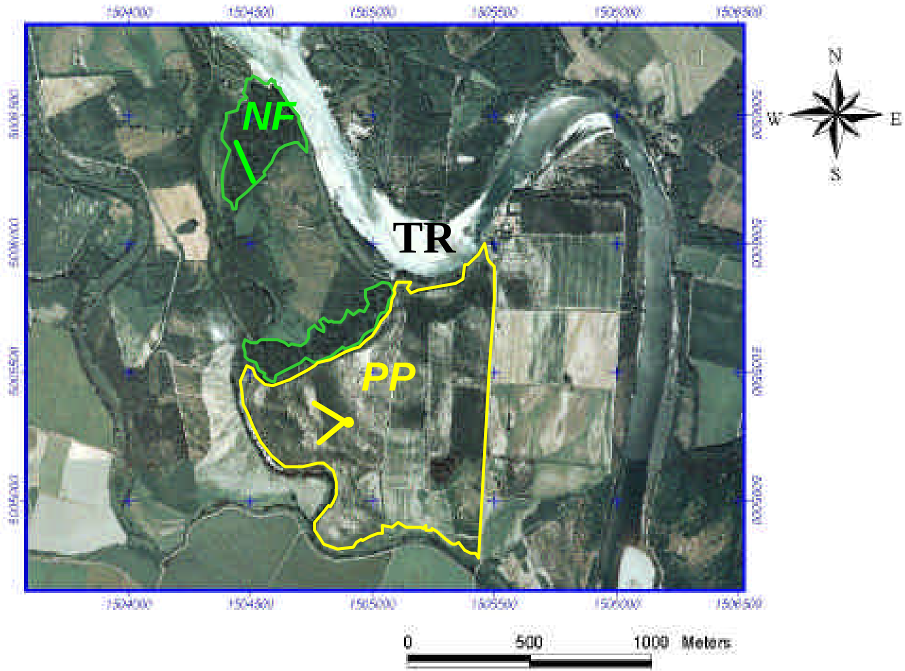

Fig. 1. Study site indicating the Natural forest (NF) and Poplar plantation (PP) sites and the Ticino River (TR). The transects along which the measurement plots were installed are indicated by green (NF) or yellow (PP) lines. The eddy tower at the PP site is indicated by a dot.

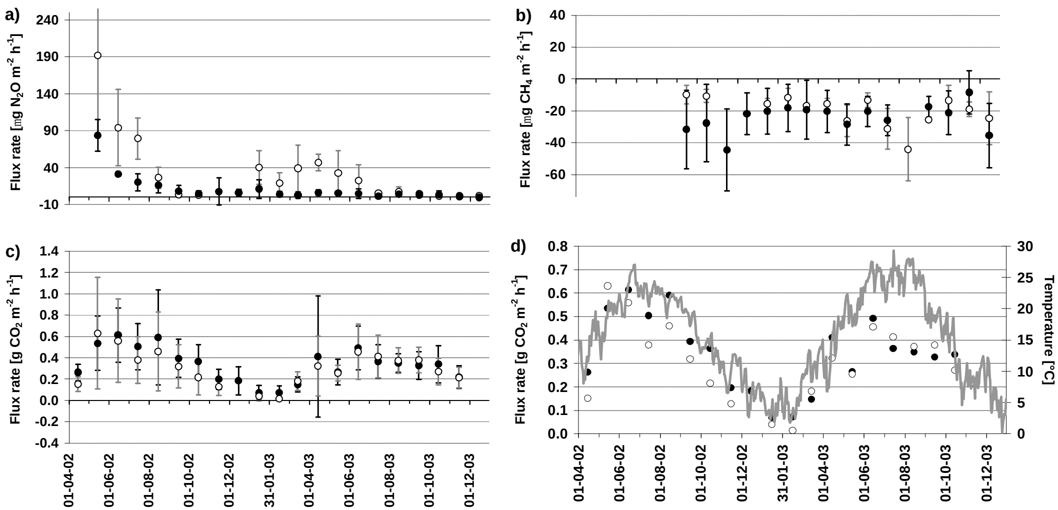

Fig. 2. Monthly mean (±1 SD) of **(a)** N_2O, **(b)** CH_4, and **(c)** CO_2 fluxes from the soil surface at the sites PP (open circles) and NF (closed circles) from May 2002 until March 2004. In **(d)** the mean CO_2 fluxes from the soil surface are plotted in comparison with soil temperature at 5 cm depth.

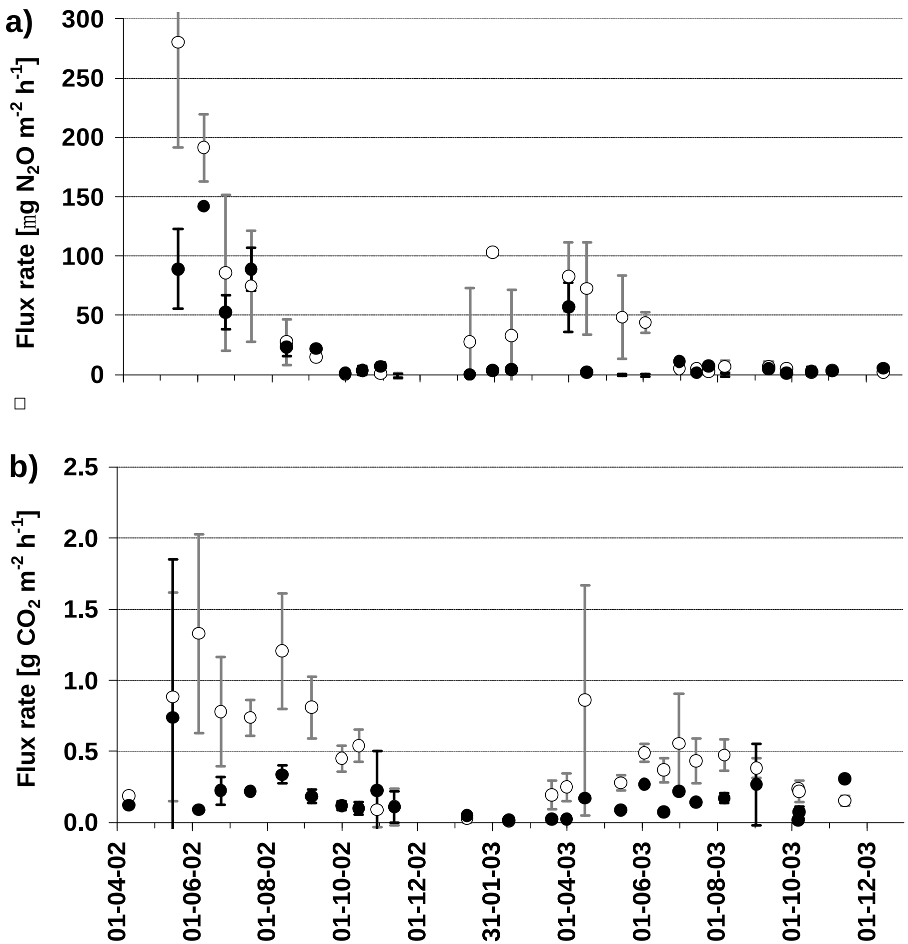

Fig. 3. Measured fluxes (± 1 SD) of **(a)** N_2O and **(b)** CO_2 at the Poplar Plantation, reported separately for plot #10 (closed circles) and plots #1 and #6 (open circles).

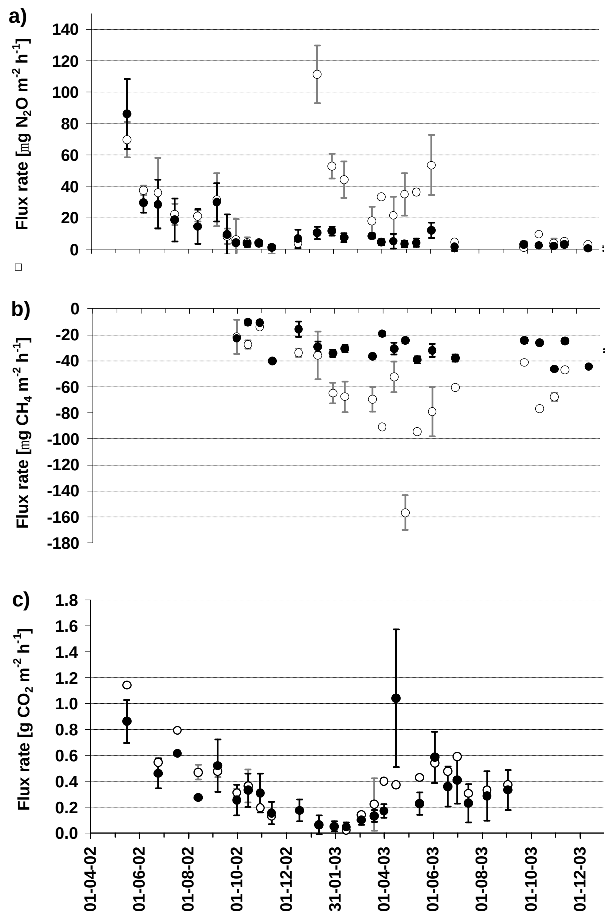

Fig. 4. Fluxes of **(a)** N_2O, **(b)** CH_4 and **(c)** CO_2 in the Natural Forest. The plot affected by the flooding event in November 2002 (plot #19, open circles) is depicted separately from the other plots (closed circles) used for the N_2O flux measurements.

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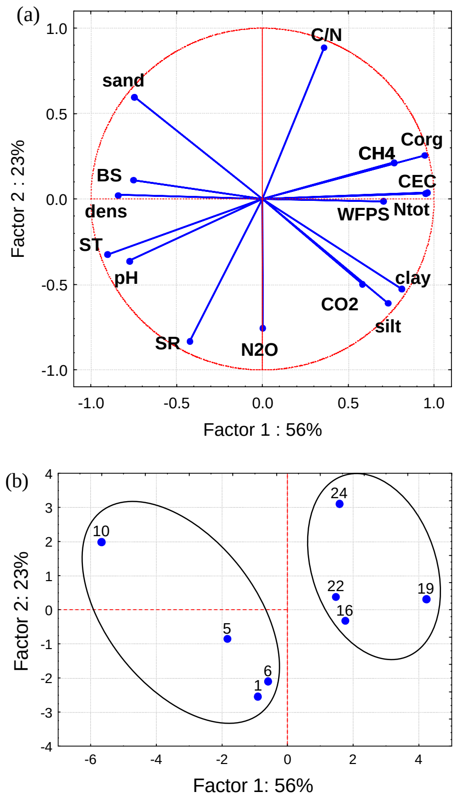

Fig. 5. Loadings plot **(a)** and scores plot **(b)** of the PC Analysis. PCA variables: Corg: carbon content; Ntot: nitrogen content; C/N ratio, sand/silt/clay content; BS: base saturation; dens: bulk density; ST: soil surface temperature; pH (water); SR: specific soil respiration; WFPS; water-filled pore space; CEC: cation exchange capacity; N_2O: N_2O emissions; CO_2: CO_2 emissions; CH_4:CH_4 consumption. *PCA cases*: NF site (plots 16, 19, 22, 24); PP site (plots 1, 5, 6, 10).

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Impact of poplar cultivation on SOC and GHG fluxes

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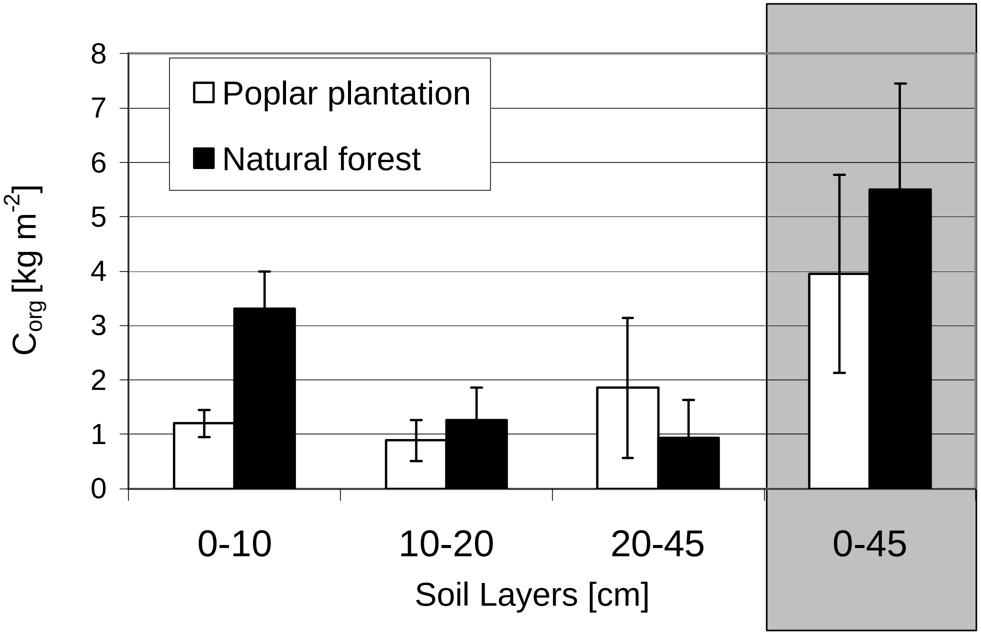

Fig. 6. Content of soil organic carbon in three different soil layers and in the soil profile down to 45 cm at the NF and the PP site.

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