Management effects on net ecosystem carbon and GHG budgets at European crop sites

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A B S T R A C T

The greenhouse gas budgets of 15 European crop sites covering a large climatic gradient and corresponding to 41 site-years were estimated. The sites included a wide range of management practices (organic and/or mineral fertilisation, tillage or ploughing, with or without straw removal, with or without irrigation, etc.) and were cultivated with 15 representative crop species common to Europe. At all sites, carbon inputs (organic fertilisation and seeds), carbon exports (harvest or fire) and net ecosystem production (NEP), measured with the eddy covariance technique, were calculated. The variability of the different terms and their relative contributions to the net ecosystem carbon budget (NECB) were analysed for all site-years, and the effect of management on NECB was assessed. To account for greenhouse gas (GHG) fluxes that were not directly measured on site, we estimated the emissions caused by field operations (EFO) for each site using emission factors from the literature. The EFO were added to the NECB to calculate the total GHG budget (GHGB) for a range of cropping systems and management regimes. N_2O emissions were calculated following the IPCC (2007) guidelines, and CH_4 emissions were estimated from the literature for the rice crop site only. At the other sites, CH_4 emissions/oxidation were assumed to be negligible compared to other contributions to the net GHGB. Finally, we evaluated crop efficiencies (CE) in relation to global warming potential as the ratio of C exported from the field (yield) to the total GHGB. On average, NEP was negative (-284 ± 228 g C m^{-2} year^{-1}), and most cropping systems behaved as atmospheric sinks, with sink strength generally increasing with the number of days of active vegetation. The NECB was, on average, 138 ± 239 g C m^{-2} year^{-1}, corresponding to an annual loss of about 2.6 ± 4.5% of the soil organic C content, but with high uncertainty. Management strongly influenced the NECB, with organic fertilisation tending to lower the ecosystem carbon budget. On average, emissions caused by fertilisers (manufacturing, packaging, transport, storage and associated N_2O emissions) represented close to 76% of

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EFO. The operation of machinery (use and maintenance) and the use of pesticides represented 9.7 and 1.6% of EFO, respectively. On average, the NEP (through uptake of CO_2) represented 88% of the negative radiative forcing, and exported C represented 88% of the positive radiative forcing of a mean total GHGB of 203 ± 253 g C-eq m^{-2} year^{-1}. Finally, CE differed considerably among crops and according to management practices within a single crop. Because the CE was highly variable, it is not suitable at this stage for use as an emission factor for management recommendations, and more studies are needed to assess the effects of management on crop efficiency.

1. Introduction

The impacts of agriculture on global climate change through greenhouse gas (GHG) emissions and changes in land cover have been summarised in the recent analyses of Desjardins et al. (2007) and Raddatz (2007). Agriculture contributes to the emission of GHGs through disturbance of soil and vegetation carbon pools (e.g., ploughing/tillage and management of crop residues) and the biospheric fluxes of other GHGs, but also through field or farm operations. (e.g., emissions of fossil fuels from energy sources needed for tillage practices or in the application of organic amendments and chemicals). Among the biospheric fluxes, net CO_2 ecosystem production (NEP) can be measured at the plot or field scale using the eddy covariance (EC) method, but additional measurements are needed to estimate net biome productivity (NBP) or the net ecosystem carbon budget (NECB) of croplands (see Smith et al., 2010). The net ecosystem carbon budget (NECB) is a term applied to the total rate of organic carbon accumulation (or loss) from ecosystems (Chapin et al., 2006). When integrated over time and space, the NECB equals the NBP (Schulze & Heimann, 1998; Buchmann & Schulze, 1999; Chapin et al., 2006). For croplands, the NBP can be assessed over the long term based on analysis of the changes in soil carbon stocks, or it can be assessed over shorter time scales (e.g., annual) by combining NEP measurements with estimates of C inputs (e.g., seeds, tubers, organic fertiliser) and C outputs (e.g., harvest, DOC) (see Aubinet et al., 2009; Smith et al., 2010). An annual approach is useful because it allows assessment of the effects of individual crops or particular climatic or management events on the NBP while the monitoring of soil C stock variations smooth out short-term effects associated with organic matter fractions presenting a rapid turnover rate.

Most studies assessing the NEP, NECB or NBP based on the EC methodology have focussed on forests or grasslands, but only a few have dealt with croplands, in part due to the difficulties and uncertainties associated with estimating the cropland carbon budget (see Osborne *et al*., 2010). Among those examined, maize/soybean rotations in North America have received the most attention (Baker and Griffis, 2005; Bernacchi et al., 2005; Hollinger et al., 2005; Pattey et al., 2002; Suyker et al., 2005; Suyker et al., 2004; Verma et al., 2005). Although rice (Saito et al., 2005), sugar beet (Moureaux et al., 2006), winter wheat and triticale (Ammann et al., 1996; Anthoni et al., 2004; Baldocchi, 1994; Moureaux et al., 2008; Béziat et al., 2009), and sunflower, rapeseed or maize for silage (Béziat et al., 2009) have also been investigated, these studies do not provide a comprehensive assessment that accounts for the impact of regional differences in crops and cropping systems or management practices. In a modelling study at the European scale by Janssens et al. (2003), the NECB for croplands was estimated to be 90 ± 50 g C m^{-2} year^{-1}. However, this contrasts with more recent studies based on modelling and carbon inventories that suggest that European cropland soils are close to equilibrium, acting as either small sources (3 g C m^{-2} year^{-1} in Schulze et al., 2009; see also Smith et al., 2005; Bondeau et al., 2007;) or small sinks (16 ± 15 g C m^{-2} year^{-1} in Gervois et al., 2008).

To deepen our present understanding of cropland GHG fluxes, the CarboEurope-IP project (2004–2008) has provided a unique opportunity to extend studies of the NEP to assess the NECB and NBP and their variations with climate and management for representative croplands in Europe (see Eugster et al., 2010; Kutsch et al., 2010; Osborne et al., 2010; Moors et al., 2010). Other GHGs were measured at some sites but rarely in a continuous or systematic way.

Other experimental studies and analyses have addressed C and GHG emissions associated with field or farm operations (Koga et al., 2003; Lal, 1997, 2004; Gaillard, 1997; ADEME, 2007; St Clair et al., 2008; Hillier et al., 2009; Eugster et al., 2010). Such studies can be used to recommend management practices that limit carbon loss-based operations and products, including associated off-farm or external inputs (Pimentel, 1992; Marland et al., 2003; IPCC, 2006). Considering the contribution of field operations together with assessments of GHG emissions and sinks (CO_2, CH_4 and N_2O), it is possible to estimate a large part of the net radiative forcing due to crop growth and management. This examination can be done using the concept of a global warming potential (GWP) (Houghton et al., 2001). However, to evaluate the full radiative forcing for crops, albedo effects should be accounted for, but this is beyond the scope of our paper.

Only a few studies have presented combined measurements of the biospheric fluxes and emissions of GHGs caused by field or farm operations. Robertson et al. (2000) and Robertson and Grace (2004) compared the GWPs for several crop rotation and management regimes based on changes in soil carbon stocks, CH_4 and N_2O chamber measurements and estimates of the emissions associated with some field operations, such as inputs (fertilisers, lime) and soil work (fuel consumption). Byrne et al. (2007) estimated C sequestration and the net greenhouse gas budget of a grassland in Ireland using eddy covariance data combined with a farm-scale carbon budget. Allard et al. (2007) and Soussana et al. (2007) also estimated the effects of management on NBP and the GHG budget (GHGB) of grasslands. To our knowledge, no comparable studies have been published for croplands using eddy covariance measurements.

In this paper, we (1) analysed the variability of Net Ecosystem Production (NEP) measured with the eddy covariance technique, as well as carbon inputs (mainly through organic fertilisation) and carbon exports (during harvest), and examined their relative contributions to the NECB for croplands at several European crop sites; (2) evaluated the effect of management on the NECB; (3) estimated the emissions caused by field operations reported at the plot scale; (4) combined the NECB and emissions caused by field operations to estimate the total GHGB for a range of cropping systems and evaluated the effects of management; and finally, (5) evaluated crop efficiency in relation to the total net GHGB as the ratio of C exported from the field (yield) to total GHGB. For points 1, 2, 4 and 5, data from 15 European cropland sites were available (41 site-years), and for point 3, data from 17 sites (51 site-years) were used.

2. Material and methods

In this section, we will describe the methods used to assess the main biospheric and non-biospheric fluxes (emissions and sinks) contributing to the GHGB at the plot scale (see Fig. 1). Albedo effects were not considered. The crop species studied cover more than

Table 1
Main characteristics of the sites, management practices, net primary productivity (NPP), amount of residue left on the field, and date of the starting period for calculation of the NEP.

Site Name - Country	Years	Latitude, Longitude	Crop history	Crop	Soil preparation	Fertilisation	Irrigation (mm)	NPP (T DM ha^{-1})	Residues (T DM ha^{-1})	NEP starting date
Auradé FR	2004–2005	43°54′97″N, 0°10′61″E	Cultivated, mineral fertiliser > 30 years	Rapeseed	Direct	Mineral	0	12.7	7.4	
	2005–2006			winter wheat	Multiple	Mineral	0	13.4	7.4	01/10/2005
	2006–2007			Sunflower	Ploughing	Mineral	0	8.4	6.2	21/09/2006
Avignon FR	2003–2004	43°55′00″N, 4°52′47″E	Cultivated, mineral fertiliser > 20 years. Main cropswere durum wheat, maize and sunflower	Durum wheat	Multiple	Mineral	0	19.9	9.2	01/01/2004
	2004–2005			Peas	Multiple	Mineral	60	6.5	4.3	01/09/2004
	2005–2006			Durum wheat	Multiple	Mineral	20	20.7	13.5	01/09/2005
	2006–2007			Sorghum	Multiple	Mineral	80	18.7	13.5	06/11/2006
Cioffi IT	2004–2005	40°31′26″N, 14°57′27″E	Cultivated, organic manure > 40 years on a former swamp	Rye-grass/maize	Multiple	Mineral + organic	330	10.1–14.1	5.8–1.6	01/09/2004
	2005–2006			Fennel/maize	Multiple	Mineral	300	5.7–18.6	3.7–1.7	25/08/2005
	2006–2007			Fennel/maize	Multiple	Mineral	416	7.3–17.7	6.3–1.4	25/08/2006
Carlow IR	2004–2005	52°51′29″N, 6°55′1″O		Spring barley	Ploughing	Mineral	0	12.4	7.3	01/03/2004
	2005–2006			Spring barley	Ploughing	Mineral	0	13.5	7.9	01/12/2004
	2006–2007			Spring barley	Ploughing	Mineral	0	13.5	8.1	01/02/2006
Dijkgraaf NL	2006–2007	51°59′31″N, 5°38′45″E	Cultivated, organic manure > 10 years. Maize and grassland	Maize	Ploughing	Organic	0	20.3	4.0	01/10/2006
Gebesee GE	2003–2004	51°06′00″N, 10°54′51″E	Cultivated > 40 years. Organic manure until the 1980s	Rapeseed	Tillage	Mineral	0	10.9	6.8	05/08/2003
	2004–2005			Winter barley	Tillage	Mineral	0	12.4	6.2	05/08/2004
	2005–2006			Sugar beet	Deep tillage	Mineral	0	22.8	5.7	28/10/2005
	2006–2007			Winter wheat	Tillage	Mineral + organic	0	15.5	7.3	28/10/2006
Grignon FR	2004–2005	48°50′38″N, 1°57′03″E	Cultivated, organic and mineral fertiliser > 50 years. Present crop rotation since 1999	Mustard/maize	Multiple	Mineral + organic	0	NA–15.9	NA–8.4	
	2005–2006			Winter wheat	Tillage	Mineral	0	17.0	1.3	01/10/2005
	2006–2007			Winter barley	Tillage	Mineral + organic	0	15.0	3.2	20/07/2006
Klingenberg GE	2003–2004	50°53′34″N, 13°31′21″E	Cultivated, organic and mineral fertiliser since 1975. Partially grassland before 1975	Winter barley	Tillage	Mineral	0	18.2	5.2	
	2004–2005			Rapeseed	Tillage	Mineral + organic	0	11.6	1.9	21/08/2004
	2005–2006			Winter wheat	Tillage	Mineral	0	11.1	0.7	23/09/2005
	2006–2007			Maize	Tillage	Mineral + organic	0	16.9	5.5	20/10/2006
	2007–2008			spring barley	tillage	Mineral	0	6.7	2.2	03/10/2007
Lamasquère FR	2004–2005	43°49′65″N, 1°24′19″E	Cultivated, organic manure > 30 years	Triticale	Tillage	Mineral + organic	0	18.4	6.6	
	2005–2006			Maize	Ploughing	Mineral + organic	147.8	20.9	3.6	01/09/2005
	2006–2007			Winter wheat	Ploughing	Mineral + organic	0	15.7	7.3	15/09/2006
Langerak NL	2005–2006	51°57′13″N, 4°54′10″E	Cultivated, organic and mineral fertiliser. Maize and grassland production	Maize	Ploughing	Organic	0	18.8	1.7	01/02/2005
Lonzée BE	2003–2004	50°33′08″N, 4°44′42″E	Cultivated, mineral fertiliser > 70 years. The same 4-year rotation for 10 years	Sugar beet	Multiple	Mineral + organic	0	25.6	10.4	

Table 1 (*Continued*)

Site Name - Country	Years	Latitude, Longitude	Crop history	Crop	Soil preparation	Fertilisation	Irrigation (mm)	NPP (T DM ha^{-1})	Residues (T DM ha^{-1})	NEP starting date
	2004–2005			Winter wheat	Tillage	Mineral	0	20.4	7.2	01/10/2004
	2005–2006			Seed potato	Multiple	Mineral	0	9.3	1.8	16/09/2005
	2006–2007			Winter wheat	Tillage	Mineral	0	17.6	6.7	16/09/2006
Lutjewad NL	2005–2006	53°23′56″N, 6°21′22″E	Not cultivated (fallow) before 2005	Winter wheat	Ploughing	Mineral	0	20.7	1.9	15/10/2005
	2006–2007			Winter wheat	Ploughing	Mineral	0	19.6	1.9	15/10/2006
Molenweg NL	2004–2005	51°39′00″N, 4°38′21″E	Cultivated, organic and mineral fertiliser > 30 years. Agricultural crops and vegetables	Potato	Direct	mineral + organic	0	52.1	10.1	
Oensingen CH	2003–2004	47°17′11″N, 7°44′01″E	Cultivated according to Swiss Integrated Pest Management system with 4-year crop rotation cycle since 1994, 3-yr grassland interspersed	Winter wheat	Ploughing	Mineral	0	10.8	1.8	20/09/2003
	2004–2005			Winter barley/ cover crops	Ploughing	Mineral	0	9.1–NA	1.5	09/11/2004
	2005–2006			Potato	Multiple	mineral + organic	0	NA	9.0	09/11/2005
	2006–2007			Winter wheat	Ploughing	Mineral	0	11.2	1.9	18/10/2006
Risbyholm DK	2003–2004	55°31′49″N, 12°05′50″E	Cultivated, mineral fertiliser > 20 years for grain production on a former drained bog	Winter wheat	Ploughing	Mineral	0	15.2	2.5	02/01/2004
	2004–2005			Winter wheat	Ploughing	Mineral	0	15.7	2.6	15/10/2004
	2005–2006			Winter wheat	Ploughing	Mineral	0	14.7	2.4	11/09/2005
	2006–2007			Winter wheat	Ploughing	Mineral	0	NA	10.1	11/09/2006
	2007–2008			Rapeseed	Ploughing	Mineral	0	3.2	0.5	01/09/2007
El Saler Sueca ES	2004–2005	39°16′32″N, 0°18′55″E	Paddy rice crop > 100 years	Rice	Multiple	Mineral	600	18.0	9.5	03/05/2004
	2005–2006			Rice	Multiple	Mineral	600	19.0	10.0	01/10/2004
	2006–2007			Rice	Multiple	Mineral	600	18.0	9.9	01/10/2005
	2007–2008			Rice	Multiple	Mineral	600	19.2	10.4	01/10/2006
Vredepeel NL	2005–2006	51°31′54″N, 5°50′39″E	Cultivated, organic and mineral fertilizer >30 years. Agricultural crops and vegetables	Sugar beet	Ploughing	Mineral + organic	0	22.2	1.6	10/04/2006

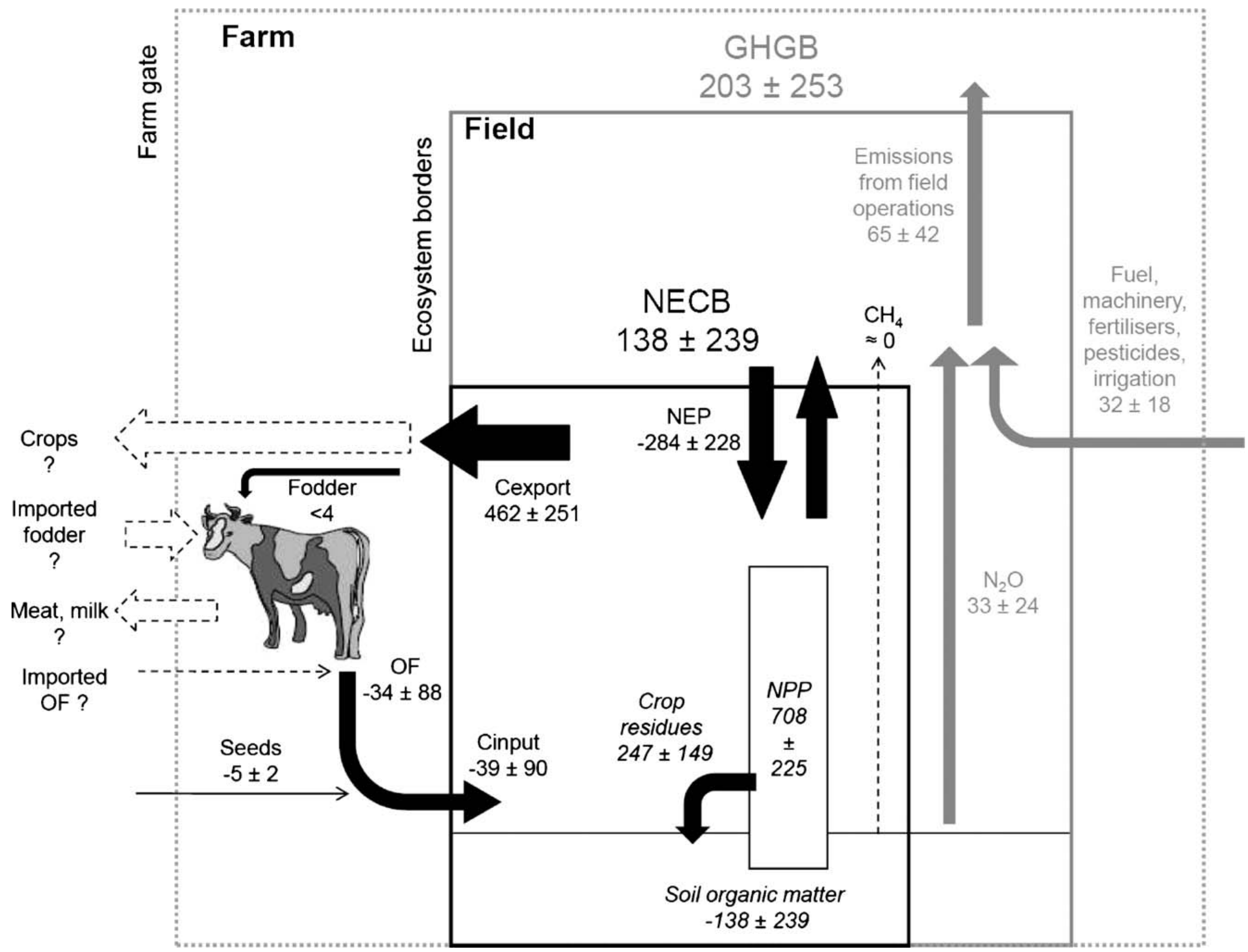

Fig. 1. Schematic representation of the various components of the net ecosystem carbon budget (NECB, in the black box) and of the greenhouse gases budget (GHGB, in the grey box) for the 41 site-years of our study for which net ecosystem production (NEP) data were available. This figure shows that some of the components of the GHGB are located "offsite". Numbers summarising our results are expressed in g C-eq m^{-2} $year^{-1}$. In our study, NEP, Cexport and Cinput represent 36, 59 and 5% of the NECB, respectively. On many European farms, livestock cannot be maintained by the production of the farm itself; thus, fodder has to be imported. Manure produced by the livestock is brought to the field and is an important factor in the soil carbon balance. In this study, we assumed that 100% of the harvest was sold as cash crops, neglecting the part of carbon from the Cexport term actually returning to the field as manure. At most, this could lead to a 4 g C m^{-2} $year^{-1}$ overestimation of the NECB at sites receiving organic fertilisation. On average, the soil was not balanced because organic fertiliser and NEP did not compensate for carbon losses at harvest. Another possibility for balancing carbon fluxes in agriculture is based on farm gate balances, but that is beyond the scope of this paper. Moreover, additional GHG emissions on site and "off site" associated with field operations have to be accounted for. They represent 32% of the GHGB. N_2O emissions from crop residues and fertilisers alone represented close to 16% of the GHGB. On average, croplands were GHG sources (after Kutsch et al. 2008, modified). Other abbreviations: NPP: net primary production; Cexport: carbon exported at harvest; OF: organic fertiliser; Seeds: seeds or tubers imported; Cinput: the sum of Seeds and OF.

59% of the arable lands (see EUROSTAT, 2008 and Swiss Federal Statistical Office, 2008) of the nine countries represented in this study and more than 73% of the cropping areas of the EU 27 (FAOSTAT).

2.1. Sites and biospheric fluxes

We used CarboEurope-IP Level 4 net ecosystem exchange (NEE) data and management information from different cropland sites that provided flux measurements for at least one year during the 2004–2007 project period (see Table 1). Level 4 data are the result of high-frequency eddy covariance information that has been processed to obtain NEE fluxes at 30-minute intervals following CarboEurope-IP recommendations (in terms of rotation, spectral and air density corrections; see Aubinet et al., 2000). The NEE data were then quality checked, filtered and gap-filled following the methodology described in Reichstein et al. (2005). NEE Level 4 data were then integrated over one year (365 days) to obtain annual NEP. The period used to calculate the NEP always included the harvest date.

The starting (and ending) dates of the one-year periods varied by crop and site according to Table 1. The start date was defined either as the time between harvest of the previous crop and ploughing/tillage for the next crop or as the time between harvest of the previous crop and sowing when there was no soil preparation before sowing. At some sites, the period used to calculate the yearly NEP overlapped by a few days with the period used to calculate the NEP for the following year because there was less than one full year between two ploughing events (see Table 1). Conversely, some gaps existed between the periods used to calculate the NEPs for two successive crops due to missing flux data during this interval. In a few cases (e.g., Avignon 2005–2006 and Oensingen 2004–2005), we tested different starting dates for the same crop to assess the effects of either including or omitting volunteer re-growth events (+ weeds) or cover cropping (during intercropping periods) on the carbon budgets.

To assess the influence of the length of the growing season on the NEP, the number of days of active vegetation cover (NDAV), defined as the number of days when daily gross primary production (GPP) was above zero (using a 1 g C m^{-2} d^{-1} threshold), was calculated based on Level 4 data from the CarboEurope database. The NDAV may include periods of significant GPP from voluntary re-growth (+ weeds) or the presence of a cover crop.

Methane fluxes were not measured at these sites and could not be included in the C budget calculations. Methane emissions and oxidation at upland sites were considered negligible compared to the other source and sink terms in the total cropland GHGB. However, because methane fluxes are expected to significantly affect GHGB at the El Saler Sueca site (intermittently flooded paddy rice), they were estimated using data from the literature (Pathak et al., 2005; IPCC, 2006; Zou et al., 2009). We estimated those fluxes to be 20 g CH_4 m^{-2} $year^{-1}$ (within the range of 10–40 g CH_4 m^{-2} $year^{-1}$), corresponding to an emission of 125 g C-eq m^{-2} $year^{-1}$ (within the range of 63–250 g C-eq m^{-2} $year^{-1}$) at the El Saler Sueca site.

2.2. Net Ecosystem Carbon budget calculations

Non-CO_2 carbon losses corresponding to harvest (Cexport, i.e., grains, straw, tubers) or fires (F) and C gains corresponding to organic fertilisation or addition of sugar beet lime (OF) or seeds/mother tubers (S) were accounted for along with the NEP to obtain the net ecosystem carbon budget. Hereafter, the sum of OF and S is referred to as Cinput. The NECB was considered as the total rate of organic carbon accumulation or loss from ecosystems (see Smith et al., 2010). Carbon losses by erosion, as volatile organic compounds, as a result of dissolved or particulate organic and inorganic carbon leaching and due to microbially produced methane (CH_4) were neglected (except for El Saler Sueca, see above). Additionally, C gains by deposition of organic dust particles and pollen and by deposition of dissolved carbon in rain and fog were neglected due to lack of data (see Eugster et al., 2008 for uncertainties introduced by those approximations). Therefore, the NECB was defined as follows:

$$\mathrm{NECB} = \mathrm{NEP} + \mathrm{C\,export} + \mathrm{F} + \mathrm{OF} + \mathrm{S}. \tag{1}$$

We used the micrometeorological convention by which NEP is negative when the ecosystem is fixing carbon and positive when it is losing carbon. Analyses of plant carbon and nitrogen content and, for some sites, analyses of exported biomass and carbon and nitrogen in residues were performed just before and after harvest, respectively. The amount of residue was calculated as the difference between NPP and exported biomass (see Table 1), or it was estimated using a mean of the residues for similar crops from other sites when data were missing. Carbon exported (Cexport) from the plot during harvest was either calculated by subtracting the carbon content in crop residues from the carbon content in above-ground biomass or was obtained directly by multiplying the biomass exported by its carbon concentration. As in Hollinger et al. (2005), Cexport was considered a positive term corresponding to a rapid carbon release to the atmosphere. Prescribed fire events occurred only in 2004 and 2005 at the El Saler Sueca site, where rice was cultivated. Carbon lost during fire events (F) was estimated assuming that after a fire, all of the carbon contained by the residues left on the ground that had burned (approximately 40% of total residues) was lost, although this is clearly a simplification (see Osborne et al., 2010). The proportion of burned residues was estimated visually. OF was calculated from analyses of the carbon content in organic fertiliser (or sugar beet lime) provided by the farmers. Because OF was a carbon input to the plot, it was negative. Finally, S was small and sometimes neglected, but in some cases, such as for potato crops, it was calculated after analysis of the seed/mother tuber carbon content. Carbon lost in soil adhering to root/tuber crops was ignored, although this could be significant (see Osborne et al., 2010).

Table 2
Estimates of the primary (burned fuel) and tertiary (manufacture, maintenance, amortisation) emissions in kg C-eq ha^{-1} for a range of field operations.

Field operations	Primary emissions	Tertiary emissions
Ploughing 30–50 cm	20.4–33.3	0.547
Field cultivation	1.82	0.168
Disking	5.43	0.155
Harrowing	1.36	0.091
Rotary hoeing	5.43	0.091
Ridging	2.71	0.182
Sowing	2.71	0.155
Potato planter	6.83	0.168
Rolling	5.80	0.155
Mineral fertiliser application	1.43	0.091
Organic fertiliser application	3.05	0.137
Pesticide application	1.15	0.046
Harvest	14.1	0.764
Cutting	5.5	0.155
Haying	5.5	0.155

2.3. Emissions from field operations (EFO)

Each site's principal investigator (PI) was in charge of interviewing the farmers or field managers and collecting information on field operations at the site that could affect the C or net GHGB. Field operations were then sorted according to Gifford (1984) into primary, secondary and tertiary sources of C or GHGs. Primary sources of C emissions were either mobile operations (e.g., tillage, sowing, harvesting and transport) or stationary operations, such as pumping water for irrigation. Secondary sources of GHGs converted to C equivalent emissions were comprised of manufacturing, packaging and storing of fertilisers (mineral as well as organic) and pesticides and N_2O emissions caused by fertilisers and residues on the field. Tertiary sources of C emissions included manufacturing and maintenance of equipment (e.g., tractors and farm machinery). We did not include emissions associated with farm buildings and local roads that only served to drive farm equipment from the farm to the cropland site.

2.3.1. Primary sources

Direct emissions associated with tractors and farm machinery (mobile operations) are due to the fuel burned in internal combustion engines. We considered the carbon emission coefficient for burned fuel to be 0.814 kg C-eq l^{-1}. Emission factors (EF) were obtained for each operation after interviews with the farmers (see Table 2), and the same EF values were applied to all European sites. This method assumes that the same machines and tools were used on all sites and that each type of operation lasted the same amount of time, whatever the soil type, soil conditions, etc. Emission factors were, however, consistent with those reported in Lal (2004). Emissions caused by irrigation were estimated using EFs of 0.516 and 0.029 kg C-eq ha^{-1} $year^{-1}$ mm^{-1} recalculated from Dvoskin et al. (1976) for centre-pivot (Cioffi), frontal mobile ramp (Avignon), traveller (Lamasquère) and static (Vredepeel) sprinklers, assuming the energy for irrigation is taken from fossil fuels, and flood irrigation systems (El Saler Sueca), respectively. The equivalent C emissions for installation of irrigation systems were calculated based on Lal (2004) (recalculated from Batty and Keller, 1980). Equivalent C emissions for the installation of centre-pivot (Cioffi) or frontal mobile ramp (Avignon), traveller (Lamasquère) or static (Vredepeel) sprinklers, and flood irrigation systems (El Saler Sueca) were 21.6 kg C-eq ha^{-1} $year^{-1}$, 23.3 kg C-eq ha^{-1} $year^{-1}$ and 9.4 kg C-eq ha^{-1} $year^{-1}$, respectively.

2.3.2. Secondary sources

2.3.2.1. Pesticides. Equivalent C emissions for pesticides were calculated using the EFs reported in Gaillard et al. (1997) and Lal (2004) (see Table 3). These emissions correspond to manufacturing, packaging, transport and storage of pesticides. When data on the amount of active substance of pesticides applied and specific emission factors were available, C-eq emissions were calculated by multiplying the EFs by the amount of active substance. When no specific EF was found in the literature for the active substance present, a mean EF per type of pesticide (fungicide, insecticide, herbicide, growth regulator) was used (see Table 3). In some cases, only the type of pesticide or the brand applied was known, but not the amount of active substance. In such cases, mean C-eq emissions per type of pesticide application were calculated using data from the other sites in this study.

2.3.2.2. Fertilisers. EFs corresponding to manufacturing, packaging, transport and storage of mineral and organic fertilisers were found in Kramer and Moll (1995), Gaillard et al. (1997) and Lal (2004) (Table 4). It is possible that a portion of the C exported from the plot that was considered as going directly back into the atmosphere was used to produce organic manure that may come back to the plot. If this occurred, the source amount of C could have been considered twice. The first consideration would be when it left the plot at harvest, and the second consideration would be when the material was oxidised by the cows, producing GHG that we estimated using the emission factors mentioned here for producing organic fertiliser. We have estimated that this overestimation of the GHG emissions could represent up to 4 g C-eq m^{-2} $year^{-1}$. This number is less than the uncertainty range of the NEP and Cexport (see Béziat et al., 2009). It should be noted that in our study, only a small part of the harvest was used as fodder because most of the production was used for cash crops or was sold to typical livestock-orientated farms.

Emissions of N_2O caused by fertiliser applications were estimated following the methodology recommended in the IPCC (2006) report: we calculated that 1.7% of the nitrogen applied as fertiliser was converted into N_2O (direct plus indirect emissions), and N_2O emissions were converted into C-eq values (1 kg N_2O corresponding to 81.3 kg EC). It should be noted that lower EFs for N_2O emissions from fertilisers (ranging from 0.26% and 0.87%) were reported in Cioffi on the basis of chamber studies (data not shown). N_2O emissions caused by crop residues were estimated in the same way after determining the N content of the residues (see above). Based on an EF of 2.7 kg t^{-1} (uncertainty range 1.4–4.2 kg t^{-1}) for slurry applications to grasslands (Chadwick et al., 2000), CH_4 emissions from manure and slurry applications were calculated. As they never exceeded 0.2 g C-eq m^{-2} $year^{-1}$ at our sites (less than 0.5% of NECB), these emissions were omitted.

2.3.2.3. Tertiary sources. Emissions caused by the manufacture, amortisation and maintenance of machinery were calculated using EFs per hour of use that were found in the ADEME (2007) report and from interviews with several farmers from our studied sites to evaluate the time of use of the different machines for each type of operation (ploughing, harvest, etc.). EFs by type of operation are reported in Table 2. These factors are consistent with the EFs found in Lal (2004) and with the emissions from machines reported in Robertson et al. (2000).

2.4. Total GHG budget and crop efficiency

The global warming potential figures for N_2O, CH_4 and CO_2 were 296, 23 and 1, respectively (relative to an equivalent mass of CO_2), assuming a 100-year time horizon (IPCC, 2006). They were then converted to C equivalents (C-eq) using a carbon to CO_2 mass ratio

Table 3
Estimates of equivalent carbon emission (kg C-eq kg^{-1}) for production, transportation, storage and transfer of pesticides.

Type	Emission factor	Source
Herbicides		
2, 4, 5-T	2.7	Lal (2004)
2, 4-D	1.7	Lal (2004)
Alachlor	5.6	Lal (2004)
Amidosulfuron	2.91	Gaillard et al. (1997)
Asulame	2.45	Gaillard et al. (1997)
Atrazine	1.55–3.8	Gaillard et al. (1997), Lal (2004)
Bentazon	8.7	Lal (2004)
Bifenox	0.79	Gaillard et al. (1997)
Butlylate	2.8	Lal (2004)
Carbetamide	2.45	Gaillard et al. (1997)
Chloramben	3.4	Lal (2004)
Chlorosulfuron	7.3	Lal (2004)
Chlortoluron	2.91	Gaillard et al. (1997)
Cyanazine	4	Lal (2004)
Dicamba	5.9	Lal (2004)
Dinoseb	0.67–1.6	Gaillard et al. (1997), Lal (2004)
Diquat	8	Lal (2004)
Diuron	5.4	Lal (2004)
EPTC	3.2	Lal (2004)
Ethofumesate	2.6	Gaillard et al. (1997)
Fluazifop-butyl	10.4	Lal (2004)
Fluometuron	7.1	Lal (2004)
Fluroxypyr	5.95	Gaillard et al. (1997)
Glyphosate	4.77–9.1	Gaillard et al. (1997), Lal (2004)
Ioxynil	2.6	Gaillard et al. (1997)
Isoproturon	2.91	Gaillard et al. (1997)
Linuron	5.8	Lal (2004)
MCPA	1.27–2.6	Gaillard et al. (1997), Lal (2004)
MCPB	2.35	Gaillard et al. (1997)
Mecoprop P	2.35	Gaillard et al. (1997)
Metamitrone	2.46	Gaillard et al. (1997)
Metolachlore	2.71–5.5	Gaillard et al. (1997), Lal (2004)
Paraquat	9.2	Lal (2004)
Pendimethaline	1.1	Gaillard et al. (1997)
Phenmediphame	2.45	Gaillard et al. (1997)
Propachlor	5.8	Lal (2004)
Pyridate	2.6	Gaillard et al. (1997)
Rimsulfuron	2.91	Gaillard et al. (1997)
Tebutame	2.59	Gaillard et al. (1997)
Terbuthylazine	2.46	Gaillard et al. (1997)
Trifluralin	3	Lal (2004)
Mean ± S.D.	3.92	
Fungicide		
Benomyl	8	Lal (2004)
Captan	2.3	Lal (2004)
Carbendazime	4.17	Gaillard et al. (1997)
Chlorothalonil	0.99	Gaillard et al. (1997)
Fenpropimorphe	1.68	Gaillard et al. (1997)
Ferbam	1.2	Lal (2004)
Flusilazole	1.68	Gaillard et al. (1997)
Mancozèbe	0.77	Gaillard et al. (1997)
Manèbe	0.81	Gaillard et al. (1997)
Maned	2	Lal (2004)
Prochloraze	1.68	Gaillard et al. (1997)
Tebuconazole	1.68	Gaillard et al. (1997)
Mean ± S.D.	2.25	
Insecticide		
Carbaryl	3.1	Lal (2004)
Carbofuran	9.1	Lal (2004)
Chlorodimeform	5	Lal (2004)
Cypermethrine	7.02–11.7	Gaillard et al. (1997), Lal (2004)
Lambda-cyhalothrine	7.02	Gaillard et al. (1997)
Lindane	1.2	Lal (2004)
Matlathion	4.6	Lal (2004)
Methoxychlor	1.4	Lal (2004)
Methyl parathion	3.2	Lal (2004)
Parthion	2.8	Lal (2004)
Phorate	4.2	Lal (2004)

Table 3 (*Continued*)

Type	Emission factor	Source
Taxaphene	1.2	Lal (2004)
Mean ± S.D.	4.73	
Molluscicide		
Methiocarbe	2.45	Gaillard et al. (1997)
Growth regulator		
Chlormequat (CCC)	2.37	Gaillard et al. (1997)
Ethephon	2.37	Gaillard et al. (1997)
Trinexapac-éthyle	2.37	Gaillard et al. (1997)
Mean ± S.D.	2.37	

of 0.2727. Therefore, the total GHGB was calculated in C-eq units for each site-year and for each plot and crop considering the sum of the NECB and of all the emissions caused by field operations:

$$\mathrm{GHGB} = \mathrm{NEP} + \mathrm{Exp} + \mathrm{F} + \mathrm{OF} + \mathrm{S} + \sum \mathrm{EFO}. \tag{2}$$

The annual GHGBs were then summed for the six sites where crop rotation was complete or that were cultivated with a monoculture (Carlow, El Saler Sueca). Full crop rotations occurred at four sites, with cycles of three (Cioffi), four (Avignon, Oensingen) and five years (Risbyholm).

Crop efficiency was calculated as the ratio between the C exported at harvest (grain and eventually straw) and total GHGB:

$$\mathrm{CE} = \frac{\mathrm{Cexport}}{\mathrm{NEP} + \mathrm{Cexport} + \mathrm{F} + \mathrm{OF} + \mathrm{S} + \sum \mathrm{EFO}}. \tag{3}$$

CE is expressed in g C exported g^{-1} C-eq emitted and equals 1 if NEP, F, OF, S and $\sum$EFO are balanced. The CE was calculated for each crop species and for different management regimes for the same crop.

3. Results

Overall, NEP, NECB and the total GHGB were calculated for 15 sites comprising 41 site-years, and emissions from agricultural practices were calculated for 17 sites comprising 51 site-years (see Table 5).

3.1. Net ecosystem production

NEP varied strongly between sites and crops and between crops at each site (see Table 5 and Fig. 2), but, on average, NEP was negative (mean ± SE of −284 ± 228 g C m^{-2} $year^{-1}$), with large variability among sites. Part of this variability in NEP was a result of the difficulty of defining budgeting years for cropland sites with intensive management. Thus, the NEP values reported here differ slightly from those presented by Kutsch et al. (2010) and Moors et al. (2010) because of differences in the integration periods.

3.1.1. Winter crops

On average, winter crops had rather similar NEPs, with −292 ± 170 (n = 13), −358 (n = 2), −303 ± 159 (n = 3) and −214 (n = 2) g C m^{-2} $year^{-1}$ for winter wheat, durum wheat, winter barley and rapeseed, respectively (Fig. 2). However, NEP variability within the same crop grown at different sites was high. This was largely caused by differences in latitude and climate variability influencing the length of the growing season and the amount of C assimilated. Additional factors, such as management (e.g., fertilisation, amount of residues decomposing from the previous crop) and soil properties, may also be involved and are discussed in Kutsch et al. (2010), Moors et al. (2010) and Eugster et al. (2010).

Fig. 3 represents NEP as a function of the number of days of active vegetation cover (NDAV). In general, NEP increased in absolute value with NDAV for winter crops. However, two points corresponding to winter wheat grown at Oensingen and Risbyholm in 2006–2007 seem to lie outside the relationship between the sum of days when vegetation was active and NEP. The case of Oensingen is explored in more detail by Dietiker et al. (2010). They used the DNDC model to simulate net CO_2 uptake and found the greatest discrepancy between the measurements and the model in 2007. This result may indicate that the mild winter of 2006/2007, which led to the warmest January on record in large parts of Switzerland (MeteoSwiss, 2008), did not automatically lead to strong increased yields despite the substantial increase in NDAV in this year.

There was only one positive NEP value for winter crops, corresponding to winter wheat at Gebesee in 2006–2007. It should also be noted that in Fig. 3, rapeseed grown at Risbyholm was considered a summer crop because it was sown in May and harvested in July, which is atypical for rapeseed in southern Europe. The fact that the growing season was much shorter for the rapeseed grown at Risbyholm compared to the crop grown at Klingenberg partly explains why the NEP at Risbyholm was much lower than it was at Klingenberg (see Table 5). Another explanation is that no harvest occurred that year at Risbyholm because of flooding. Similarly, at Oensingen in the previous year, potato (2005–2006) was not harvested because of a fatal accident in the farmer's family. Therefore, for those two sites, large amounts of decomposing crop residues increased ecosystem respiration and reduced the NEP of the current and subsequent years, respectively.

3.1.2. Spring and summer C3 crops

NEP for spring- and summer-grown C_3 crops varied between 278 (peas) and -652 ± 41 g C m^{-2} $year^{-1}$ (rice) (Fig. 2). Positive NEP values for pea (Avignon) can be explained by the very short growing season, which left the soil without vegetation cover for a large part of the year. The potatoes grown at Oensingen were subject to a hail event that destroyed part of the crop. Finally, spring barley (Klingenberg) also had positive NEP values, but the reasons for this are more difficult to identify. Considering the C_3 summer crops as a whole, NEP tended to decrease with increasing NDAV (Fig. 3). However, for individual crops, such as sugar beet, rice and spring barley, NEP tended to increase with increasing NDAV.

El Saler Sueca was the only site in this study where rice was grown and where the values for NEP in Table 5 and Fig. 2 do not include methane emissions. Those fluxes were estimated at 20 g CH_4 m^{-2} $year^{-1}$, corresponding to a 15 g C m^{-2} $year^{-1}$ loss from the ecosystem. Even with the inclusion of estimated methane emissions, El Saler Sueca remains the site with the lowest mean NEP value (between −591 and −678 g C m^{-2} $year^{-1}$). Only Gebesee (−655 g C m^{-2} $year^{-1}$), when sugar beet was grown (year 2005–2006), and Lonzée (−605 g C m^{-2} $year^{-1}$), when winter wheat was grown (2006–2007), had similar negative NEP values.

Table 4
Emission factors for production of mineral fertilisers (kg C-eq kg^{-1}), solid manure (kg C-eq t^{-1}) and liquid manure (kg C-eq m^{-3}).

Fertiliser	Emission factor
N (NH_4NO_3)	1.11
N (KAS)	1.35
N (Urea)	1.29
P	0.42
K	0.15
Ca	1.35
Mg	0.15
S	0.15
B	1.11
Sugar beet lime	0.032
Solid manure	0.88
Liquid manure	0.90

Table 5
The different terms[a,b] composing annual GHG budgets (GHGB) calculated in g C-eq m^{-2} $year^{-1}$ for each site of the CarboEurope-IP network.

Site Name and country	Years	Crop	NEP	Cinput		Cexport		Fire	NECB	Machines	Pesticides	Fertilisers	N_2Of	N_2Or	Irrigation	Total emissions	GHGB
				Seeds	OF	Grain	Straw										
Auradé FR	2004–2005	rapeseed	NA	−0.1	0	242		0	NA	3.1	0.6	26.2	29.4	8.3	0	67.5	NA
	2005–2006	winter wheat	−305	−6.3	0	277		0	−34.0	4.7	0.4	13.7	14.6	4.0	0	37.5	4
	2006–2007	sunflower	−8.5	−0.2	0	97.6		0	88.9	2.6	0.5	3.6	4.5	4.5	0	15.7	105
Avignon FR	2003–2004	durum wheat	−255	−6.3	0	279	177	0	198	5.2	0.8	27.3	26.5	5.1	0	64.8	262
	2004–2005	peas	278	NA	0	97.9		0	375	5.3	0	6.5	0	10.5	5.3	27.6	403
	2005–2006	durum wheat	−461	−6.3	0	189	120	0	−158	7.5	0.8	20.4	19.0	7.4	4.2	59.3	−99
	2006–2007	sorghum	−170	NA	0	222		0	52.3	6.4	0.1	4.3	0	12.2	5.3	28.3	81
Cioffi IT	2004–2005	rye-grass/maize	−412	NA	−19.2	191–720		0	480	9.6	0.4	47.2	45.8	9.8	19.2	132	612
	2005–2006	fennel/maize	−274	NA	0	68–725		0	519	12.3	0.9	72.8	83.1	10.6	18.6	198	717
	2006–2007	fennel/maize	−342	NA	0	34–953		0	645	12.2	0.9	96.2	87.7	15.4	22.6	235	880
Carlow IR	2004–2005	spring barley	−144	−6.3	0	225		0	75.0	5.4	1.8	20.8	20.1	4.0	0	52.1	127
	2005–2006	spring barley	−200	−6.3	0	248		0	41.6	5.5	1.3	23.6	23.6	4.3	0	58.3	100
	2006–2007	spring barley	−236	−6.3	0	290		0	48.2	6.0	0.8	23.6	23.6	4.4	0	58.5	107
Dijkgraaf NL	2006–2007	maize	NA	−0.4	0	758		0	NA	4.2	0.8	7.2	35.5	3.6	0	51.3	NA
Gebesee GE	2003–2004	rapeseed	NA	−0.1	0	342		0	NA	3.2	0.6	32.7	33.1	7.7	0	77.3	NA
	2004–2005	winter barley	−123	−6.2	0	370		0	241	4.7	0.4	9.7	10.3	3.4	0	28.5	269
	2005–2006	sugar beet	−655	NA	0	787 (tuber)		0	132	3.7	0.8	5.6	4.5	11.5	0	26.0	158
	2006–2007	winter wheat	25.5	−6.3	−34.7	362		0	346	5.7	0.5	14.8	15.9	4.0	0	41.0	387
Grignon FR	2004–2005	mustard/maize	NA	NA	0	338		0	NA	5.8	1.3	21.8	20.1	7.6	0	56.7	NA
	2005–2006	winter wheat	−179	−6.3	0	696		0	497	3.8	1.4	14.9	15.8	0.7	0	36.6	534
	2006–2007	winter barley	−363	−6.3	0	505		0	164	3.7	1.4	12.0	15.5	1.7	0	34.4	198
Klingenberg GE	2003–2004	winter barley	NA	−6	0	276	237	0	NA	3.6	0.4	41.4	12.9	2.8	0	61.2	NA
	2004–2005	rapeseed	−306	−0.1	−256	243	317	0	−1.8	4.1	1.2	30.6	52.8	2.1	0	91.0	89
	2005–2006	winter wheat	−145	−6.3	0	253	203	0	304	4.1	1.6	30.5	29.8	0.4	0	66.4	371
	2006–2007	maize	88.9	−0.4	−176	515		0	448	3.5	0.6	16.0	29.5	5.0	0	54.6	482
	2007–2008	summer barley	10.5	−6.3	0	118	93	0	217	2.8	0.2	8.5	6.9	1.2	0	19.5	237
Lamasquère FR	2004–2005	triticale	NA	−6.3	66.7	270	243	0	NA	2.9	0.3	15.3	28.9	3.6	0	51.1	NA
	2005–2006	maize	−240	−0.3	−249	479	327	0	316	5.4	0.4	13.8	33.9	3.3	7.6	64.4	381
	2006–2007	winter wheat	−387	−6.3	−80.9	184	199	0	−90	5.2	0.1	27.9	43.9	4.0	0	81.1	−9
Langerak NL	2005–2006	maize	−271	−0.4	−26.8	794		0	496	4.2	0.4	2.2	5.4	1.5	0	13.7	509
Lonzée BE	2003–2004	sugar beet	NA	NA	−66	630 (tuber)		0	NA	5.4	1.9	63.7	22.4	14.9	0	108	NA
	2004–2005	winter wheat	−460	−6.0	0	388	172	0	93.8	2.7	1.1	24.3	29.0	4.0	0	61.1	155
	2005–2006	seed potato	−42.9	−4.0	0	290 (tuber)		0	243	6.8	5.4	26.3	16.8	4.0	0	59.0	302
	2006–2007	winter wheat	−605	−6.0	0	310	140	0	−161	3.1	0.7	26.3	28.0	3.7	0	61.7	−99
Lutjewad NL	2005–2006	winter wheat	NA	−6.3	0	869		0	NA	4.9	0.6	21.8	23.2	1.0	0	51.5	NA
	2006–2007	winter wheat	−455	−6.3	0	818		0	356	4.9	1.7	25.3	26.9	1.0	0	59.9	416
Molenweg NL	2004–2005	potato	NA	NA	−53.5	1583 (tuber)		0	NA	2.9	2.7	29.7	38.6	21.8	0	95.6	NA
Oensingen CH	2003–2004	winter wheat	NA	−5.3	0	284	158	0	0	5.3	0.2	17.9	18.3	1.0	0	41.7	0
	2004–2005	winter barley/cover crops	−424	−6	0	260	69	0	−101	5.6	0.4	14.0	14.4	0.8	0	34.3	−66

Table 5 (*Continued*)

Site Name and country	Years	Crop	NEP	Cinput		Cexport		Fire	NECB	Machines	Pesticides	Fertilisers	N_2Of	N_2Or	Irrigation	Total emissions	GHGB
				Seeds	OF	Grain	Straw										
	2005–2006	potato	217	−4.4	−121	0	0	0	92	3.7	0	31.8	22.7	19.4	0	58.2	150
	2006–2007	winter wheat	−173	−7.7	0	223	180	0	222	2.8	0.2	19.9	20.1	1.0	0	43.0	265
Risbyholm DK	2003–2004	winter wheat	−403	−6.3	0	375	206	0	173	4.9	1.3	29.2	29.1	1.4	0	65.9	239
	2004–2005	winter wheat	−306	−6.3	0	388	214	0	290	4.7	0.7	28.3	28.2	1.4	0	63.4	354
	2005–2006	winter wheat	−260	−6.3	0	363	199	0	296	4.8	0.4	28.9	28.8	1.3	0	64.1	360
	2006–2007	winter wheat	−197	−6.3	0	0		0	−204	4.7	0.7	30.8	31.1	5.5	0	72.9	−131
	2007–2008	rapeseed	−123	−0.1	0	153		0	30	4.2	0.2	19.9	19.1	0.6	0	44.0	74
El Saler Sueca ES	2004–2005	rice	−679	NA	0	431		89	−248	3.2	0.2	19.3	29.1	13.7	3.6	73.4	−133
	2005–2006	rice	−693	NA	0	448		90	−245	3.0	0.2	19.3	29.2	14.4	3.6	74.7	−122
	2006–2007	rice	−606	NA	0	357		0	−249	3.1	0.2	19.3	29.5	14.2	3.6	74.5	−175
	2007–2008	rice	−630	NA	0	372		0	−258	3.2	0.2	19.3	30.4	10.5	3.6	76.0	−182
Vredepeel NL	2005–2006	sugar beet	−486	NA	−433	850 (tuber)		0	−69.3	4.2	3.2	28.7	61.3	0.9	4.1	103	33

[a] NEP: cumulative net ecosystem production; Cinput: carbon inputs as seeds and organic fertilisers; Cexport: carbon exported during harvest (yield). When data were available, Cexport as grain or straw was specified. Fire: carbon lost during fire events (at El Saler Sueca only); NECB: net ecosystem carbon budget, calculated as the sum of the three previous terms; Machines: emissions caused by direct use, maintenance and amortisation of machines; Pesticides: emissions associated with production, transportation, storage and transfer of pesticides; Fertilisers: emissions associated with production, transportation, storage and transfer of organic and mineral fertilisers; N_2Of: N_2O emissions caused by the use of fertilisers; N_2Or: N_2O emissions caused by the decomposition of crop residues left on the field; Irrigation: emissions caused by irrigation; Total emissions: the sum of emissions caused by field operations; GHGB: the sum of NECB and all emissions caused by field operation.

[b] Methane emissions of approximately 15 g C-eq m^{-2} $year^{-1}$ at El Saler Sueca are not included.

Spring barley had a less negative NEP (-193 ± 46 g C m^{-2} $year^{-1}$) than winter cereals, partly because NDAVs and yields were smaller (see Figs. 2 and 3). Sugar beet had very negative NEP values at Gebesee (-655 g C m^{-2} $year^{-1}$) and Vredepeel (-486 g C m^{-2} $year^{-1}$) in 2005 and 2006. Even when the potato crop had a similar NDAV to sugar beet, for instance at Gebesee, the NEP was very positive. Peas had the highest positive NEP value, but the growing season was the shortest, and LAI was low compared to other crops (data not shown). Finally, the NEP for sunflower was close to equilibrium (-8.5 g C m^{-2} $year^{-1}$).

3.1.3. Summer C_4 crops

For C_4 summer crops alone, there was no clear trend of increasing NEP with NDAV. The NEP for sorghum was -170 g C m^{-2} $year^{-1}$, which is comparable to the mean NEP value for maize. NEP for maize alone was on average -141 ± 200 g C m^{-2} $year^{-1}$, but variability between sites was very large, ranging between -271 and 89 g C m^{-2} $year^{-1}$ at Langerak and Klingenberg, respectively (Fig. 2). At Klingenberg, a hail event occurred in July 2007 (half-hourly precipitation of 38 mm). This event caused significant damage to the maize plants, probably inducing a reduction in LAI and net C fixation.

3.1.4. Effects of cover crops or voluntary re-growth and weeds on NEP

When considering C_4 crops combined with fennel or rye-grass (Cioffi site), the NEP tended to increase with NDAV. The NEP for maize alone was always smaller than when maize was combined with either rye-grass or fennel (see Table 5). Indeed, when maize is combined with another crop, bare soil periods are shorter and soil C losses are compensated by net C assimilation from the cover crop. In a similar way, the growth of volunteer seedlings and weeds after the harvesting of winter wheat at Avignon (2005–2006) and the sowing of a mixture of phacelia (*Phacelia tanacetifolia*), alexandrine clover (*Trifolium alexandrinum*) and oat (*Avena sativa*) (named "cover crops" in Tables 1 and 5) at Oensingen after winter barley (2004–2005) had a noticeable effect on NEP (see also Dietiker et al., 2010). Our comparison of NEP estimates including and excluding the period of re-growth that occurred after winter barley at Oensingen revealed NEP values of -424 and -144 g C m^{-2} $year^{-1}$, respectively. However, it should be noted that this is not a measure of the accuracy of flux measurements but an indication of the problems with defining budgeting years for cropland sites. In this case, the integration period changed from 09 November 2004–08 November 2005 to 05 August 2004–04 August 2005. Similarly, by including in or omitting from the calculation the periods encompassing re-growth events and weeds development that occurred after the growth of winter wheat at Avignon (periods between 01 September 2005–31 August 2006 and 06 November 2005–05 November 2006, respectively), the NEP value changed from -461 to -478 g C m^{-2} $year^{-1}$, respectively.

3.2. Carbon exports

Carbon exports showed a wide variation among sites and crop types, ranging from 0 g C m^{-2} $year^{-1}$ at Oensingen and Risbyholm (no harvest, see above) to 987 and 1583 g C m^{-2} $year^{-1}$ at Cioffi (fennel/maize, 2006–2007) and Molenweg (potato, 2004–2005). Considering sites where NEP was measured, the crops associated with the biggest Cexport terms were maize and sugar beet, at 705 ± 165 and 818 ± 44 g C m^{-2}, respectively (Fig. 2). At those sites, all of the aboveground parts of the maize plants were exported, mostly for silage, whereas in most of the countries represented in this study (except Switzerland), 53% of the surface area where maize is grown is used for grain production only (EUROSTAT, 2008). As the CarboEurope data set did not include this latter variant of

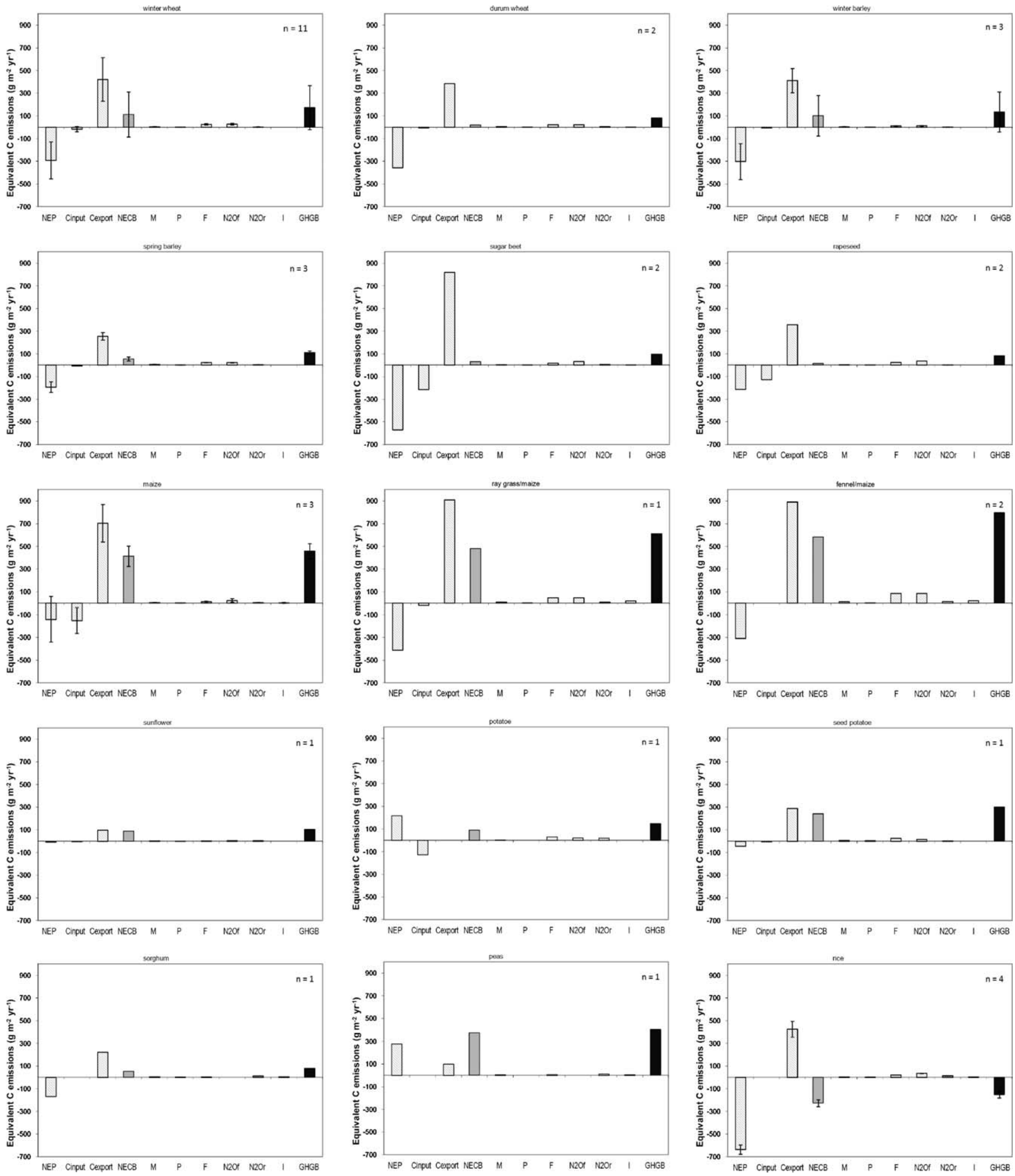

Fig. 2. Mean for each crop species of the different terms composing the annual GHG budgets (GHGB) calculated in C-eq at European crop sites: net ecosystem production (NEP), carbon inputs (Cinput) as seeds and organic fertilisers, carbon exports corresponding to harvest and fire (Cexport) and net ecosystem carbon budget (NECB), calculated as the sum of the three previous terms. The emissions associated with field operations are as follows: emissions caused by direct use, maintenance and amortization of the machines (M), emissions associated with production, transportation, storage and transfer of pesticides (P), emissions associated with production, transportation, storage and transfer of fertilisers (F), N_2O emissions caused by the use of fertilisers (N_2Of), N_2O emissions caused by the decomposition of crop residues left on the field (N_2Or) and emissions caused by irrigation (I). Finally, the GHGB, the sum of NECB with emissions caused by field operations, is presented. Vertical full lines (error bars) are ± the standard deviation of each measurement mean. They were calculated when the number of sites per crop species was ≥3.

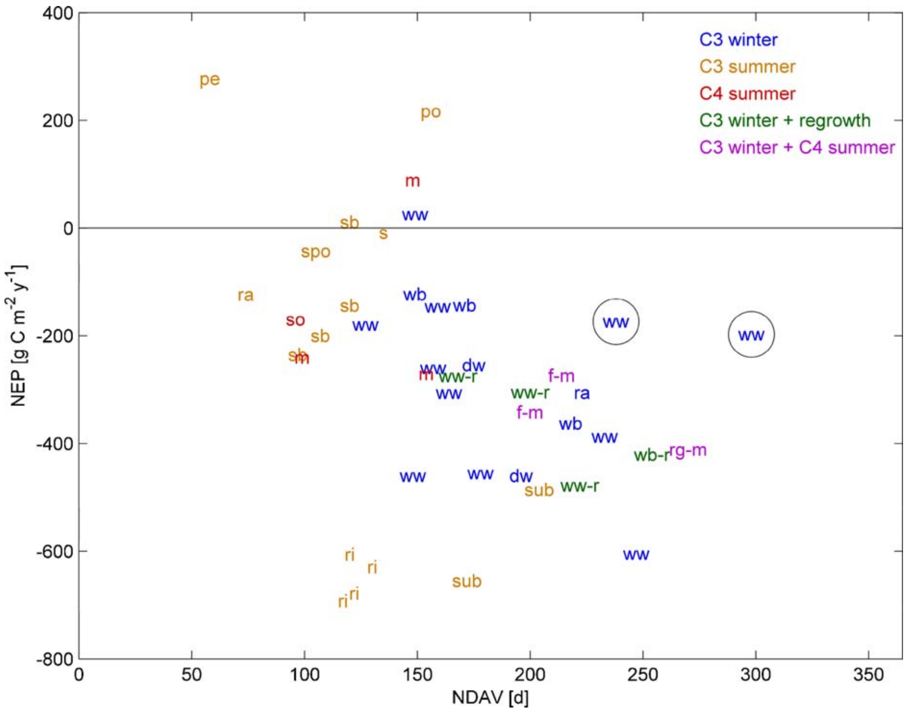

Fig. 3. Net ecosystem production (NEP) as a function of number of days of active vegetation (NDAV). Each point represents one site year. C3 winter crops are presented in blue; C3 summer crops are in orange; C4 summer crops are in red; C3 winter crops that were followed by re-growth events (volunteer seedlings) and weed development are represented in green; a combination of a C3 winter crop and a C4 summer crop on the same site in the same year is represented in violet. The different crop species presented are winter wheat (ww), durum wheat (dw), winter barley (wb), rapeseed (ra), sugar beet (sub), spring barley (sb), potato (po), seed potato (spo), pea (pe), sunflower (s), sorghum (so), maize (m), rye-grass/maize (r-m), fennel/maize (f-m) and rice (ri). Data for winter wheat grown at Oensingen and Risbyholm in 2006–2007 are circled.

maize cropping, this study cannot claim to be representative of the NBP for all types of maize growing in Europe.

3.3. Carbon inputs

Carbon inputs, mainly through organic manure amendments, also varied considerably among sites and sometimes between years for the same site (see Table 5). The Lamasquère site received solid and liquid organic manure each year, corresponding to a Cinput ranging between 67 and 249 g C m^{-2}. Cioffi also received solid and liquid organic manure, but only for the rye-grass/maize cropping system, corresponding to a Cinput of only 19.2 g C m^{-2}. Vredepeel (sugar beet), Klingenberg (rapeseed, maize) and Oensingen (potato) received 433, 256, 176 and 121 g C m^{-2}, respectively, as solid manure. Molenweg (potato), Gebesee (winter wheat), Langerak (maize), Cioffi (fennel/maize), and Grignon (mustard/maize) received 53.5, 34.7, 26.8, 19.2, and 9.8 g C m^{-2}, respectively, as liquid manure. Lonzée received 66 g C m^{-2} year^{-1} as sugar beet lime in 2003-2004 just before sugar beet was grown. The amounts of Cinput through seeds and mother tubers were small (0.1 and 7.7 g C m^{-2}, respectively) in comparison with those from organic fertiliser or sugar beet lime and were smaller than the uncertainties associated with the estimation of NEP and Cexport (Béziat et al., 2009).

3.4. Net ecosystem carbon budget

3.4.1. General results

On average, the NECB was 138 ± 239 g C m^{-2} year^{-1}, corresponding to a C loss ranging from −258 g C m^{-2} year^{-1} at El Saler Sueca (rice, 2007–2008) to 645 g C m^{-2} year^{-1} at Cioffi (fennel/maize, 2006–2007) (Table 5). Seventy percent of the site-years had positive NECB values, corresponding to carbon losses, even though negative NEP values were observed for most of them. The reason for this observation is that Cexport was, on average, higher (462 ± 251 g C m^{-2} year^{-1} considering only sites where at least one-year fluxes were measured) than those associated with NEP (−284 ± 228 g C m^{-2} year^{-1}) and Cinput (−38 ± 90 g C m^{-2} year^{-1}). To assess the relative contributions of NEP, Cinput and Cexport to NECB, their absolute values were summed, and their relative contributions to the total were calculated. NEP, Cinput and Cexport represented, on average, 36.2, 4.9 and 58.9% of the NECB, and NEP (through uptake of CO_2) represented 88% of the C inputs. Therefore, NEP and Cexport had the greatest impacts on the annual C budget of the croplands examined. Even when considering only those crops grown with organic fertilisers, NEP and Cexport were usually the two primary factors driving the NECB (see Table 5).

3.4.2. NECB variability among sites

Even when methane emissions were considered, El Saler Sueca remained the site with the lowest (most negative) average NECB. These low NECB values can be explained by the low NEP values for rice, as discussed above. On average, rice fixed the most C, with a mean NECB of −228 ± 30 g C m^{-2} year^{-1} or −213 ± 30 g C m^{-2} year^{-1} depending on whether or not methane emissions were included, but these estimates did not take into account C losses associated with fire, horizontal transport of crop residues and DOC by water flows (during winter flooding). For 2007, we estimated that aboveground crop residues totalled 264 g C m^{-2}. Considering that close to 50% of aboveground crop residues can be exported with water flows, the NECB would be only −126 g C m^{-2} in 2007. The NECB was also very low at Risbyholm in 2006–2007 (winter wheat) because, exceptionally, no harvest occurred in that year (see above).

3.4.3. NECB variability among crops

Winter wheat had a mean NECB of 112 ± 198 g C m^{-2} year^{-1}, corresponding to a non-significant C loss because of high vari-

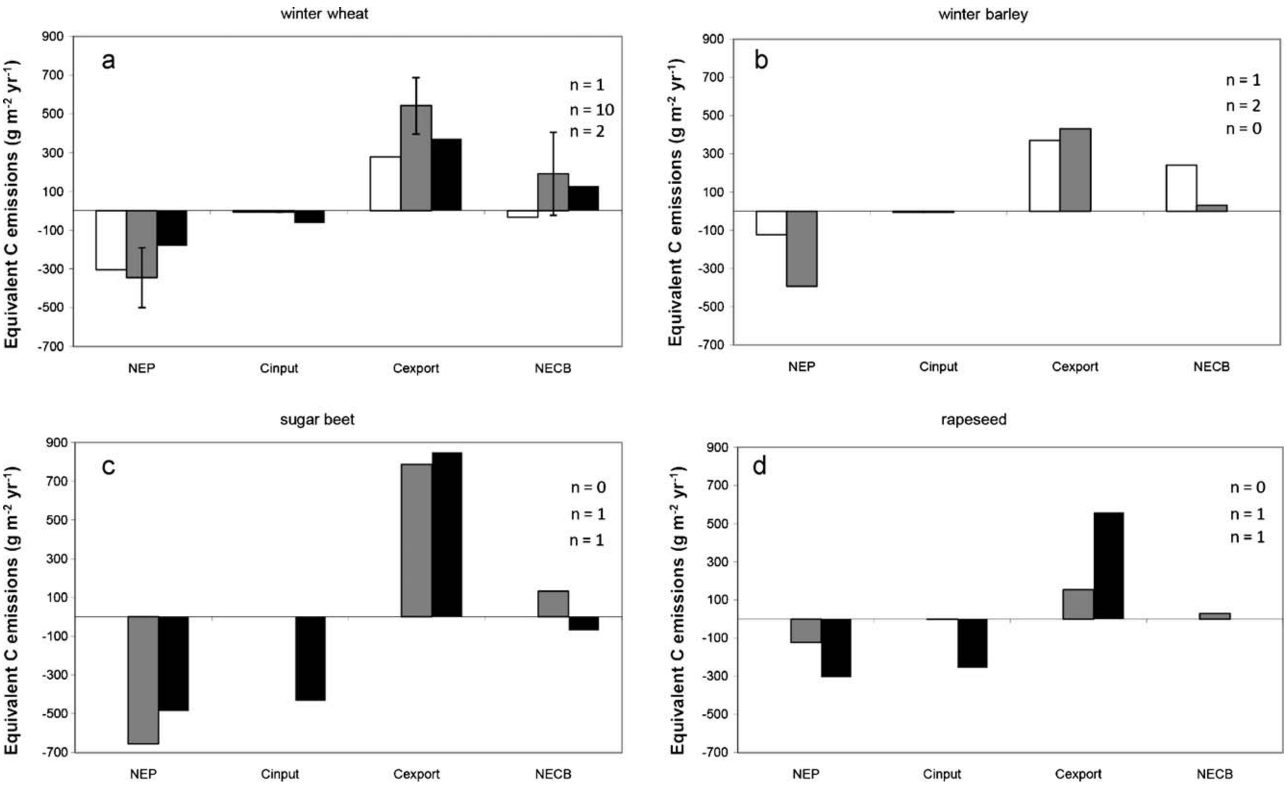

Fig. 4. Management effects on net ecosystem production (NEP), carbon inputs (Cinput), carbon exports corresponding to harvest and fire (Cexport) and net ecosystem carbon budget (NECB) for winter wheat (a), winter barley (b), sugar beet (c) and rapeseed (d). Data for winter wheat at Risbyholm in 2006–2007 are not included. White bars correspond to crops that only received mineral fertilisation and for which only grains were removed. Grey bars correspond to crops that received mineral fertilisation and for which grains and straw or tubers were removed. Black bars correspond to crops that received both mineral and organic fertilisation and for which grains and straw or tubers were removed. The number of datapoints used to calculate each bar is represented in the upper right corner: the upper value is for white bars and the lower for black bars. Vertical full lines (error bars) are ± the standard deviation of each measurement mean and were calculated when the number of sites per crop species was ≥3.

ability. NECB values were negative in 2006–2007 at Lonzée and Lamasquère, when winter wheat was grown. The 2006–2007 winter was, however, exceptionally warm in western and south-western Europe. Temperatures in January and February were close to 4.4 °C, which was 3 °C above the normal values at Lonzée and Lamasquère in 2007, and the NEP values observed at those sites were much lower than the ones observed at Lonzée in 2004–2005 and at Auradé (12 km from Lamasquère) in 2005–2006 for a winter wheat crop (see Table 5). Other crops, such as durum wheat, rapeseed, winter barley, spring barley, sugar beet, potato, sunflower and sorghum, were small sources of C with NECBs below 102 g C m^{-2} $year^{-1}$ (Fig. 2). Additionally, when comparing the different crops, the three terms contributing to the NECB were much larger for sugar beet compared to the other crops.

Finally, fennel/maize, rye-grass/maize, maize, pea and seed potato had large positive NECB values (Fig. 2) of 582 (n = 2), 480 (n = 1), 413 ± 91, 375 (n = 1) and 243 (n = 1) g C m^{-2} $year^{-1}$, respectively. The net carbon loss was significant for maize only (see Fig. 2) because the number of samples available for the other crops was too low. However, as discussed above, almost all of the aboveground biomass of the maize crops was exported for silage. Therefore, these results are not representative of maize fields used for grain production only.

3.4.4. NECB variability with management

Management practices varied considerably across the different sites (see Tables 1 and 5 and Fig. 2). Some sites exported only grain, while others also exported straw, and some received only mineral fertilisers, while others received both mineral and organic fertilisers. The assessed effects of management on the NECB are presented in Fig. 4 for crops (1) receiving only mineral fertiliser where grains were exported, (2) receiving only mineral fertiliser where grains and straw or tubers were exported, and (3) receiving both mineral and organic fertilisers where grains and straw or tubers were exported.

For winter wheat, NECB was negative when only grain was exported. Organic fertilisation could not compensate for C losses when all of the biomass was exported. For winter wheat receiving mineral fertilisation but for which grain and straw were exported, the variability in NEP and Cexport caused large variations in NECB (from −161 to 497 g C m^{-2} $year^{-1}$).

The NECB for winter barley was close to equilibrium when aboveground biomass was removed and was surprisingly positive (241 g C m^{-2} $year^{-1}$) when only grains were removed. In the latter case, NEP was small, and Cexport was rather similar in both treatments.

For all crops, organic fertilisation tended to reduce the NECB (see Fig. 4). For sugar beet and rapeseed, crops receiving both organic and mineral fertilisation had small negative NECBs, whereas those receiving mineral fertiliser had small positive NECBs. It was not possible to generalise the results due to a small number of samples. Moreover, the comparison of the effects of different fertiliser types on NECB for rapeseed is uncertain because the rapeseeds grown at Klingenberg (mineral plus organic fertiliser) and Risbyholm (mineral fertiliser only) were cultivated as winter crops and spring crops, respectively, due to very different climate conditions (see above).

3.5. Emissions from field operations

In this section, when emissions from field operations (EFO) are presented and discussed for the sites where NEP was measured, the numbers corresponding to all of the sites (including those where NEP was not measured) also appear in parentheses.

3.5.1. Emissions from machines

Emissions caused by the use of farm machinery represented, on average, only 5.0% (4.8%) of EFO and ranged between 2.6 g C-eq m^{-2} $year^{-1}$ (Auradé, sunflower) and 12.3 g C-eq m^{-2} $year^{-1}$ (Cioffi, fennel/maize). At Auradé, only five operations involving machinery were performed (ploughing, fertiliser application, sowing, pesticide application and harvest), while at Cioffi, where two crops were grown per year, nineteen operations were carried out. Soil preparation and harvest often represented a large part of the emissions associated with the use of farm machinery. For instance, at Carlow (spring barley, 2005–2006), ploughing (30 cm) and harvesting represented both 25.8% of machinery emissions. At Avignon (pea, 2004–2005), soil preparation (ploughing and multiple tillage events) and harvest represented 36.2% and 25.0% of emissions, respectively. Finally, at Lonzée (winter wheat, 2004–2005), tillage and harvesting represented 10% and 48.8% of emissions, respectively.

3.5.2. Emissions caused by fertiliser use

The manufacturing, transport, storage and application (causing N_2O emissions) of fertilisers represented between 15 and 94% of EFO, with, on average, 51.4 ± 34.9 (51.0 ± 31.9) g C-eq m^{-2} $year^{-1}$. Crops where only organic (Dijkgraaf and Langerak) or mineral fertilisers (see Table 1) were applied represented 25.2 ($n = 2$) and 49.1 ± 33.9 g C-eq m^{-2} $year^{-1}$, or 70.3% and 75.4% of EFO, respectively. For those same sites where only organic or only mineral fertilisers were used, fertiliser manufacturing accounted for 4.7 and 23.0 ± 16.9 g C-eq m^{-2} $year^{-1}$, respectively. This is not very surprising because according to Stout (1990), energy input associated with nutrients derived from animal manure is less than that when chemical fertilisers are used (energy for application of fertilisers is not included).

Generally, winter crops had higher emissions for manufacturing, transport and storage of fertilisers than summer crops because of higher fertiliser inputs. By contrast, emissions at Klingenberg (spring barley), Gebesee (sugar beet), Dijkgraaf (maize), Avignon (peas and sorghum), Auradé (sunflower) and Langerak (maize), all being spring or summer crops, were the lowest (below 9 g C-eq m^{-2} $year^{-1}$). Emissions were highest at Cioffi because two crops were grown per year at that site.

The emissions of N_2O from fertilisers represented, on average, 40.4% (40.6%) of EFO, or 27.4 ± 18.9 (26.8 ± 17.3) g C-eq m^{-2} $year^{-1}$. Emissions from fertilisers ranged between 0 g C-eq m^{-2} $year^{-1}$ at Avignon for pea and sorghum due to no fertilisation to 87.7 g C-eq m^{-2} $year^{-1}$ in 2006–2007 for fennel/maize at Cioffi, both of which received fertilisers. However, similar EFs were used for all sites and types of fertilisers, even though Kuikman et al. (2006) showed that EFs can vary spatially and with fertiliser type. Moreover, summer crops had lower emissions than winter crops because of lower fertiliser inputs. However, Skiba et al. (1996) reported that N_2O emissions for winter crops are lower than for summer crops because the latter are fertilised under soil temperature conditions more conducive to denitrification. Therefore, it is very likely that these results do not represent the real variability in N_2O emissions. Still, as GHG emissions associated with fertilisers can represent up to 94% of EFO (including manufacturing; Hillier et al., 2009), efforts should be made to enhance nutrient use efficiency by minimising losses caused by erosion, leaching and volatilisation, perhaps by including in the rotation crops that can fix atmospheric nitrogen and improve the recycling of nutrients contained in the crop residue. Indeed, N_2O emissions corresponding to the mineralisation of crop residues represented, on average, 11.0% (10.6%) of EFO, or 5.8 ± 5.1 (6.0 ± 5.4) g C-eq m^{-2} $year^{-1}$. Considering all sites, winter barley and winter wheat had the lowest N_2O emissions associated with crop residues, with 2.2 ± 1.1 and 2.5 ± 1.7 g C-eq m^{-2} $year^{-1}$, respectively (see Fig. 2). Finally, considering all sites, potato, pea, rice, fennel/maize and sorghum had mean N_2O emissions caused by crop residues above 10 g C-eq m^{-2} $year^{-1}$.

In total, N_2O emissions represented 51.4% (51.2%) of the EFO. In the future, efforts should be made to systematically and continuously measure the N_2O emissions and NEP in the field to reduce uncertainties in the EFO and total GHGB for specific crops. Such an effort was made at the Cioffi and Grignon sites. At Cioffi, it was found that the emission factors (this refers to the amount of N_2O emitted from the various mineral and organic N applications to the soil) were 0.87% in 2007 and 0.26% in 2008, both consistently lower than the reference IPCC (2006) value used in this study. Therefore, we might have overestimated N_2O emissions at some sites.

3.5.3. Emissions caused by pesticide use

The manufacturing, transport, packaging and storage of pesticides represented only 1.6% (1.5%) of the EFO. However, the number of treatments varied considerably among crops and sites. There were no treatments for pea and potato (at Avigon, 2004–2005 and Oensingen, 2005–2006), only 1 and 2 for rapeseed and triticale, respectively (Risbyholm, 2007–2008 and Lamasquère, 2004–2005), and up to 7 and 13 for sugar beet and seed potato, respectively (Lonzée, 2003–2004 and 2005–2006), usually combining several chemicals at once. The cost in C-eq corresponding to the use of pesticides therefore varied greatly (see Table 5) depending on the number of treatments and the chemicals used. On average (including all sites), the emissions corresponding to pesticide applications were higher for seed potato and sugar beet (5.4 and 2.0 g C-eq m^{-2} $year^{-1}$, respectively) and were less than 1.4 g C-eq m^{-2} $year^{-1}$ for the other crops (Fig. 2). The maximum emissions from pesticide applications represented 9.1% of the EFO (for seed potato at Lonzée in 2005–2006). Therefore, for most crops, any efforts to improve the accuracy of EFO estimates should focus on these additional contributions.

3.5.4. Emissions caused by irrigation

For irrigated sites, irrigation only represented 9.5% (8.8%) of the EFO, although the methods differed between sites. Gravimetric techniques were used at El Saler Sueca, and these are less energy-consuming than sprinklers, centre-pivot, frontal ramps or solid rolls, which were used at Vredepeel, Cioffi, Avignon and Lamasquère, respectively. Therefore, even if the amount of irrigation was high at El Saler Sueca, C-eq emissions were low compared to Cioffi. Cioffi received a large amount of irrigation (between 300 and 416 mm), and this represented between 9.4 and 14.5% of the EFO. However, Avignon was the site where irrigation represented the largest part of the EFO, with a value of 19% for a pea crop. The reason for this observation is that neither pesticide nor nitrogen fertiliser was added to the peas; therefore, there were no N_2O emissions caused by fertilisers. Overall, our results are consistent with emissions reported in the literature (see Lal, 2004).

3.5.5. EFO variability among sites

For the different site-years or crops, values for the EFO could be sorted into three groups. In the first group, EFO was low, between 0 and 30 g C-eq m^{-2} $year^{-1}$. This group included sorghum, pea and sunflower, which have short growing seasons and require few inputs. Maize cultivated at Langerak and receiving only organic fertilisation also belongs to this group, along with some cereals, such as winter barley (Gebesee, 2004–2005) and spring barley (Klingenberg, 2007–2008). On average, however, winter and spring barley belong to the second group. In the second group (most of the site-years), the EFO and mean EFO per crop ranged between 30 and 100 g C-eq m^{-2} $year^{-1}$ and between 40 and less than 80 g C-

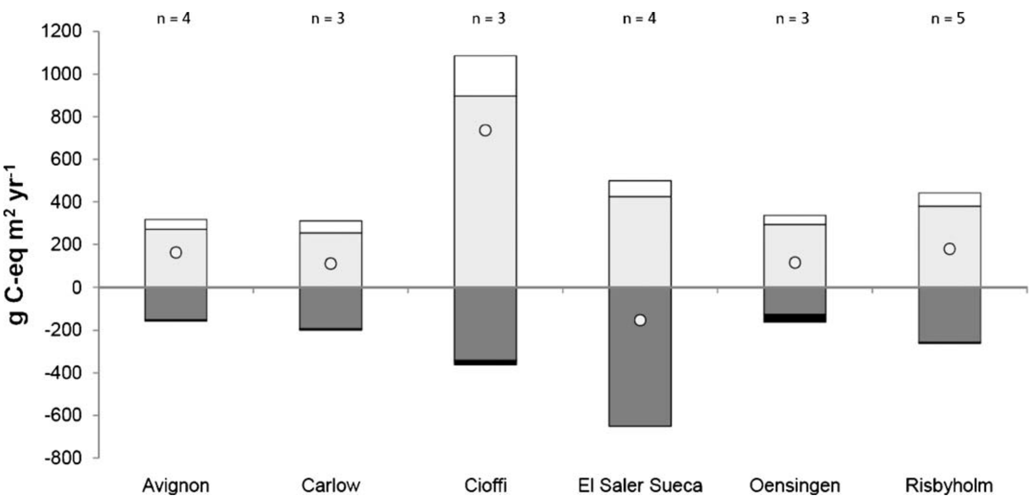

Fig. 5. Annual mean values of the different terms composing GHGB (white dots) for sites where the crop rotation was completed or where monoculture was grown: Cinput (black bars), NEP (light grey bars), Cexport (dark grey bars) and EFO (black bars). The number of years corresponding to the full rotation for each site is shown.

eq m^{-2} $year^{-1}$, respectively. Winter barley is often in the low range of that group. In the third group (five site-years), the EFO was above 100 g C-eq m^{-2} $year^{-1}$. High EFOs were found for Cioffi and Lonzée (sugar beet, 2003–2004). At Cioffi, two crops were cultivated in the same year; therefore, the inputs were high. For instance, both maize and fennel or rye-grass received fertilisers. At their maximum in 2006–2007, the EFO represented 26.7% of the total GHGB. Even this result has restricted generality, and it would be interesting to investigate further the increase in EFO for systems with two crops per year instead of one to evaluate their environmental impact. Indeed, it is likely that similar systems will become more frequent in the future because of projected lengthening of the growing season due to global warming (allowing two main crops per year) and because of increasing pressure to produce more food and energy per cropland area and to promote cover crops during winter to reduce soil erosion, enhance carbon sequestration and reduce nitrate leaching.

3.6. Total GHG budget

To assess the relative contributions of the different terms to the total GHGB, their absolute values were summed, and their relative contributions were calculated. Overall, the EFO represented only 7.6% of the total GHGB compared to 53.4, 33.4 and 4.5% for Cexport, NEP and Cinput, respectively. Therefore, the NEP (through uptake of CO_2) represented 88% of the negative radiative forcing, and Cexport represented 88% of the positive radiative forcing. The use of machinery, manufacturing, transport and storage of pesticides and fertiliser, N_2O emissions from fertilisers and from residues and irrigation made only small contributions (0.6, 0.1, 2.8, 3.2, 0.7 and 0.3% of total GHGB, respectively). However, when EFO was directly compared to GHGB (without considering the absolute values of all of the terms), it represented 32% of the GHGB (64.8 g C-eq m^{-2} $year^{-1}$ for EFO; over 203 g C-eq m^{-2} $year^{-1}$ for GHGB). N_2O emissions alone represented nearly 16.4% of the GHGB.

With a mean GHGB of 203 ± 253 g C-eq m^{-2} $year^{-1}$ for all site-years where NEP could be estimated, crops, on average, acted as GHG sources. Overall, the total GHGB ranged from -182 g C m^{-2} $year^{-1}$ for rice at El Saler Sueca (2007–2008) to 880 g C m^{-2} $year^{-1}$ at Cioffi (fennel/maize, 2006–2007) (see Table 5). Nine site-years (four of them being rice) over a total of 41 had a negative total GHGB, meaning that they were acting as GHG net sinks. For most site-years, Cexport and emissions of GHGs associated with field operations exceeded net carbon fixation through the NEP and C inputs from organic fertilisers and seeds.

On average, rice was a net GHG sink, with a mean GHGB of -153 ± 30 g C-eq m^{-2} $year^{-1}$ (-138 ± 30 g C-eq m^{-2} $year^{-1}$ when considering methane emissions) (see Figs. 2 and 5). All other crops had mean positive values for GHGB. As for the NECB, crops having very negative NEPs did not always have the best potential for fixing C (low or negative NECBs) and were not necessarily the most efficient crops in terms of total GHG emissions. This is particularly obvious when considering maize. Maize alone or in combination with fennel or rye-grass had the highest positive GHGB values at 457 ± 68 g C-eq m^{-2} $year^{-1}$, 798 ($n = 2$) g C-eq m^{-2} $year^{-1}$ and 612 ($n = 1$) g C-eq m^{-2} $year^{-1}$, respectively (see Fig. 2). Although the results from the Cioffi site cannot be generalised, it is interesting to note that longer periods with vegetation cover do not necessarily improve the total GHGB. However, when net assimilation from a fallow crop or voluntary re-growth was accounted for in the NEP (and in the total GHGB) at Oensingen (winter barley, 2004–2005) and Avignon (winter wheat, 2005–2006), the C budgets were improved by 280 and 18 g C m^{-2} $year^{-1}$, respectively.

Values for GHGB are presented in Fig. 5 for sites where the crop rotation was complete or that were cultivated with a monoculture. El Saler Sueca was the only site acting as a GHG sink, and Cioffi was the largest source, with an annual emission of 730 g C-eq m^{-2}. The NEP and organic inputs could not compensate for the large C exports occurring twice a year, and the EFO was always the highest at that site because large inputs were required to grow two crops a year. Indeed, emissions associated with irrigation, use of machines, fertiliser manufacturing, and N_2O emissions caused by nitrogen fertilisation were the highest at that site. At other sites, GHG emissions ranged between 111 and 179 g C-eq m^{-2} $year^{-1}$. Carlow was the smallest GHG source, even though its NEP was below average, as only grains were exported, and the EFO was below average. Results from Oensingen and Risbyholm should be considered special cases because at those two sites, one of the crops was not harvested (exported) during the crop rotation. However, in the case of Oensingen, the situation would also have occurred under normal conditions. The hail damage to the potato crop would have been covered by hail insurance; therefore, the farmer would not have harvested the potatoes had the researchers not insisted on keeping them growing to examine the effect of hail damage on the NEP.

3.7. Assessment of crop efficiency

Crop efficiency (CE) was calculated as the ratio between Cexport (yield) and total GHGB. Results are presented in Fig. 6 for

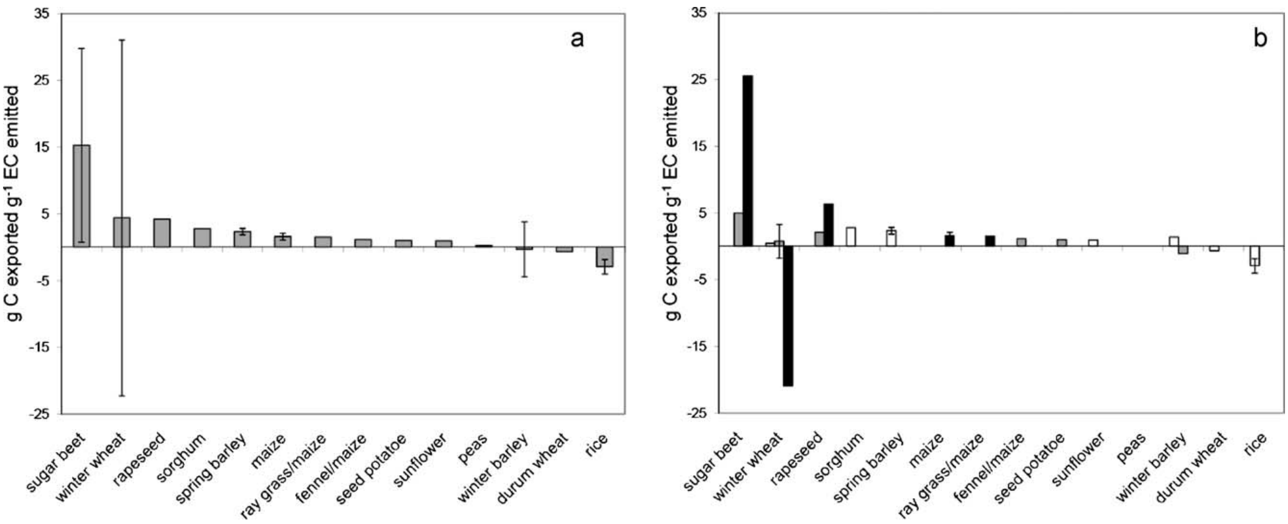

Fig. 6. (a) Mean crop efficiency per crop species, calculated as the ratio between Cexport during harvest and annual GHG budget calculated in C-eq. (b) Mean crop efficiency per crop species for crops that only received mineral fertilisation and for which only grains were removed (white bars), for crops that only received mineral fertilisation and for which grains and straw were removed (grey bars) and for crops that received both mineral and organic fertilisation and for which grains and straw were removed (black bars). Vertical full lines (error bars) are ± the standard deviation of each measurement mean. They were calculated when the number of sites per crop species was ≥3. Data for winter wheat and potato at Risbyholm in 2006–2007 and Oensingen in 2005–2006 are not included because no harvest was performed.

crops where at least one year of NEP was calculated and a harvest was taken by the farmer (Oensingen 2005–2006 and Risbyholm 2006–2007 are excluded from this analysis). On average, winter crops were much more efficient than summer crops (Fig. 6a), with CEs of 4.6 ± 6.4 and 0.8 ± 1.9 g C exported g^{-1} C-eq emitted, respectively (rye-grass/maize and fennel/maize were not included in the calculations). Durum wheat and winter barley had negative mean CEs (−0.7 ± 4.1 and −0.3 ± 4.1 g C exported g^{-1} C-eq emitted, respectively), indicating that they were acting as GHG sinks, but variability was high for winter barley. Sugar beet, winter wheat and rapeseed were among the most efficient crops, with mean CEs of 15.3 ± 14.5, 4.4 ± 26.7 and 4.2 (n = 2) g C exported g^{-1} C-eq emitted, respectively (Fig. 6a). Most crops had a CE above 1 or below 0 g C exported g^{-1} C-eq emitted. However, seed potato, sunflower and pea had mean CEs below 1 g C exported g^{-1} C-eq emitted. These results indicate that these crops were producing more GHGs than yield on a C basis. Of course, these results should be treated with caution due to the small number of sites studied and differences in management practices.

In general, the high variability in the CEs was to a large extent explained by differences in management (see Fig. 6b), and organic fertilisation improved the CEs for all crops whose straw was removed. For winter wheat, the CE was negative when organic plus mineral fertilisation was used, even if all aboveground biomass was exported. This result shows that for winter wheat, net GHG fixation was possible, especially when organic fertilisation was used, despite higher Cexport. For sugar beet and rapeseed, the CEs were also higher when organic plus mineral fertilisation was used. Considering crops producing oil, rapeseed (4.2 g C exported g^{-1} C-eq emitted) was, on average, more efficient than sunflower (0.9 g C exported g^{-1} C-eq emitted), but the methods of fertilisation and the proportion of total biomass exported differed.

Maize alone or in combination with fennel or rye-grass never exceeded a CE of 1.6 g C exported g^{-1} C-eq emitted. In this study, combining maize with rye-grass or growing maize alone did not affect the CE on an annual basis. Finally, sorghum produced 2.8 g C g^{-1} C-eq emitted, and rice had a negative CE, meaning that it was a GHG sink producing biomass. For the reasons mentioned above, this latter result should be considered with much caution.

4. Discussion

In this study, we assessed, for the first time, the effects of management practices on GHG emissions by analysing the NEP obtained from eddy covariance determinations, lateral fluxes due to harvest and manure, and GHG emissions produced by field operations and decomposition of crop residues for 15 European cropland sites. These sites covered a large climate gradient and a variety of crops and cropland management practices, including 14 different crop species encompassing 41 site-years. Most of those sites were converted to cropland several decades ago, and it could have been expected that the soils would be close to equilibrium with respect to carbon.

The crops examined in the current work are representative of more than 73% of the cropping areas (FAOSTAT) in Europe (EU 27) and more than 59% of the arable land (see EUROSTAT, 2008 and Swiss Federal Statistical Office, 2008) in the nine countries covered by this study. While these sites may be broadly representative of the area covered by wheat (25.1% of arable lands versus 26.8% in this study), winter and spring barley (14.8% versus 14.6% in this study) and rapeseed (4.4% versus 4.9% in this study), they overestimated maize, rice, potato and sugar beet (differences in surface area of 8.3, 9.1, 3.2 and 2.2%, respectively; EUROSTAT, 2008). Moreover, the maize sites were not representative of maize grown in Europe because, at the sites where NEP was measured, maize was used for silage and only represented 46% of the maize area in the countries contributing to our study. Representativity is also discussed in Kutsch et al. (2010), who focus on the NECB of full crop rotations.

4.1. Net ecosystem production

In this study, most NEPs were negative, corresponding to a sink for atmospheric CO_2 by the crops, which is consistent with other studies on maize/soybean rotations in North America (Baker and Griffis, 2005; Bernacchi et al., 2005; Hollinger et al., 2005; Pattey et al., 2002; Suyker et al., 2005; Suyker et al., 2004; Verma et al., 2005), rice (Saito et al., 2005), and winter wheat and triticale (Ammann et al., 1996; Anthoni et al., 2004; Baldocchi, 1994). However, NEPs were positive for several crops. The reasons for positive or negative NEPs are various and can include a combination of several factors, as discussed above and in other papers in this issue (see Eugster et al. (2010), Kutsch et al. (2010). Decomposition of crop material from previous years may explain part of the variability. The number of days of active vegetation cover was also identified as one of the factors influencing the NEP in this study.

In general, NEP increased in absolute value with NDAV for winter crops. However, two points corresponding to potato and winter wheat grown at Oensingen and Risbyholm, respectively, in 2006–2007 seem to lie outside of this relationship between NDAV and NEP. Potatoes were not harvested in the previous year at Oensingen, and winter wheat was not harvested that year at Risbyholm. Therefore, heterotrophic respiration may have been a significant component in 2006–2007 at Oensingen and during the late season at Risbyholm, thereby decreasing the NEP. There was only one positive NEP value for winter crops, corresponding to winter wheat at Gebesee in 2006–2007. This result was probably the consequence of a late sowing because the previous crop (sugar beet) was harvested in late autumn. Finally, rapeseed grown at Risbyholm was considered a summer crop because it was sown in May. Additionally, the NEP at Risbyholm was much lower than at Klingenberg, probably due to a much shorter growing season for rapeseed at Risbyholm compared to Klingenberg.

The pea crop had the highest positive NEP value, and sunflower was close to equilibrium ($-8.5\,g\,C\,m^{-2}\,year^{-1}$). The likely reason is that both crops had a rather short growing season with low LAI values (data not shown). Moreover, sunflower had a rather low photosynthesis rate compared to most other crops (see Béziat et al., 2009). Therefore, C assimilated during the growing season was compensated by small C losses of a longer duration during the extended period with bare soil or limited vegetation cover.

Average NEP values for maize were $-141 \pm 200\,g\,C\,m^{-2}\,year^{-1}$, but variability between sites was large. At Klingenberg, a hail event in July 2007 caused significant damage to the maize plants, resulting in a reduction in LAI and probably a reduction in C net fixation. Overall, the values observed for the sites examined were lower than those found in the literature, which vary between −381 and $-572\,g\,C\,m^{-2}\,year^{-1}$ in Verma et al. (2005) for the Mead sites in Nebraska, USA. However, as discussed in Béziat et al. (2009) for Lamasquère, the sites in this study were rain-fed or received less irrigation (see Table 1) compared to the Mead site, with irrigation ranging between 302 and 378 mm. Therefore, more irrigation would probably have improved the C budget for the maize, but other factors, such as soil types, crop varieties and density, may also cause differences in the NEP.

Rice was the crop with the lowest mean NEP value. The presence of water covering the ground at the El Saler Sueca site during the vegetation period reduced both ecosystem respiration (the lowest values of all sites; see Eugster et al., 2010) and photosynthesis limitation (high stomatal conductance), thereby enhancing the NEP. Algal and cyanobacterial photosynthesis associated with the water column may also have contributed to increased carbon uptake.

On some occasions, re-growth events and weed development increased the number of days of active vegetation cover, but those events are very dependent on climate and are usually interrupted by soil preparation prior to the sowing of the next crop. Béziat et al. (2009) estimated that re-growth events and weed growth caused a net fixation of approximately $50\,g\,C\,m^{-2}$ after triticale at Lamasquère in 2005–2006. This re-growth occurred because the summer was relatively wet and because soil preparation occurred late in the season. Soil preparation, disking, stubble cultivation and use of herbicides may delay, prevent or interrupt voluntary re-growth and kill weeds. Therefore, postponing the operations or encouraging cover crops such as fennel or rye-grass, as in Cioffi, or a Phacelia/clover-based mixture, as in Oensingen, can improve the carbon budget of agricultural ecosystems. When maize was combined with cover crops, as in Cioffi, NDAV and NEP increased compared to other sites where no cover crop was grown.

Because NEP is the second most important term in the NECB and GHGB calculation, it is important to estimate it accurately. Differences in integration periods or gap-filling methods produce differences in NEP. In Béziat et al. (2009), NEP for sunflower was found to be $28\,g\,C\,m^{-2}\,year^{-1}$ compared to $-8.5\,g\,C\,m^{-2}\,year^{-1}$ in this study because of small differences in the integration periods. With different integration periods, Aubinet (2009) and Prescher et al. (2010) also calculated slightly different NEP values for Lonzée and Klingenberg, respectively. Differences in gap-filling methods also produced differences in NEP values for sugar beet between this study and Moureaux et al. (2006). Uncertainties in the NEP measurements by means of the eddy covariance caused by systematic and random errors (see Osborne et al., 2010) have been discussed in recent years (Hollinger and Richardson, 2005; Richardson and Hollinger, 2007; Richardson et al., 2006; VanGorsel et al., 2007; Aubinet, 2008; Finnigan, 2008; Lasslop et al., 2008; Moureaux et al., 2008; Béziat et al., 2009) and are summarised in Kutsch et al. (2010).

4.2. NECB and NBP

On average, the NECB for the crops examined was $138 \pm 239\,g\,C\,m^{-2}\,year^{-1}$, corresponding to a C source, but the uncertainty surrounding this estimate was larger than the source itself. Considering a mean soil organic C content of $5300\,g\,C\,m^{-2}$ (53 t of organic C ha^{-1} to a depth of 30 cm; Smith et al., 2000) in European agricultural soils, the mean NECB would correspond to an annual loss of $2.6 \pm 4.5\%$ of the soil organic C content. Of course, this value should be considered with caution because the crop species, soil conditions and management practices in this study are probably not fully representative of all croplands found in Europe (see Osborne et al., 2010). However, the variability around this mean is probably rather representative of the variability in NECB for European croplands and reflective of the short-term (year-to-year) variability in the NECB. Clearly, determinations made over longer periods would be required before a robust assessment of the sustainability of current land use practices could be quantified (see also Eugster et al., 2010). If our results are to be considered representative of European croplands, this result may be surprising as cropland soils in Europe are expected to be near equilibrium with respect to carbon because the sites have been managed as croplands for many years.

Kutsch et al. (2010) found slightly lower but still positive values for the NECB ($91 \pm 203\,g\,C\,m^{-2}\,year^{-1}$) at eight sites with at least four years of continuous measurements. Our results support a previous study by Janssens et al. (2003), who estimated a value for NECB of $90 \pm 50\,g\,C\,m^{-2}\,year^{-1}$ for European croplands based on longer term studies of soil C stock inventories (10 years or more). However, our results contrast with more recent studies based on modelling and carbon inventories that suggest that European cropland soils are close to equilibrium, being either small sources (Smith et al., 2005; Bondeau et al., 2007; Schulze et al., 2009) or small sinks (Gervois et al., 2008). This difference may be explained by the fact that the soil characteristics, management practices, and climatic conditions of the sites used in the current study are not representative of those found across the EU and/or by the difficulty that models have in representing the variability of the NEP or the different management practices used. Moreover, it is worth noting that uncertainties relating to Cexport (59% of NECB) and Cinput (5% of NECB) are proportionally bigger than the uncertainty in the NEP for most sites. In a recent study, Béziat et al. (2009) found that uncertainties in C removal by harvest and in C inputs as organic fertilisation caused larger uncertainties for NBP than for NEP. Moreover, Aubinet et al. (2009) found an overall C budget error of $\pm$140 g C m^{-2} over four years for a crop rotation, which might be exceeded for some of the sites examined in the current study.

In the literature, as in our study, contrasting NECBs and NBPs were found in association with different management practices, crops and cropping systems. In the USA, for example, some maize/soybean rotations were found to be small albeit non-significant, carbon sinks (Baker and Griffis, 2005; Dobermann et al., 2006; Hollinger et al., 2005; Hollinger et al., 2006), and others were small but non-significant carbon sources (Grant et al., 2007; Verma et al., 2005). At the USA sites, the crops received mineral fertilisation, and only grains were exported. In contrast, a carbon source was reported in north China (Jun et al., 2006) over winter wheat/maize rotations (grain exportation, fertilisation not specified) with NBP ranging from 108 to 341 g C m^{-2} $year^{-1}$. Finally, at Lonzée, for the full four-year rotation receiving mineral fertilisation, Aubinet et al. (2009) observed a mean NBP of 42 g C m^{-2} $year^{-1}$. However, they concluded that the warm 2006–2007 winter may have led to an underestimation of what might be regarded as more typical NBP values; by substituting the 2004–2005 winter for that of 2006–2007, the NBP was found to be 90 g C m^{-2} $year^{-1}$.

Climate and management can cause large differences in yield, NEP and NECB among sites, even for the same or similar crops. For all crops, organic fertilisation tended to lower the NECB (see Fig. 4) of the present crop, but its effect on the subsequent crops is more difficult to assess. In most cases, the harvest index and the fate of the harvestable product drives the proportion of NPP that will be exported, thereby influencing the NECB. For farms specialising in cereal production, it is more likely that only the grains will be exported so that most of the biomass (approximately two thirds of the total biomass including roots) produced in the field could potentially remain there, with most of it being progressively decomposed and a small part of it increasing the soil carbon pool (see Osborne et al., 2010). However, in a number of situations, baled straw may be removed for commercial and/or local reasons. Fields where the biomass is exported for silage or biomass energy will lose most of the C fixed by the plant during the growing season at harvest. If this loss of C is not compensated for by animal manure application, it is more likely that the NECB will correspond to C losses from the soil. Other alternatives, such as reduced tillage or the introduction of a legume cover crop, and their effects on soil C stocks and GHGB have been investigated (see Robertson et al., 2000). More studies assessing the effects of various management regimes at one site and their impacts on NPP, NEP, NBP and GHGB are needed to provide general recommendations. Additionally, these studies should involve the same crops at a range of locations.

4.3. Emissions from field operations

Collecting information and estimating the GHG emissions for the field operations for all sites and years represented a huge task. Some emission factors could not be found in the literature for some pesticides. There was also a lack of updated emission factors for pesticide and fertiliser production, the manufacture of machinery, etc., making it difficult to quantify the actual C cost with complete accuracy. For instance, many EFs used in this study for pesticides were from a 13-year-old study (Gaillard et al., 1997). However, our estimates of fluxes from machinery, N_2O emissions and fertiliser manufacture, transport and storage, and their respective contributions to GHGB, were consistent with figures found in Robertson et al. (2000).

Unfortunately, because N_2O and CH_4 emissions were not measured continuously at the sites examined, they had to be estimated from EFs found in the literature. However, EFs for N_2O emissions may vary considerably depending on soil conditions and sources of nitrogen (Kuikman et al., 2006). Because we estimated that N_2O emissions represented close to 50% of the EFO and close to 16% of the GHGB, efforts should be made to generalise N_2O measurements at crop sites to assess the GHGB. In Robertson et al. (2000), N_2O emissions represented close to 45% of the GHGB, but emissions from pesticides, maintenance, manufacture and amortisation of the machines were not accounted for. Similarly, because methane emissions may have represented close to 10% of the GHGB for rice crops according to EFs found in the literature, measuring them for rice paddies would improve assessment of their GHGB. Conversely, Roberston et al. (2000) measured methane fixation for a maize/soybean/wheat rotation representing close to 4 g C-eq m^{-2} $year^{-1}$. Therefore, methane oxidation at other crop sites in addition to El Saler Sueca may have counterbalanced a small part of the GHG emissions, thus improving the GHGB. Methods for measuring N_2O and CH_4 fluxes are listed in Smith et al. (2010).

In the current study, most sites are representative of medium- to high-input farm types, but the share of agricultural area managed by farms identified as low- and medium-input farm types increased slightly between 1990 and 2000 across the EU-12 (EUROSTAT, 2002). Low input farms represented 26% of total utilised agricultural area (UAA) in 1990 compared to 28% in 2000. Although a high proportion of the agricultural area is still managed by high-input farms, these are decreasing in importance. They represented 44% of the UAA in 1990 compared to 37% in 2000 for the EU-12 (EUROSTAT, 2002). Generally, high-input farm types are predominant in the Netherlands, Belgium, southeastern United Kingdom, northern France, northern Italy and northern Greece. However, trends of increasing use of inputs have also been identified in regions dominated by low-input farm types, such as in Mediterranean Member States and Scotland. None of our sites was cultivated with organic crops, but these only represented 3.2% of the UAA of the EU-15 in 2002, with a quarter of it being located in Italy (EUROSTAT, 2002). However, this number is increasing each year, and organic farming should represent 10% of the UAA in Belgium and the Netherlands and up to 20% of the UAA in Germany in 2010, according to their respective national action plans (EUROSTAT, 2002). Moreover, in the EU-15, the irrigated area increased by 14.5% between 1990 and 2000 and represented 11.1% of the UAA in 2000, compared to 29% of the site-years in our study (EUROSTAT, 2002). Finally, for the EU-15, the amount of mineral and organic nitrogen fertilisers represented 76 and 57 kg N ha^{-1} in 1990 and 74 and 57 kg N ha^{-1} in 2000, respectively, compared to 152 and 25 kg N ha^{-1} in this study. For these reasons, the mean EFO reported in the current study may overestimate the EFO at the European scale by close to 30%. Finally, emissions from field operations represented 33% of the final GHGB. A realistic 30% error in EFs or a 30% overestimation of the EFO would change the GHGB by 10%. Therefore, efforts should be made in future studies to improve the estimates of the emission factors for the different field operations and to cover the main management regimes for Europe.

4.4. GHG budgets

The mean total GHGB was 203 $\pm$ 253 g C-eq m^{-2} $year^{-1}$; therefore, the combined effect of the crops examined was that they acted as a GHG source. This number is comparable to the 114 g C-eq m^{-2} $year^{-1}$ found for a conventional tillage maize/soybean/winter wheat rotation in the USA but is much lower than the 893 to 1189 g C-eq m^{-2} $year^{-1}$ for a rice/wheat/cowpea rotation in India reported in Robertson and Grace (2004). Combining bottom-up and top-down modelling approaches at the European scale, Schulze et al. (2009) found a value of 40 $\pm$ 40 g C-eq m^{-2} $year^{-1}$ for croplands between 2000 and 2005 (some of the emissions associated with farm operations such as use of pesticides or irrigation are not accounted for). This number is within the range of our observations, and our study emphasises that the modelling of NECB or total GHGB for croplands at regional to continental scales is challenging because if NEP is to be represented as accurately as possible, it is also essential to have good representations of (1) the variability in management that determines C inputs and exports and (2) the variability in emissions caused by field operations.

NEP and Cexport represented 88% of the negative and positive radiative forcing and close to 140 and 228% of the GHGB, respectively. The NEP is the result of two large and opposite terms, primarily photosynthesis (GPP) and ecosystem respiration (R_E). Therefore, a small increase in GPP or reduction in R_E would improve NEP, NECB and GHGB in a noticeable way. Similarly, even a small reduction in Cexport would substantially improve the NECB and GHGB. The biggest proportion of additional emissions from field operations came from fertilisation and N_2O emissions from fertilisers, representing close to 12 and 14% of GHGB, respectively. Comparing sites where only organic or only mineral fertilisers were used, we confirmed the results by Stout (1990), who claimed that the energy input associated with nutrients derived from animal manure is less than that from the use of chemical fertilisers (energy for application of fertilisers is not included).

The cost of transformation of the products leaving the field must also be taken into account, but that is beyond the scope of this study (see Hillier et al., 2009). Crops having very negative NEPs did not consistently have the best potential for fixing C (low or negative NECBs) and were not necessarily the most efficient crops in terms of total GHGB. However, when net assimilation from fallow crops or voluntary re-growth was accounted for in the NEP (and therefore in total GHGB), the C budgets improved. A possible 50 g C m^{-2} assimilation by voluntary re-growth, as in Lamasquère in 2006, would compensate for the GHG emissions associated with fertiliser use. Therefore, encouraging or preserving vegetation cover on croplands could improve C or GHG budgets, assuming this does not generate additional emissions from heterotrophic respiration, N_2O emissions or EFO, which could counteract this C benefit.

From the results of the present study, it is obvious that calculating C budgets for crops and associated agricultural activities without considering biospheric fluxes, and particularly net CO_2 exchanges between crops and the atmosphere, would strongly overestimate total GHG emissions. In this study, this result corresponds to a 240% overestimation of the mean GHG emissions. Therefore, taking into account the NEP is essential when assessing C or GHGB for crops, especially energy crops being compared to other energy sources. However, the accuracy of such estimates is strongly related to the limited information on GHG emissions from management activities.

4.5. Crop efficiency

At the European scale, inputs tended to increase between 1990 (EU-12) and 2000 (EU-15) for cereal farms. High-input areas also increased in size for permanent crops, but at the same time, cereal yield increased from about 5.2 t ha^{-1} to more than 6.4 t ha^{-1}; it also increased for mixed cropping systems from 6.0 to 6.5 t ha^{-1} (EUROSTAT, 2002). These trends point to increased efficiency in the use of farm inputs. Of course, our data are too limited to confirm this trend, but calculation of the GHGB allowed us to compare crop efficiencies (CE) and their variability by management regime. CEs varied greatly among crops as well as between management regimes. Additionally, organic fertilisation increased the CE for all crops. Indeed, organic fertilisation reduced GHG emissions because the benefits from the amount of C imported as manure were not offset by the N_2O emissions associated with fertiliser application or manure production. From our data, it was not possible to draw general conclusions regarding which fertilisation method had a more beneficial effect on crop production because none of the sites fertilised with organic amendments was harvested for grain only and because more data are needed to perform a statistical analysis.

Of course, part of the variability in the CE is caused by differences in factors such as soil, climate and management. Studies comparing the CEs for different crops, in similar climatic and soil conditions and for comparable management regimes are needed to better understand this variability. For instance, it would be interesting to compare CE for sunflower and rapeseed grown for biofuel at different locations with similar management practices. Similarly, the effects of different management regimes on CE for a single crop at a single site should be investigated. For these reasons, and because our dataset was too small for most crop species, CEs from this study should not be considered definitive. Finally, it would be useful to develop a framework for the comparison of crops' efficiency with respect to the GHGB or other criteria, such as water use efficiency, to develop a broader vision of the impacts of crop production on the environment.

5. Conclusions

In this study, combining NEE flux measurements integrated over one year with lateral C flux inventories at the plot scale allowed us to estimate yearly net cropland carbon budgets for a range of sites in Europe. EU croplands as a whole proved to be C sources (138 ± 239 g C m^{-2} $year^{-1}$), but the variability of this estimate was larger than the estimate itself. This variability was caused by differences in climatic conditions, management regimes and crop species. Of course, longer integration periods are necessary to assess the NBP and to evaluate climatic variability effects on the NECB and NBP; however, because the detection of short-term changes in soil C stocks using conventional means is problematic (Garten and Wullschleger, 1999) and generally requires even longer integration periods to detect significant soil C changes (Smith, 2004), there is a real need for similar studies to evaluate the potential of croplands to store or release carbon under different soil conditions, crop species and management regimes.

Additionally, efforts should be made to systematically measure other GHG fluxes at the plot scale and to update emission factors for a range of field operations to reduce uncertainties in the total GHGB of croplands (see Smith et al., 2010 and Osborne et al., 2010). Using a relatively simple but exhaustive approach to evaluate GHG emissions caused by field operations, we were able to estimate the GHG budget for 41 site-years covering most of the common crops grown in Europe and the main management regimes. The mean total GHG budget was estimated to be 203 ± 253 g C-eq m^{-2} $year^{-1}$. Taking into account all of these terms is essential when assessing GHG budgets for crops, especially energy crops being compared to other energy sources.

Finally, crop efficiency, or the ratio between C exported (yield) and the total GHGB, was compared for several crop species and management regimes. Data for most crop species and management regimes are currently too scarce to use as emission factors to assess the impact of crop production on climate; however, in the future, this approach could have much wider applicability.

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References

ADEME, 2007. Bilan Carbone, Calcul des facteurs d'émissions et sources bibliographiques utilisées Version 5.0, Agence de l'Environnement et de la Maîtrise de l'Energie.
Allard, V., et al., 2007. The role of grazing management for the net biome productivity and greenhouse gas budget (CO_2, N_2O and CH_4) of semi-natural grassland. Agriculture, Ecosystems & Environment 121 (1–2), 47–58.
Ammann, C., Meixner, F.X., Busch, J., Losch, R., 1996. CO_2 and H_2O gas exchange of a triticale field: II. Micrometeorological flux studies and comparison with upscaling from porometry. Physics and Chemistry of the Earth 21 (3), 151–155.
Anthoni, P.M., Freibauer, A., Kolle, O., Schulze, E.-D., 2004. Winter wheat carbon exchange in Thuringia, Germany. Agricultural and Forest Meteorology 121 (1–2), 55–67.
Aubinet, M., 2008. Eddy covariance CO_2 flux measurements in nocturnal conditions: an analysis of the problem. Ecological Applications 18 (6), 1368–1378.
Aubinet, M., et al., 2000. Estimates of the annual net carbon and water exchange of forests: the EUROFLUX methodology. Advances in Ecological Research 30, 114–175.
Aubinet, M., et al., 2009. Carbon sequestration by a crop over a 4-year sugar beet/winter wheat/seed potato/winter wheat rotation cycle. Agricultural and Forest Meteorology 149 (3–4), 407–418.
Baker, J.M., Griffis, T.J., 2005. Examining strategies to improve the carbon balance of corn/soybean agriculture using eddy covariance and mass balance techniques. Agricultural and Forest Meteorology 128 (3–4), 163–177.
Baldocchi, D.D., 1994. A comparative study of mass and energy exchange rates over a closed C3 (wheat) and an open C4 (corn) crop: II. CO_2 exchange and water use efficiency. Agricultural and Forest Meteorology 67, 291–321.
Batty, J., Keller, J., 1980. Energy requirements for irrigation. In: D. Pimentel (Editor), Handbook of energy utilization in agriculture, Florida, pp. 35–44.
Bernacchi, C.J., Hollinger, S.E., Meyers, T., 2005. The conversion of the corn/soybean ecosystem to no-till agriculture may result in a carbon sink. Global Change Biology 11 (11), 1867–1872.
Béziat, P., Ceschia, E., Dedieu, G., 2009. Carbon balance of a three crop succession over two cropland sites in South West France. Agricultural and Forest Meteorology 149 (10), 1628–1645.
Bondeau, A., et al., 2007. Modelling the role of agriculture for the 20th century global terrestrial carbon balance. Global Change Biology 13 (3), 679–706.
Buchmann, N., Schulze, E.-D., 1999. Net CO_2 and H_2O fluxes of terrestrial ecosystems. Global Biogeochemical Cycles 13, 751–760.
Byrne, K.A., Kiely, G., Leahy, P., 2007. Carbon sequestration determined using farm scale carbon balance and eddy covariance. Agriculture, Ecosystems & Environment 121 (4), 357–364.
CarboEurope-IP, 2009. CarboEurope-IP Ecosystem component Database. http://gaia.agraria.unitus.it/database/carboeuropeip/.
Chadwick, D.R., Pain, B.F., Brookman, S.K.E., 2000. Nitrous oxide and methane emissions following application of animal manures to grassland. Journal of Environmental Quality 29 (1), 277–287.
Chapin, F., et al., 2006. Reconciling carbon-cycle concepts terminology, and methods. Ecosystems 9 (7), 1041–1050.
Desjardins, R.L., Sivakumar, M.V.K., de Kimpe, C., 2007. The contribution of agriculture to the state of climate: Workshop summary and recommendations. Agricultural and Forest Meteorology 142 (2–4), 314–324.
Dietiker, D., Buchmann, N., Eugster, W., 2010. Testing the ability of the DNDC model to predict CO2 and water vapour fluxes on cropland and simulation of spring wheat yield under realistic future climate conditions. Agriculture, Ecosystems and Environment, 139, 396–401.
Dobermann, A., Baker, J.M., Walters, D.T., 2006. Comment on "Carbon budget of mature no-till ecosystem in North Central Region of the United States". Agricultural and Forest Meteorology 136 (1–2), 83–84.
Dvoskin, D., Nicol, K., Heady, E., 1976. Irrigation energy requirements in the 17 western states. In: W. Lockeretz (Editor), Agriculture and energy. Academic, New York, pp. 103–112.
Eugster, W., et al., 2010. Management Effects on European Cropland Respiration. Agriculture, Ecosystems and Environment, 139, 346–362.
Eugster, W., Sven Erik, J., Brian, F., 2008. Wind Effects, Encyclopedia of Ecology. Academic Press, Oxford, pp. 3794–3803.
EUROSTAT, 2002. http://epp.eurostat.ec.europa.eu/portal/page/portal/agri_environmental_indicators/indicators_overview.
EUROSTAT, 2008. http://epp.eurostat.ec.europa.eu/portal/page/portal/region_cities/regional_statistics/data/database.
FAOSTAT, 2009. Crop statistics. http://faostat.fao.org/site/567/default.aspx.
Finnigan, J., 2008. An introduction to flux measurements in difficult conditions. Ecological Applications 18 (6), 1340–1350.
Gaillard, G., Hausheer, J., Crettaz, P., 1997. Inventaire Environnemental des Intrants Agricoles en Production Végétale. Base de Données pour l'Etablissement des Bilans Energétiques et Ecologiques en Agriculture., Station Fédérale de Recherche en Economie et Technologies Agricoles, Tänikon, Switzerland, 49 pp.
Garten Jr., C.T., Wullschleger, S.D., 1999. Soil carbon inventories under a bioenergy crop (Switchgrass): measurement limitations. Journal of Environmental Quality 28 (4), 1359–1365.
Gervois, S.b., et al., 2008. Carbon and water balance of European croplands throughout the 20th century. Global Biogeochemical Cycles, 22.
Gifford, R., 1984. Energy in different agricultural systems: renewable and nonrenewable sources. In: S.G (Ed.), Energy and agriculture. Springer-Verlag, Berlin, pp. 84–112.
Grant, R.F., et al., 2007. Net biome productivity of irrigated and rain fed maize soybean rotations: modelling vs. measurements. Agronomy Journal 99 (6), 1404–1423.
Hillier, J., et al., 2009. The carbon footprints of food crop production. International Journal of Agricultural Sustainability 7, 107–118.
Hollinger, D., Richardson, A.D., 2005. Uncertainty in eddy covariance measurements and its application to physiological models. Tree Physiology 25, 873–885.
Hollinger, S.E., Bernacchi, C.J., Meyers, T.P., 2005. Carbon budget of mature no-till ecosystem in North Central Region of the United States. Agricultural and Forest Meteorology 130 (1–2), 59–69.
Hollinger, S.E., Bernacchi, C.J., Meyers, T.P., 2006. Corrigendum to "Carbon budget of mature no-till ecosystem in North Central Region of the United States [Agric. For. Meteorol. 130 (2005) 59–69]". Agricultural and Forest Meteorology, 136(1–2): 88–89.
Houghton, J.T., et al., 2001. Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge, UK, 881 pp.
IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4, Agriculture, Forestry and Other Land Use.
Janssens, I.A., et al., 2003. Europe's Terrestrial Biosphere Absorbs 7 to 12% of European Anthropogenic CO_2 Emissions. Science 300 (5625), 1538–1542.
Jun, L., et al., 2006. Carbon dioxide exchange and the mechanism of environmental control in a farmland ecosystem in North China Plain. Science in China Series D 49 (Suppl. II), 226–240.
Koga, N., Tsuruta, H., Tsuji, H., Nakano, H., 2003. Fuel consumption-derived CO_2 emissions under conventional and reduced tillage cropping systems in northern Japan. Agriculture, Ecosystems and Environment 99, 213–219.
Kramer, K.J., Moll, H.C., 1995. Energie-intensiteiten van voedingsmiddelen. IVEM-onderzoeksrapport no. 77 (in Dutch). Centre for Energy and Environmental Studies, university of Groningen, the Netherlands.
Kuikman, P.J., Hoek, K.W.v.d., Smit, A., Zwart, K., 2006. Update of emission factors for direct emissions of nitrous oxide from agricultural soils on the basis of measurements in the Netherlands, Alterra, 40 pp.
Kutsch, W.L., et al., 2010. The net biome production of full crop rotations in Europe. Agriculture, Ecosystems and Environment, 139, 336–345.
Lal, R., 1997. Residue management, conservation tillage and soil restauration for mitigating greenhouse effect by CO_2 enrichment. Soil and Tillage Research 43, 81–107.
Lal, R., 2004. Carbon emission from farm operations. Environment International 30 (7), 981–990.
Lasslop, G., Reichstein, M., Kattge, J., Papale, D., 2008. Influences of observation errors in eddy flux data on inverse model parameter estimation. Biogeosciences 5 (5), 1311–1324.
Marland, G., West, T., Schlamadinger, B., Canella, L., 2003. Managing soil organic carbon in agriculture: the net effect on greenhouse gas emissions. Tellus 55b, 613–621.
MeteoSwiss, 2008. Witterungsbericht Jahr 2007 (the weather conditions during year 2007). Swiss Federal Office for Meteorology and Climatology, Zurich, 7 pp.
Moors, E.J., et al., 2010. Variability in carbon exchange of European croplands. Agriculture, Ecosystems and Environment, 139, 325–335.
Moureaux, C., Debacq, A., Bodson, B., Heinesch, B., Aubinet, M., 2006. Annual net ecosystem carbon exchange by a sugar beet crop. Agricultural and Forest Meteorology 139 (1–2), 25–39.
Moureaux, C., et al., 2008. Carbon balance assessment of a Belgian winter wheat crop (Triticum aestivum L.). Global Change Biology 14 (6), 1353–1366.
Osborne, B., Saunders, M., Walmsley, D., Jones, M., Smith, P., 2010. Key Questions and Uncertainties Associated with the Assessment of the Cropland Greenhouse Gas Balance. Agriculture, Ecosystems Environment, 139, 293–301.
Pathak, H., Li, C., Wassmann, R., 2005. Greenhouse gas emissions from Indian rice fields: calibration and upscaling using the DNDC model. Biogeosciences 2 (2), 113–123.
Pattey, E., Strachan, I.B., Desjardins, R.L., Massheder, J., 2002. Measuring nighttime CO_2 flux over terrestrial ecosystems using eddy covariance and nocturnal boundary layer methods. Agricultural and Forest Meteorology 113, 145–158.
Pimentel, D., 1992. Energy inputs in production agriculture. Energy for World Agriculture 6, 13–29.
Prescher, A.-K., Grünwald, T., Bernhofer, C., 2010. Land use regulates carbon budgets in eastern Germany: From NEE to NBP. Agricultural and Forest Meteorology.
Raddatz, R.L., 2007. Evidence for the influence of agriculture on weather and climate through the transformation and management of vegetation: Illustrated by examples from the Canadian Prairies. Agricultural and Forest Meteorology 142 (2–4), 186–202.
Richardson, A.D., Hollinger, D.Y., 2007. A method to estimate the additional uncertainty in gap-filled NEE resulting from long gaps in the CO_2 flux record. Agricultural and Forest Meteorology 147 (3–4), 199–208.

Richardson, A.D., et al., 2006. A multi-site analysis of random error in tower-based measurements of carbon and energy fluxes. Agricultural and Forest Meteorology 136 (1–2), 1–18.
Robertson, G., Grace, P., 2004. Greenhouse gas fluxes in tropical and temperate agriculture: the need for a full-cost accounting of global warming potentials. Environment Development and Sustainability 6, 51–63.
Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922–1925.
Saito, M., Miyata, A., Nagai, H., Yamada, T., 2005. Seasonal variation of carbon dioxide exchange in rice paddy field in Japan. Agricultural and Forest Meteorology 135 (1–4), 93–109.
Schulze, E.D., Heimann, M., 1998. Carbon and water exchange of terrestrial systems. In: G.J. N and J. Melillo (Editors), Asian change in the context of global change. Cambridge University Press, Cambridge, pp. 145–161.
Schulze, E.D., et al., 2009. Importance of methane and nitrous oxide for Europe's terrestrial greenhouse-gas balance. Nature Geosciences 2, 842–850.
Smith, J., et al., 2005. Projected changes in mineral soil carbon of European croplands and grasslands, 1990-2080. Global Change Biology 11 (12), 2141–2152.
Smith, P., 2004. How long before a change in soil organic carbon can be detected? Global Change Biology 10, 1878–1883.
Smith, P., et al., 2010. Measurements necessary for assessing the net ecosystem carbon budget of croplands. Agriculture, Ecosystems and Environment, 139, 302–315.
Smith, P., Powlson, D.S., Smith, J.U., Falloon, P., Coleman, K., 2000. Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. Global Change Biology 6 (5), 525–539.
Soussana, J.F., et al., 2007. Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites. Agriculture, Ecosystems & Environment 121 (1–2), 121–134.
St Clair, S., Hillier, J., Smith, P., 2008. Estimating the pre-harvest greenhouse gas costs of energy crop production. Biomass and Bioenergy 32 (5), 442–452.
Stout, B., 1990. Handbook of energy for world agriculture, New York, 504 pp.
Suyker, A., Verma, S., Burba, G., Arkebauer, T.J., 2005. Gross primary production and ecosystem respiration of irrigated maize and irrigated soybean during a growing season. Agricultural and Forest Meteorology 131, 180–190.
Suyker, A.E., et al., 2004. Growing season carbon dioxide exchange in irrigated and rainfed maize. Agricultural and Forest Meteorology 124 (1–2), 1–13.
Swiss Federal Statistical Office, 2008. http://www.bfs.admin.ch/bfs/portal/en/index/infothek/lexikon.html.
VanGorsel, E., Leuning, R., Cleugh, H.A., Keith, H., Suni, T., 2007. Nocturnal carbon efflux: reconciliation of eddy covariance and chamber measurements using an alternative to the u*-threshold filtering technique. Tellus B 59 (3), 397–403.
Verma, S.B., et al., 2005. Annual carbon dioxide exchange in irrigated and rainfed maize-based agroecosystems. Agricultural and Forest Meteorology 131 (1–2), 77–96.
Zou, J., Liu, S., Qin, Y., Pan, G., Zhu, D., 2009. Sewage irrigation increased methane and nitrous oxide emissions from rice paddies in southeast China. Agriculture, Ecosystems & Environment 129 (4), 516–522.