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Greenhouse gas emissions from four bioenergy crops in England and Wales: Integrating spatial estimates of yield and soil carbon balance in life cycle analyses

JONATHAN HILLIER*, CARLY WHITTAKER†, GORDON DAILEY‡, MATTHEW AYLOTT§, ERIC CASELLA¶, GOETZ M. RICHTER‡, ANDREW RICHE‡, RICHARD MURPHY†, GAIL TAYLOR§ and PETE SMITH*

*Institute of Biological and Environmental Sciences, School of Biological Sciences, University of Aberdeen, Cruickshank Building, St. Machar Drive, Aberdeen AB24 3UU, UK, †Department of Life Sciences, Sir Alexander Fleming Building, Imperial College London, South Kensington Campus, London SW7 2AZ, UK, ‡Rothamsted Research, Harpenden, Hertfordshire AL5 2JQ, UK, §School of Biological Sciences, University of Southampton, Bassett Crescent East SO16 7PX, UK, ¶Centre for Forestry and Climate Change, Forest Research, Farnham, Surrey GU10 4LH, UK

Abstract

Accurate estimation of the greenhouse gas (GHG) mitigation potential of bioenergy crops requires the integration of a significant component of spatially varying information. In particular, crop yield and soil carbon (C) stocks are variables which are generally soil type and climate dependent. Since gaseous emissions from soil C depend on current C stocks, which in turn are related to previous land management it is important to consider both previous and proposed future land use in any C accounting assessment. We have conducted a spatially explicit study for England and Wales, coupling empirical yield maps with the RothC soil C turnover model to simulate soil C dynamics. We estimate soil C changes under proposed planting of four bioenergy crops, *Miscanthus* (*Miscanthus* × *giganteus*), short rotation coppice (SRC) poplar (*Populus trichocarpa* Torr. & Gray × *P. trichocarpa*, var. Trichobel), winter wheat, and oilseed rape. This is then related to the former land use – arable, pasture, or forest/seminatural, and the outputs are then assessed in the context of a life cycle analysis (LCA) for each crop. By offsetting emissions from management under the previous land use, and considering fossil fuel C displaced, the GHG balance is estimated for each of the 12 land use change transitions associated with replacing arable, grassland, or forest/seminatural land, with each of the four bioenergy crops. *Miscanthus* and SRC are likely to have a mostly beneficial impact in reducing GHG emissions, while oilseed rape and winter wheat have either a net GHG cost, or only a marginal benefit. Previous land use is important and can make the difference between the bioenergy crop being beneficial or worse than the existing land use in terms of GHG balance.

Keywords: Bioenergy, greenhouse gas, LCA, *Miscanthus*, poplar, RothC, short rotation coppice

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Introduction

The amount of energy consumed globally has increased more than 10fold during the 20th century to an estimated 451 exajoules (EJ) in 2002, which is approximately 10,800 million tonnes of oil equivalent (Boyle, 2004). With the emerging economic growth areas such as China and other Asian countries continuing to develop, this demand is projected to increase with time. However, fossil fuel reserves are declining. Fossil fuel reserves vary depending on the fuel; proven coal reserves may last for around to 130 years, oil for approximately 40 and natural gas for about 60 years (BP, 2008). Renewable energy sources may be used to replace part of the energy currently provided by fossil fuels, and provide a degree of energy security for those countries lacking in fossil fuel resources.

Correspondence: Dr Jonathan Hillier, tel. + 44 1224 273810, fax + 44 1224 272703, e-mail: j.hillier@abdn.ac.uk

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Bioenergy crops form one part of a suite of options for renewable energy sources. However, in order to determine the sustainability of production of bioenergy, rigorous life cycle analyses (LCAs) are required. Full greenhouse gas (GHG) LCA of bioenergy production chains are often constrained by a lack of information on preharvest GHG costs related to land use change, either from the soil or from management of the energy crop (Elsayed *et al*., 2003). Recent studies on liquid biofuels have considered displaced land use effects and the potential costs related to converting existing forest land to bioenergy crop production (Fargione *et al*., 2008; Searchinger *et al*., 2008), and concluded that the energy crops they considered are at best of marginal value in mitigating GHG emissions. However, this is not the only scenario, and others may potentially be more positive in GHG balance – for example, the conversion of existing poor quality arable land to relatively low input biomass crops such as *Miscanthus* and short rotation coppice (SRC). St Clair *et al*. (2008) calculated stock changes of soil organic carbon (SOC) according to Ogle *et al*. (2005) and included estimates of farming-related emissions; allowing several baseline land uses and bioenergy crops to be compared in terms of farm gate emissions. Their findings indicated that, although the savings may be small for liquid biofuel crops (e.g. wheat, oilseed rape), and negative when replacing existing natural or forest land, replacing grassland or arable land with *Miscanthus* or SRC can yield significant GHG savings, even before consideration of the fossil fuel carbon (C) displaced at end use.

In this study, we consider emissions associated with the production of energy from winter wheat (for bioethanol), oilseed rape (for biodiesel), *Miscanthus*, and SRC poplar. Liquid biofuels from wheat and oilseed rape are typically blended with conventional fossil fuels such as petroleum spirit and diesel, and can be consumed in most modern vehicles. *Miscanthus* and SRC are generally low-input biomass crops that are grown over periods of up to 20 years (e.g. DEFRA, 2002, 2007). After 1–3 years of establishment growth, the crops are cut either annually (*Miscanthus*) or in cycles of 3–4 years (SRC). After harvesting, both crops can be used to generate heat, electricity, or combined heat and power (CHP) and in the future could even be converted to liquid biofuels (Sims *et al*., 2006). Although, there have been some UK-based LCA studies performed for *Miscanthus* and SRC [ElSayed *et al*., 2003; BEAT v2, 2008, AEA Technology & North Energy Associates Ltd. Available at http://www.biomassenergycentre.org.uk/portal/page?_pageid=74,153193&_dad=portal&_schema=PORTAL (Accessed 20 March 2009)], and for both bioenergy and food crops (e.g. Adler *et al*., 2007; St Clair *et al*., 2008; Hillier *et al*., 2009), and in some cases emissions from soil incorporated, variability in this aspect has not as yet been given detailed consideration.

One might expect that production of winter wheat and oilseed rape would have a similar effect on SOC whether grown for energy or food. Thus when grown for bioenergy, they may be considered as typical arable crops. In general, arable production reduces soil C stocks relative to natural ecosystems and those which are less disturbed (e.g. permanent grasslands – Guo & Gifford, 2002; Bellamy *et al*., 2005; Smith *et al*., 2005), principally through breeding for high grain/seed yields (reducing soil C inputs), and through regular disturbance of the soil (tillage). This decline is limited in duration – in time the SOC reaches a new equilibrium and no further decline occurs (Jenkinson, 1990).

However, since *Miscanthus* and SRC are perennial crops the soil is less disturbed after establishment, which, in principle, should help to maintain or increase SOC stocks. Additionally, these crops have thus far not received the same breeding focus for above ground productivity, which may mean that a greater proportion of their biomass is directed towards belowground production – leading to relatively high C inputs to the soil from roots, rhizomes, and litter (Himken *et al*., 1997; Neukirchen *et al*., 1999; Riche & Christian, 2001). It has been suggested that annual soil C gains of the order of 0.5–1 t ha^{-1} are reasonable when converting arable land to *Miscanthus* or SRC, e.g. Hansen (1993), Hansen *et al*. (2004), Hoosbeek *et al*. (2004), or see Rowe *et al*. (2009) for a recent review. However, since there is a lack of consistent information over a range of climates and soil types, the evidence cannot be taken as absolute. Indeed in some cases, depending on previous land use, and plantation age, little or no gain has been found, e.g. Guo & Gifford (2002), Saurette *et al*. (2008).

The IPCC Tier 1 method (IPCC, 2004) to estimate soil C emissions from land use change, although robust and easy to employ, lacks spatial precision, and thus the effects of geographical variation in crop yield, soil texture, and climate remain unaccounted for. Process-based soil C models, which are now abundant (e.g. SOMNET, Smith, 2001), are the ideal tools to integrate such effects, although the variation in complexity and scope of the models is large. In this study, we used the RothC model (Coleman & Jenkinson, 1996), which is one of the most widely used soil C turnover models. It has been evaluated for a range of climates and vegetation types (cropland, grassland, and forests, e.g. Coleman *et al*., 1997; Smith *et al*., 1997; Falloon & Smith, 2002; Skjemstad *et al*., 2004), and has been previously used for prediction at both regional and global scales (e.g. Jenkinson *et al*., 1991; Wang & Polglase, 1995; Falloon *et al*., 1998; Tate *et al*., 2000; Smith *et al*., 2005, 2006, 2007; Falloon *et al*., 2006). The model allows integration of climate and soil type and expected crop yields to predict soil C turnover. We have conducted a spatially explicit study for England and Wales, using yield maps of *Miscanthus* (*Miscanthus* × *giganteus*), SRC poplar (*Populus trichocarpa* Torr. & Gray × *P. trichocarpa*, var. Trichobel), winter wheat and oilseed rape to simulate soil C turnover over a 20-year period.

The GHG emissions from soil are placed in context with life cycle emissions by integrating simulated emissions from the soil with those from crop management and cultivation, and then quantifying the potential fossil fuel C that could be displaced. It is not within the scope of this study to consider the various LCA implications of the many options available for the land use, the use of coproducts from biofuel production (e.g. use of rape meal for fuel or for animal feed), or the various options available for conversion technology. Indeed such results will also change as yields, technology and biomass supply chain scales are optimized. Here, we study currently representative systems for the four bioenergy crops, in order to consider soil-related emissions in the context of full LCAs, and hence explore the relative GHG savings that could be achieved with each crop.

Materials and methods

Spatial datasets

For spatial estimation of yields and emissions, we have used predictive empirical yield models for SRC (Aylott *et al*., 2008) and *Miscanthus* (Richter *et al*., 2008), and regional Defra yield statistics for winter wheat and oilseed rape, to estimate annual C inputs to the soil under each of these crops. We employ relationships previously developed for use in the SUNDIAL model (Smith *et al*., 1996) to estimate soil C inputs from the plants, as previously described in Smith *et al*. (2005). These figures were input into RothC along with climatic and soil data described below to estimate spatial emissions under each of the crops for England and Wales.

For the predictive yield models and RothC, the same spatial datasets were used to ensure consistency in our methodology:

- 1 : 250 000 scale map of soil series in England and Wales (NATMAP vector, Cranfield University, UK). Soil variables were clay content (%) and SOC (scaled to $t\,ha^{-1}$ in the top 25 cm of the soil).
- 1 km^2 resolution land cover map LCM2000 (Centre for Ecology and Hydrology, NERC). Land uses were reclassified as either arable (AR), managed grassland (GR), or forest/seminatural (FO).
- 5 km^2 resolution climatic data map providing long-term averages of monthly minimum and maximum temperatures, precipitation and saturation vapour pressures (Perry & Hollis, 2005).

Yield maps

The empirical yield model for *Miscanthus* was derived from harvestable dry matter yields at 14 field trials in the United Kingdom and site-specific meteorological variables and soil available water (Richter *et al*., 2008). Measured harvestable yields of crops established for at least 3 years at these arable sites ranged from 5 to 18 $t\,ha^{-1}$ dry matter, averaging 12.8 ($\pm$ 2.9) $t\,ha^{-1}$ dry matter. Harvestable yields across all sites depended on the following input variables: average seasonal air temperature ($P < 0.01$) and cumulative precipitation ($P < 0.001$) (both for the period April to September), and soil available water capacity (AWC; $P < 0.001$). Rainfall between maturity and harvest (October–mid-February) had a small negative effect on yields ($P < 0.01$). The error of the fitted model was approximately 15% (2 $t\,ha^{-1}$) and a total of 50% of the observed yield variation could be accounted for. Low yields tended to be overestimated and high yields underestimated.

The amount of litter measured in one *Miscanthus* experiment at Rothamsted between 1994 and 2002 ranged from 2.2 to 7.5 $t\,ha^{-1}$ dry matter, corresponding to 17–35% of the total (harvestable plus litter) biomass yield (Christian *et al*., 2008). The average weight loss as a result of litter fall, 27%, is in reasonable agreement with similar data for litter elsewhere in the literature (Himken *et al*., 1997; Clifton-Brown & Lewandowski, 2002; Jorgensen *et al*., 2003).

A similar empirical yield model was developed for three SRC species – one poplar and two willow – of which only the poplar model Trichobel is used here. The model was derived from harvestable dry matter yields measured at 49 field trials in the United Kingdom and used site-specific meteorological, soil and topographical variables (Aylott *et al*., 2008). The sites were not irrigated or fertilized. Predicted harvestable yields for Trichobel over two crop rotations each of 3 years, ranged from 5 to 16 $t\,ha^{-1}$, averaging 9.3 ($\pm$ 3.0) $t\,ha^{-1}$ dry matter. By using partial least squares regression, spring temperatures and summer rainfall were identified as principal limiting factors of yield ($P < 0.01$). The error of the fitted model was approximately 15% (1.4 $t\,ha^{-1}$ dry matter) and a total of 72% of the observed yield variation could be accounted for. As with *Miscanthus*, low yields tended to be overestimated and high yields underestimated.

For winter wheat and oilseed rape, regional Defra yield statistics from 2001 by NUTS 2 region were used. The mean dry matter yields were 7.7 and 2.9 $t\,ha^{-1}$ for winter wheat and oilseed rape, respectively. Since the data represent measured yields from sizeable farm holdings, they may not correspond precisely with the yield maps for *Miscanthus* and SRC. In general, since farmers are presumably more likely to grow crops where they give reasonable yields, it is likely that there is a positive bias associated with winter wheat and oilseed rape yields in comparison with *Miscanthus* and short rotation poplar. It should also be noted that there is considerably less variation in the winter wheat and oilseed rape maps than in those for SRC or *Miscanthus* (indeed there is no regional variation at all in the oilseed rape map). A consequence of this relative lack of variation will be explored in the 'Results' and 'Discussion'.

Soil C emissions

The soil C turnover model RothC requires information regarding C inputs to the soil from the plant (including those from leaf litter and from the root system). We employ a characterization as a function of yield as employed in SUNDIAL (Bradbury *et al*., 1993; Smith *et al*., 1996, 2005) as follows:

$$\mathrm{C_{input}}(\mathrm{t\,ha^{-1}\,yr^{-1}}) = c_1(c_2 + c_3(1\mathrm{e}^{c_4.\mathrm{Yield}(t\,ha^{-1})})).$$

Thus, for (hypothetical) zero yield, C input is equal to c_1c_2, and as yield increases, C inputs increase monotonically up to an asymptotic value of $c_1(c_2 + c_3)$ at a rate determined by the shape factor c_4.

For oilseed rape and winter wheat these have previously been established as presented in Smith *et al*. (2005). For winter wheat we have

$$\mathrm{C_{input}} = 1.346(1.23 + 1.4(1 - \mathrm{e}^{-0.24.\mathrm{Yield}})), \tag{1}$$

and for oilseed rape

$$\mathrm{C_{input}} = 1.12(1.56 + 3.7(1 - \mathrm{e}^{-0.21.\mathrm{Yield}})). \tag{2}$$

For *Miscanthus* and SRC, we searched available literature in order to construct analogous relations. Suitable data are as yet limited for these crops and thus some assumptions regarding the parameterization were required. Firstly, for both crops we assumed that the variation in yield accounted for half of the variation in C inputs – thus setting $c_2 = c_3 = 1/2$. Similarly, a lack of data led us to assume the shape factor c_4 to be the average of those for winter wheat and oilseed rape. This left the parameter c_1 to be determined, and this was performed via calibration runs of RothC on the following:

- *Miscanthus*: Hansen *et al*. (2004). Predicting for the 25 cm topsoil with initial soil C of 45 t ha^{-1}, an average annual sequestration (balance between C inputs and gross emissions) rate of 0.47 t ha^{-1}, for an average yield of 7.13 t ha^{-1}, with soil clay of 5%.
- SRC: Gielen *et al*. (2005) and Hoosbeek *et al*. (2004): Predicting for the 25 cm topsoil with initial soil C of 30.2 t ha^{-1}, an average annual sequestration rate of 0.73 t ha^{-1}, for an average yield of 10.1 t ha^{-1}, with average soil clay of 14.6%.

This gave the following parameterizations for *Miscanthus*

$$\mathrm{C_{input}} = 6.85(0.5 + 0.5(1 - \mathrm{e}^{-0.23.\mathrm{Yield}})), \tag{3}$$

and SRC

$$\mathrm{C_{input}} = 8.01(0.5 + 0.5(1 - \mathrm{e}^{-0.23.\mathrm{Yield}})). \tag{4}$$

Yield data were used to estimate annual C inputs according to Eqns (1)–(4). All urban areas and inland waterways (according to Great Britain boundary data, Collins Bartholomew, UK), in addition to elevations above 250 m (Land-Form Panorama DTM, Ordnance Survey, UK) were filtered out of the model. The *Miscanthus* model was felt to be unreliable for predicted yields of <5 t ha^{-1} and >15 t ha^{-1} so these points were also filtered.

This yield data, together with the climatic data, soil clay percentage, top SOC, and previous land use were used to initialize RothC. Open pan evaporation (OPE) was estimated via Hamon's equation (Hamon, 1963);

$$\mathrm{OPE} = 0.75\left(\frac{715.5\Lambda e_\mathrm{s}(T_\mathrm{m})}{T_\mathrm{m} + 273.2}\right),$$

where Λ is the fraction of the day that has daylight, T_m is the mean daytime temperature ($^\circ$C), e_s is the saturation vapour pressure at temperature T_m.

Simulations were then run for 20 years under each of the four bioenergy crops and average yearly SOC changes were output. As such, the method consists of balancing yield dependent soil C inputs with, soil type and climate-dependent emissions. Averages of soil GHG balance aggregated by previous land use class for each bioenergy crop were then incorporated into a fuller LCA analysis.

LCAs

Using a similar methodology to that of St Clair *et al*. (2008), we first estimated a farm gate GHG balance by considering annual emissions associated with management and offsetting the emissions associated with management of the land under its previous usage. We then also estimated per hectare emissions from combustion, and compared this with emissions associated with obtaining the equivalent amount of energy from fossil fuels to complete the LCA.

Figures for cultivation, before and after the bioenergy crop was planted, were determined as follows:

Management of grassland and forest were assumed as in St Clair *et al*. (2008). Nitrogen (N) application rates of 150 kg ha^{-1} were assumed for grassland (according to Davies, 2005). For natural/forest land, we assumed management according to the establishment of broadleaf forest as given in Willoughby *et al*. (2004).

Figures for *Miscanthus* were mainly derived from Bullard & Metcalf (2001), assuming no fertilizer was applied. SRC emissions were obtained from growers and Matthews *et al*. (1994); in this study we assumed that N requirements were met by pig slurry application rather than from mineral fertilizers.

For winter wheat cultivation, we assumed N fertilizer application rates of 175 kg N ha^{-1} according to BEAT v2, 2008, AEA Technology & North Energy Associates Ltd. Available at http://www.biomassenergycentre.org.uk/portal/page?_pageid=74,153193&_dad=portal&_schema=PORTAL (Accessed 20 March 2009), with emissions from fertilizer production based on spreadsheets developed by North Energy Associates Ltd., NF0614 (2006). It is assumed that 3.4 t of wheat are required for the production of 1 t of bioethanol (BEAT v2, 2008, AEA Technology & North Energy Associates Ltd. Available at http://www.biomassenergycentre.org.uk/portal/page?_pageid=74,153193&_dad=portal&_schema=PORTAL (Accessed 20 March 2009)]. Alongside bioethanol production, 1.5 t of straw and 1.4 t of dry distillers grains and solubles (DDGS) are also produced. Both of these coproducts can be used as a fuel source to displace fossil fuels, but here it is assumed that straw is used in other markets (e.g. animal bedding) and DDGS is used as an animal feed. The emissions from cultivation and processing are allocated between the coproducts on the basis of their economic price. Thus, for example, assuming a yield of 8 t ha^{-1} for wheat and 3.5 t ha^{-1} for wheat straw, and prices per tonne of wheat grain and straw at £69 and £25, respectively, 86.3% $(=(8\times69)/((8\times69)+(3.5\times25)))$ would be allocated to wheat grain and the remainder to straw.

Similarly, oilseed rape cultivation was also based on BEAT v2, 2008, AEA Technology & North Energy Associates Ltd. Available at http://www.biomassenergycentre.org.uk/portal/page?_pageid=74,153193&_dad=portal&_schema=PORTAL (Accessed 20 March 2009) with fertilizer application rates of 189 kg N ha^{-1}. It is assumed that 2.9 t of oilseed rape is required for production of 1 t of biodiesel, and alongside this around 2.6 t of straw, 1.6 t of rape meal, and 0.1 t of glycerine are produced. Again, these coproducts can have various uses, including combustion, although there may be more economically viable options. For emissions allocation, economic values of £152, £25, £323, £84, £268, and £50 t^{-1} were used respectively for rapeseed, rape straw, crude rapeseed oil, rape meal, biodiesel, and glycerine.

Also, in accordance with BEATv2, the heat for the bioethanol and biodiesel production processes is provided by a natural-gas CHP plant, generating excess electricity which is exported to the National Grid. This displaces up to 642 KWh$_e$ t^{-1} (kilowatt hours of electricity per tonne) of bioethanol, and 401 KWh$_e$ t^{-1} of biodiesel.

Emissions from farm machinery construction were based on data from Matthews *et al*. (1994), Lal (2004), Nix and Hill (2004), and North Energy Associates Ltd., NF0614 (2006). Direct N_2O emissions from soil due to N application are based on the IPCC Tier 1 GHG Inventory Guidelines (2006), which state that 1% of all applied N is emitted in the form of nitrous oxide (N_2O) (de Klein *et al*., 2006). Conversion to C equivalents was performed using 100-year time horizon global warming potentials [298 for N_2O and 25 for CH_4 (methane) – Forster *et al*. (2007)]. The emissions from management are summarized in Table 1.

To calculate fossil fuel displacement, we first assume that *Miscanthus* and SRC displace coal (through cofiring), bioethanol displaces petrol, and biodiesel displaces diesel. The calorific value for *Miscanthus* and SRC was calculated using the Milne equation (Phyllis, 2003). *Miscanthus* is assumed to be combusted at a moisture content of 15% with a low heating value (LHV) of 15.1 GJ t^{-1}, and SRC at 30% (after a period of natural drying) with a LHV of 12.1 GJ t^{-1}. LHV is the energy released on combustion of a given quantity of fuel excluding the heat obtained by condensing the water vapour produced by its combustion. In comparison, we assumed an energy density for coal of 30.5 GJ t^{-1} (DEFRA, 2008).

Table 1 Farm management related emissions for bioenergy and reference land uses

	Management emissions (kg CE ha^{-1} yr^{-1})	Average crop yield (t dry matter ha^{-1} yr^{-1})	Average final product yield (t ha^{-1} yr^{-1})
Miscanthus (bales)	75	9	9
SRC poplar (billets)	40	9.6	9.6
Winter wheat	1005	7.7	2.4
Oilseed rape	1056	3.0	1
Grassland	450	na	na
Forest/seminatural	40	2.8	2.8

CE, carbon equivalent; SRC, short rotation coppice.

Table 2 Energy densities and emissions per unit energy for bioenergy and reference fossil fuel sources

	Displaces	Energy density ($GJ\,t^{-1}$)	Emissions per unit energy produced ($t\,CE\,GJ^{-1}$)
Miscanthus	Coal (via cofiring)	15.1	na
SRC poplar	Coal (via cofiring)	12.1	na
Winter wheat (Bioethanol)	Petrol	26.7	na
Oilseed rape (Biodiesel)	Diesel	37.3	na
Coal	na	30.5	0.0256
Petrol	na	44.7	0.0205
Diesel	na	43.4	0.0210

CE, carbon equivalent; SRC, short rotation coppice.

Carbon dioxide emissions from biomass combustion are considered 'carbon neutral', though CH_4 and N_2O emissions are accounted for according to data from Elsayed *et al*. (2003); 0.002 kg CH_4 GJ^{-1} and 0.005 kg N_2O GJ^{-1} used as default values. For bioethanol and biodiesel, energy densities of 26.7 and 37.3 $GJ\,t^{-1}$ were assumed, respectively (Elsayed *et al*., 2003), as compared with 44.7 $GJ\,t^{-1}$ for petrol and 43.4 $GJ\,t^{-1}$ for diesel (DEFRA, 2008). Emissions related to energy production are given in Table 2.

Results

Figure 1 shows spatial representations of the predicted yields for *Miscanthus* and SRC, and the regional yields for winter wheat and oilseed rape.

In general, *Miscanthus* yields are highest in the South and East, while coppice poplar yields are highest in the North and West. Poplar yields are highest in areas with an abundant water supply (i.e. high rainfall) rather than in areas with longer hours of sunshine and higher temperatures. *Miscanthus* yield maps reflect the large effect of soil AWC expressing low yields on shallow soils, e.g. in the southwest and the Cotswolds whereas highest yields were estimated for The Wash and Kent. Surprisingly, summer rainfall is not high or efficient enough to meet the water needs in the warm areas of the SW of England and Wales.

Figure 2 presents average soil C emissions for each of the four energy crops. It is apparent that, on average, under the perennials (*Miscanthus* and SRC), more C is sequestered or retained in the soil in comparison with the annuals (winter wheat and oilseed rape). We also note that, in general, soil tends to lose C in regions where there are high initial soil C stocks, e.g. more westerly regions, the Fens (the northern part of East Anglia), and to gain C (or be neutral for winter wheat and oilseed rape) in areas which have been under long-term arable production, e.g. South-East, Midlands.

This strong relationship between initial soil C stocks and net SOC balance is further evidenced in Fig. 3, where net annual emissions are plotted against initial SOC. The intersecting horizontal and vertical lines in each case show the average equilibrium soil C content over all simulated cells for each of the four energy crops. The vertical lines show the average current SOC stocks for each previous land use classification. The order of the SOC equilibria for the bioenergy crops from high to low SOC is SRC, *Miscanthus*, oilseed rape, and winter wheat, obtained by intersecting the displayed regression line with the average calculated inputs (inputs associated with the average yield for each crop over all cells). Although the estimated equilibria for both perennial crops are higher than the annuals, the relatively higher equilibrium SOC for SRC in these simulations is mainly due to the differences in calibration of the relative C input curves rather than any differences in prediction yields. The relative lack of spread about the trend-line for oilseed rape and winter wheat, compared with *Miscanthus* and SRC is due to the lesser degree of spatial variation in their yield maps.

For *Miscanthus* and SRC, the average SOC equilibria are between those for improved grassland and forest/seminatural, meaning soil C would tend to increase when replacing the former (and arable land) and decrease when replacing the latter. Although the equilibrium for oilseed rape corresponds closely to the average SOC in the arable soils, it is perhaps a little surprising that for winter wheat it is lower. We believe there are two likely reasons for this: Smith *et al*. (2005) previously observed that differences between soil C inputs for oilseed rape and winter wheat could probably be attributed to the additional breeding that winter wheat has historically received to improve the harvest index (sacrificing biomass that can be returned to the soil for biomass in the grain). This justified in part the parameterization of C inputs for these two crops. One may also observe that although we typically suppose arable soils to have quite low SOC levels, this is not always the case. As an example, the soils in the Fens have higher initial SOC levels, which biases the initial SOC upwards. To evaluate the reliability of our results, we checked our data using only the South-East and Midlands NUTS2 regions and found in this case that the arable equilibrium SOC was between that for winter wheat and oilseed rape as expected based on the above.

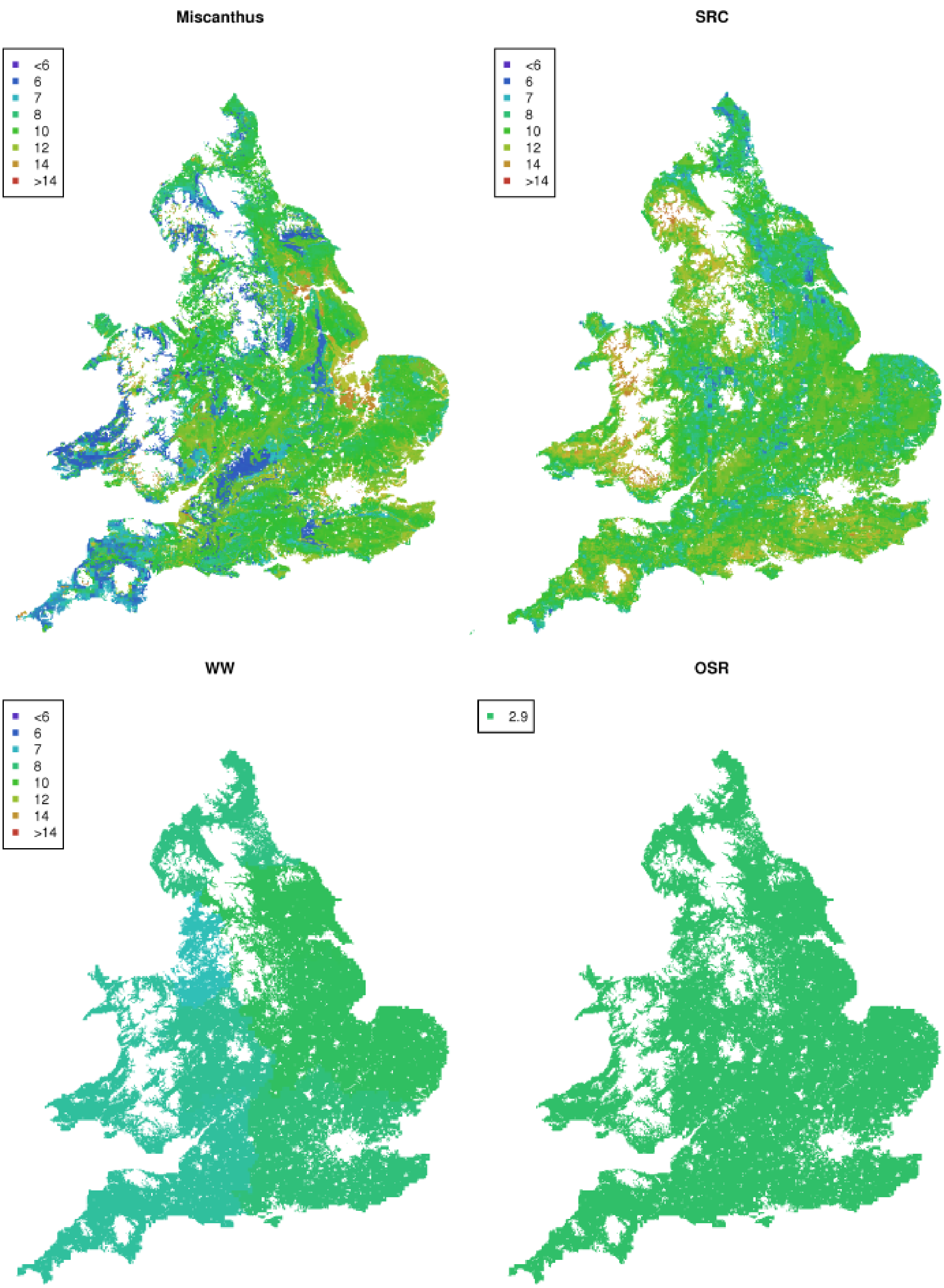

Fig. 1 Yield ($t\,ha^{-1}\,yr^{-1}$) maps for *Miscanthus*, short rotation coppice poplar, winter wheat, and oilseed rape.

One further observation that is reinforced by Figs 1–3 is that soil C changes are finite, with changes to SOC levels occurring until a new equilibrium is attained. For example replacing a hectare of 'average' grassland with *Miscanthus* is predicted to yield soil C gain until the *Miscanthus* average equilibrium level is reached. This

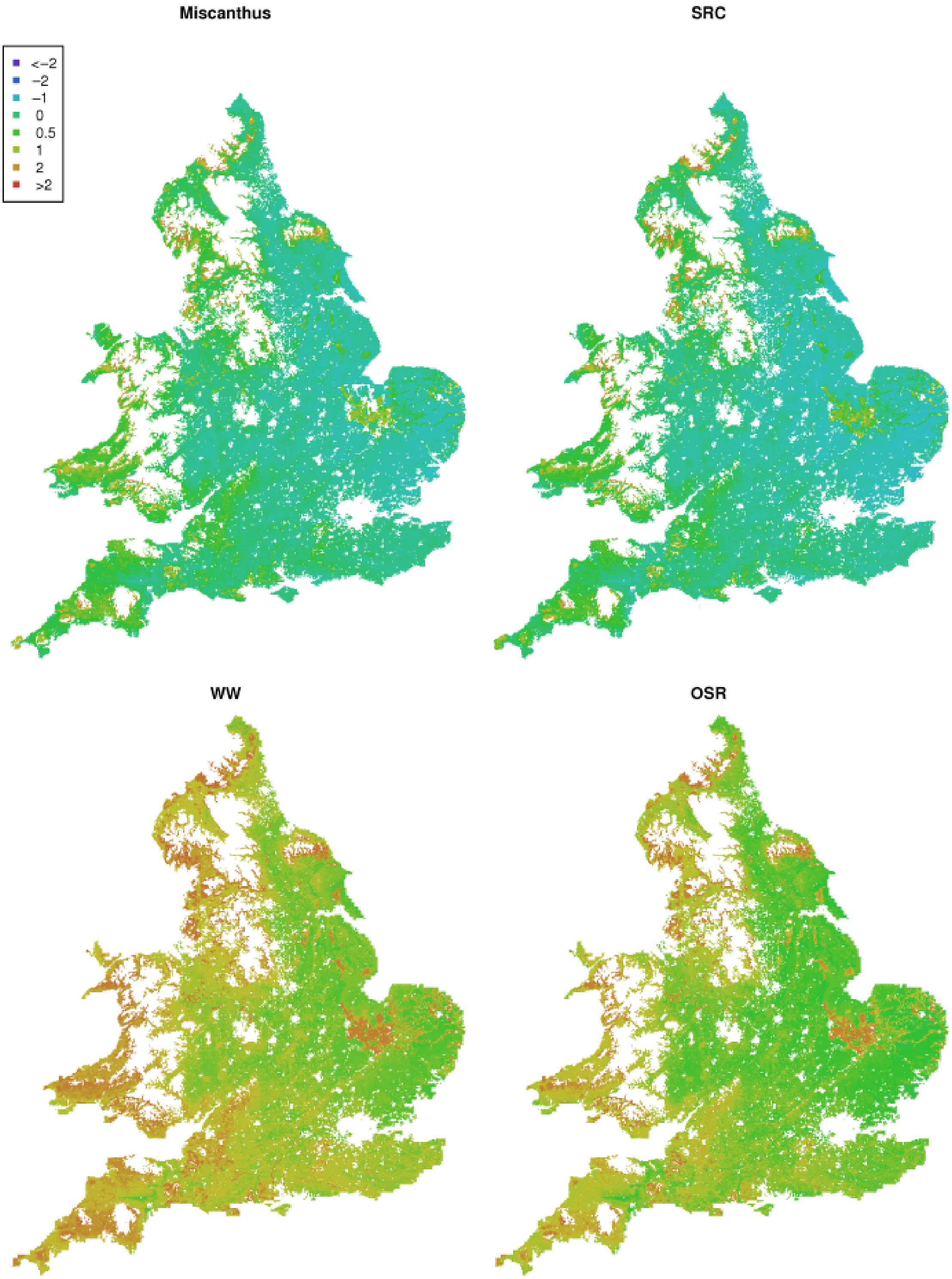

Fig. 2 Predicted soil emissions/sequestration (t CE ha^{-1} yr^{-1}) for *Miscanthus*, short rotation coppice poplar, winter wheat, and oilseed rape: Annualized 20-year averages from RothC.

will occur after around 15 t of C is sequestered. Once this has been achieved no more significant changes are expected to the soil C stocks.

Figure 4 displays the predicted annual C stock changes for each of the four bioenergy crops on each of the three land classes (black bars). Error bars for the

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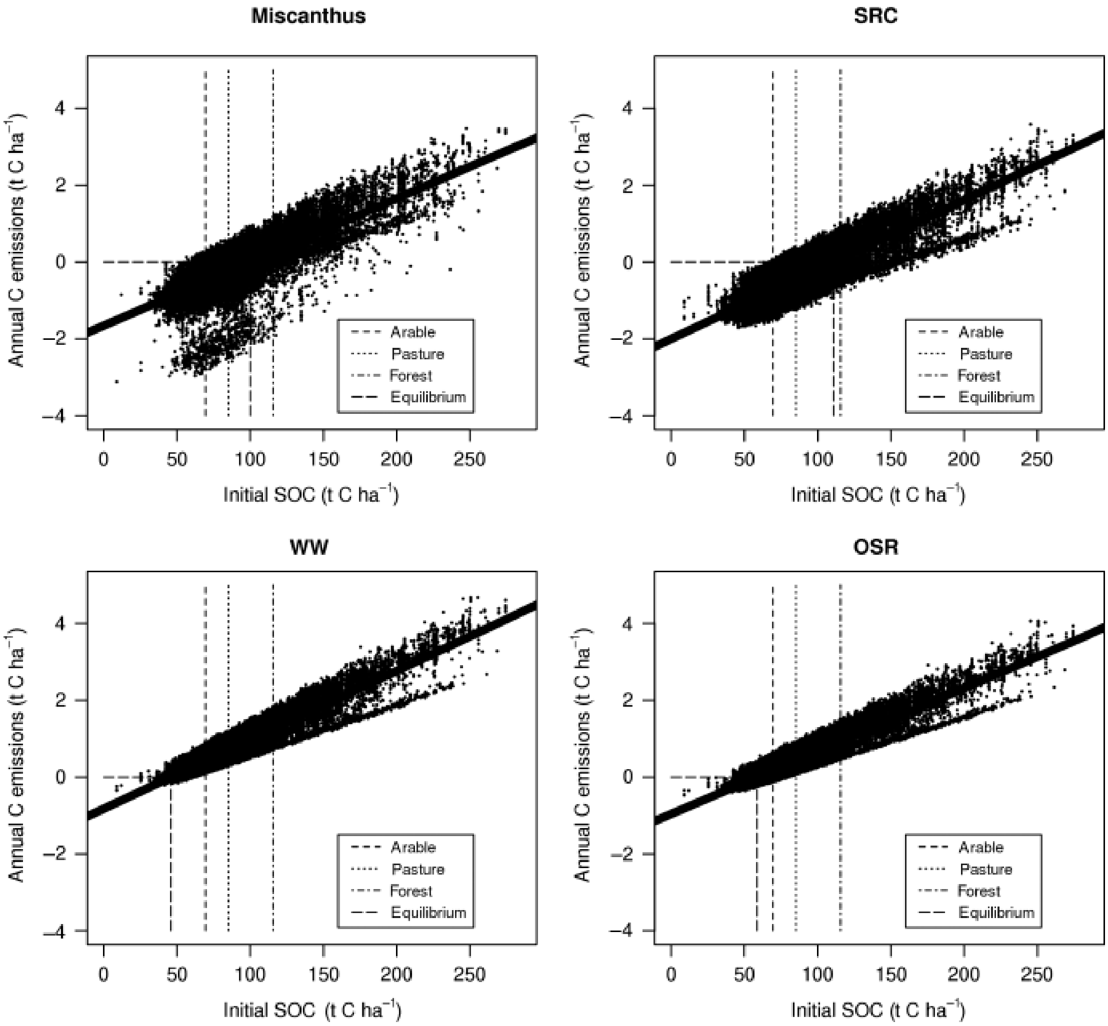

Fig. 3 Relationship between initial soil carbon (C) and emissions (t CE ha^{-1} yr^{-1}) for the four bioenergy crops. Vertical lines represent the average soil C for the three land classes arable, grassland, and forest/seminatural, from left to right, respectively. Intersecting horizontal and vertical lines with the regression relation represent the average equilibrium soil C for each bioenergy crop.

soil emissions represent ± 2 SD, so that if we selected a grid cell for a particular land use change at random there is an approximately 95% chance that the emissions would lie within the range shown. The dark grey bars show the effect of changes in management-related emissions, when added to the mean soil emissions/sequestration for each land use class to give a farm gate GHG balance estimate. For mean aboveground C sequestration rates of forest, we used a figure of 2.8 t ha^{-1} (Smith *et al*., 2000), to offset against any gains. In this case, the effects of the soil balance are reinforced;

- replacement of arable and grassland by perennial crops leads to GHG saving,
- replacement of forest or natural land is marginal at best and results in substantial net emissions for winter wheat and oilseed rape,
- replacement by oilseed rape and winter wheat of arable land is broadly neutral, and
- replacement of grassland by oilseed rape and winter wheat results in net emissions.

The main expected benefit of bioenergy crops is that they should displace C released from fossil fuel use. Under the assumption that *Miscanthus* and SRC would replace coal, winter wheat petrol (through bioethanol), and oilseed rape diesel (through biodiesel), these scenarios have been incorporated into Fig. 4 (light grey bars). In this case, some benefits are now expected for winter wheat replacing arable and grassland, since offsetting fossil fuel displaced tends to improve the GHG balance. Replacing forest/seminatural land with *Miscanthus* or SRC is also expected to be broadly neutral, since the coal displaced offsets any losses from soil,

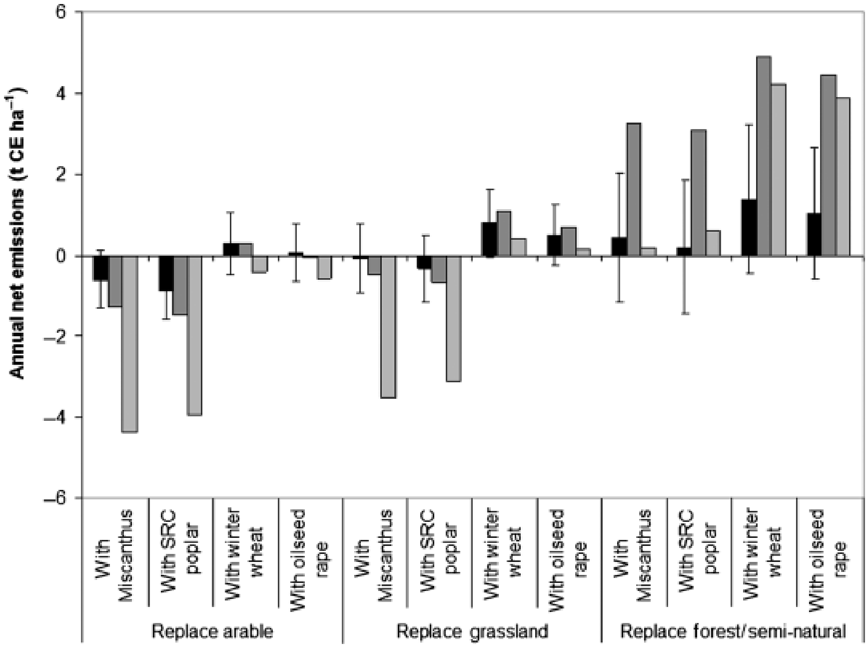

Fig. 4 Net annual greenhouse gas (t CE ha^{-1}) balance for all replacement scenarios. Black – soil emissions, error bars represent 2 SD. Grey – incorporating before and after management emissions. Light grey – incorporating fossil fuel displaced.

Table 3 Break even yields and upper and lower 95% confidence intervals for the 12 land use change scenarios

Replace	With	Break even crop yield (t ha^{-1})	Lower 95% confidence level	Upper 95% confidence level
Arable	*Miscanthus*	−3.8(+)	5.5	12.5
	SRC (poplar)	−5.8(+)	7.1	12.1
	Winter wheat	3.3(+)	7.1	8.4
	Oilseed rape	−0.2(+)	na	na
Grassland	Miscanthus	−1.4(+)	5.1	12.9
	SRC (poplar)	−2.5(+)	6.7	12.5
	Winter wheat	12.3(−)	7.1	8.4
	Oilseed rape	3.8(−)	na	na
Forest/seminatural	Miscanthus	9.5	4.5	13.5
	SRC (poplar)	12.0	6.0	13.2
	Winter wheat	55.0(−)	7.1	8.4
	Oilseed rape	24.0(−)	na	na

Negative break even yields indicate that the bioenergy crop is beneficial even before fossil fuel displaced is accounted for. (+) and (−) indicate that the bioenergy crop chain has positive or detrimental GHG impact respectively for 95% of modelled yields.
GHG, greenhouse gas; SRC, short rotation coppice.

management or previous aboveground sequestration in forests. There is now a clear expected benefit in the case of *Miscanthus* and SRC on existing arable and grassland, with expected GHG savings of around 4–5 t C ha^{-1} yr^{-1} and 3–4 t C ha^{-1}, respectively.

In Table 3 the break-even yields (the crop yield resulting in zero net emissions) are shown for each of the bioenergy crops on the three land use classes. Negative break-even yields indicate that even without consideration of fossil fuel C displaced, cultivation of the bioenergy crop has a beneficial GHG effect, as is the case for *Miscanthus* and SRC poplar replacing arable or grassland. In addition, when considering fossil fuel C displaced, winter wheat replacing arable results in expected GHG benefits for more than 95% of predicted yields. It is also expected that oilseed rape replacing arable would yield a net GHG benefit although, this is partly because the management emissions are slightly lower than for winter wheat (which was used as the baseline arable management). Also, since the yield map displayed no variation we were unable to construct meaningful confidence intervals in the case of oilseed rape. For winter wheat and oilseed rape replacing grassland, break-even would be achieved for yields of 12.3 and 3.8 t ha^{-1} respectively. It is probably unlikely that such high yields could be reliably achieved in practice, and indeed, any GHG benefits would be relatively small even is this were so. In both these cases, GHG benefits would have been predicted had soil emissions not been taken into account, which demonstrates the importance of incorporating this component. When replacing forest or seminatural land, break-even yields are above the average for each bioenergy crop, although only slightly so for *Miscanthus* (9.5 t ha^{-1} break-even compared with a mean of 9.0 t ha^{-1}). Indeed, for *Miscanthus* and SRC these values would appear to be achievable in many locations, so there may be cases where reasonable GHG benefits could be achieved. This indicates that there may be instances where replacing Forest or seminatural land with *Miscanthus* or SRC poplar can result in a GHG benefit, although, given the uncertainties in the model inputs, further investigation would be needed before this interpretation could be regarded as reliable. For winter wheat and oilseed rape, implausibly high yields of 55 and 24 t ha^{-1} would be required.

Discussion

The GHG balance of bioenergy crop production depends on crop yield (through its impact on soil C inputs, and fossil fuel displaced), on crop type (through differing yields and conversion routes), and on previous land use (through the relation with soil C stocks). In terms of crop yields the results are highly dependent on the conversion route for the biofuel, and also the calibration of C inputs to the soil as a function of crop yield. For the former, both *Miscanthus* and SRC poplar have significantly lower processing costs than winter wheat and oilseed rape and substantially higher final product yields which results in higher fossil fuel displacement. It should be noted however, that although the yield maps for winter wheat and oilseed rape are based on observed farm data, the yield maps for *Miscanthus* and SRC poplar use empirical models developed from limited datasets. Since the sensitivity of the overall GHG balance on crop yield is high the potential for error in the empirical models should be considered (additionally, extreme yield values were filtered from the analysis which acted to reduce the spread of outcomes explored). For example, for the *Miscanthus* model of Richter *et al*. (2008) the water availability in the soil (AWC) was a key driver of yield. In a recent re-evaluation of the empirical model, AWC from the NATMAP database was recalculated using a pedo-transfer function based on primary and hydrological data, which increased the yield estimate on average by 3 t ha^{-1} (Lovett *et al*., 2009). This indicates the uncertainty in model predictions, and the need for more thorough long-term monitoring of *Miscanthus* and SRC in order to reduce this uncertainty.

Previous land use is also highly important for prediction of the C balance for bioenergy crops, due to its impact on initial soil C stocks, and to emissions associated with cultivation. In the cases of *Miscanthus* and SRC, it can make the difference between the effect of bioenergy crop production being positive (e.g. replacing existing arable land), neutral, or even negative (seminatural land/forest). When using winter wheat and oilseed rape for bioenergy, the effect on the C balance changes from a neutral or marginal benefit (existing arable land), to significantly detrimental for the C balance of the whole bioenergy production chain (forest/seminatural). In the case of oilseed rape or winter wheat replacing grassland, accounting for soil emissions alters a GHG benefit in to a GHG cost. When the soil component is removed from the LCA, our results indicated that (when replacing rotational set-aside) both *Miscanthus* and SRC can offer savings up to 99%, over the equivalent fossil fuels – which after considering storage, transportation, and power plant maintenance would be reduced to about 98% and 92%, respectively, whereas BEAT v2, 2008, AEA Technology & North Energy Associates Ltd. Available at http://www.biomassenergycentre.org.uk/portal/page?_pageid=74,153193&_dad=portal&_schema=PORTAL (accessed 20 March 2009) identifies savings of 37% for bioethanol compared with petrol, and only 17% for biodiesel compared with diesel fuel.

The results suggest that, on average, replacement of both arable crop land and pasture land with *Miscanthus* or SRC should lead to overall annual increases in SOC. However, these results are highly dependent on the calibration of plant inputs to soil vs. yield. The calibrations have been performed from limited datasets, and required assumptions concerning the proportion of C inputs attributable to yield variation. Although for *Miscanthus* the measurements were taken over a 16-year period (Hansen *et al*., 2004), this was done for only 3 years under poplar (Hoosbeek *et al*., 2004; Gielen *et al*., 2005). The *Miscanthus* experiment showed most of the gain in SOC to have occurred in the 9–16-year interval with relatively little before that. The poplar dataset showed considerable variance, part of which the authors attributed to the 'priming effect' (the acceleration of soil C decomposition by fresh C input to soil, e.g. Fontaine *et al*., 2004). Also, since cultivation of grassland and forest/seminatural land results in an initial soil disturbance, there is a potentially significant early soil C loss, which we have not explicitly explored here (initial disturbances are integrated into the 20 years rotation); for example, Guo & Gifford (2002), reviewing soil stock changes under land use change found an average decrease in soil C levels when replacing pasture land with plantations. We note here that such an outcome lies within the confidence intervals we obtained, the variation in GHG balance which is due to variations in yield, initial soil C and clay content, and climatic conditions. There is a clear need for SOC balance data over a range of soil and climate conditions, lengths of crop production cycles, and for the long-term implications of reversion to arable (i.e. extra C stored while crop maintained).

There is generally better data available for winter wheat and oilseed rape (as well as less need for model predictions since long-term trends are better understood). Although it is unlikely that winter wheat or oilseed rape would be grown exclusively for periods of 20 years, the predicted SOC equilibria are close to the mean of the arable land class and thus we might expect all farm gate components to be quite representative of a standard arable rotation.

In the more extreme cases uncertainty in calibration of soil C inputs does not greatly influence the broad outcomes of the LCA. For the greatest savings, *Miscanthus* and SRC poplar on arable or grassland, or the greatest costs, winter wheat or oilseed rape on woodland/seminatural, the soil component forms a relatively small part of the overall savings or costs. In other cases, where uncertainty in the calibration of inputs or variation in soil emissions can significantly impact outcomes (winter wheat and oilseed rape on arable or grassland, *Miscanthus* and SRC poplar on forest/seminatural), further and close monitoring of yields and soil C stocks would be required before reliable predictions could be made.

Since, overall GHG balance is significantly dependent on (spatially variable) crop yield, as well as previous land use, careful consideration is needed for how bioenergy crop production is to be incorporated into the landscape. An additional consideration is that the best cases observed here involved the replacement of land which is currently used for food production – the scenario which is most likely to require displacement of food crop production onto noncultivated land. One potentially optimal strategy is to initially replace low productivity land currently under arable production with *Miscanthus* or SRC, since in comparison with high productivity arable land, this is less likely to result in food crop displacement. For example, although a full consequential LCA is required for rigorous examination of displacement effects, a recent scenario allocation study (Lovett *et al*., 2009) estimated that *Miscanthus* production of 350 000 ha – sufficient to meet the renewable obligation for biomass – could result in as little as 6% reduction of arable crops (mainly wheat and oilseed rape).

For the LCAs conducted here, the N_2O emissions model (IPCC Tier 1 method) is very simple, supposing that 1% of all N applied is emitted as N_2O, or equivalently, application of 100 kg of N results in N_2O emissions equivalent to around 296 (44/28) = 465 kg CE. When converted into C equivalents, the N_2O emissions form a substantial part of the overall C emissions for the bioenergy chain in the case of winter wheat and oilseed rape. Crutzen *et al*. (2007) observe that when considering the fate of all synthetically produced N, as much as 3–5% may eventually end up as atmospheric N_2O. If using such a figure instead of the 1% IPCC guideline, the lower N requirements of *Miscanthus* and SRC in comparison with winter wheat and oilseed rape mean that the differences observed in this study would be further increased. There are also more refined approaches to treating N_2O emissions, ranging from the empirical model of Bouwman *et al*. (2002) through to more complex, process-based models such a NEMIS (Henault & Germon, 2000) or DNDC (Li *et al*., 1994). However, the use of such models would introduce additional uncertainty into our results due to difficulties in finding appropriate driving data, and we wished to avoid this confounding factor in our analysis.

Aside from SOC balance, N use remains the most significant contributor to the preharvest C footprint for many arable crops, both through embodied C and N_2O emissions. According to the latest UK GHG inventory (Choudrie *et al*., 2008) around 75% of the UK's agriculture based emissions are from N fertilizer use. Similarly, Hillier *et al*. (2009), analysing a range of farm types and arable crops, estimated that around 75% of the farm gate emissions (and more than 95% of the variation in emissions) can be attributed to N use. This is true regardless of whether fertilizer is applied in the form of organic or inorganic N. As such, timing and control of N applications, use of fertilizers with nitrification inhibitors, together with strategies to incorporate N, e.g. N-fixing leguminous crops, into rotations, and controlled traffic farming (Vermeulen & Mosquera, 2009) remain important avenues of exploration for reducing the footprint of both food and bioenergy crops alike.

In contrast to oilseed rape and winter wheat, the *Miscanthus* and SRC crops did not receive mineral fertilizers in this LCA study. SRC was assumed to receive slurry applications during establishment and after harvesting, but in this instance the emissions were not attributed to the crop, since the slurry requires disposal and this could occur on any other land. Slurry could also be applied to *Miscanthus* if required, although current research suggests that it would not be, since negligible yield responses to fertilizers are observed (e.g. Christian *et al*., 2008). However, not enough is known about the site-specific nutrient requirements of *Miscanthus* or SRC and if the United Kingdom sees future expansion of energy crops, slurry may become a limited resource, meaning that mineral fertilizers may eventually be required. To partly explore this uncertainty, we repeated our analysis assuming application of $100\,kg\,ha^{-1}\,yr^{-1}$ of synthetic N for both *Miscanthus* and SRC. Once fertilizer production and soil N_2O emissions were taken into account the farm-gate footprint increased (in both cases) by around 700 kg $CO_2\,ha^{-1}$, which still implies substantial difference between these crops and winter wheat and oilseed rape, and leaves our main findings unaltered.

Emissions from machinery use are relatively small in comparison with those from the soil and from N fertilizer production and application, although for *Miscanthus* and SRC, which have lower N requirements, they are more significant. Improvements in yield would increase the fuel requirement of harvesting, effectively diluting the GHG saving achieved during the establishment, first year maintenance and site termination processes (when compared with winter wheat and oilseed rape). However, we would suggest that in respect to greenhouse gases it would be more beneficial to focus future work on further developing our understanding of bioenergy crop performance, soil C management and the role/fate of N fertilizer in the soil.

Conclusions

Spatial variation in yield and soil GHG emissions form significant components of the total emissions in a bioenergy crop LCA, and can make the difference between a bioenergy chain being beneficial or detrimental in terms of GHG balance. For the replacement scenarios considered here, the most important single factor affecting potential gains from biomass crop production is still final product yield (and when replacing forest land, this must be offset against previous above ground C sequestration rates). However, there is still relatively little known about the effect of production of *Miscanthus* and SRC on soil C stocks under various land replacement scenarios, and it is important to collect more comprehensive data on the above if accurate LCAs are to be constructed.

The results clearly indicate a distinction between the perennial crops, *Miscanthus* and SRC, and the annual crops, winter wheat and oilseed rape. This is due to differences in N requirements, soil C balance and the energy conversion route. There is considerable scope for regional and national optimization of food and energy supply chains, e.g. through different uses of wheat and oilseed rape coproducts, or different landscape deployment strategies. Future work should help provide accurate characterization and accounting of all components in the supply chain, which is important if reliable data is required for policy and decision making.

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