1. Title

GREENHOUSE GAS BALANCE OF CROPLAND CONVERSION TO BIOENERGY POPLAR SHORT ROTATION COPPICE

Running title: From cropland to bioenergy SRC: a GHG balance

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Keywords: land use change, GHG budget, climate mitigation, bioenergy SRC, eddy covariance, SOC change, GHG offset

Paper type: Original Research

2. Abstract

The production of bioenergy in Europe is one of the strategies conceived to reduce greenhouse gas (GHG) emissions. The suitability of the land use change from a cropland (REF site) to a short rotation coppice plantation of hybrid poplar (SRC site) was investigated by comparing the GHG budgets of these two systems over 24 months in Viterbo, Italy. This period corresponded to a single rotation of the SRC site. The REF site was a crop rotation between grassland and winter wheat, i.e. the same management of the SRC site before the conversion to short rotation coppice. Eddy covariance measurements were carried out to quantify the net ecosystem exchange of CO_2 (F_{CO2}), whereas chambers were used to measure N_2O and CH_4 emissions from soil. The measurements began two years after the conversion of arable land to SRC: for that an older poplar plantation was used to estimate the soil organic carbon (SOC) loss due to SRC establishment, and to estimate SOC recovery over time. Emissions from tractors and from production and transport of agricultural inputs (F_{MAN}) were modelled. The C emission rate of heat produced from natural gas was then used to credit to the SRC site the GHG emission offset due to its substitution . Emissions generated by the use of biomass (F_{EXP}) were also considered. Suitability was finally assessed by comparing the GHG budgets of the two sites. Cumulative F_{CO2} at the SRC site was -3512 ± 224 $gCO_2\ m^{-2}$ in two years, while in the REF site it was -1838 ± 107 $gCO_2\ m^{-2}$ in two years. F_{EXP} was equal to 1858 ± 240 $gCO_2\ m^{-2}$ in 24 months in the REF site, thus basically compensating F_{CO2}, while it was 1118 ± 521 $gCO_2\ m^{-2}$ in 24 months in the SRC site. This latter could offset -379.7 ± 175.1 $gCO_2eq\ m^{-2}$ from fossil fuel displacement. Soil CH_4 and N_2O fluxes were negligible. F_{MAN} weighed 2% and 4% in the GHG budgets of SRC and REF sites respectively, while the SOC loss weighed 455 ± 524 $gCO_2\ m^{-2}$ in two years. Overall, the REF site was close to neutrality in a GHG perspective (156 ± 264 $gCO_2eq\ m^{-2}$), while the SRC site

was a net sink of -2202 ± 792 $gCO_2eq\ m^{-2}$. In conclusion the experiment led to a positive evaluation from a GHG viewpoint of the conversion of cropland to bioenergy SRC.

3. Introduction

In the articulated regulation concerning energy and climate change policies, the European Union (EU) established two targets for 2020: (i) reduction of 20% of greenhouse gas (GHG) emissions relative to the levels of 1990, and (ii) share of 20% renewable energy use in gross final energy consumption (European Commission, 2007; European Commission, 2008). For Italy the latter is modulated to 17% (European Commission, 2009).

In the context of climate mitigation, bioenergy crops are expected to play a key role in the renewable energy supply in the EU in the next coming decades (Djomo et al., 2013). Short rotation coppice (SRC) of fast growing trees, and especially of poplar (*Populus* spp.), is a promising culture in this context. SRC has the potential to reduce GHG emissions to the atmosphere during both its production (by capturing CO_2 from the atmosphere and storing it in aboveground biomass and soil) and use (by avoiding CO_2 emissions from fossil fuel burning). However, the management of SRC requires energy inputs, and converting land for SRC production (i.e. land use change, LUC) may alter the equilibrium of the existing ecosystems, causing an impact that in some cases can counterbalance the positive effects on climate mitigation of the SRC (Zona et al., 2013; see also Crutzen et al., 2008; Fargione et al., 2008 for bioenergy crops in general). The LUC to SRC may imply losses of soil organic carbon (SOC) at the installation (Don et al., 2012), especially in C-rich soils, and the management of SRC requires the use of fossil fuels which in some cases can outweigh part of the benefits of the supposed carbon neutral SRC systems (Abbasi and Abbasi, 2009). A recent study (Djomo et al., 2011), however, showed that poplar and willow SRCs biomass use can save up to 80%-90% of GHG emissions compared to using coal for energy production. Studies on the climate mitigation potential of poplar cultivations constitute an important tool in supporting energy and environmental policies at different scales. In recent years researchers approached poplar SRCs from different perspectives: ecological (Jaoudé et al., 2010; Zhou et al., 2013), economic (Strauss and Grado, 1997; Mitchell et al., 1999; El Kasmioui and Ceulemans, 2012, 2013), energetic and environmental (Jungmeier and Spitzer, 2001; Cherubini et al., 2009; Devis et al., 2009; Nassi o Di Nasso et al., 2010; Arevalo et al., 2011; Don et al., 2012; Dillen et al., 2013; Djomo et al., 2013). However, these studies often used different approaches making it difficult to compare their results (Migliavacca et al., 2009; Djomo et al., 2011). Furthermore, emphasis was mainly given to emissions from fossil fuels compared with the biogenic emissions due to the LUC (Djomo et al., 2013). Including the different contributions of the LUC in the assessments of emission savings related to energy crops is crucial (Davis et al., 2009). A full GHG budget based on long-term measurements of CO_2 and non-CO_2 GHGs via eddy covariance (EC) methodology (Aubinet et al., 2012) and soil chambers measurements (Allard et al., 2007), can be used to assess the GHG mitigation potential of land conversion to SRC (Byrne et al., 2007; Ceschia et al., 2010). Several authors (e.g. Ceschia et al., 2010; Osborne et al., 2010) highlighted the need for a more consistent number of studies on GHG budgets, including different types of management practices, climate conditions, and soil characteristics, in order to reduce the uncertainty in GHG budgets at large scale (Smith et al., 2010). A GHG budget approach was used by Gelfand et al., 2011 in a conversion of unmanaged lands to herbaceous biofuel crops in the US. In Europe, Zona et al., 2013 estimated the GHG balance in the first year after the conversion from agricultural lands to a poplar SRC in Belgium, focusing on biogenic contributions. The present study considered a conversion of a cropland (hereafter indicated as “REF site”) to a poplar SRC (hereafter indicated as “SRC site”) for bioenergy production in the Mediterranean area (Viterbo, Central Italy). The aim was to extend the GHG balance to emissions generated by field management and to the offset of GHG from fossil fuels substitution. The number of studies on SRC systems cultivated in Mediterranean areas is limited, where water availability can constitute a limiting factor for biomass yield and thus climate mitigation (Cherubini et al., 2009). Given that the warming mitigation potential of energy crops is the main reason for subsidies to arable land conversion, our study aimed to assess the suitability of the LUC to SRC in terms of mitigation of GHG emissions.

4. Materials and methods

4.1 GHG budgets assessment

The GHG budgets were calculated for the SRC and for the REF sites on a temporal basis of two years (24 months), corresponding to the second rotation cycle of the SRC site. They included several positive and negative GHG contributions, with the following sign convention: a positive flux indicates a release into the atmosphere, while a negative flux represents an uptake from the atmosphere. For the SRC site, the net GHG budget (B_{SRC}) was calculated as the algebraic sum of all GHG contributions as indicated in Eq. (1):

$$B_{SRC} = F_{CO2} + F_{CH4} + F_{N2O} + F_{MAN} + F_{SOC} + F_{SAV} + F_{EXP} \qquad (1)$$

In equation (1), F_{CO2} represents the flux of CO_2, i.e. the net ecosystem exchange (NEE) of CO_2, while F_{CH4} and F_{N2O} represent the biogenic methane and nitrous oxide soil-atmosphere exchanges. F_{MAN} includes the GHG emissions related to the management of the SRC site, and F_{SOC} is the loss of soil organic carbon content due to the installation of the cuttings. F_{SAV} represents the GHG offsets, i.e. avoided GHG emissions due to the substitution of natural gas by biomass in the heat production, and F_{EXP} the biomass exported from the site at the end of the cycle and reemitted as CO_2 at burning.

Similarly, the net GHG budget of the REF site (B_{REF}) was estimated with the algebraic sum indicated in Eq. (2), where in respect to Eq. (1) there is not F_{SOC} and F_{SAV}, and F_{EXP} is the portion of the exported biomass that returns to the atmosphere as CO_2 or CH_4:

$$B_{REF} = F_{CO2} + F_{CH4} + F_{N2O} + F_{MAN} + F_{EXP} \qquad (2)$$

All the contributions of B_{SRC} and B_{REF} were expressed as CO_2-equivalent (CO_2eq) fluxes per unit of surface, as the functional unit of the study was one square meter of land. Finally, the net GHG cost or benefit of converting the cropland to a SRC plantation was calculated by comparing B_{SRC} and B_{REF}. Displacement of food and feed production related to SRC cultivation on cropland was beyond the scope of this study.

4.2 Site description

Two sites close to each other located in a private farm (*Gisella ed Elena Ascenzi S.A.A.S.*) in Castel d'Asso, Viterbo, Italy (coordinates: 42°22' N, 12°01' E), were selected during the summer 2011. Two EC towers were installed in the two sites to measure the exchanges of CO_2 and H_2O between the ecosystem and the atmosphere following the methodology reported in Aubinet et al., 2000. The climate of the area is Mediterranean, with a yearly average rainfall of 766 mm, mean temperature of 13.76 °C and weak summer aridity in July-August (Blasi, 1993). The SRC site was a 2-year rotation cycle managed poplar plantation of 11 ha planted in 2010 to produce biomass for energy (heat). Poplar cultivar was *Populus x canadensis* – clone AF2 selected in *Alasia Franco Vivai*'s nurseries. According to the regional law (Rural Development Programme of Latium 2007-2013, Latium Region, 2015) 12 years is the maximum period to get subsidies for SRC, and corresponded to the time the farmer decided to cultivate the SRC site (personal communication). For that reason the calculations of the present study will be referred to a 12-years lifespan. The site was previously managed with a 2-year rotation between a clover grassland (*Trifolium incarnatum L.*) in mixture with ryegrass (*Lolium multiflorum Lam*) and winter wheat (*Triticum aestivum L. emend. Fiori et Paol*). The REF site was a 9 ha grassland-winter wheat rotation located at a short distance (300 m). Having the identical land use and management of the SRC site before the installation of the poplars, it was selected to assess the GHG effects of the LUC. GHG balances were calculated over 24 months in both sites. However, these 24-month periods did not completely overlap, as the two cultivations had different beginning times: for the SRC site the GHG budget estimation went from 12 January 2012 (immediately after the first harvest of the SRC site) to 11 January 2014, corresponding to the second cycle of cultivation. The period of calculation of the GHG budget for the REF site went instead from 1 September 2011 until 31 August 2013. However, manual chamber measurements of CH_4 and N_2O in the REF site started at the beginning of April 2012. The 24 months considered for the SRC site corresponded to the second cycle of the short rotation coppice, and thus did not include the period right after the conversion of agricultural land. This rotation was supposed to terminate with the harvest. However, due to unfavourable climate conditions (a strong drought during summer), the harvest of the SRC site planned for 2014 was postponed to 2015.

The SRC site had a planting density of around 5300 cuttings per hectare, that were planted in rows spaced 2.5 m at a distance of 0.75 m between each other. The first harvest occurred in January 2012. The SRC site was irrigated during the driest periods in summer using a system of tubes installed 35 cm belowground on alternate inter-rows, summing up to about 210 mm in 2012 and 80 mm in 2013 of equivalent precipitation added to the soil. No fertiliser was provided to the SRC site in 2012, while 40 kg per hectare of urea were dissolved in the irrigation water in a single event in 2013. Insecticide (DECIS) was used in May 2012 against *Chrysomela populi* L. In the REF site a shallow tillage (15 cm) was performed in September 2011 with a rotary harrow, and clover and ryegrass were sown. At the end of April 2012 half of the crop was converted to sorghum (*Sorghum vulgare Pers.*) after a period of aridity in spring time. Both the clover and the sorghum were grazed during the growing season, with grazing removing all the above-ground biomass from the sorghum, while the clover was harvested at the end of the cycle. At the end of October 2012 the land was tilled at 40 cm depth, and winter wheat was sown in November. In April 2013 herbicide was distributed over wheat (Buctril at a rate of 1 l ha^{-1}), which was harvested at the beginning of July 2013 and no other operation was performed until the end of August. Sorghum was irrigated in several days in summer using a sprinkler with a total amount of 275 mm of equivalent precipitation, while no irrigation was applied to the winter wheat. Sorghum was also fertilised twice with 150 kg ha^{-1} of ammonium nitrate, while 200 kg ha^{-1} of the same fertiliser were provided once to the wheat. Apart from irrigation and fertigation, all the operations described above were performed using two different types of tractors, generating different diesel consumptions associated to each operation (Table 3).

An older SRC site (indicated hereafter as O_SRC site) located alongside the other one and subjected to the same type of management, but planted in 2007, was used in the estimation of SOC content loss caused by the LUC. This was necessary as the expected SOC loss following the conversion (i.e. during the first rotation) was not measured.

In the 24 months considered for the GHG budget of the SRC site, precipitations summed up to 1078 mm, with an average temperature of 14.72 °C, while in the 24 months used for the REF site precipitations were 1157 mm and average temperatures 15.31 °C. In both cases yearly values of precipitation were lower than the long-term average of 766 mm (Blasi, 1993). An intense drought occurred in summer 2012, with no rain from the beginning of June until the end of August, in contrast to the long-period average of cumulate rainfall in these months (110 mm, Blasi, 1993). Soils were classified as *Chromic Luvisol* according to the World Reference Base classification (USS 2014), with a clay-loam texture. Values of pH ranged between 5.88 in the REF site, 6.66 in the O_SRC site and 6.69 in the SRC site. The stock of nitrogen (N) up to 70 cm was not significantly different between sites, ranging from 3.16 ± 1.60 MgN ha^{-1} to 3.19 ± 1.47 MgN ha^{-1} and 3.25 ± 1.47 MgN ha^{-1} respectively for SRC, O_SRC and REF sites. See Fig. 1 for a schematic representation of land cover and management events of the two sites.

4.3 F_{CO2}: eddy covariance measurements

The EC technique was used to determine the turbulent vertical fluxes of momentum, CO_2, latent and sensible heat. A 3-D sonic anemometer was installed in each site for high-frequency measurements of wind speed, wind direction and sonic temperature. CO_2 and water vapour mass densities were collected using a fast-response open-path infrared gas analyser (see Table 1 for models and manufacturers). These instruments were mounted on towers located in the centre of the fetches. On the REF site the mast was 3 m high, while an extendible telescopic pole was used in the SRC site in order to always measure turbulences above the roughness layer (Foken, 2008).Several meteorological variables above and belowground were continuously measured on a 30 min basis to properly calculate fluxes and characterise the two sites. In Table 1 the complete instruments setup is described, including both meteorological and high-frequency variables.

Half-hourly fluxes were calculated with EddyPro® software (LI-COR, Lincoln, NE, USA). Several corrections to the time series (Aubinet et al., 2012) were applied as reported in Table 2. Post-processing included spike removal and friction velocity (u_*) filtering (Papale et al., 2006), gap-filling using the marginal distribution sampling (MDS) approach and partitioning of F_{CO2} into gross primary production (GPP) and ecosystem respiration (R_{eco}) components (Reichstein et al., 2005). The gap-filled F_{CO2} and its components were then cumulated along the 24-month period considered.

Uncertainty in F_{CO2} was calculated on the basis of the uncertainty in the u_* filtering, assuming that the main potential systematic error is due to advection and thus linked to the u_* filtering. One hundred thresholds were calculated using a bootstrapping technique and then applied to filter the data. For each half-hour, the median of the distribution of F_{CO2} obtained using the 100 thresholds was used for the GHG budget (Gielen et al., 2013), and the range of uncertainty was derived as half the range 16^{th}-84^{th} percentile.

4.4 Soil characteristics and SOC stock and changes

To better characterize the soil properties and to quantify the changes in SOC stocks due to the installation of the poplar plantation, a number of soil analyses were performed in the three sites in two different periods. In a first phase on February 2012 three soil trenches 150 cm wide were opened randomly in each site and the soil sampled by depth (0-5, 5-15, 13-30, 30-50, 50-70, 70-100 cm) at both the opposite sides of the profiles to have six replicate samples per depth. The bottom layer (70 cm – 100 cm) was absent in the REF site due to the presence of the bedrock at 80 cm, rather than 100 cm as in both the SRC sites. Samples were collected using a cylinder to determine also the bulk density. Main goals of this first sampling campaign were to describe the soil characteristics and to determine the number of replicates necessary to detect with statistical significance a change in SOC content of 0.5 gC kg^{-1} soil (Conen et al., 2003). In the SRC and O_SRC sites ten samples of the organic layer were also taken removing all the material present over the mineral surface within a squared frame with an area of 361 cm^2. In the REF site this sampling was not performed because a permanent organic layer was missing. All samples were air-dried at room temperature and then sieved at 2 mm to separate the coarse fraction, and the analyses performed on the fine earth. The pH was measured in deionised water with a sure-flow electrode, using a ratio soil-solution of 1:2.5 (w/w), and texture was determined after destruction of the cement using sodium hypochlorite adjusted at pH 9 (Mikutta et al., 2005). The sand fraction was separated by wet sieving at 53 µm while the silt and the clay fractions were separated by time sedimentation according to the Stokes law. Total carbon (C) and nitrogen concentrations were measured on finely ground samples by dry combustion (ThermoFinnigan Flash EA112 CHN), while SOC and N stocks were determined taking into account soil C and N concentrations and a weighed mean of bulk density, depth of sampling and stoniness (Boone et al., 1999). During the second phase in March 2014 a new sampling was performed in the REF, SRC and O_SRC sites. The number of samples necessary to detect statistically a SOC change was 50, as derived from the first phase. Samples were taken from the first 15 cm of soil, as most of the changes in a short period occur in the shallower layers. C concentration was measured and SOC stocks re-calculated. The normality of the distributions was checked using a Chi-squared test (Pearson, 1900). An ANOVA test (Fisher, 1919), combined with a Tukey multiple comparison test were used to check if SOC stocks were different between the sites. As data of F_{CO2} from the beginning of the cultivation are missing, SOC changes due to the installation of the poplar cuttings were calculated building a linear regression between SOC content of the SRC site (4 years old) and the O_SRC site (7 years old), then estimating the SOC at the time of plantation (year "0"). Following the "free-intercept model" described by Anderson-Teixeira et al., 2009, the SOC content change due to the plantation of the SRC was then extrapolated considering the difference between the SOC content at year 0 and the one measured in the REF site, assuming the SOC content in the REF site in equilibrium, as this type of land use was constant in the last 30 years. Uncertainties in SOC concentration and stock were calculated as standard deviations from the mean values of each repeated measure, while errors were estimated using the law of error propagation as reported by Goodman, 1960.

4.5 Soil CH_4 and N_2O fluxes

On-site measurements of CH_4 and N_2O soil fluxes were combined with laboratory incubation analyses, where soil samples were tested at different water contents and N addition levels. Field measurements of soil N_2O and CH_4 fluxes were carried out in the two sites using nine manual, dark, static PVC chambers (15 cm diameter, 20 cm height, and total volume 0.0039 m^3) per site, placed over as many PVC collars (7 cm height, 15 cm diameter) permanently inserted into the soil at 5 cm depth for the period of observation. At the SRC site, three collars were distributed along the tree line (between two trees), three along the irrigated inter-rows and three along the non-irrigated inter-rows, while in the REF site collars were placed in three different blocks of three collars each. Gas samples were collected from each chamber at the closure time, and 30 and 60 minutes after closure. Samples were stored in glass vials provided with butyl rubber air tight septum (20 ml) and concentration of N_2O and CH_4 was measured using a trace Ultra gas chromatograph (GC) (Thermo Scientific, Rodano, IT). The flux detection limit of the GC was in the order of about 0.1 mg of CH_4 or N_2O m^{-2} day^{-1}, and the analytical precision of the GC for standards at ambient concentration was approximately 3-5 %, using one standard deviation as a measure of mean error. Further details on GC set in Castaldi et al., 2013. Measurements started two weeks after collar insertion and samples were collected every 2-4 weeks, depending on land management practices and weather conditions, for a total of 30 dates in the SRC site and 24 for the REF site. Similar frequencies were used in previous studies (e.g. Pihlatie et al., 2007; Weslien et al., 2009), and considered pertinent on the basis of the low magnitude of the measured fluxes. To test if fertilisation could trigger a peak of N_2O emission as found in previous studies (e.g. Gauder et al., 2012), measurements in both sites were carried out more frequently in occasion of fertilisation events (on average every two days), starting from the day before the application of fertiliser and for a week. Measurements also covered different soil and meteorological conditions, including periods of drought and rewetting. Measured average daily soil CH_4 and N_2O fluxes were cumulated over the 24 months by linear interpolation as described by Marble et al., 2013, and uncertainty calculated propagating the standard deviations of the replicates. Intergovernmental Panel on Climate Changes (IPCC) 100-year global warming potential (GWP) weighed estimates of GHGs (Forster et al., 2007) were used to convert F_{N2O} and F_{CH4} into CO_2 equivalents: factors 298 and 25 respectively.

4.5.1 Laboratory incubations

Due to the fact that we don't have continuous measurements of non-CO_2 fluxes from soil, we performed laboratory analysis to verify the accuracy of field campaigns. Laboratory incubations were carried out to assess the GHG emission rates under controlled laboratory conditions in soil treated with both water and nitrogen addition, and to quantify the rates of soil mineralization and nitrification. The rational of the incubation was to assess whether the fluxes were driven by limiting conditions like water and/or nitrogen, or slow rate of organic N mineralization, as found in a Mediterranean coppice site in the same region (Castaldi et al., 2009; Gundersen et al., 2012). Addition of N allowed to check if short-time peaks of emissions occurred that could escape due to the selected frequency of sampling. Soil cores (7 cm diameter, 10 cm height) sampled in the two ecosystems were incubated at 20 °C. Water was then added to reach three different ranges of Water Filled Pore Space (WFPS%): 20% (i.e. the value estimated at sampling), 50% and 90%, each of them replicated five times. The sample at the highest WFPS% was also replicated with or without nitrogen supply (100 kgN ha^{-1} of NH_4NO_3). Cores were placed in gas-tight 1-litre jars and 6 ml air samples were collected immediately after closure and after 3 hours of incubation for N_2O production determination. Gas concentration was determined by gas chromatography the day after the treatment and in the following 5 days, leaving the jars open during this period and closing them only when N_2O production needed to be determined, in order to avoid developing of liquid oxygen tension conditions. Net mineralization and nitrification, and net potential nitrification rate were determined on sieved (2 mm mesh) soil samples over 14 days of incubation, while for the determination of potential nitrification soil was amended with ammonium sulphate $(NH_4)_2SO_4$ (100 µgN g^{-1} dry soil). A modified method (Kandeler, 1996; Castaldi and Aragosa, 2002) was used to extract NH_4^+ and NO_3^- from the soil at T_0 and T_{14} days for further concentration determination with calibrated specific electrodes after the addition of a pH and ionic buffer 0.4 ml di ISA (Ionic Strength Adjustor; Orion cat. No. 951211 e Orion cat No. 930711). Mineralization rates were calculated as the total soil mineral N (µg of N-$NH4^+$ + N-$NO3^-$ per gram of dry soil) measured after 14 days of incubation (T_{14}) minus total mineral N measured at incubation start (T_0) divided by the number of days of incubation. Nitrification rates were calculated similarly, considering only the amount of N-$NO3^-$ produced at T_{14} minus the amount of N-$NO3^-$ present at T_0.

In order to compare results obtained with soil cores to field conditions, in situ *WFPS%* was calculated for the whole period of field monitoring:

$$WFPS\% = \frac{M_{SOIL}}{1-\left(\rho_{BULK}/\rho_{PART}\right)} * 100 \qquad (3)$$

where M_{SOIL} is the volumetric soil moisture (m^3 m^{-3}), ρ_{BULK} is the bulk density (Mg m^{-3}) and ρ_{PART} is the particle density (Mg m^{-3}). For mineral soil ρ_{PART} is approximated to that of common silicate materials (2.65 Mg m^{-3}, Chesworth, 2008).

4.6 Emissions due to management

Life cycle assessment (LCA) was used to estimate the anthropogenic GHG emissions due to farming operations (Robertson et al*.,* 2000) in both sites (Table 3), and the GHG emissions due to grazing in the REF site (Table 4). The present study is not a full LCA, but the LCA approach was used to estimate emissions caused by field management as described in the following. Fossil fuel emissions associated with the cultivation of the SRC and REF sites included on-site emissions from tractors (used to carry out all the main operations: ploughing, seeding, solid fertilisation, harvesting) and irrigation, as well off-site emissions from the production and transport of agricultural inputs (fertiliser, insecticide, herbicide). Emissions due to the production of tractors were considered negligible as in Budsberg et al., 2012 and Caputo et al., 2014. On-site GHG emissions due to diesel consumption were calculated as the product of the amount of fuel diesel consumed to carry out a given farm activity (e.g. harvesting) and the emissions factor of diesel, 90 $gCO_2eq\ MJ^{-1}$ (Table 3). This factor includes emission costs due to the combustion of diesel (74 $gCO_2eq\ MJ^{-1}$), and emissions due to its production and transportation (16 $gCO_2eq\ MJ^{-1}$) (Edwards et al., 2007). Considering energy density of diesel to be 38.6 $MJ\ l^{-1}$ (Alternative Fuel Data Center, 2014), producing, transporting and burning 1 l of diesel emitted 3474 gCO_2eq. An exception was made for harvesting in the SRC site, for which emissions for diesel consumption relative to the previous harvest (2012) were considered, as the harvest at the end of the cycle was postponed. Emissions due to irrigation were calculated by multiplying the electricity consumed in powering the pumps by an emissions factor of 750 $gCO_2\ kWh^{-1}$, calculated as the average of different emission factors for different sources of electricity (Bechis and Marangon, 2011) weighted on the Italian electricity grid mix, derived from the Italian energetic balance 2012 (Italian Ministry of Interior, 2013). Off-site emission costs for fertilisers and insecticides were estimated as the product of applied amount of fertiliser or insecticide and the emission factors for manufacturing 1 kg of fertiliser/insecticide: 4018.9 $gCO_2\ kg^{-1}$ N for urea (NPK rating 40-0-0), 4812 $gCO_2\ kg^{-1}$ N for diammonium phosphate (NPK 18-46-0)[1], 7030.8 $gCO_2\ kg^{-1}$ N for ammonium nitrate (NPK 33-0-0) and 7481.9 $gCO_2\ kg^{-1}$ N for calcium ammonium nitrate (NPK 27-0-0) (Wood and Cowie, 2004). Although emission factors differ among insecticide types, in this analysis we assumed that the difference is negligible as the use of insecticides was limited, and thus considered the emission factor of insecticide (active ingredient: deltamethrin) as the product of energy required to produce 1 kg of insecticide (310 $MJ\ kg^{-1}$) and the emission rate of insecticide (60 $gCO_2\ MJ^{-1}$) (Barber, 2004; Liu et al., 2010). The emission factor of herbicide was taken from literature (Ceschia et al., 2010): 3.92 kgC per kg of product. Fuel used for the application of chemical products was included in the on-site

[1] This includes production and transport costs of the overall fertiliser, including P.

calculations described above. All the contributions listed above were converted on a surface basis (Table 3).

4.7 Biomass use and GHG offset

During the first year of cultivation the REF site was grazed by sheep, which were brought to the field in defined periods (Table 4). Hence, the aboveground biomass (AGB) from the REF site was rather grazed by sheep or provided as hay to other livestock, destined to meat and milk production, or in the case of wheat used in food (grains) and feed (foliage) production. Due to the different species cultivated throughout the two years and to the different uses of the biomass, F_{EXP} of the REF site (Eq. (2)) includes the following:

$$F_{EXP} = E_{CH4,on} + E_{CO2,on} + E_{CH4,off} + E_{CO2,off} \tag{4}$$

where the first subscript indicates whether the exported C is reemitted to the atmosphere as CO_2 or CH_4, and the second subscript distinguishes between emissions occurring on-site (*on*) and off-site (*off*). In fact, the percentage of AGB ingested by herbivores on grassland varies with the intensity of management (Soussana et al., 2010). In the present study, however, what was left in the field by the sheep was then harvested and provided them off-site. We assumed then that, apart from the grains in wheat ears, all the AGB was ingested by sheep or other livestock, and that the digestible portion of the organic C ingested was respired back to the atmosphere as CO_2 or emitted as CH_4 via enteric fermentation (Eq. (4)) (Soussana et al., 2007). Biomass in the REF site was sampled every 2-3 weeks in five plots (0.5 m x 0.5 m) randomly selected within the field. In three dates samples were collected immediately after grazing in a grazed area and in an undisturbed area to quantify the intensity of mowing (68%) and identify the C ingested on-site and off-site. Biomass samples were oven-dried at 70 °C to constant mass and weighed. Total AGB was obtained cumulating dry weights measured immediately before each grazing event, subtracting each time the 32% of the dry weight of the previous sample to consider mowing intensity. IPCC methodology (Dong et al., 2006) was then used to estimate $E_{CH4,on}$ (Eq. (4)), adjusting the methane emission factor per animal considering the average weight (55 kg) of sheep (19 gCH_4 $head^{-1}$ day^{-1}), and multiplying it by the daily number of sheep present on-site. The method in Soussana et al., 2007 (their Eq. (4)) was then adapted to estimate the other three components in Eq. (4): $E_{CH4,off}$ was estimated applying to the C ingested off-site the ratio between the C weight in $E_{CH4,on}$ and the C ingested on-site. The C emitted as CH_4 was subtracted from the digestible portion of the C ingested, assumed to be 65%, and the remaining converted in CO_2 as to estimate $E_{CO2,on}$ and $E_{CO2,off}$. The remaining, non-digestible C (35%) was assumed to be returned to the SOC of the grassland (for the on-site part) or of other systems (for the off-site part) as faeces, thus not contributing to the GHG balance. The portion of C that was stock in the body mass of animals was considered negligible (Soussana et al., 2007). For the sake of simplicity, we assumed that also the C content of wheat ears will be shortly respired back to the atmosphere as CO_2, and thus included in $E_{CO2,off}$ (Eq. (4)).

At the end of the cycle, poplar aboveground woody biomass (AGWB) of the SRC site was supposed to be harvested and burnt, thus from one side releasing C back to the atmosphere, and from the other offsetting GHG emissions for fossil fuels displacement. To estimate poplar AGWB diameters were measured at the end of the cycle, after the leaves fall. Three rows of trees were selected inside the plantation and the diameters of these trees were measured (minimum threshold 0.5 cm) at 1 m height. A simple model considering the regression between individual shoot dry weight (W_D) and 1 m diameter (D) was used:

$$W_D = b * D^c \quad (5)$$

where b and c are empirical parameters, W_D is in kg DM (kg of Dry Mass), and D is in cm. Parameters were set as b = 0.0847 and c = 2.112 following Mareschi, 2008 (see also Paris et al., 2011) for the second rotation cycle of clone AF2 of the plantation located in Bigarello

(Mantua province), as the one with the more similar climatic and soil characteristics, and also with the same root and shoot age. Dry combustion (1108EA, Carlo Erba, Milan, IT) was used to determine the C concentration for both sites. Regarding the GHG emissions offset, it was assumed that heat produced from SRC biomass will substitute heat produced from natural gas. The GHG offset (F_{SAV}) was estimated based on the yield of the SRC site, the energy density of poplar, the conversion efficiency of typical biomass boiler in Italy, and the emission rate of heat production from natural gas in Italy:

$$F_{SAV} = Y * H_L * \eta_{CONV} * I_{NG} \tag{6}$$

where Y is the biomass yield (kg m^{-2}), H_L is the low heating value of poplar (13 MJ kg^{-1} at 30% moisture content, Boundy et al., 2011), η_{CONV} is the efficiency of conversion of poplar chips to heat, assumed in this study to be 84% (Saidur et al., 2011), and I_{NG} is the carbon emission rate (intensity) of heat produced from natural gas (i.e. 55.862 gCO_2eq MJ^{-1}) for Italy (Romano et al., 2014).

5. Results

5.1 *Biogenic fluxes of CO_2*

The cumulative F_{CO2} in the REF site for the two years considered was -1838 ± 107 $gCO_2\ m^{-2}$, partitioned in 8032 ± 313 $gCO_2\ m^{-2}$ absorbed through photosynthesis (GPP) and 6216 ± 338 $gCO_2\ m^{-2}$ emitted by total R_{eco}. In the SRC site cumulative F_{CO2} summed up to -3512 ± 224 $gCO_2\ m^{-2}$, with GPP equal to 8717 ± 298 $gCO_2\ m^{-2}$ and R_{eco} equal to 5205 ± 425 $gCO_2\ m^{-2}$ (Fig. 2). Hence, the SRC site was a larger CO_2 sink compared to the REF site over the measuring period, due to both the higher GPP and the lower ecosystem respiration of the SRC site relative to the REF site.

Seasonal differences between the sites in the net flux of CO_2 were observed (Fig. 3). The main dissimilarity was the timing of the peak of CO_2 uptake, during the spring in the REF site and in summer for the SRC site. In both sites, peaks of CO_2 uptake were higher in 2013 than in 2012. In the latter, however, a minor peak of uptake were observed in early fall in the SRC site. Periods with positive net fluxes of CO_2 appeared longer and with higher values in the REF site (Fig. 3, top). Air temperatures registered in the two sites were similar, but higher in summer 2012, while the SWC at 30 cm depth was higher in the REF than in the SRC site (Fig. 3, bottom).

5.2 *Soil CH_4 and N_2O fluxes*

Daily average of both F_{N2O} and F_{CH4} were very low in almost every measurement (Fig. 4), leading to low total cumulative soil F_{N2O} and F_{CH4} for both the sites: overall soil non-CO_2 fluxes were 15.5 ± 4.7 $gCO_2eq\ m^{-2}$ in two years for the SRC site and 0.5 ± 1.6 $gCO_2eq\ m^{-2}$ in two years for the REF site. Both sites were small sources of N_2O and small sinks of CH_4. CH_4 sink at the SRC site was not significantly different from the one at the REF site, although on average slightly higher, and significantly higher N_2O emissions were observed at the SRC site, although still very low. Measurements carried out in occasion of fertilisation events showed no significant increase in the emission rates of N_2O in respect to non-fertilisation periods: fluxes in the SRC site in the period of the unique fertilisation occurred in the two years of study remained low, and in the REF site none of the four measurements taken in the period of the fertilisation event of June 2012 exceeded the detection limit of the GC.

5.2.1 Laboratory incubations

N_2O emissions determined in laboratory incubations confirmed that over most of the analysed WFPS% values both soils were producing little N_2O in absence of N addition, even at WFPS% normally considered to trigger N_2O emission (WFPS% 60-80%) (Fig. 5). Addition of N did not seem sufficient to stimulate N_2O production. In contrast, very high WFPS%, close to saturation, was able to trigger a strong increase of N_2O production in the soil of the REF site. Comparing the data reported in Fig. 5 with the field data of WFPS% for the REF site (Fig. 6), it can be seen that most of the time WFPS% was significantly below 70% in the whole profile and that at 5 cm, where most of the interaction with added fertilizer might have occurred, the WFPS% never exceeded 50%. Mineralization and nitrification rates were quite low in both sites, with slightly positive mineralization rates in the SRC site (0.28 ± 0.05 μgN g^{-1} d^{-1}) and a very small net immobilization in the REF samples (-0.2 ± 0.2 μgN g^{-1} d^{-1}). Net nitrification rates calculated in the control (no N addition) were also quite low and varied between 0.5 ± 0.05 μgN g^{-1} d^{-1} and -0.1 ± 0.2 μgN g^{-1} d^{-1} in the REF site, that might suggest either a quite slow ammonification phase as a limiting step of the nitrification or a slow nitrification rate. However, when ammonium sulphate was added to soil samples the potential nitrification rates significantly increased reaching 1.8 ± 0.1 μgN g^{-1} d^{-1} and 1.4 ± 0.3 μgN g^{-1} d^{-1} in the SRC and the REF sites respectively, suggesting that mineralization might be the limiting step of subsequent nitrification and denitrification processes in the field.

5.3 Emissions due to management

The GHG emissions due to management practices were in total 100.9 ± 20 $gCO_2eq\ m^{-2}$ for the SRC site and 135.7 ± 27.1 $gCO_2eq\ m^{-2}$ for the REF site. Analysing the single contributions, differences arose between the two sites (Fig. 7): fertilisation was the main source of emission of GHGs in the REF site, while its contribution to GHG emissions of the SRC site was limited. Irrigation constituted a big portion of the GHG emissions from management operations in the SRC site, while in the REF site, despite similar amounts of water provided, irrigation played a smaller role, similar to harvesting and tillage. Emissions due to the latter were more relevant in the REF site than in the SRC site.

5.4 SOC content changes

In the first 15 cm of soil total C stocks were 1603 ± 376 $gC\ m^{-2}$ in the REF site, 1169 ± 442 $gC\ m^{-2}$ in the SRC site and 1403 ± 279 $MgC\ ha^{-1}$ in the O_SRC site. The statistical analysis performed on the SOC stocks showed that there were statistically significant differences between SOC data of the three sites (Table 5; p-value = $2.05*10^{-7}$). The linear regression between SOC content of SRC and O_SRC sites led to the relation:

$$SOC(t) = 78 * t + 857 \tag{7}$$

where *t* are the years from plantation and *SOC* is the soil organic carbon content expressed in $gC\ m^{-2}$. Estimated uncertainty was 25 $gC\ m^{-2}$ for the slope value, and 139 $gC\ m^{-2}$ for the intercept (Fig. 8), meaning that the yearly SOC accumulation after poplar plantation was 78 ± 25 $gC\ m^{-2}$ and the initial value (t=0) was 857 ± 139 $gC\ m^{-2}$, 746 ± 858 $gC\ m^{-2}$ lower than the REF value, corresponding to the SOC content loss due to the installation of the SRC. As this loss was a positive flux occurring only once in a LUC at the installation of the cuttings (Arevalo et al., 2011), and that the expected lifespan of the SRC site was 12 years, the value considered for the 24-month GHG budget was 1/6, corresponding to 124 ± 143 gC m^{-2} (455 ± 524 gCO_2 m^{-2}).

5.5 *Biomass use and GHG offset*

The dry weight of AGB in the REF site summed up to 0.72 ± 0.18 kg m^{-2} for the grassland, of which 0.35 ± 0.07 kg m^{-2} due to the clover in mixture and 0.37 ± 0.17 kg m^{-2} from the sorghum, while the winter wheat totalled 0.63 ± 0.09 kg m^{-2}, of which 0.36 ± 0.05 kg m^{-2} in the ears. The C content measured was 46% for all species, leading to a total of 621.0 ± 93.2 gC m^{-2} in AGB, of which 265.5 ± 79.2 gC m^{-2} ingested by sheep on-site, 191.2 ± 49.8 gC m^{-2} used by livestock off-site, and 163.9 ± 21.9 gC m^{-2} converted to food. The estimated emissions of CH_4 due to enteric fermentation was 4.3 ± 1.3 gCH_4 m^{-2}, equal to 3.3 ± 1.0 gC m^{-2} emitted as CH_4, and thus corresponding to 109 ± 33 gCO_2eq m^{-2} ($E_{CH4,on}$, Eq. (4)). Hence, about 1.25% of the ingested C became CH_4 in the digestive process. Using this ratio led to estimate other 2.4 ± 0.6 gC m^{-2} emitted as CH_4 off-site, i.e. 3.2 ± 0.8 gCH_4 m^{-2}, or 80 ± 20 gCO_2eq m^{-2} ($E_{CH4,off}$). Subtracting the C emitted as CH_4 on- and off-site to the respective digestible C ingested by sheep and other livestock led to 621 ± 189 gCO_2eq m^{-2} emitted on-site ($E_{CO2,on}$) and 447 ± 118 gCO_2eq m^{-2} offsite. Adding to this latter the emissions expected from wheat ears use (i.e. 601 ± 80 gCO_2eq m^{-2}) gave 1048 ± 143 gCO_2eq m^{-2} ($E_{CO2,off}$): in total 1858 ± 240 gCO_2eq m^{-2} in two years (F_{EXP}, Eq.(4)).

For the SRC site, applying Eq. (5) with the diameters distribution led to estimate AGWB (dry matter) in 0.62 ± 0.29 kg m^{-2}, which with a C content of 49%, corresponded to a F_{EXP} of

1118 ± 521 gCO_2eq m^{-2} per two years that are expected to be reemitted to the atmosphere at the combustion. This value of AGWB then corresponded to 8.1 ± 3.7 MJ m^{-2} of gross energy from biomass chips, which decreased to 6.8 ± 3.1 MJ m^{-2} of final heat obtainable from burning biomass chips when the conversion efficiency is considered. This could offset about 379.7 ± 175.1 gCO_2eq m^{-2} from final heat produced using natural gas.

5.6 GHG budgets

All the contributions reported in the previous sections were summed up to calculate the GHG budgets of the two sites. The net GHG budget of the REF site (B_{REF}, Eq. (2)) amounted to 156 ± 264 gCO_2eq m^{-2}, indicating that the REF site was close to neutrality from a GHG perspective, while for the SRC site the B_{SRC} (Eq. (1)) resulted in a cumulative sequestration of -2202 ± 792 gCO_2eq m^{-2}. The different components of the GHG budget of the two sites are summarized in Fig. 9. In the REF site the F_{CO2}, weighing about 48% in the GHG budget, was completely compensated by the emissions of CO_2 and CH_4 due to the biomass utilisation (about 44% and 5% respectively), while the other components had a minor role (F_{MAN} around 4%, soil non-CO_2 <1%). F_{CO2} was the main contribution also in the SRC site, where it represented the 63% of B_{SRC}, while F_{EXP} represented the 20%, SOC loss (8%) and the GHG offset for the fossil fuel substitution (7%) had a similar weight, and the other contributions played a minor role. As B_{REF} was almost neutral and the SRC site a sink of GHGs, the difference between the two GHG budgets was favourable to the SRC site (2358 ± 835 gCO_2eq m^{-2} saved), highlighting the advantages in terms of GHGs of the LUC from common agricultural to SRC of poplar in the study area.

6. **Discussion**

The two ecosystems behaved differently in the measuring period: they were both characterised by a seasonal uptake of CO_2 (Fig. 3), driven by the timing and duration of the growing season, occurring in spring at the REF site and in summer at the SRC site. The peak of CO_2 uptake was similar at both sites in 2012, while it was higher at the REF site in 2013. Periods with positive CO_2 fluxes were longer in the REF site, and often higher in magnitude, likely as a consequence of the shorter growing season of grasses and winter wheat compared to the poplars trees of the SRC site. Also the land cover of the two sites during the dormant periods and the shift in time between them might have played a role on this difference: some herbaceous vegetation kept growing in the SRC site in wintertime, while harvesting and ploughing in the REF site in late summer/early fall might have enhanced ecosystem respiration. Inter-annual differences were also observed at both sites. Both the higher air temperature and the more extended period of low SWC proved the strong aridity of summer 2012, responsible for the autumnal increase of CO_2 uptake at the SRC site, corresponding to the rewetting of the soil. At the REF site, autumn uptake was higher in 2011, while the springtime uptake was much higher in 2013 than in 2012 (Fig. 3). This different behaviour was mostly ascribable to the different cultivations (grassland – winter wheat), and to some extent to the different climate conditions in springtime. All these differences in ecosystems responses resulted in a net sink of GHGs from the SRC site and in a neutral GHG balance for the REF site.

A GHG balance not significantly different from zero is in agreement with the average results for a set of sites in Soussana et al., 2007, where however management costs were not considered, and on-site CO_2 emissions from grazing animals were measured with EC. C sequestered by the SRC site in our study was higher than that of the Belgian site in the study of Zona et al., 2013. In the latter the net budget was positive (in a time span of one year and a half) with a net emission of 280 ± 80 $gCO_2eq\ m^{-2}$, due to both the higher emission rates of CH_4 and N_2O fluxes from soil (350 ± 50 $gCO_2eq\ m^{-2}$), and to the lower CO_2 sink (−80 ± 60 $gCO_2eq\ m^{-2}$) as compared to the present study. Also Jassal et al., 2013 found lower F_{CO2} in a 3-year-old poplar SRC in Canada (-293 $gCO_2\ m^{-2}\ year^{-1}$) as compared to F_{CO2} of the SRC site of the present study (root age: 4 years), likely due to the lower stem density of their site. All these values lied in the range found by Arevalo et al., 2011, i.e. -77 $gCO_2\ m^{-2}\ year^{-1}$ and -4756 $gCO_2\ m^{-2}\ year^{-1}$ relative to a 2-year-old and 9-year-old poplar SRC respectively. These results show that even in a Mediterranean area, where plants are subjected to drought stress, with a proper use of irrigation there is the potential for a positive effect on climate mitigation.

Several studies (Grigal and Berguson, 1998; Price et al., 2009) confirmed that converting agricultural land to SRC resulted in an initial release of SOC due to SRC establishment, and then in a slow and continuous accumulation of SOC due to vegetation activity and wood encroachment (Arevalo et al., 2011). Despite the deep tillage at the SRC establishment, and the fact that the REF site was ploughed every year at different depths, a gradient decreasing with depth in the C distribution of the vertical profile was evident in the three sites (not shown). This suggests that the changes in SOC were attributable only to the plantation of the SRC due to the effects of tillage (Anderson-Teixeira et al., 2009), and not to the mechanical redistribution of SOC. This study indicates a SOC loss of 47% in respect to the value measured in the REF site, due to the installation of poplar cuttings. This loss was not measured at the time it occurred, i.e. right after the conversion of arable land to poplar short rotation coppice, but was estimated with data from the O_SRC site. The reported value was close to the range maximum reported in the review by Post and Kwon, 2000 (20%-50%), but was higher than the results found by Arevalo et al., 2011 (7%). The absolute value, however, was close to the one of this latter study (8 $MgC\ ha^{-1}$), where though the initial SOC was one order of magnitude higher (114.7 MgC ha^{-1}). To correctly interpret this rapid loss of SOC for a conversion of a cropland to a SRC the low degree of disturbance that characterises the REF site must be taken into account. Furthermore the loss of SOC found in the present study has to be considered along with its own uncertainty that was as large as the estimated value: in the purposes of the GHG balance, where the uncertainty of the single components are propagated to the net budget, this result is correctly interpreted as a range. We highlight that a loss of SOC close to the minimum of the abovementioned range by Post and Kwon, 2000, e.g. 321 gC m^{-2}, would have changed B_{SRC} (-2202 ± 792 $gCO_2eq\ m^{-2}$) by only -259 $gCO_2eq\ m^{-2}$. Thus, even if a measured value would have probably been more accurate, the sensitivity of the total GHG budget to this loss was shown to be relatively low. The estimated annual SOC accumulation rate was in the range reported by Don et al., 2011 for SRCs (0.44 ± 0.43 MgC ha^{-1} y^{-1}), which explained how the frequent harvest of above ground biomass was likely to facilitate the die off of the roots that contributes to SOC accumulation. In our study, the low biomass yield supports the hypothesis that a big fraction of C taken up via photosynthesis was transferred to roots and soil. In our study the break-even point, where the initial SOC content would be restored and a net SOC accumulation would start, was 10 years, in agreement with findings from other studies (e.g. Hansen, 1993; Arevalo et al., 2011 found a value of 7 years, while Grigal and Berguson, 1998, calculated a break-even point of 15 years). This result, not directly involved in the 24-month GHG budget, is relevant considering that the SRC of the present study is expected to be used for 12 years, thus enough to allow the complete recovery of the SOC loss occurred at the plantation. Different previous land uses, soil types (in particular clay content), climate conditions, fertilisation rates may be the main causes of differences between studies, as shown in a meta-analysis by Laganière et al., 2010.

Our results showed that CH_4 and N_2O soil fluxes were not relevant in the GHG budgets due to the combination of soil characteristics and climatic trends at both sites. Low values are reported in other studies for SRCs: e.g. Gauder et al., 2012 found that the soil of different energy crops acted as weak sink of CH_4 even in case of fertilisation, while emissions of N_2O turned out to be higher for annual than perennial (willow) crops, which showed no significant effect of fertilisation on N_2O fluxes. Agricultural sites usually have higher N_2O effluxes from soil, though their magnitude depends on cultivations and on management practices, as shown by Ceschia et al., 2012. The SRC site as a perennial woody crop was subjected to low soil disturbance during its lifespan, while the REF site was ploughed once per year, with an impact on the ecosystem respiration. Zona et al., 2012 found high N_2O emissions in the first growing season of a poplar SRC in Belgium: 197 ± 49 $gCO_2eq\ m^{-2}$ in six months, which drastically decreased to 42 ± 17 $gCO_2eq\ m^{-2}$ for the whole following year. This suggested an influence of soil disturbance during land conversion on the stock of N in soil, which was almost 1/3 lower in our study sites than in the one of Zona (9.1 ± 2.1 $MgN\ ha^{-1}$). In the present experiment however, N_2O fluxes were low both in the SRC and REF sites, even during periods of fertilisation, with no clear patterns. The low N_2O fluxes were confirmed by laboratory analyses, as the presence of extra N did not affect the emission rates of N_2O, and only very high WFPS% could trigger significant N_2O fluxes. The needed conditions of soil humidity were never reached in the REF site and reached only for a few days at 35 cm depth in the SRC site (Fig. 6). At this depth fertilizer was added as fertigation in the SRC site: we hypothesize that the very low porosity, the compaction and strength of the soil might have favoured slow gas release and further N_2O reduction, thus leaving little N_2O to escape to the atmosphere from soil surface. In the REF site, winter fertilisation was also associated with low temperatures, a further constraint to microbial activity. These results provide further evidence of how the simple application of the IPCC N_2O emission factor to the analysed systems might have led to an overestimation of the field GHG contribution to the overall GWP in both sites. Laboratory estimates of mineralization and nitrification rates suggested that N mineralization might be the limiting process of the chains of mineral N microbial transformations, that contributed to maintain N_2O emissions low even during events of intense rainfall and soil saturation. The clay content and compaction of the analysed soils might be an important factor in limiting oxygen and substrate diffusion that are both necessary to have optimal rates of soil organic matter mineralization. From a methodological point of view, the low emissions of both CH_4 and N_2O from soil also suggest that using 4 samples of gas concentration per chamber instead of 3 would have not dramatically improved the accuracy of the calculated fluxes, as a slight variation in the slope would have not induced significant changes in the results. The relevance of this result lies in the fact that fertilising a poplar SRC in a Mediterranean area and in this kind of soil does not necessarily lead to increased emissions of N_2O, with the requirement that the correct equilibrium is found between irrigation and WFPS%. Thus it is possible to maximise yield and GHG mitigation with the right management practices (Nassi o Di Nasso et al., 2010). CH_4 and N_2O fluxes might have been enhanced by the land conversion in the first period of cultivation of the SRC site, as found for CO_2. However, measurements carried out in the REF site, ploughed every year, and the incubation experiment showed very low fluxes, mostly related to soil characteristics and not to management activities. Thus, a low sensitivity of the total GHG budget to these components can be expected.

Other components of the GHG budget related to N compounds (e.g. aerosol NH_4NO_3, N deposition and leaching) were considered negligible in this study as compared to the role of N_2O emissions from soil and related to fertiliser production.

Regarding the use of biomass, comparisons with other studies for the REF site are complicated because half of the field was converted to sorghum in spring for the low productivity experienced during the drought. However, the productivity of the clover in mixture was found highly variable by Martiniello, 1999, and the results of the present study are comparable with the lower values found by this author in non-irrigated stands in Mediterranean climate (0.39 kg m^{-2}). Sorghum productivity was lower than that reported by Nassi o Di Nasso et al., 2011 (around 0.75 kg m^{-2}) in a similar climate, likely due to the short period of cultivation and to grazing. The productivity of winter wheat was similar to that of Anthoni et al., 2004 (0.32 ± 0.03 kg m^{-2}). The drought in summer 2012 had an important influence on the AGWB of the SRC site, which was lower compared to other studies (e.g. Scholz and Ellerbrock, 2002, 0.4 to 0.7 kg m^{-2} $year^{-1}$), and to the F_{CO2} values found with EC. Our hypothesis is that the period of drought had influenced the aboveground/belowground ratio, and that the herbaceous vegetation contributed to increase the F_{CO2}. In terms of C, the difference $F_{CO2} - F_{EXP}$ represents to a first approximation the C stocked by each ecosystem that does not return shortly to the atmosphere after utilisation, minus heterotrophic respiration (Rh). While in the SRC site that difference was negative (C sink of 650 gC m^{-2}), the REF site acted like a small source of C (120 ± 98 gC m^{-2}). Small sources were also found by Anthoni et al., 2004 (between 50 gC m^{-2} and 100 gC m^{-2}), while Aubinet et al., 2009 reported a 4-year rotation crop being a source of 340 gC m^{-2}. For poplar, Deckmyn et al., 2004 found a similar behaviour in a poplar SRC in Belgium. Concerning the fraction of the exports that is emitted as CH_4 from enteric fermentation, our estimates were in agreement with those of Dengel et al., 2011. Several studies (e.g. Gilmanov et al., 2007) used EC to measure CO_2 and CH_4 fluxes from grazed systems. Some included in the GHG budget only F_{CO2}, F_{CH4} and F_{N2O}, and made a C budget for lateral fluxes like biomass export (e.g. Allard et al., 2007). However, the EC method is not capable of measuring point sources of trace gases moving inside and outside the footprint (data discarded by QA/QC procedures: see also Baldocchi et al., 2012). Thus we adapted the method described in Soussana et al., 2007 for off-site emissions, extending it also to on-site emissions, to include the effects of aboveground biomass use in the GHG budget.

Different studies (e.g. Cherubini et al., 2009; Djomo et al., 2013) confirmed the advantages of using biomass from SRC over fossil fuels in mitigating the increase of atmospheric GHG concentrations, while Abbasi and Abbasi, 2010 found that the SRC management led to GHG emissions that compensate the gain due to the fossil substitution. The low yield of the SRC site led to lower GHG savings compared to those found by Cherubini et al., 2009 for production of heat from woody products (379.7 ± 175.1 gCO_2eq m^{-2} in two years against 600 gCO_2eq m^{-2} per year). These latter found GHG mitigation to be directly proportional to crop yield for dedicated bioenergy crops. In a GHG budget perspective, however, the yield is also proportional to C emissions from combustion, and correlated with F_{CO2}. The same study reported GHG savings of other bioenergy systems, showing that the performance of wood-based systems is lower in terms of GHG offset than the one of other bioenergy crops, e.g. switchgrass (1300 gCO_2eq m^{-2} $year^{-1}$), Miscanthus (1600 gCO_2eq m^{-2} $year^{-1}$) and fibre sorghum (1800 gCO_2eq m^{-2} $year^{-1}$). In the present study the role of GHG offset was relevant in the GHG balance; however it's important to consider that natural gas, while being the most used fossil fuel for heating systems in Italy, has also a lower carbon intensity for heat production (55.862 gCO_2eq MJ^{-1}) as compared to coal (76.188 gCO_2eq MJ^{-1}) and oil (73.693 gCO_2eq MJ^{-1}) (Romano et al., 2014). A different scenario, where biomass would substitute the use of other energy sources with higher emission factors (like coal) would lead to a higher GHG offset.

Our study confirmed that farming operations have only a limited importance in the overall GHG budget when conditions of relevant CO_2 uptake by vegetation are met, and the values we found were similar to the ones found by Gelfand et al., 2011. In the SRC site irrigation was more important than other contributions and caused more emissions than irrigation in the

REF site. This suggests that belowground irrigation was less efficient in terms of GHG emissions than the sprinkler. Fertilisers and other chemical products often have a higher impact on the GHG balance as compared to other field operations due to the off-site GHG emissions (Ceschia et al., 2010). At the study sites the amount and frequency of applications were relatively small, and this justifies the minor role of fertilisation in the total GHG budget. Thus the importance of farming operations can vary from year to year, depending on climate conditions and on farmer decisions.

This study reports on the GHG budget of poplar SRC in Mediterranean areas. However, when considering the implications of SRC in a wider perspective, other factors should also be considered to assess the overall sustainability of this type of LUC. Among them, irrigation is one of the most important (Dougherty and Hall, 1995), as poplar cultivations in Mediterranean climate require considerable amounts of water. In the LUC presented here, both the SRC and the REF sites were irrigated with similar amounts of water, using a less efficient technique at the REF site (sprinkler system) than at the SRC site (belowground drip system; e.g. Camp, 1998). The impact of the LUC on the local water balance is thus expected to be small in this particular case, but not in general. An appropriate design of these systems is also crucial to avoid water dispersion: in the present study we observed that irrigation could not compensate the drought stress experienced by the SRC site in 2012, thus concerns arise on the proper location of the belowground tubes and on the amounts of water applied.

7. Conclusions

The comparison of the two net GHG budgets led to conclude that poplar SRC cultivation for biomass production in the analysed sites of Central Italy was suitable from the point of view of the climate mitigation at farm level when this is performed converting former agricultural land. The cultivation and use of the SRC site in the place of traditional crop rotation led to a reduction of GHG concentration in the atmosphere, even taking into consideration the disadvantages of the SOC content loss at the installation of the SRC. This result was in agreement with previous studies on Mediterranean climate, where the cultivation of poplar SRC may be critical for its dependence on water availability, but with possibility of success (see for example Gasol et al, 2009). In our study, however, the inclusion into the net GHG budget of all the contributions, from the management and biological activities to the use of the biomass and the effects of the land use change on the SOC content, highlighted the importance of the C distribution in respect to the biomass use, whereas the SOC loss at the installation, while being an important part of the budget, did not result to be crucial in the evaluation of LUC suitability. Estimated uncertainty was quite large, underlining the high variability of the GHG budgets and confirming the need of large efforts in terms of data collection to correctly estimate the different components. Furthermore in this type of analyses there is a set of factors – like climatic conditions, irrigation and farmer needs – that influence the sensitivity of the net GHG balance, acting on the F_{CO2}, the biomass yield, the emissions from management activities and the offset of GHG (Cherubini et al., 2009). The magnitude of the benefits deriving from the LUC from common agriculture to SRC of hybrid poplar for biomass production, thus, depends on the interaction between the diverse components of the budget and their variability.

7. Acknowledgments

This research was supported by the GHG Europe Project (EU Seventh Framework Programme, FP7) and the MIUR-PRIN project "Climate change mitigation strategies in tree crops and forestry in Italy" (CARBOTREES). The authors wish to acknowledge the technical employees of DIBAF (A. Baiocco, A. Boschi, M. Tomassucci), CNR IBAF (G. Santarelli) and CNR IBIMET (A. Zaldei) for their irreplaceable support and logistic help during the field campaign, and the database staff (E. Canfora, A. Ribeca, D. Polidori, C. Trotta) for their support in the data post-processing. S. Sabbatini is supported by the ICOS-Inwire EU project and S. Njakou Djomo was supported by the ERC Advanced Grant 233366 POPFULL. We further thank G. Parisi for gas chromatography analysis. We are also grateful to the farm *Gisella ed Elena Ascenzi S.A.A.S.*, and in particular to A. Trani, the farmer that kindly allowed us to install our instruments on his fields and to access his farm whenever we needed it and that supported our scientific activities with his interest and availability. All the data collected are available at the European Ecosystem Fluxes Database at www.europe-fluxdata.eu, site codes IT-CA1, IT-CA2 and IT-CA3.

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9. Tables

Table 1 – Instrumental setup of the two towers. SRC = short rotation coppice site; REF = reference site; T = temperature; RH = relative humidity; PAR = photosynthetically active radiation; M_{SOIL} = soil water content; P = precipitation; EC = eddy covariance; prof = profile. 4-component radiometers were used to measure short- and long-wave radiations, and derive net radiation. SRC site soil profiles were located in irrigated and not-irrigated inter-rows. Precipitation was assumed to be consistent in the two ecosystems.

	SRC	REF
Meteo		
Air T and RH	MP-100, Rotronic AG, Bassersdorf, CH	MP-100, Rotronic AG, Bassersdorf, CH
PAR	Li-190, LI-COR, Lincoln, NE, USA	-
Radiations	CNR-1, Kipp&Zonen, Delft, NL	NR01, Hukseflux, Delft, NL
M_{SOIL}	CS616, Campbell Scientific, Logan, UT, USA (2 prof.)	CS616, Campbell Scientific, Logan, UT, USA (1 prof.)
Soil T	107, Campbell Scientific, Logan, UT, USA (2 prof.)	107, Campbell Scientific, Logan, UT, USA (1 prof.)
Soil heat flux	HFT3, REBS Inc., Seattle, WA, USA	HFP01, Hukseflux, Delft, NL
P	-	ARG100, EML, North Shield, UK
Logger	CR3000, Campbell Scient., Logan, UT, USA	CR1000 Campbell Scient. Logan, UT, USA
EC		
Anemometer	CSAT3, Campbell Scientific, Logan, UT, USA	USA-1, Metek GmbH, Elmshorn, DE
Gas-Analyser	LI-7500, LI-COR, Lincoln, NE, USA	LI-7500A, LI-COR, Lincoln, NE, USA

Table 2 – Correction steps applied to the time series using LICOR EddyPro software.

Correction	Reference
Despiking	Vickers and Mahrt (1997)
Density fluctuations	Webb et al. (1980)
Maximisation of covariance for time lag compensation	Aubinet et al. (2000)
Linear detrending for trend removal	Gash and Culf (1996)
2-D coordinate rotation	Wilczak et al. (2001)
High-pass filtering effect	Moncrieff et al. (1997)
Low-pass filtering effect	Ibrom et al. (2007)

Table 3 – Farming activities. Three tractors of different power were normally used to collect chips: two of the type 1, and one of the type 2. DAP = diammonium phosphate; AN = ammonium nitrate; CAN = calcium ammonium nitrate. SRC and REF as defined previously.

Operation	Fuel consumption (unit ha^{-1})	Input rates (unit ha^{-1})	Site
Harvesting – wood chipper	30 l diesel	-	SRC
Harvesting – Tractor type1	20 l diesel	-	SRC
Harvesting – Tractor type 2	10 l diesel	-	SRC
Shallow tillage	8 l diesel	-	SRC, REF
Application of insecticide	1.125 l diesel	1.25 kg DECIS®	SRC
Mechanical weeding	4 l diesel	-	SRC
Ploughing	8 l diesel	-	SRC, REF
Sowing	2 l diesel	-	REF
Seed covering	4 l diesel	-	REF
Application of fertiliser	2 l diesel	a. 150 kg DAP b. 150 kg AN c. 200 kg CAN d. 40 kg Urea	a. REF b. REF c. REF d. SRC
Reaping	20 l diesel	-	REF
Chemical weeding	1.125 l diesel	1 l Buctril®	REF
Bale	7.5 l diesel	-	REF
Irrigation	a. 471 kWh electricity b. 149 kWh electricity	a. 16 l H_2O b. 46 l H_2O	a. SRC b. REF

Table 4 – Grazing calendar and methane emissions in the REF site. Graz_days = number of days with grazing; Num = number of sheep in the cropland

Months	**Graz_days**	**Num (per 9 ha)**
Dec 2011	10	800
Jan 2012	7	400
Jun 2012	2	580
Aug 2012	1	580
Sep 2012	2	580
Oct 2012	5	400

Table 5 – Soil characteristics of the ecosystems of the layer 0-15 cm. SRC and REF as previously defined; SOC = soil organic carbon; ρ_{BULK} = bulk density. Superscripts a-c indicate statistically significant differences between the means of SOC.

Site	Variable	Value ± dev. std.
REF	C (%)	1.46 ± 0.34
	ρ_{BULK} (Mg m^{-3})	1.00 ± 0.11
	SOC (MgC ha^{-1})	16.03 ± 3.76$^{(a)}$
SRC	C (%)	1.05 ± 0.40
	ρ_{BULK} (Mg m^{-3})	1.12 ± 0.15
	SOC (MgC ha^{-1})	11.69 ± 4.42$^{(b)}$
O_SRC	C (%)	1.38 ± 0.27
	ρ_{BULK} (Mg m^{-3})	1.02 ± 0.11
	SOC (MgC ha^{-1})	14.03 ± 2.79$^{(c)}$

10. Figures

Fig. 1 – Scheme of the chronological land cover during the cultivation cycle taken into account for GHG budget calculation in the two ecosystems. The expected harvest of poplar at the beginning of 2014 was postponed of one year: for that reason data from the previous harvest (beginning 2012) were taken into account for GHG budget calculation. Textures indicate different land cover type, symbols mark the most important management practices, straight lines indicate the periods in which sites were irrigated, dashed line period of grazing. SRC = short rotation coppice site; REF = reference site; in the x axis dates are reported as month-year (mm-yy)

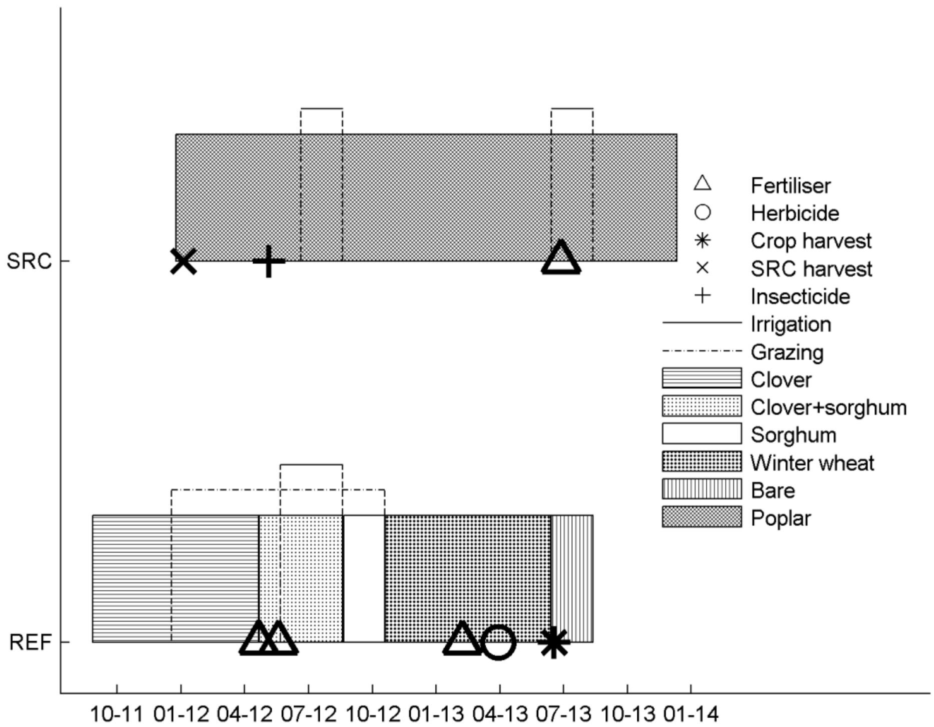


Fig. 2 – Boxplot of the 24-month cumulative fluxes of net ecosystem exchange of CO_2 (F_{CO2}) (a), gross primary production (GPP) (b) and ecosystem respiration (Reco) (c) from eddy covariance (EC) data in the REF and SRC sites. Each box represents the range 16th-84th percentile: the central mark is the median, while the whiskers extend to the 5th and 95th percentiles

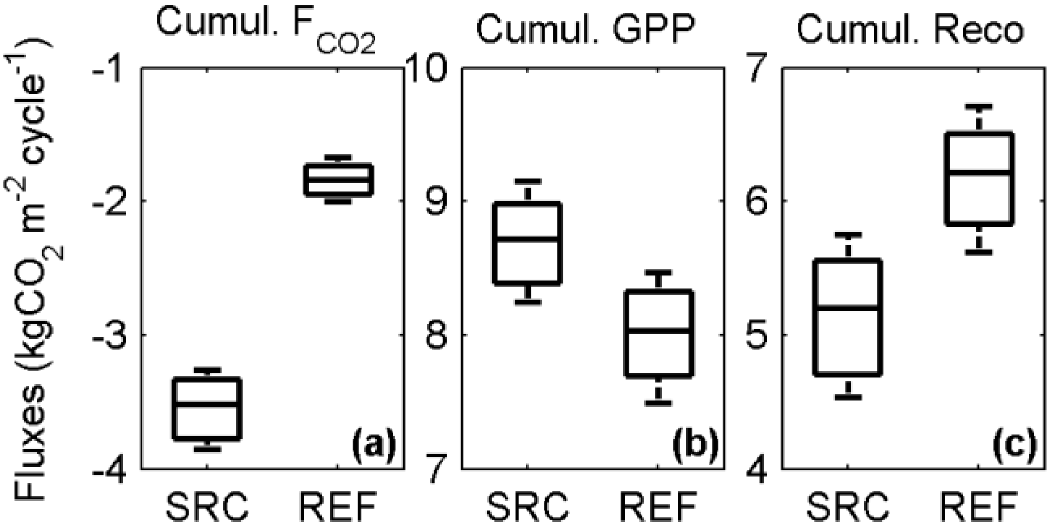


Fig. 3 – Monthly averages of F_{CO2} in the REF and SRC sites (top panel). The bottom panel shows monthly averages of air temperature (Tair) and soil water content (SWC) at 30 cm depth. In both subplots dotted lines are used for the SRC site and continuous lines for the REF site, while in the bottom panel SWC is in grey and the Tair in black.

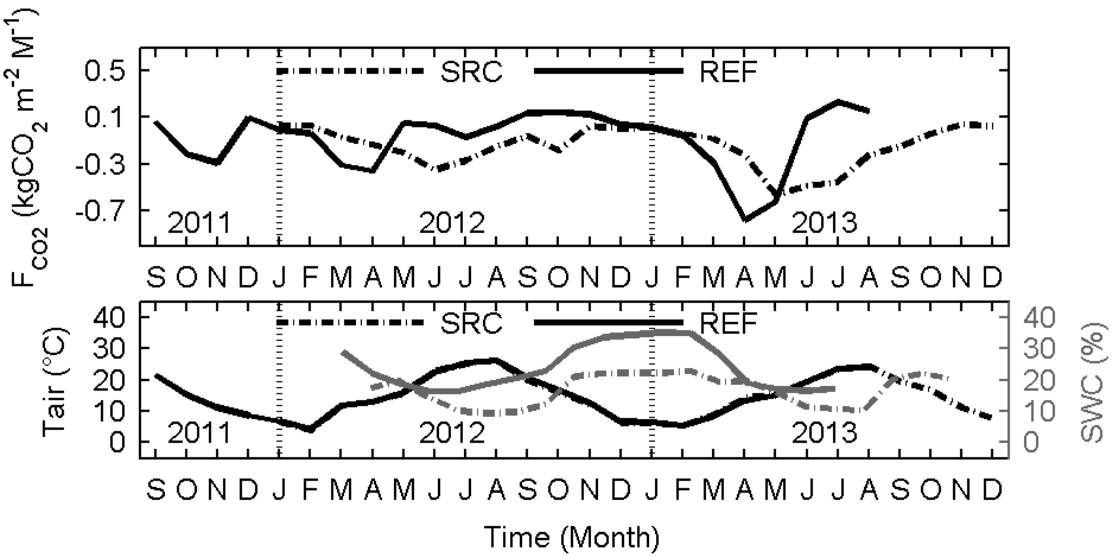


Fig. 4 – Fluxes of soil N_2O (crosses) and CH_4 (circles) in the SRC (a –c) and the REF (b – d) sites. Each marker represents the average of the nine chambers, with bars indicating their standard deviation. First letter of month in the x-axis

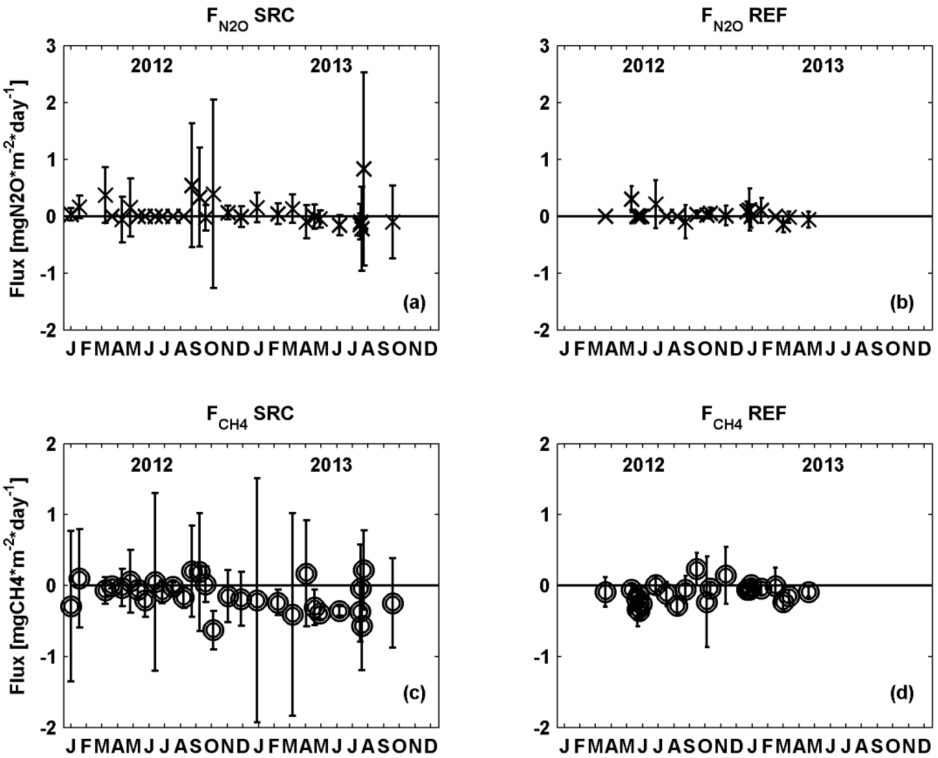
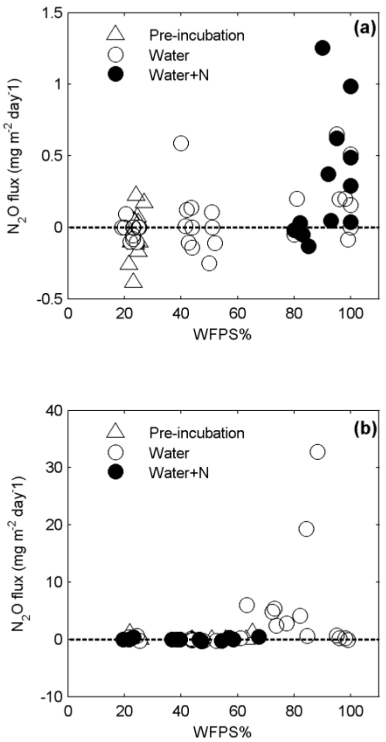


Fig. 5 – N_2O fluxes from incubation experiment reported in function of the water filled pore space estimated for each single replicate. In (a) data from samples taken in the SRC site are shown, in (b) data from REF site samples



1140 **Fig. 6 – WFPS% in the REF site at three different depths for the 24-month integration periods. Dashed line points to**

1141 **the threshold (70%) unleashing N_2O from lab incubations. First letter of month in the x-axis**

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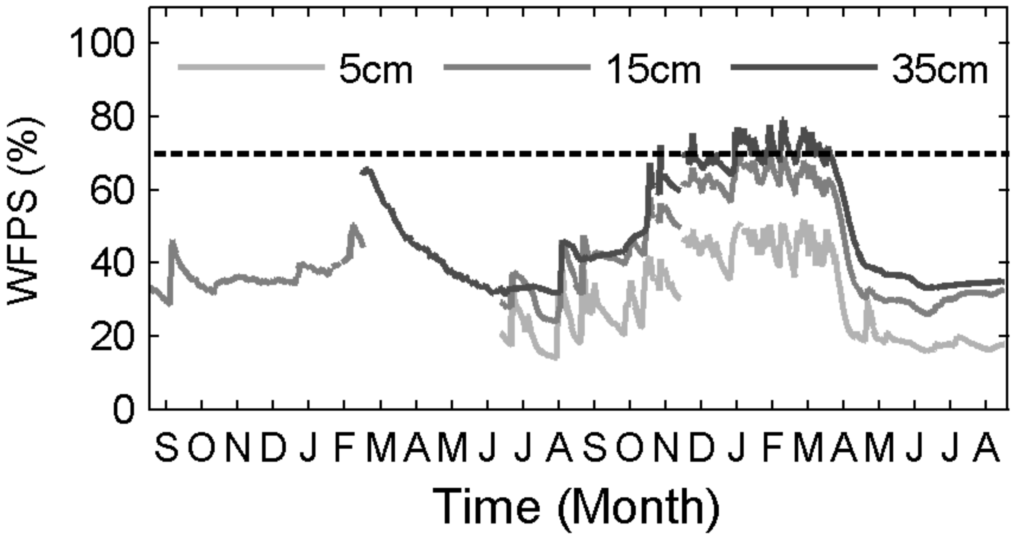

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Fig. 7 –GHG emissions of the different farming operations. Harv = harvesting; plow = ploughing; sow = sowing; irr = irrigation; fert = fertilisation; othe= minor contributions. SRC and REF as previously defined

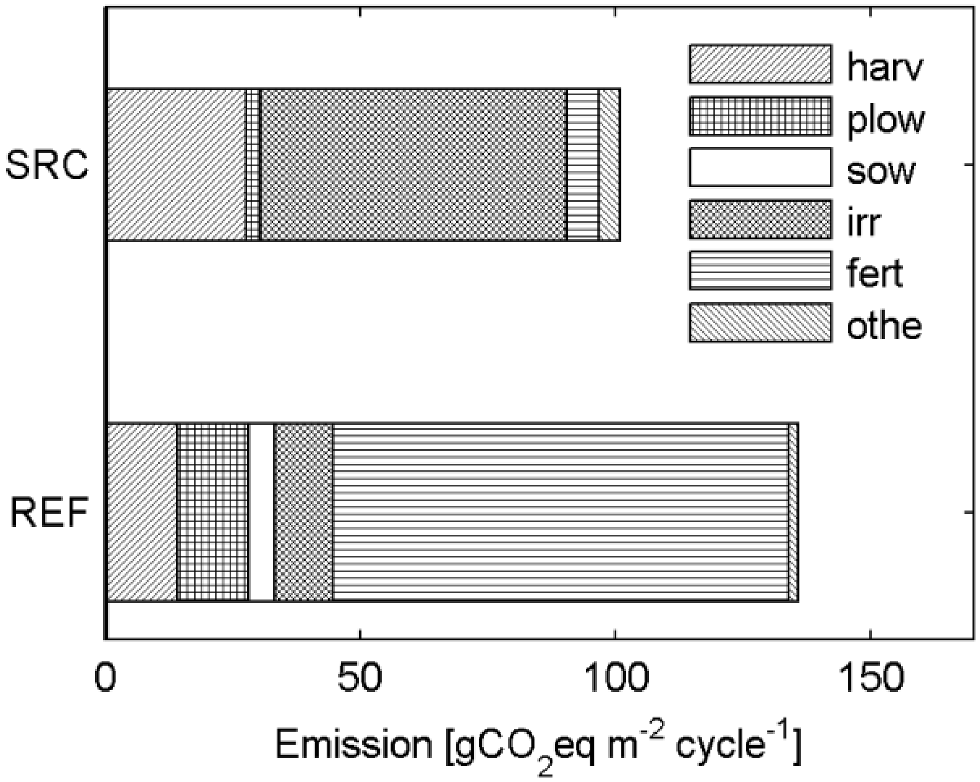

Fig. 8 – Regression line of SOC content in time *t* (years). The gap between SOC(0) and SOC content in the REF site represented the loss of SOC for the land use change. Est = estimated values; meas = measured values; SRC and REF as previously defined; O_SRC is the older short rotation coppice site used to build the regression

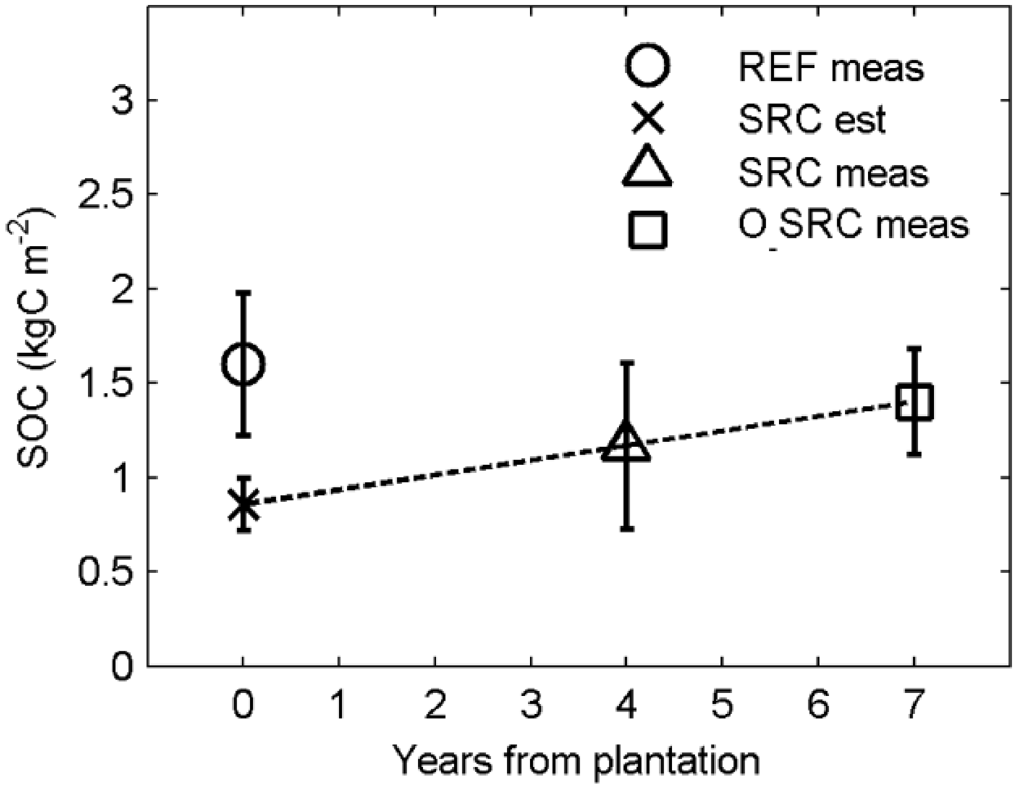

Fig. 9 – GHG balances of the SRC and the REF sites: components (left) and net (right). F_{CH4} and F_{N2O} from soil are negligible and not inserted in the graph. F_{MAN} = management; E_{CH4} = exported biomass reemitted as CH_4 by enteric fermentation; E_{CO2} = exported biomass reemitted as CO_2 by sheep respiration; F_{SOC} = initial SOC change at the installation of cuttings; F_{SAV} = GHG savings for replacement of fossil fuel use; F_{CO2} as previously defined