



Measurements necessary for assessing the net ecosystem carbon budget of croplands

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ABSTRACT

There are a number of methods that can be used to help assess carbon budgets at the site to continental scales. Eddy covariance (EC) networks have been developed over the last decade and have been used to make many advances in our understanding. However, eddy covariance measurements of CO₂ and water vapour exchanges quantify the fluxes only on short time scales, but do not assess the impacts of long-term processes that contribute to biogeochemical cycling in croplands, such as harvest or residue removal and other management practices, so many other supplementary measurements are required to attribute different components of the carbon flux. Such methods include isotope studies, chamber flux measurements of C and other greenhouse gases, inventories of above- and below-ground biomass as well as management in- and outputs, book-keeping modelling, process modelling, experimental manipulation and earth observation (e.g. remote sensing). In this review, we summarise the component fluxes that make up the total cropland carbon budget, describe the key fluxes and methods used to estimate them, and examine how they need to be integrated to obtain the net ecosystem carbon budget of European croplands. We describe the uncertainties and difficulties inherent at each stage and how these can be minimised.

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1. Introduction

During the past decade, the eddy covariance technique has become the most important method for measuring trace gas exchange between terrestrial ecosystems and the atmosphere (Baldocchi, 2003). The reasons for this success are the small impact of the measurements on the observed systems and the fact that the measurements integrate over a larger part of the landscape and over all ecosystem processes. Eddy covariance can be used to measure the net ecosystem exchange (NEE) of CO₂, or the net ecosystem production (NEP), that is to say, the net CO₂ flux or balance of all CO₂ entering an ecosystem and all of the CO₂ leaving the ecosystem during a time period of interest, typically a year (Chapin et al., 2006).

Since the eddy covariance method measures a net flux, different processes contributing to the measured net ecosystem exchange cannot be distinguished using eddy covariance techniques. The non-CO₂ exchanges of carbon are not typically included in NEE and NEP, but can also be quantified with the eddy covariance method, if adequate fast gas analysers for the specific compounds are used (e.g. CH₄, VOCs), though VOC losses are thought to be low from croplands (compared to forest systems). Moreover, all carbon gains and losses that are not associated with turbulent fluxes, such as for example advective losses, harvest removals or manure applications are often also not included in ecosystem-scale carbon budgets. Estimates of all these components are however needed for the assessment of carbon budget. In this paper, we describe the additional measurements that are needed to estimate the net ecosystem carbon budget (NECB), and focus especially on the measurements needed for assessing the NECB of croplands. We briefly describe and evaluate the components of the carbon budget in croplands before

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describing the various methods available for measuring these components. In the final section, we examine some of the tools used for scaling the carbon budget of croplands from ecosystem to continental levels.

2. Components of the carbon budget of croplands

In a series of papers assessing the carbon budget of Europe, Ciais et al. (2010) and Luyssaert et al. (2010) compiled estimates of the NECB of European grasslands, croplands and forests. Here we follow the same notation, and summarise the components of the NECB of croplands.

The gross primary production (GPP) of an ecosystem represents the gross uptake of CO₂ that is used in photosynthesis (Ciais et al., 2010). About half of the photo-assimilates from GPP are consumed by autotrophic respiration (R_a), which is required for the synthesis of new plant tissues and the maintenance of living tissues (Luyssaert et al., 2007). The quantity of photosynthates not used for respiration and therefore available for other processes, is defined as net primary production (NPP) and relates to GPP and R_a as:

$$\text{GPP} = \text{NPP} + R_a \quad (1)$$

Accurate direct measurements of total NPP are impossible. Part of the organic material produced during NPP is lost via emission of volatile organic compounds (VOC; more important in forestry than in croplands), exudation from roots, or carbon transfer to root symbionts. Although the bulk of NPP is allocated to the production of above- and below-ground biomass, the less quantified fractions associated with exudation and volatile losses are difficult to assess. Among the reasons for this, is that not all of the biomass produced remains on site at the point at which the measurements are made, for example due to harvest, or due to losses to pests and/or herbivory. These losses are extremely difficult to quantify. If estimating from measured biomass, corrections need to be made for the biomass removed before measurement. Further, fine root turnover occurs throughout the year and can be very difficult to estimate. Because of this, NPP estimates are highly uncertain (Lauenroth et al., 2006). In croplands, NPP is estimated as follows:

$$\text{NPP} = \text{NPP}_{\text{foliage}} + \text{NPP}_{\text{seeds/fruits}} + \text{NPP}_{\text{roots}} + \text{NPP}_{\text{residual}} \quad (2)$$

In estimates of NPP, NPP_{residual} (the sum of all unknown fractions of total NPP) is never included. In addition to the high uncertainty in the NPP estimates, this NPP_{residual} term also implies that there is a systematic underestimation of all reported estimates of NPP (Ciais et al., 2010), though the size of the under-estimation, is by its nature, unknown.

Each year, part of the biomass produced is transferred to litter and soil carbon pools (each of which has different residence times; Zimmermann et al., 2007). These carbon pools are subject to decomposition by microbial activity, a process defined as heterotrophic respiration (R_h). The decomposition processes that contribute to R_h include decomposition of biomass from the current year, but also contain decomposition of organic matter that accumulated in the ecosystem over decades, centuries or millennia. The difference between NPP and R_h is termed the net ecosystem productivity (Ciais et al., 2010):

$$\text{NEP} = \text{NPP} - R_h \quad (3)$$

The sum of R_h and R_a represents the total ecosystem respiration (R_e). By definition, GPP = NPP + R_a, and the sum of the belowground fraction of R_a and R_h is termed soil respiration. Thus, in practice NEP can be determined by the difference between GPP and R_e:

$$\text{NEP} = \text{GPP} - R_e \quad (4)$$

However, NEP may differ from the net rate of organic carbon accumulation (or loss) in ecosystems as determined by other methods. The carbon fluxes observed at eddy covariance monitoring sites differ from the long-term carbon budget mainly because non-CO₂ carbon losses and non-respiratory CO₂ losses, which occur at a range of time scales, are typically ignored (Ciais et al., 2010). On short timescales of less than a year, for example, carbon can be lost from the ecosystem (e.g. as VOCs; small in croplands), whereas over longer timescales, in excess of one year part of the annually accumulated NEP may leave the ecosystem to the ground water as dissolved or particulate organic, or inorganic, carbon (DOC, DIC and POC; collectively termed D here) or to the atmosphere as microbially-produced methane (CH₄) and via exports of carbon in pollen and seeds carried away by the wind (e.g. Eugster, 2008), though the losses from the latter source are likely to be very small, and within the error of NEE measurement. In addition, all or part of the carbon that has been built up over the years by the accumulation of the annual NEP can leave the ecosystem and eventually return to the atmosphere or be deposited outside the ecosystem, by fires (F), harvests (H) and/or erosion (E) (Ciais et al., 2010).

In croplands, organic carbon can enter the ecosystem through fertilisation with manure, re-deposition of eroded sediments from elsewhere, and deposition of dissolved carbon in water (I). The eroded carbon that does not leave the cropland landscape is merely redistributed and does not appear as a loss term. Therefore, non-CO₂ and non-respiratory carbon losses and gains should be accounted for in Eq. (4) to obtain the carbon budget. The net ecosystem carbon budget is the term applied to the total rate of organic carbon accumulation (or loss) from ecosystems (Chapin et al., 2006) and relates to NEP as follows:

$$\text{NECB} = \text{NEP} - \text{D} - \text{F} - \text{H} - \text{VOC} - \text{CH}_4 - \text{E} + \text{I} \quad (5)$$

When integrated over time and space the NECB equals the net biome production (NBP; Schulze and Heimann, 1998; Buchmann and Schulze, 1999; Chapin et al., 2006). Assuming that most of the harvest is respired as CO₂ over short time periods, it is the NBP that is reflected in the long-term atmospheric concentration of CO₂ and other atmospheric carbon-compounds.

In croplands, lateral fluxes, particularly the terms harvest (H) and manure (part of I) are important parts of the NECB. Fig. 1 shows how the various components of the carbon budget of croplands are related and symbolizes the fact that some important fluxes are located 'off-site'. If fodder for livestock is produced on a cropland and the harvest is fed to the animals at the farm the livestock and related fluxes should be added to the cropland NECB which is virtually extended to the farm gate (Kutsch et al., 2001, 2008a,b).

Although Eqs. (1)–(5) above are based on solid theoretical principles, many methodological issues remain. For example, gross primary productivity estimates derived from eddy covariance measurements do not account for light induced inhibition of foliar or stem respiration which can reduce above-ground autotrophic respiration by over 50% (Atkin et al., 2000; Tcherkez et al., 2005), and are sensitive to biased NEP estimates due to underestimation of night time fluxes (Gu et al., 2005; Moureaux et al., 2006) that can occur, notably from advection (Finnigan, 2008; Aubinet, 2008; Kutsch et al., 2008a,b). NEP measurements by eddy covariance techniques can suffer from incoming or outgoing advective fluxes, which could result in over- or under-estimation of NEP (Feigenwinter et al., 2008). Conversely, NPP may be underestimated as only its major biomass components are measured, whereas its minor components (including senescence) are often ignored (e.g. Eq. (2); Clark et al., 2001; Scurlock et al., 2002). Also, depending on the methodology applied to estimate R_h, respiration from mycorrhizae can either be considered as auto- or as heterotrophic respiration. Despite these methodological shortcomings, some studies have shown that independent measurements

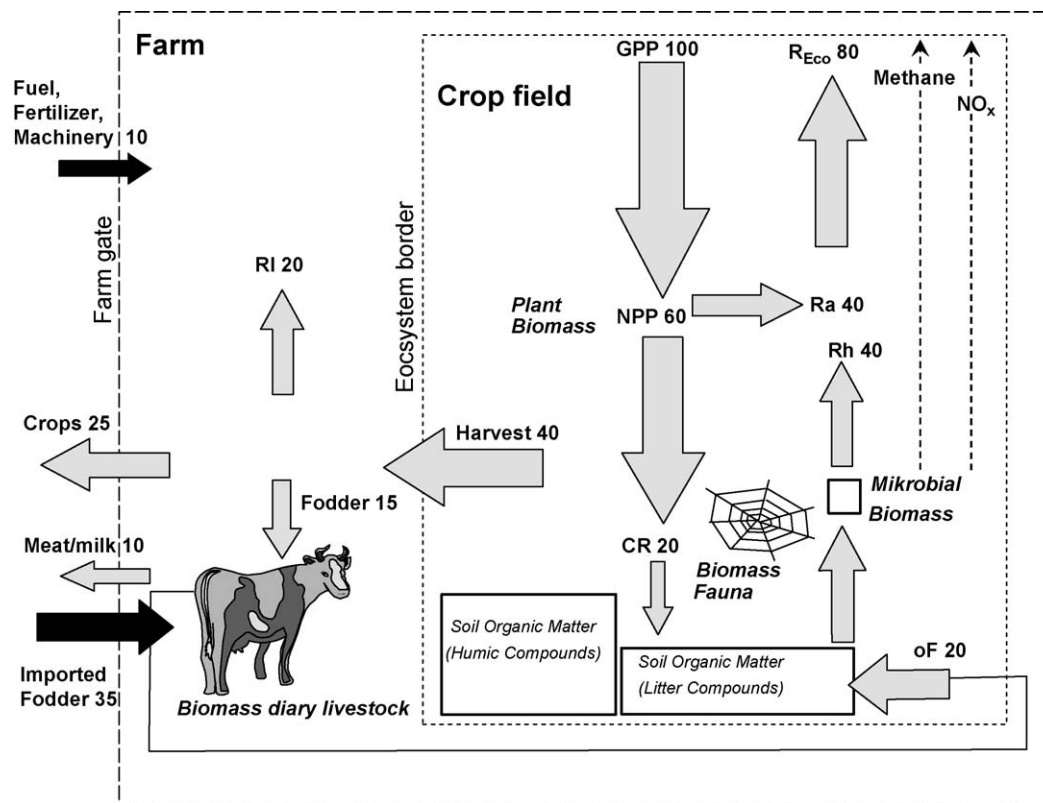


Fig. 1. Schematic representation of the various components of the carbon balance of croplands are related, showing that some of the important fluxes are located 'off-site'. In this example about 40% of gross primary production (GPP = 100%) is respired by the plants, while 60% are used for biomass production (net primary production, NPP), of which two thirds are harvested and one third remains on the site as residues. In this example, parts of the harvest are sold as cash crops (25%) while others are used to feed the livestock on the farm (15%). In 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 back to the field and is an important factor of the soil carbon balance. The soil of this exemplary field is balanced only due to the application of organic fertilizer. Another possibility to balance carbon fluxes in agriculture are farm gate balances. In this approach only the imports and exports through the farm gate are balanced. The farm in this example has an input/output ratio of 0.78, which means that it is a net source of GHG, that might also be increased by methane and nitrous oxide fluxes that are not represented in this calculations (after Kutsch et al., 2008a,b, modified). Abbreviations: GPP: gross primary production, R_{Eco} : ecosystem respiration, NPP: net ecosystem production, CR: crop residues, R_a : autotrophic respiration (plants and mycorrhizae), R_h : heterotrophic respiration (microbial biomass and fauna), R_l : respiration of the livestock at the farm, oF: organic fertilizer.

of the different components are often consistent, indicating that these issues have either a small effect or compensate each other (Luyssaert et al., 2007; Moureaux et al., 2008; Aubinet et al., 2009). Fig. 2 shows how the various components of the carbon budget of croplands are related.

3. Measurement methods for different components of the C budget of croplands

3.1. Gross primary production

The gross primary production of an ecosystem represents the gross uptake of CO_2 through photosynthesis (Ciais et al., 2010). GPP is not easily measured at large scales, but can be estimated from the net ecosystem exchange measured by eddy covariance (see below). In this procedure, the night-time respiration is extrapolated to daytime (R_d) and GPP (Fig. 3) is calculated by:

$$GPP = NEE - R_d \quad (6)$$

Also gas exchange measurements have been used to estimate GPP (e.g. Kutsch et al., 2001). In this approach, usually models of leaf photosynthesis are run with continuously measured microclimate data from a site. Gas exchange measurements have to be performed regularly throughout the growing season to derive seasonal courses of the model parameters.

Further methods to calculate total C uptake are based on biomass assessment combined with pulse-labelling with C isotopes

in the air in a chamber surrounding the plant/soil, which can either be radioactive (^{14}C), or stable (^{13}C), such as that used in stable isotope profiling (SIP; e.g. Rangel-Castro et al., 2004), though it is very difficult to assess the differences in biomass between isotope measurements and to measure the losses via exudation and respiration of all components. A range of remote sensing products offer estimates of GPP (e.g. MODIS, 2009), but the estimates are derived from spectral data, often using algorithms (models) and/or fixed GPP/NPP ratios (MODIS, 2009). Given this very indirect method, the associated uncertainties may be large.

More commonly, micrometeorological techniques are used for estimating component fluxes. The partitioning of NEP into its gross component fluxes, GPP and TER, can be derived from eddy covariance data by a myriad of methods which fall into three broad categories: Non-linear regressions, look-up tables and statistical analyses (Desai et al., 2008).

Non-linear regression techniques depend on relating NEE with environmental variables. Daily values of GPP can be calculated indirectly as the difference between NEE and R_e (see Eq. (4)). Ecosystem respiration, in turn, may be solely derived from the correlation between night-time NEE with soil temperature and soil moisture over time (where GPP is assumed to be zero). This relationship is most commonly described as either an exponential or logistic correlation with temperature and other influencing factors (Lloyd and Taylor, 1994; Falge et al., 2002; Barr et al., 2004; Reichstein et al., 2005). These drivers of respiration (such as leaf area index and moisture) may be directly incorporated in the expression

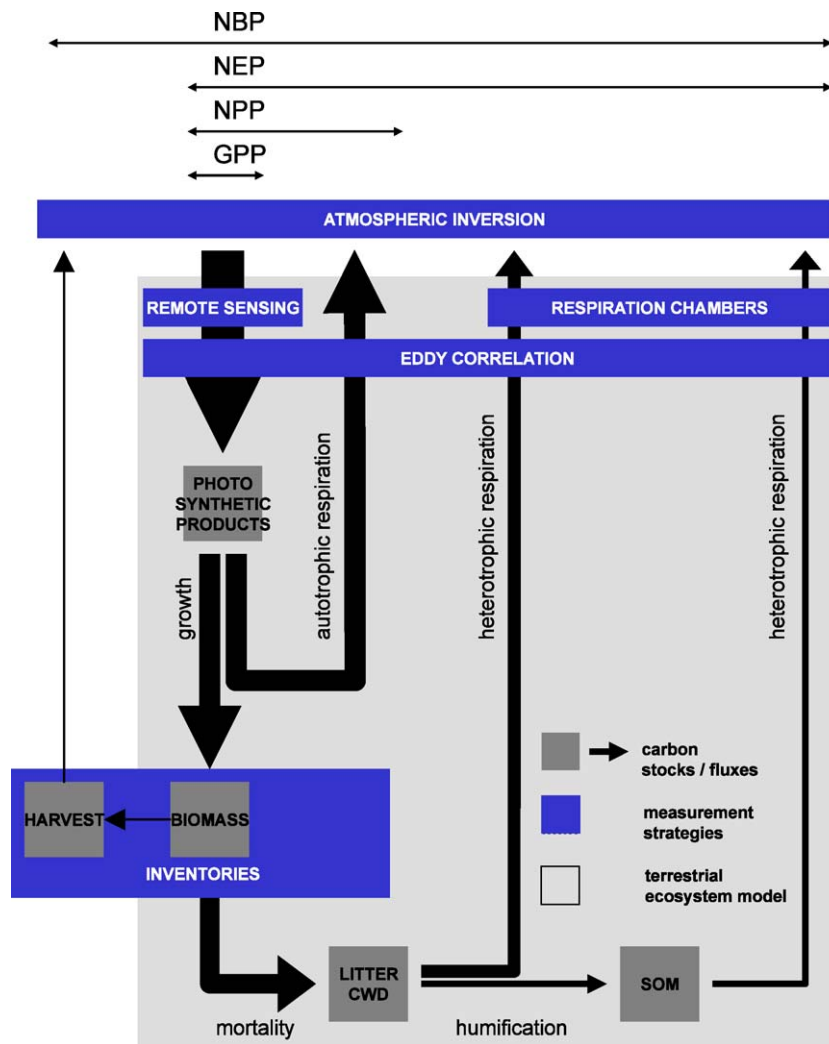


Fig. 2. Schematic representation of the components of the carbon balance of croplands.

(Reichstein et al., 2002) or indirectly via variations in the reference temperature over time (Falge et al., 2002). In addition, R_e can be further constrained by generating daytime values from the y-intercept from light-response curve of GEP (Falge et al., 2002; Gilmanov et al., 2003). Whilst this only allows for one value of daytime R_e to be gen-

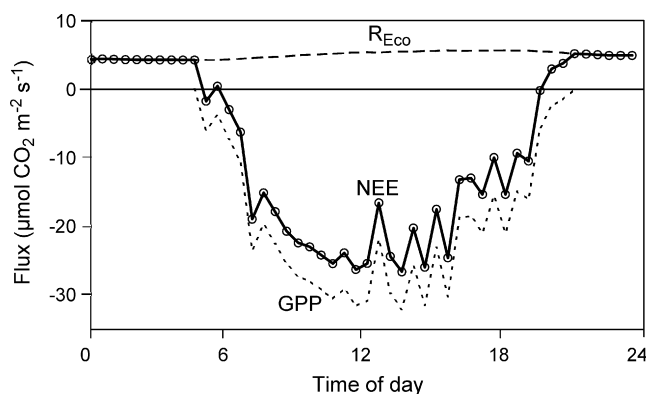


Fig. 3. Diurnal course of CO_2 fluxes as derived from eddy covariance to show the procedure to derive GPP and R_{Eco} . NEE is measured continuously by the eddy covariance system. R_{Eco} is extrapolated from night time fluxes to daytime by using temperature and soil moisture response functions. After that GPP is calculated by subtracting R_{Eco} from NEE.

erated, it accounts for any light-inhibition of foliar respiration that may occur. Further advances have been made with the incorporation of an algorithm to account for short-term (hourly) temperature sensitivity of ecosystem respiration (Reichstein et al., 2005). This further reduced the bias in estimates of gross component fluxes of NEE, especially in croplands where ecosystem dynamics alter over short timescales. Béziat et al. (2009) show that it is very important to apply these method during defined crop functioning periods (CFP), which are the periods between dates of sowing, maximum crop development, harvest and tillage, to take account of fast and discontinuous changes in canopy structure.

Statistical techniques have also been developed to partition net carbon fluxes. These can be process-based models driven by both, meteorological data and ecosystem-specific data such as leaf area index (LAI), light use efficiency (LUE), canopy structure, soil type, etc. Alternatively, artificial neural networks utilise empirical non-linear regressions incorporated into neural learning patterns (Papale and Valentini, 2003). These networks depend on using good quality datasets to 'train' the network and to resolve the relationships. The advantage of this method is that there are no *a priori* assumptions about functional relationships between fluxes and controlling factors.

Comparisons between different partitioning methods have shown good agreement between techniques and no particular technique has been shown to be more robust across a variety of

timescales (Desai et al., 2008). Indeed, the effect of technique on calculated fluxes has been estimated to be less than $25 \text{ gC m}^{-2} \text{ yr}^{-1}$ (Falge et al., 2002).

Combining natural abundance isotopic and micrometeorological techniques to resolve component fluxes was first proposed by Yakir and Wang (1996) who utilised measurements of ^{18}O in CO_2 to partition the NEE of crop stands. More recent studies have attempted to partition NEE based on isotopic flux densities or 'isofluxes' ($F\delta$), which were defined as the product of total carbon flux (NEE) and the $\delta^{13}\text{C}$ of the atmosphere for that flux ($\delta^{13}\text{C}_a$). By measuring the isotopic signatures associated with F_R and F_A , gross fluxes could be obtained by isotopic mass balance (e.g. Knohl and Buchmann, 2005). One difficulty with this technique is that while eddy covariance can measure CO_2 fluxes at 10 Hz, it is difficult to directly measure the $\delta^{13}\text{C}$ signal of that flux. However, advances in tuneable diode laser absorption spectroscopy (TDLAS) allow measurement of ^{13}C isofluxes at 10 Hz (Bowling et al., 2004). Also the utilisation of isotope techniques to partition fluxes is highly dependent on accurate estimates of the isotope ratio of both GPP and R_e . However, there have been problems associated with obtaining accurate estimates of the $\delta^{13}\text{C}$ associated with photosynthesis ($\delta^{13}\text{C}_p$; Knohl and Buchmann, 2005). These issues will need to be addressed before the information contained in the isotopic ratios of fluxes can be utilised to their full potential.

3.2. Net primary production

As mentioned above, direct measurements of total NPP are practically impossible. Like GPP, a range of remote sensing products offer estimates of NPP (e.g. MODIS, 2009), but the same limitations apply (see Section 3.1). At large spatial scales, "bottom-up" estimates of NPP are often made from measured crop yield using allometric equations such as the harvest index (ratio of harvested product such as grain, to non-harvested product such as straw), measured or estimated ratios of above- and below-ground biomass (Goudriaan et al., 2001) and harvest losses, and proportional losses due to, for example, herbivory (Haberl et al., 2007). These allometric equations differ according to crop and by region (Haberl et al., 2007). At site level, the quantity of biomass in each above ground crop component, and the C content of each component can be readily measured. Root biomass and C content is also relatively easily measured in croplands.

3.2.1. $\text{NPP}_{\text{seeds/fruits}}$

The harvestable yield of crops is commonly recorded, and most farmers know the yield per hectare of their fields in a given year. Many harvesting machines measure yield, but pre-harvest and harvesting losses are not recorded. Many countries record yield statistics at varying degree of spatial resolution, and the FAO holds an electronic database of production and yield at country level (FAOSTAT, 2009). Many EU countries hold these statistics at very high spatial resolution (e.g. Parish Records of England and Wales; National Digital Archive of Datasets, 2009; Agreste in France) and at EU level, yield statistics are available at sub-country level, for example at administrative NUTS2 level (Rounsevell et al., 2006).

From the harvested weight, dry matter yields are estimated from crop specific-dry matter contents, and these are converted to carbon contents either by crop specific carbon contents determined on a representative subsample of the harvest, or based on the generic assumptions common to most plant material that 0.45 (or for simplicity often rounded to 0.5) of the dry matter is made up of carbon (e.g. Haberl et al., 2007; Goudriaan et al., 2001; Smith et al., 1997; Zheng et al., 2003). The harvested yield ($\text{NPP}_{\text{seeds/fruit}}$) is then converted to total NPP using the allometric relationships as described above (Haberl et al., 2007), or at site level, from the measured biomass and C content.

3.2.2. $\text{NPP}_{\text{foliage}}$

In the absence of measured biomass and C content at site level, the same simplification is also made for foliage (including the stem) NPP (the total above-ground, non-harvested plant material) which is often calculated using a fixed fraction of the harvest yield, known as the harvest index. The harvest index is the ratio of harvested commercial yield (e.g. grain for cereals) compared to the above-ground, non-harvested material (e.g. straw and stubble of a cereal crop; Goudriaan et al., 2001). The harvest index differs for different crop plant species, different cultivars and varies also across regions, with more productive countries having a higher harvest index for the same crops. Further, soil and climatic conditions alter allocation of carbon in the crop. Generic species-specific, region- or country-specific factors are often used (e.g. Haberl et al., 2007). This ratio, sometimes expressed as the proportion of total above-ground dry matter to harvested yield (Haberl et al., 2007), can be used in the absence of measured site data, to calculate total above-ground NPP from the measured harvested yield.

3.2.3. $\text{NPP}_{\text{roots}}$

$\text{NPP}_{\text{roots}}$ can be measured in plots at site level, but as with $\text{NPP}_{\text{foliage}}$, the below-ground NPP can be calculated using a fixed fraction of above-ground to below-ground biomass (Goudriaan et al., 2001), derived in turn from measurements using rhizotrons, mini-rhizotrons or sequential coring. $\text{NPP}_{\text{roots}}$, when not measured directly, is a relatively uncertain figure, and varies greatly in previous estimates from fixed factors of 0.5, to species- and region or country-specific factors (Haberl et al., 2007) closer to 0.25.

3.2.4. $\text{NPP}_{\text{residual}}$

In estimates of NPP, $\text{NPP}_{\text{residual}}$ (the sum of all expected but unquantified [unknown] fractions contributing to the total NPP) is never included. Some authors have tried to estimate $\text{NPP}_{\text{residual}}$ losses due to pests and herbivory which would contribute to NPP (Haberl et al., 2007), but by definition, the residual term cannot be fully quantified. The residual term might also include the residues not removed as yield and straw, which when collected from the soil surface, can be substantial (Dietiker et al., 2010).

3.3. Total ecosystem respiration (R_e)

Total ecosystem respiration (R_e) can be derived from night-time eddy covariance fluxes by means of a temperature and soil water response model that is used to calculate daytime respiration flux, or the intercept of the day flux response to light flux (see Section 3.1). Since R_e is the sum of heterotrophic respiration (R_h) and autotrophic respiration (R_a ; see Section 3.6), it can also be calculated by adding these terms, which can be measured as described in Sections 3.4 and 3.5.

The below-ground components of R_e , due to sub-surface heterotrophic (R_h) and autotrophic respiration (R_a) contribute collectively to the measured soil respiration, and methods for measuring soil respiration are described below in Section 3.6.2, and in the section on partitioning fluxes using micrometeorological techniques (Section 3.1).

3.4. Autotrophic respiration (R_a)

Autotrophic respiration (R_a) is due to metabolic processes associated with plant metabolism, and gives the budget between GPP and NPP. The above-ground component of autotrophic respiration can be measured using gas-exchange methods by enclosing the plant (or part of the plant) in a chamber whilst excluding the contribution of soil respiration (Hubbard et al., 1995). High species-specific variability in autotrophic respiration, often driven by nutrient status or ontogenetic stage of the plants (young plants

or tissues respire more than old tissues, photosynthesizing tissues more than storage tissues) or by the proportion of living tissue in the plant, makes generalisation challenging (Reich et al., 2006; Amthor and Baldocchi, 2001). Since respiration rates also scale with nitrogen concentrations, some models use such auxiliary data to constrain autotrophic respiration of plants (Potter et al., 2001). Mechanistic models (e.g. Thornley and Cannell, 2000; McCree and Fernandez, 1989) of autotrophic respiration can also be used to partition R_a from R_e .

3.5. Heterotrophic respiration (R_h)

To understand the underlying processes and dependencies, it is useful to separate cropland heterotrophic respiration (R_h) into two components, (1) the microbial consumption of root exudates, and decomposition of dead roots and soil organic matter (R_{h1}) in the field, and (2) the microbial decomposition of crop products ingested by humans and animals (R_{h2}) off site. At the continental scale, the annual storage of harvest (H) is a negligible fraction of the total flux (Ciais et al., 2010), so R_{h2} can be often regarded as equal to the harvested carbon (H ; see Section 3.12), though the C returned to the cropland as manure has to be accounted for (I_{fert} ; see Section 3.14). Heterotrophic respiration (R_{h1}) is the respiration from free living soil microbes, and gives the budget between NPP and the NEP measured by the eddy covariance technique (see Section 3.6). R_{h1} can be measured using soil chambers that do not contain plant cover or roots (Hanson et al., 2000), but this can be complicated in soils because of significant spatial variability, and removing roots from the soil always introduces substantial disturbance effects. Methods to measure the below-ground component of heterotrophic respiration (R_h), as a component of soil respiration (and how to separate autotrophic and heterotrophic below-ground fluxes) are described below in Section 3.6.2.

3.6. Soil respiration (sum of the belowground fraction of R_a and R_h)

3.6.1. Total soil respiration measurements in the field

Lankreijer et al. (2003) and Smith et al. (2008a) recently reviewed a number of techniques for measuring soil respiration in the field. Here we summarise these techniques. Field measurements of soil CO_2 efflux are usually made by enclosing a known area of soil, cleared of green vegetation, in a sealed chamber (Ryan and Law, 1995). The CO_2 evolved from the soil is measured quantitatively by one of three different methods now known as static absorption, dynamic open and dynamic closed chambers, the latter two methods also being known as the steady-state and non-steady state methods, respectively (Livingston and Hutchinson, 1995). Static absorption is reviewed in Smith et al. (2008a), and since it is no longer commonly used, it is not discussed further here.

3.6.1.1. Dynamic open (or steady-state) chambers. In a dynamic open chamber method, air is passed through the chamber and passes to the gas analyser and is then evacuated (Pumpanen et al., 2010); the efflux of CO_2 from the soil (S_r) covered by the chamber is obtained as a function of the difference in CO_2 concentration between air entering and leaving the chamber (Smith et al., 2008a; Pumpanen et al., 2010):

$$S_r = \Delta c \cdot \frac{f}{A} \quad (6')$$

where c is the difference in CO_2 mass fraction in the incoming and outgoing air streams; f is the gas flow rate through the chamber and A is the surface area covered by the chamber (Nakayama, 1990). The difference in CO_2 concentration is usually measured by an infrared gas analyser (IRGA). Many studies have focused on ensuring

that pressure differences between inside and outside the chamber are eliminated; if air is blown into the chamber, an overpressure within the chamber will be established and the natural efflux of CO_2 from the soil will be suppressed; conversely, drawing air out of the chamber will induce relative negative pressure in the chamber and an increase in the soil efflux rate. Kanemasu et al. (1974) showed that the measured CO_2 efflux was about an order of magnitude larger when air was drawn out of a chamber ($\Delta P = -2.5$ Pa) compared to when air was blown in ($\Delta P = +1$ Pa). Fang and Moncrieff (1998) suggested it was necessary to keep the pressure difference to within ± 0.2 Pa with the dynamic chamber method for reliable measurements to be made, but this is difficult to achieve. A number of designs have been published recently that seek to overcome some of the difficulties in the dynamic method (Smith et al., 2008a).

3.6.1.2. Dynamic closed chamber (non-steady state) methods. If a closed chamber is placed on the soil, the concentration of CO_2 respired from the soil will build up inside the chamber and this enrichment can be used to estimate the efflux from the soil. This method is the basis of many of the successful commercial designs on the market today. The soil efflux can be expressed by:

$$S_r = \frac{(\Delta c / \Delta t) V}{A} \quad (7)$$

where c is the CO_2 concentration increment in the chamber in the time interval t , V is the volume of air within the chamber and A is the soil surface area covered by the chamber. The CO_2 content of a sample taken at discrete intervals can be measured by alkali absorption, by gas chromatography (GC) (Crill, 1991; Castro et al., 1994), by IRGA (Parkinson, 1981) or mass spectrometer (Clymo and Pearce, 1995). From Eq. (7) it can be seen that chamber dimensions must be known accurately, and the chamber design must be adapted to the magnitude of the expected flux density S_r (Livingston and Hutchinson, 1995). In addition, as samples are withdrawn from the chamber, it is important to replace the sampled air by an equal volume of air so as to avoid any artefacts of pressure imbalance within the chamber as an under-pressure will draw more gas from the soil to compensate, thereby artificially increasing the soil efflux S_r . The latter point has been addressed by the major manufacturers of such systems based on the designs by Parkinson (1981) and Norman et al. (1992). Norman et al. (1992) also recommended that a narrow piece of vent tube be located on one of the walls of the closed chamber to minimise the pressure differential between inside and outside; if the tube is narrow, the diffusive pathway is small and no CO_2 leakage from the chamber should occur. This conclusion is supported by the results of Longdoz et al. (2000) and more recent review by Davidson et al. (2002) who also conclude that when pressure differentials are kept small (~ 0.1 Pa) then errors in flux estimates are reduced to around 15% (Pumpanen et al., 2004). They further conclude that for typical chambers of 10–20 cm height, most non-steady state methods on typical soils will underestimate fluxes by about 15%. The use of a vent to equalise pressure inside and outside a closed chamber is not without controversy, however, as Conen and Smith (1998) argued that wind blowing over the vent induced a Venturi effect which actually caused air to come into the chamber from the soil thus increasing the efflux. The effect of changing wind-speed outside even a vented chamber on soil efflux inside the chamber remains difficult to predict and it is argued that any new design of chamber needs to be tested thoroughly for such possible sources of error. A new design of vent tube described by Xu et al. (2006) suggests it is possible to eliminate the impact of fluctuating wind speed outside the chamber on soil efflux measurements (a result also suggested by Bain et al., 2005). An alternative approach consists in placing the vent close to the soil (Suleau et al., 2009). Closed chambers may also use a fan to mix the air within the chamber so that a representative sub-sample can be taken, and various

designs of fan and mixing arrangements exist, with the intention of minimising disturbance of the soil boundary layer (Welles et al., 2001), though the use of fans also remains controversial. A further complication of using chambers enclosing vegetation is the difficulty of estimating the actual air-space volume.

3.6.2. Separating autotrophic and heterotrophic respiration; manipulation methods in the field

Autotrophic and heterotrophic respiration are likely to respond differently to environmental drivers, and thus it is desirable to be able at times to separate out the two sources of respired soil CO₂ (Subke et al., 2006). Various manipulation experiments have been designed to disentangle the different components of the total efflux; some of the methods are destructive, others can be used *in situ* (Kuzakov and Larionova, 2005; Kutsch et al., 2010). The four most widely used approaches are shortly introduced below.

3.6.2.1. Component integration. Component integration involves separating the roots from the soil and then measuring respiration separately from each accurately weighed component under controlled conditions in the laboratory. The basic assumption is that the weighted sum of the respiration rates measured after this separation is the same as if the soil was intact and within its original undisturbed environment (Hanson et al., 2000).

3.6.2.2. Root exclusion zones. In croplands, root exclusion zones (zones without vegetation; Hanson et al., 2000), can be created easily, as it requires only that seeds are not sown in the exclusion zone. Comparison of soil respiration in these zones and in zones covered by vegetation allow heterotrophic and autotrophic components to be differentiated. The temperature and soil humidity as well as easily available labile carbon sources (from root exudates) will be different in the root exclusion zone so supplementary micrometeorological measurements are needed.

3.6.2.3. Compound-specific chamber design. Heinemeyer et al. (2007) and Moyano et al. (2007) used a multiplexed soil chamber system to estimate the fraction of the total CO₂ efflux coming from autotrophic or heterotrophic respiration. They used a novel mesh collar design to isolate the three main soil CO₂ efflux components: root, extraradical mycorrhizal hyphae, and soil heterotrophic respiration. Their experiments also showed that the depth to which the soil collars were inserted had a significant bearing on the measured soil efflux. Given that most soil collars are inserted about 5 cm into the soil, cutting any roots and mycorrhizal hyphae in this layer will lead to an underestimate in the fluxes (Raich and Nadelhoffer, 1989). This is likely to be true of many previously published measurements of soil efflux.

3.6.2.4. Isotope methods. The ¹³C or ¹⁴C contents in soil-respired CO₂ can be used to distinguish between autotrophs and heterotrophs given that they discriminate between CO₂ containing one of these heavier-than-normal isotopes and CO₂ that contains the most abundant lighter ¹²C isomer. If this is the case, then the ratio between ¹³C or ¹⁴C and ¹²C in respired CO₂ reflects its origin e.g. photosynthates, or other carbohydrates in the case of roots. Methods using isotope discrimination can be based on measuring the natural abundance of these isotopes in soil or by injecting known quantities of the isotopes into the system and monitoring its evolution some time later (Lin et al., 1999; Subke et al., 2004).

(a) *Natural abundance.* Plants with the C₃ photosynthetic pathway have a ¹³C isotopic signature of about –26‰; C₄ plants have a signature of about –12‰. By growing C₄ plants in a soil that previously grew only C₃ plants, we can estimate the contribution of root respiration to soil respiration. Rochette et al. (1999), for

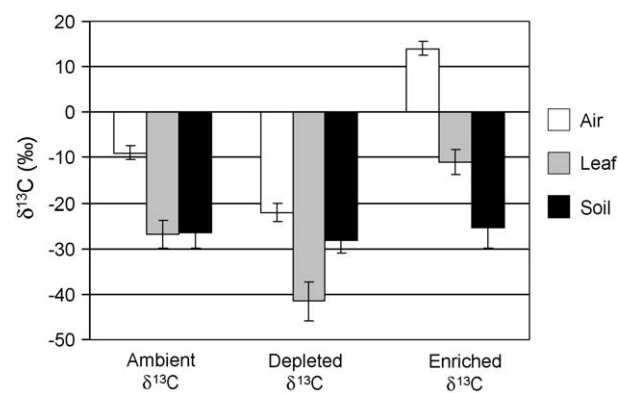


Fig. 4. Carbon isotope ratio of root- and soil respired CO₂ from *Hordeum vulgare* stands exposed to CO₂ with both depleted and enriched δ¹³C relative to atmospheric CO₂.

example, showed that root respiration contributed about 43% of total soil respiration in a maize crop at the height of the growing season, using this approach.

(b) *Labelling.* For C₃ plants growing on C₃-based organic matter in the soil, the isotopic difference may be too small to use the natural abundance technique and an alternative is to label the plant with isotopically distinct air that can be traced in time (Meharg, 1994). Isotopes of carbon can be applied to plants either in a single one-off pulse or continuously throughout the lifetime of the plant as in FACE experiments (Paterson et al., 1997). The isotopic signature of the gas coming from the roots is slightly different from that entering the plant (Duranceau et al., 1999, 2001; Ghashghaie et al., 2003; Gessler et al., 2007; Bathellier et al., 2008), and very different from that coming from the soil heterotrophs (Ekblad and Hogberg, 2000; Fig. 4). However, difficulties arise for FACE as well since no (industrial, labelled) CO₂ is applied to the control rings. For labelling two options are available: either (a) artificially increased concentrations in ¹³C or ¹⁴C or (b) strongly depleted in the heavier isomers, be it in the gas used for fumigation or in the organic matter supplied for decomposition.

3.7. Net ecosystem production

3.7.1. The eddy covariance method for total net ecosystem production

Soil chambers (see Section 3.6) are the simplest method of measuring soil efflux but suffer from limited spatial representativeness and in some configurations, cannot be left for extended periods on the ground since they may alter the soil itself by chamber effects (see Smith et al., 2008a) and they may also be confounded by the influence of vegetation cover and diffusional constraints. The eddy covariance method can measure the net exchange of CO₂ over areas that are typically of the order of several hundred square metres, and the instrumentation is sufficiently robust and energy-efficient that it can be used for extended periods from seasons to years (Baldocchi, 2003). The technique uses the covariance between rapid fluctuations in vertical wind-speed measured with a three-dimensional ultrasonic anemometer and simultaneous measurements of the rapid fluctuations in the CO₂ concentration, as measured by a fast-response infra-red gas analyser (IRGA; Moncrieff et al., 1997; Aubinet et al., 2000) or similar equipment. The eddy covariance technique provides a measure of net ecosystem production only; to partition NEP into GPP or autotrophic or heterotrophic respiration, the ancillary measurements described elsewhere in this paper are required (though GPP and TER can be partitioned on the basis of analysis of night flux mea-

surements; see Section 3.1). The uncertainties associated with eddy covariance are well known and widely discussed but should not detract from the fact that any experimental method has an empirical uncertainty associated with it. Thus, as with other methods one should be aware of the specific strengths and limitations. Where chamber measurements are most suited for small-scale investigations, it has become widely accepted that the eddy covariance method is most valuable for obtaining NEE over spatial areas ranging from the field and ecosystem scale to regional scales (10^4 km^2 or larger; see Kustas et al., 2003; Cleugh et al., 2004; Schneider and Eugster, 2007). Eddy covariance measurements of CO_2 flux at night can be compared to soil chamber measurements of the CO_2 efflux, provided atmospheric turbulence is sufficient to mix the air from the ground surface up to the level where the eddy covariance sensors are (Smith et al., 2008a). During night time, when there are high wind speeds and these conditions are met, the two techniques can, together, be used to derive response surfaces relating soil efflux to soil temperature, a vital component in modelling the carbon budget of ecosystems. In some studies, the agreement is less good even when data are rejected for low wind speed conditions (when nocturnal advection may be present at a site, thus removing CO_2 before it can be carried past the eddy covariance sensors, or when turbulent processes are not adequately represented by the typical 30-min averages used for quantifying eddy covariance fluxes) and differences of up to 20–30% can be seen (e.g. Lavigne et al., 1997).

Because plants exchange most of their carbon as CO_2 , eddy flux-derived NEP is an ideal variable for C budgeting from local to regional scales. Smith et al. (2008b) review the use of flux networks for assessing regional sectoral carbon budgets. Over time, net C fluxes are good proxies for ecosystem total biomass stock change (Baldocchi, 2003). There are hundreds of eddy covariance towers monitoring continuously and organized in a global network (www.fluxnet.org), including the cropland sites in Europe described in the other papers in this issue. Despite the dominance of towers in forest ecosystems which is a result of the historical development of the European network, the proportion of cropland and grassland sites represented in CarboEurope-IP roughly matches the proportion of the land surface covered by croplands and grasslands (CarboEurope-IP, 2009).

The main limitations of the eddy covariance technique are related to the fact that this is a point-in-space measurement. Thus, the relationship between this point measurement of a flux crossing the sensor's reference surface and the footprint area upwind of the sensors requires certain atmospheric conditions that involve well-developed and continuous turbulence, stationary wind field and thus turbulence conditions, and a homogeneous distribution of sources and sinks of CO_2 in the footprint area. These conditions are not always met. For example, during the night-time CO_2 can accumulate under plants (Baldocchi, 2003), such that the soil efflux and the flux measured at the height of the eddy covariance instruments are no longer directly related to one another. Moreover, the CO_2 accumulated near the ground surface may be carried away via lateral cold-air drainage flow under such conditions. Since it is rare for a flux tower to be sited on completely flat terrain, even gentle topography can lead to errors in measurements of C, either through lateral drainage of accumulated CO_2 or drainage into the footprint, which produces a displacement that can result in an inaccurate estimate of the net sink (CO_2 absorbed by photosynthesis is well measured during the day; CO_2 released by respiration during the night is not totally accounted for in the fixed measurement location). It may even be possible that such lateral CO_2 drainage is more pronounced at apparently flat sites than in steeper terrain, where thermo-topographic wind systems reduce the probability of low-turbulence conditions at night (Hiller et al., 2008). These shortcomings mean that the energy balance closure is rarely com-

plete and present significant challenges, but some recent studies promise to, at least partly, address these issues (Kruijt et al., 2004), and eddy flux gap-filled data can often match stock change measurements satisfactorily (Saleska et al., 2003; Black et al., 2007). Process-based models benefit greatly from validation with eddy flux data (e.g. Morales et al., 2005).

The eddy covariance technique (and indeed most of the measurement techniques listed) encounters a number of additional problems when used in agricultural ecosystems, particularly croplands. Firstly, croplands are intensively managed and exposed to frequent and persistent disturbance, perhaps more so than grasslands and unlike many forest ecosystems, even those subject to some management. This means that disturbance can be a particular issue on cropland sites. Secondly, since nearly all crops are sown and harvested annually, and are often grown in rotation, the impact of the previous crop or past management can have a far greater effect in croplands than in perennial grasslands or forest ecosystems. The impact of recent management history is therefore far more significant for cropland sites. Thirdly, most crops are grown in much less than 12 months, especially winter crops, which makes it difficult to report an annual emission for a specific crop. The most important issue, however, relates to the diversity of croplands and how representative the cropland flux towers can be for the regional to European scale. In forest systems, towers can be placed in similar age stands of similar species in a number of regions. This ensures some degree of homogeneity that allow sites to be compared (Morales et al., 2005) and even for results to be directly up-scaled (Papale and Valentini, 2003). For croplands, however, the range of crops, tillage practices, crop management practices and recent land management histories is so large that no two sites are likely to be comparable, i.e. even if they have comparable crops; they are likely to use different tillage regimes, fertilisation practices and sowing/harvesting dates, whilst also occurring as part of a different rotation. In a recent synthesis, Eugster et al. (2008) counted 442 management practises over 4 years (2004–2007) in 15 cropland sites. Homogeneity within a single field of a single eddy covariance tower is much higher than in any other ecosystem type, but the cropland landscape is more heterogeneous over larger spatial scales, with individual fields often growing different crops. This diversity makes direct comparison between sites or direct up-scaling extremely difficult. Instead, process-based models are necessary to interpret the contributions of different processes to the measured net ecosystem productivity at each site (see Wattenbach et al., 2010). The main uncertainties in eddy covariance data arise from poorly-constrained footprints, disturbance within the footprint, gap filled data and in-canopy CO_2 storage or drainage flow leaving the footprint area during the night-time. These uncertainties have been discussed previously (e.g. Baldocchi, 2003), and methods to estimate these uncertainties have been recently been proposed (Béziat et al., 2009; Dragoni et al., 2007; Hollinger and Richardson, 2005; Richardson and Hollinger, 2007; Richardson et al., 2006).

3.8. Volatile organic compounds

VOC losses in croplands are relatively small compared to losses from forests. Variants of the eddy covariance system can be used to measure VOCs, and these can also be collected and measured using the chamber measurements described above. The Relaxed Eddy Accumulation system (hereafter REA) is an alternative to perform mass flux measurements when fast-response sensors are not available to determine the fluctuations of the species concentration in the air mass. The principle of the REA method was originally proposed by Businger and Oncley (1990) and ever since it has been widely applied for flux measurements of VOC (Pattey and Desjardins, 1999; Haapanala et al., 2006; Graus et al., 2006).

The REA technique relies on conditional sampling: air samples are collected at a constant flow rate into two different reservoirs depending on the sign of the vertical velocity of air. After a suitable sampling period the reservoirs are analysed with slow-response instruments, and the scalar flux is related to the concentration difference in the two reservoirs by:

$$F = b\sigma_w(c^+ - c^-)$$

where F is the scalar flux ($\text{kg m}^{-2} \text{s}^{-1}$); b is a theoretical or empirical coefficient (dimensionless), σ_w is the standard deviation of the vertical velocity, and c^+ and c^- are the mean concentration (kg m^{-3}) of the scalar in the up- and downdraft reservoirs, respectively. A threshold on the vertical velocity w_0 is usually applied to perform the conditional sampling: it corresponds to a wind-deadband for which air samples whose velocity is close to zero are rejected. It increases the concentration difference between the two reservoirs thus decreasing the precision requirements for the chemical analysis. It also avoids selecting air near small and unreliable vertical velocities (Brut et al., 2004). Measurements of VOC fluxes using REA systems were mainly operated over forests but less is known about croplands emissions. The emissions are vegetation species specific, and therefore, landscape-scale emissions are highly variable. However, whilst VOC fluxes are highly important for photochemistry of the atmosphere, the mass of C involved is within the noise and uncertainty of any component of the overall C budget of a site, and can thus safely be neglected.

3.9. Carbon lost to water, D (DOC, DIC and POC)

The carbon lost to water courses as dissolved organic and inorganic carbon (DOC and DIC) and as particulate organic carbon (POC) can be measured in field drains (Byre et al., 2001) but is more regularly measured in stream and river waters at catchment scale. Many studies now use suction cups permanently located in the fields to measure directly under the root zone (Weihermuller et al., 2007). There is a large uncertainty on the value of D , since the biome of origin for measured river carbon is unknown. These approaches are, however, more difficult in cropland because of frequent disturbance caused by farm management activities and the use of different rotations with variable fallow periods. There is an even larger uncertainty associated with the component of D eroded from old soil organic matter, as opposed to that derived from C recently added to the soil. Most of river transported carbon originates from ecosystems, but can have different lifetimes through the river filters, being either degassed to the atmosphere within a year, or sequestered in long-lived organic sediments. DIC can increase in alkaline soils through CO_2 adsorption. Care must be taken to avoid double counting of POC with the soil C erosion flux (Section 3.13). Meybeck and Ragu (1996) and Ciais et al. (2008) estimated values for D for the EU-25. Ciais et al. (2008) estimated the flux D for EU-25 and included losses of DIC by mineral alteration and weathering processes, and export of DOC, POC from cropland soils. It is generally accepted that D is small for croplands on mineral soils, compared to the measured D fluxes from organic soils in upland catchments (Evans et al., 2006). A recent data compilation by Ciais et al. (2008) suggests that a minimum fraction of 70% of the ecosystem carbon transported by rivers is returned rapidly to the atmosphere. Schulze et al. (2010) conclude that, as a first approximation, roughly 30% of D is a component of NBP at the EU-25 scale.

3.10. Methane (CH_4)

Methane can be measured using variants of the eddy covariance system or through chamber measurements. Chamber measurements are widely used as they are easy to apply in field trials with multiple small plots. Manual chamber measurements are usually

made on a weekly basis, and rarely more frequently than daily, whereas automatic chamber measurements allow continuous and frequent measurements. For example, Skiba et al. (1992) used static chambers from which air samples were taken after one-hour of incubation. Samples were then collected by syringe and transferred to Teflar bags. Samples can then be analysed for methane by gas chromatography using a flame ionisation detector for the analysis of methane (Skiba et al., 2002).

Automated chamber methods are expected to produce more reliable results compared to manual chamber measurements, as diurnal variations are captured. Automatic chamber measurements may suffer from underestimation of fluxes due to the chamber effects upon soil moisture during rainfall events (Yao et al., 2009). Yao et al. (2009) compared the methane emissions from manual and automated chamber measures in a rice–wheat rotation ecosystem over a non-waterlogged period. No significant difference was reported in cumulative methane emissions from the two methods in non-waterlogged conditions, but significant differences were reported in CO_2 and N_2O .

Recent developments in laser spectroscopy also provide new measurement devices to measure CH_4 concentrations at high temporal resolution (10 Hz), adequate for eddy covariance flux calculations (e.g. Verma et al., 1992; Kroon et al., 2007; Hendriks et al., 2008). Werle and Kormann (2001) used a high-frequency-modulation spectrometer with a lead-salt diode laser operating in the ν_4 band of CH_4 at $7.8 \mu\text{m}$ as a fast chemical sensor for eddy covariance measurement of methane emission from rice fields in Italy. Eddy covariance measurements were compared with simultaneously recorded emission data from closed-chambers and showed that the closed-chamber measurements were 60–90% higher than those measured using the eddy covariance method. The higher apparent methane emission from automatic chambers might be due to strong turbulence introduced by the mixing fan inside the chamber that does not allow natural gradients to form inside the box (Werle and Kormann, 2001).

Methane losses from croplands are expected to be a significant component in rice paddies of Mediterranean countries (Eugster et al., 2010), but might be negligible, or even a net sink (i.e. methane oxidation outweighs methane production), in other cropland systems on well aerated soils. In Europe, soils that are not well aerated are typically used as grasslands, not croplands. Moreover, fertiliser is mostly applied in the form of dry mineral fertiliser, whereas slurry and liquid manure that might lead to high CH_4 losses for a short period after application are not as commonly spread on croplands as grasslands.

3.11. Carbon lost in fires (F)

Ciais et al. (2010) attempted to quantify carbon lost in fires from croplands in Europe. They noted that most fire emissions now occur in Eastern European member states, where harvest residues are more frequently burnt in the field. In EU-25, the deliberate burning of agricultural residues was officially banned in 1993 (except for specific crops). Because agricultural fires are human-prescribed, their reported inter-annual variation appears quite large, with only small random inter-annual variability. A decline in the loss of carbon from Europe's croplands through fire was reported by Giglio et al. (2006) for the 1990s, and Ciais et al. (2010) reported a further decline after 2000, and estimated that prescribed agricultural fires diminish the amount of carbon available for soil decomposition by only 1%.

3.12. Carbon removed at harvest (H)

The carbon removed at harvest is part of the total NPP and is described in detail in Section 3.2; see also Section 3.2.4 on

NPP_{residual} . On working farms, farmers do record harvested yield and the C content of the grain can be easily measured but Béziat et al. (2009) found that uncertainty associated with the C removed at harvest was much larger than the non-systematic uncertainty associated with the EC measurements, even if a substantial number (30) of samples were collected at harvest. Similar results were found by Moureaux et al. (2008). Whilst much effort is put into improving EC measurements, uncertainty in experimental plots could be reduced by collecting samples to better estimate harvest removals.

3.13. Carbon lost through soil erosion (E)

Soil C erosion (E) is defined as the sum of human accelerated erosion and the effect of 'natural' processes, and is a flux in the European GHG budget that cannot be neglected (Ciais et al., 2008). Soil erosion can be measured either through measuring direct soil loss, through measuring re-deposition of the eroded material, or can be estimated using fallout nucleides ^{137}Cs and ^{210}Pb as tracers (Van Oost et al., 2007). Van Oost et al. (2007) created a global map of soil erosion and showed significant cropland soil erosion rates in EU-25, of the order of $10\text{--}15\text{ g C m}^{-2}\text{ y}^{-1}$, compared to arable lands in the rest of the World which have lower rates. If the eroded soil C is simply moved from one place in the cropland landscape to another, it does not affect the net cropland C flux.

3.14. Inputs of carbon from organic fertilizers and seeds (I_{fert})

Carbon can be determined from estimates of the dry matter content of freshly added organic materials and then analysing the C content of the dry matter. If this cannot be measured directly, C is often assumed to make up 35–55% of the fresh dry matter, though the C content can vary considerably (Wim Chardon, pers. comm.; Werner Eugster, unpublished results). Chalk applications have a C content of about 9% by mass (Werner Eugster, unpublished results). Farmers often only consider the N content of fertilizer when spreading on the field, as the N content is used in the calculation of how much additional mineral N is needed for crop fertilisation, but there are literature estimates (Smith et al., 1997; MAFF, 1994) of the C content of various manures which can be used to estimate C inputs for organic fertiliser (I_{fert}). Organic manures can, however, vary greatly depending on source of manure, materials co-composted, and method and duration of composting, so at site level, the carbon inputs from manure should be measured. An additional C input occurs in the form of seed, but this is negligible except for potatoes and similar crops, which may add around 400 kg C ha^{-1} (Aubinet et al., 2009). Addition minor sources of C to croplands include C in ash and black C, as well as from aeolian dust.

3.15. Net biome production

The net biome production is the carbon remaining in the ecosystem when all other fluxes have been accounted for. The Net Ecosystem Carbon Budget is equal to the NBP when integrated over time and space (Schulze and Heimann, 1998; Buchmann and Schulze, 1999; Chapin et al., 2006). In croplands, due to the removal each year of the crop material, the NBP is estimated by measuring the long-term change in soil organic carbon (SOC).

For soil C, long-term C cycling is often studied by measuring changes in total soil organic carbon over long periods (years to decades; Smith et al., 1997). In many sites, while soil organic matter concentration has been measured over many years, calculations of total soil organic C contents has been hindered by the absence of data on soil bulk density and by discrepancies in sampling techniques (e.g. no standardisation of soil depth and of soil layers). In the last decade, individual long-term experiments have been brought

together into networks such as the Soil Organic Matter Network (SOMNET; Smith et al., 2001), EuroSOMNET (Smith et al., 2002a,b) and LTSE (Richter et al., 2007). Such networks allow the impacts of management practices on SOC stocks to be determined and for regional projections of the impact of different management strategies to be explored (e.g. Smith et al., 1997, 1998, 2000, 2007, 2008c; Freibauer et al., 2004; Ogle et al., 2005).

In addition to measurement of changes in bulk SOC, other techniques are now being used to better understand SOC turnover. Various fractionation techniques are being used to isolate different components of SOC (e.g. Six et al., 2001; Del Galdo et al., 2003) to better understand SOC turnover, and to identify sensitive indicators of SOC change. Mathematical methods to test the relationship between measured fractions and model pools are being developed (e.g. Smith et al., 2002a,b; Zimmerman et al., 2007), in order that this information can be incorporated into process-based models.

As indicated earlier the ^{13}C natural abundance tracer technique utilises the fact that plants with the C_3 photosynthetic pathway have a different ^{13}C isotopic signature to plants with the C_4 photosynthetic pathway. When C_3 plants have been replaced by C_4 plants, or vice versa, the ^{13}C is altered, allowing new C inputs to be separated from old C that is already present. The technique has been in use for some time (Balesdent et al., 1987; de Moraes et al., 1996) but is still yielding important new results, especially when coupled with modelling techniques. The $^{13}\text{C}_2$ pulse labelling technique also shows promise for improving our understanding of SOC turnover. This technique uses the stable ^{13}C isotope, pulsed as $^{13}\text{CO}_2$ for 1 or 3 days using a chamber enclosing the plants. The ^{13}C isotope signal can then be tracked in shoots, roots, and rhizosphere soil during the months following the pulse (Rangel-Castro et al., 2004). ^{14}C bomb C can also be extremely useful in examining soil C turnover, especially when coupled with models (Jenkinson and Coleman, 1994; Hahn and Buchmann, 2004).

4. Methods to integrate NECB of croplands at the continental scale

The main method for interpreting flux (and other) results, and for extrapolating temporally and spatially the data in croplands (and other systems), is the use of process-based models (Wattenbach et al., 2010). Process-based models are continually being improved, with the most significant advance in the last decade being the development, and testing of models that simulate all biogenic greenhouse gases. The main hurdle to applying such models at the regional level is data limitation. In recent years, high-resolution, spatially-explicit datasets have become more readily available. National, regional and global databases have been improved and it is now possible to run models for entire sub-continental regions (e.g. Europe, USA) at fine spatial scale such as a $10'$ by $10'$ grid (Rounsevell et al., 2005; Mitchell et al., 2004; Smith et al., 2005, 2006; Gervois et al., 2008; Wattenbach et al., 2010), at US county level (Parton et al., 2005) or even at 1 km^2 grid scale. Historical climate data have been interpolated to higher spatial resolution and future climate scenarios from global climate models have been downscaled to the same spatial resolution (e.g. Mitchell et al., 2004). Soil data are now available at high spatial resolution (e.g. at 1 km^2 for Europe; Jones et al., 2004, 2005) and historical land-use data and future land use scenarios are beginning to be constructed at high resolution (e.g. Rounsevell et al., 2005, 2006). There are many areas in which these datasets require further improvement, but significant advances have been made in some regions in recent years, including in Europe. In some other regions, however, especially in the developing world, such datasets are poor or non-existent. The development of airborne measurement platforms that are capable of measuring CO_2 fluxes is one way in which regional

models might be verified (e.g. Gioli et al., 2004; Miglietta et al., 2006).

Improvements in remote sensing capability and products have greatly improved datasets on land use and land cover and enabled improvements in modelling the consequences of recent land-use change. All land based sectors use process-based models and remote sensing products. Remote sensing data and land information systems for C accounting have been particularly successful when land is converted from forest to other land use (Nobre and Harriss, 2002). The availability of at least 30 years of frequent Landsat coverage for all the continents also allows historical land use change to be reconstructed. More recent sensors combining high temporal and high spatial resolution (such as SPOT or FORMOSAT) are of great interest for the scientific community since they can be used to monitor fast vegetation development (i.e. crop growth) and drive or constrain empirical or process based models. For instance, Leaf Area Index maps obtained from these satellites can be used to constrain crop models such as SAFY (Duchemin et al., 2007). Assessments of historical changes prior to the Landsat era are and will remain more challenging and must rely on available maps of varying quality (e.g. Schneider and Eugster, 2007). Remote sensing and terrain modelling also offer great advances in mapping of landscapes in complex terrains for C cycle assessment. The Shuttle Radar Topographic Mission (SRTM, NASA) of 2000 has produced the finest terrain model to date. Future developments include the use of instruments that are currently airborne on satellites or the space shuttle and new-generation CO₂ sensors. For example, the Laser Vegetation Imaging Sensor (LVIS) and related airborne active sensors are not yet space-borne but the latest version of the LVIS instrument allows it to be used with standard aerial photographic windows (see review in Smith et al., 2008b). Such instruments will eventually be mounted on the space shuttle or satellites. A new generation of CO₂ and CH₄ sensors has recently been deployed that might allow for large scale verification of CO₂ inversion models by monitoring the entire atmosphere (pixels are integrated columns, for example the recently launched GoSat with a resolution of 10 km by 10 km), for relatively small changes in CO₂ and CH₄ concentrations.

5. Concluding remarks

Many measurements are necessary to assess the full carbon budget of croplands. Elsewhere in this issue are many examples of how eddy covariance techniques have been coupled with auxiliary measurements to assess cropland carbon budget using a network of European cropland sites. This paper has reviewed the various components of the cropland carbon budget, how these relate to each other and how they can be measured or estimated. This synthesis highlights the complicated portfolio of evidence that needs to be gathered at each site before a full carbon budget can be assembled, and summarises the additional measurements that are necessary to assess the full GHG budget of a cropland site (see also Ceschia et al. [this issue] for details of assessing N₂O emissions from cropland sites), and we provide a section on up-scaling results to the continent of Europe. We show that croplands are so diverse in terms of crops grown, rotation, management, soil types, and climatic conditions that it is not possible to directly upscale results from individual cropland sites to the European scale, even with an extensive network of sites. Instead, the data are most valuable to calibrate and parameterise or validate ecosystem models, which are then combined with detailed spatial datasets, to allow a more process-based understanding at the plot scale to be up-scaled to the continent's total cropland area. Through this combination of detailed measurement, improved understanding, and model development, we will be able to better estimate and project European

cropland GHG fluxes, as well as advance our understanding of the factors controlling GHG fluxes in Europe's croplands.

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