



A *Miscanthus* plantation can be carbon neutral without increasing soil carbon stocks

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Abstract

National governments and international organizations perceive bioenergy, from crops such as *Miscanthus*, to have an important role in mitigating greenhouse gas (GHG) emissions and combating climate change. In this research, we address three objectives aimed at reducing uncertainty regarding the climate change mitigation potential of commercial *Miscanthus* plantations in the United Kingdom: (i) to examine soil temperature and moisture as potential drivers of soil GHG emissions through four years of parallel measurements, (ii) to quantify carbon (C) dynamics associated with soil sequestration using regular measurements of topsoil (0–30 cm) C and the surface litter layer and (iii) to calculate a life cycle GHG budget using site-specific measurements, enabling the GHG intensity of *Miscanthus* used for electricity generation to be compared against coal and natural gas. Our results show that methane (CH₄) and nitrous oxide (N₂O) emissions contributed little to the overall GHG budget of *Miscanthus*, while soil respiration offset 30% of the crop's net aboveground C uptake. Temperature sensitivity of soil respiration was highest during crop growth and lowest during winter months. We observed no significant change in topsoil C or nitrogen stocks following 7 years of *Miscanthus* cultivation. The depth of litter did, however, increase significantly, stabilizing at approximately 7 tonnes dry biomass per hectare after 6 years. The cradle-to-farm gate GHG budget of this crop indicated a net removal of 24.5 t CO₂-eq ha⁻¹ yr⁻¹ from the atmosphere despite no detectable C sequestration in soils. When scaled up to consider the full life cycle, *Miscanthus* fared very well in comparison with coal and natural gas, suggesting considerable CO₂ offsetting per kWh generated. Although the comparison does not account for the land area requirements of the energy generated, *Miscanthus* used for electricity generation can make a significant contribution to climate change mitigation even when combusted in conventional steam turbine power plants.

Keywords: bioenergy, coal, decomposition, greenhouse gas, greenhouse gas intensity, life cycle assessment, litter, natural gas, net ecosystem exchange, soil C

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Introduction

Climate change is unlikely to be solved with a short-term solution, but alternative renewable fuel sources, like bioenergy, can be a part of the long-term solution. Therefore, it is essential to ensure these bioenergy crops are helping to turn atmospheric carbon dioxide (CO₂) into stable long-lived carbon (C) forms, rather than the reverse. As alternative energy sources, bioenergy crops and lignocellulosic feedstocks often fare well against conventional fuels in both socio-economic (Paine *et al.*, 1996; Domac *et al.*, 2005; Remedio & Domac, 2003) and

environmental (Cherubini *et al.*, 2009; Smeets *et al.*, 2009;) comparisons. The bioenergy crop, *Miscanthus x giganteus* Greef et Deu (Hodkinson & Renvoize, 2001) (herein *Miscanthus*), has attracted attention in North America and Europe due to high yields (Christian *et al.*, 2008; Heaton *et al.*, 2008), low management requirements (Miguez *et al.*, 2008; Gopalakrishnan *et al.*, 2011; McCalmont *et al.*, 2015) and the potential for improved soil C stocks (Hansen *et al.*, 2004; Schneckengerger & Kuz'yakov, 2007; Poeplau & Don, 2014). These characteristics make *Miscanthus* a particularly attractive crop in the light of climate change mitigation options (Hastings *et al.*, 2009; McBride *et al.*, 2011).

A key area of uncertainty when assessing the sustainability of bioenergy crops surrounds their potential to

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sequester more C in crop residues and soils than is emitted through production, transport and end-use processes of the harvested biomass. Quantifying the complete life cycle C budget of bioenergy plantations is therefore essential to accurately determine any potential GHG savings. This GHG mitigation potential is an important part of formal life cycle assessments (LCAs) for bioenergy crops that evaluate their environmental impact from cradle to grave (e.g. Adler *et al.*, 2007; Rowe *et al.*, 2011). To date, empirical measurements of the GHG balance of *Miscanthus* cultivation have produced inconsistent outcomes (Toma *et al.*, 2011; Drewer *et al.*, 2012; Zimmermann *et al.*, 2012; Poeplau & Don, 2014). As a consequence, GHG emissions data included in *Miscanthus* LCAs are often modelled (e.g. Hamelin *et al.*, 2012) or use IPCC default emission factors (e.g. Brandão *et al.*, 2011). To address this area of uncertainty, we focused on cultivation of *Miscanthus* from the cradle-to-farm gate to quantify the C sequestration potential of *Miscanthus*. For this, we measured four years of soil GHG emissions and net ecosystem exchange (NEE) from a 3- to 7-year-old commercial *Miscanthus* plantation in the United Kingdom, also measuring soil C stocks and accumulated plant litter.

Assessing the GHG budget of *Miscanthus* requires more than estimates of C assimilation through photosynthesis as soil C sequestration can offset a large proportion of GHG emissions from the field (Lal, 2004). Temperature (Kirschbaum, 1995) and water availability (Orchard & Cook, 1983; Wood *et al.*, 2013) are both major drivers of the microbial processes that incorporate C into soils. Further, the 'quality' of plant litter (quantified by C : N ratios or lignin : N ratios) can influence how quickly that C is decomposed (Taylor *et al.*, 1989; Donnelly *et al.*, 1990; Bonanomi *et al.*, 2013). Consequently, it is important to consider these factors when evaluating soil C sequestration. Senesced *Miscanthus* biomass is typically very low in N due to nutrient translocation. This results in low litter quality (Amougou *et al.*, 2011) which has a significant impact on the rate of C turnover from the litter layer into the topsoil (Cadoux *et al.*, 2012). Root decomposition also contributes to soil C sequestration, but *Miscanthus*-specific data are limited to a few studies (Rasse *et al.*, 2005; Agostini *et al.*, 2015). The majority (>50%) of belowground biomass is found in the top 30 cm (Neukirchen *et al.*, 1999; Amougou *et al.*, 2011), with C inputs from roots and rhizomes estimated to be as high as 0.86 tC ha⁻¹ yr⁻¹ and 2.66 tC ha⁻¹ yr⁻¹, respectively (Agostini *et al.*, 2015). However, a recent study suggests that rhizosphere activity under *Miscanthus* may stimulate priming, causing a loss of native soil C and offsetting fresh C inputs (Zatta *et al.*, 2014). Long-term studies are therefore required to assess litter accumulation,

belowground biomass and soil C stock changes in *Miscanthus* plantations, in order to quantify its benefits for climate change mitigation (Poeplau & Don, 2014; Robertson *et al.*, 2015).

While C stocks in litter, standing biomass and soils are important 'pools' to quantify, their changes over time are relatively slow compared to the 'fluxes' of the system that include photosynthesis and respiration (Kuzyakov, 2011). These processes continually respond to environmental conditions and often follow diurnal patterns strongly influenced by crop physiology (Linn & Doran, 1984; Rochette *et al.*, 1999; Cheng *et al.*, 2003). At the ecosystem scale, the balance between C uptake and CO₂ efflux is described as the NEE, and within the C cycle, this is the largest flux between atmosphere and a bioenergy plantation. NEE is typically calculated using eddy covariance to continuously monitor changes in CO₂ concentration above the plantation canopy (Baldocchi, 2003). Although the C stored in aboveground biomass is often quantified for bioenergy crops when they are harvested, measurements of the NEE are required to ensure that the amount stored in pools is in excess of the amount emitted through fluxes.

In many agricultural systems, CO₂ is not the only GHG of importance with nitrous oxide (N₂O) emissions often contributing more to a crop's overall GHG balance than the NEE (Flessa *et al.*, 2002). Despite established measurement techniques, relatively few studies have measured soil GHG emissions from *Miscanthus* plantations. The limited data available show that emissions of both N₂O and methane (CH₄) from soils are low and CO₂ efflux dominates soil GHG emissions (Toma *et al.*, 2011; Drewer *et al.*, 2012; Gauder *et al.*, 2012). To accurately quantify an average annual efflux of these GHGs, data are required throughout the year and ideally over several years. In this study we measured GHG emissions and NEE in a *Miscanthus* plantation in Lincolnshire, UK, from 2009 to 2013 (growth years 3 to 7). We then used parallel measurements of climatic variables to explore the environmental controls on soil respiration (CO₂), CH₄ and N₂O emissions, including the temperature sensitivity of respiration at different stages in the crops growth cycle. The aims of the study were to quantify the relative contributions of each GHG towards the net GHG balance of the site, and to better understand their relationship to temperature and soil moisture as environmental drivers. CO₂ was expected to dominate site GHG fluxes, with warmer and wetter periods driving the greatest soil respiration rates. In addition, changes in soil C stocks and the litter layer were quantified over time, with the expectation that the dynamics of these C pools are largely responsible for sequestration rates reported for *Miscanthus* (e.g. Dondini *et al.*, 2009). These data were then used to calculate a life

cycle GHG balance of *Miscanthus* cultivation in order to compare *Miscanthus* as a source of electricity to coal and gas.

Materials and methods

Study site

The field experiment was conducted in an 11.5-ha commercial *Miscanthus* plantation near Lincoln, Lincolnshire, UK. The soil type is a compacted loam that behaved like a heavy clay, with approximately 15 %, 36 % and 49 % of clay, silt and sand, respectively, in the top 30 cm of soil. The top 30 cm of soil had a mean total C and N concentration of 1.86 % and 0.18 %, respectively, with a soil pH ranging from 6.8 to 7.3. The bulk density of the soil was $1.46 \pm 0.03 \text{ g cm}^{-3}$ for the 0- to 15-cm layer and $1.53 \pm 0.02 \text{ g cm}^{-3}$ for the 15- to 30-cm soil layer. Root biomass (live and dead) was estimated at the end of the 7th growth year: 2.61 t dry mass ha^{-1} for 0–15 cm and 1.85 t dry mass ha^{-1} for 15–30 cm. Additional soil characteristics sampled monthly for two years within this study can be found in Table S1. The deeper soil profile showed an increasing bulk density ($1.59 \pm 0.20 \text{ g cm}^{-3}$, 30–50 cm; $1.62 \pm 0.10 \text{ g cm}^{-3}$, 50–100 cm) and a clear B-horizon at the plough depth (30 cm). There was little evidence of root biomass propagation below 70 cm when trenches were dug in early 2009. The site had a mean annual precipitation of 605 mm and a mean annual temperature of 9.9 °C (30-year average 1980–2009). The *Miscanthus* was established in 2006 at a density of 10 000 rhizomes ha^{-1} . The crop was harvested annually in the spring, beginning in March 2008, but biomass was only removed from 2009 onwards; bale yields (20% moisture content) were recorded as 6.95, 10.28, 6.24, 7.58 and 6.87 dry t ha^{-1} for 2009 to 2013, inclusive. The only addition of fertilizer was in April 2010, when a phosphorus–potassium fertilizer was applied at a rate of 125 kg ha^{-1} . The land management prior to conversion to *Miscanthus* was a crop rotation of wheat and oilseed rape, with three years of wheat directly before conversion. Further site details can be found in Robertson *et al.*, 2016.

Sampling strategy and eddy covariance

In early May 2008 a meteorological tower was established in the north east corner of the *Miscanthus* plantation, along with a flux mast positioned to maximize CO_2 measurements given prevailing winds over the cropped area. The tower and mast were equipped with a number of devices to continuously (every 30 min) monitor a range of environmental conditions (Table S3), including an ultrasonic anemometer and infrared gas analyser (IRGA) to employ an eddy covariance (EC) system to examine NEE (more details can be found in S.1). Measurements were taken from 7 May 2008 until 10 March 2013 with some exceptions around the harvesting times where instrumentation was removed. NEE data were cumulated for each growth year (March to February) and an average taken over the four full years of measurements (March 2009 to February 2013), reported in $\text{g CO}_2\text{-C m}^2$.

Soil–atmosphere gas fluxes

Measurements of soil GHGs (CO_2 , CH_4 and N_2O) were taken from October 2008 until March 2013 using the static chamber method described by Livingston & Hutchinson (1995), adapted to include the use of a pressure ‘vent’. Five chambers made from PVC (40 cm diameter and 20 cm height) were inserted approximately 3 cm into the soil surface (exact volumes noted). This avoided severing many of the fine roots that were found very close to the soil surface (similar strategies have been recommended in different land uses by Heinemeyer *et al.*, 2011 and Mills *et al.*, 2011). All chambers remained in the soil except at harvest times. Chambers were replaced in the same approximate location after each harvest, with proximity to plants taken into consideration, aiming to represent the average spacing throughout the plantation. At the exact time of GHG sampling, and near the location of GHG sampling, volumetric soil moisture (0–6 cm depth) was measured using a ML2× Theta Probe and Meter HH2 (Delta T Devices, UK) as well as soil (0–7 cm depth) and air temperature measurements using a Tiny Tag temperature logger with integral stab probe (Gemini Data Loggers, UK). Measurements were not taken between December 2010 and April 2011 or in April 2012 due to funding constraints and harvest activities, respectively.

At times of sampling, chambers were closed with a reflective aluminium lid, which had a rubber seal around the edge to prevent leakage. Chambers were enclosed for 30 min with one 10-ml sample taken every 10 min for a total of four time points collected for each plot. At the time of sampling, gas samples were transferred from the chamber headspace into a 3-ml gastight exetainer (Labco Ltd, Lampeter, UK) via a needle and syringe inserted into the self-sealing septa in the chamber lid. The majority (>85%) of GHG measurements were taken between the hours of 10:30 and 14:30 with some exceptions due to field logistics. Exetainer gas samples were analysed on a Perkin-Elmer Autosystem XL Gas Chromatograph (GC) fitted with a flame ionization detector (FID) for CO_2 and CH_4 and an electron capture detector (ECD) for N_2O . All results were calibrated against certified gas standards (BOC, UK) (Case *et al.*, 2014) and converted to a total flux reported as $\text{mg CO}_2\text{-C m}^{-2} \text{ h}^{-1}$, $\mu\text{g CH}_4\text{-C m}^{-2} \text{ h}^{-1}$ or $\mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$ in accordance with methods detailed in Holland *et al.* (1999).

Carbon and nitrogen in soil, vegetation and litter

In parallel with monthly GHG measurements, soil samples were collected using PVC pipes (5 cm internal diameter) hammered into the topsoil (0–15 cm) from five locations, one each within a 10 m radius of the static chambers. These cores were taken in March 2009 and March 2010 and then at monthly intervals from May 2011. Further, in October 2011, May 2012, October 2012 and March 2013 additional 30-cm-depth cores (split into 0- to 15-cm and 15- to 30-cm layers) were taken using a 2.5-cm-diameter gouge auger (Van Walt, Haslemere, UK). All soil collected was for destructive sampling and used for C and N determination. The routine monthly 0–15 cm cores were homogenized and freeze-dried (Alpha 1-4 LD, Martin Christ, Osterode am Harz, Germany) before being gently ground by

hand to pass through a 2 mm sieve. The 0–30 cm cores were air-dried to constant weight at room temperature before being homogenized, ground and sieved. No differences in C or N concentration were seen between the freeze-dried and air-dried samples. All visible plant matter remains (e.g. roots and leaf litter) were removed before grinding. Small subsamples of the ground soil were taken for analysis of C and N concentration through combustion in an elemental analyser (Costech ECS 4010; Milan, Italy). C and N stocks were estimated by relation to fixed site bulk densities (1.46 for 0–15 cm and 1.53 for 15–30 cm) and the depth layer (Guo & Gifford, 2002). These bulk densities were taken from 15 replicates using a 4.8-cm-diameter, 40-cm-deep split-tube sampler (Eijkelkamp Agrisearch Equipment BV, Giesbeek, the Netherlands). Care was taken to avoid compaction during coring and, where necessary, bulk density was corrected for compression based on the depth of the hole. To ensure consistency when calculating C and N stocks, the resulting bulk density for 0–15 cm was checked against the PVC cores taken monthly.

In October 2011, an adjacent field was sampled to provide an estimate of soil conditions before the *Miscanthus* was planted (a paired-site approach). This allowed a comparison to be made where samples from the adjacent field represent time-zero reference values of soil C and N stocks. This field had followed the same land use as the *Miscanthus* field prior to planting in 2006, was seeded with oil seed rape in 2006 and 2010, and winter wheat all other years. Before sampling in 2011, it had recently been harvested for winter wheat before being ploughed and cultivated again. Three replicates at five random locations were cored using the same split-tube sampler (Eijkelkamp Agrisearch Equipment BV, Giesbeek, the Netherlands) and split into 0–15 cm and 15–30 cm ($n = 15$). The soil was then freeze-dried, sieved to 2 mm and analysed for C and N. The same procedure to remove plant matter remains from the soil samples was applied. Further, these cores were analysed for bulk density and corrected for compression through coring (0–15 cm, $1.13 \pm 0.17 \text{ g cm}^{-3}$; 15–30 cm, $1.41 \pm 0.15 \text{ g cm}^{-3}$). C and N stocks were calculated using the field-specific bulk density values. No carbonates were detected at either depth from either field.

Between October 2008 and March 2013 senesced above-ground biomass was collected using twenty five 2-m² litter traps. Traps were placed on top of the litter layer throughout the plantation, with senesced biomass collected and weighed on a monthly basis and values extrapolated to an average rate per hectare. Subsamples of the senesced biomass were weighed and returned to the laboratory for moisture content determination (oven-dried at 105 °C for 24 h). The resulting dried subsample was then ground by freeze-milling (6770 Freezer/Mill, SPEX SamplePrep, Stanmore, UK) before C and N concentrations were determined. The amount of biomass added to the litter layer after harvesting, termed harvesting inefficiency, was also quantified by measuring the size of the litter layer before and after harvest. This varied between years but was proportional to the aboveground yield. Using an average of the measurements taken, a standard value of 5% of the year's harvest was used in future calculations (this value was similar to that reported by Sanderson *et al.*, 1997).

After harvesting (in May 2011, March 2012 and March 2013), the size (t ha⁻¹) of the litter layer was quantified by collecting all of the O-horizon (lightly raked from the soil surface) from 1.6 m² circles at 25 random locations throughout the plantation before extrapolating to a per area average for the site (after moisture content was determined by drying in an oven at 105 °C until constant weight, ~24 h). Additionally, the litter layer was quantified at 15 locations at six time points between March 2012 and March 2013 (May, June, August, September, October and January). Subsamples of the litter layer were dried, milled and analysed for C and N concentration. The decomposition rate of this litter layer was assessed assuming first-order decay rates as per Olson (1963), deriving a constant (k) to match a line of best fit through measured litter layer points. This constant was compared to two other studies for *Miscanthus* litter, Amougou *et al.* (2012) and Yamane & Sato (1975) who reported k values of 0.776 and 0.511, respectively.

Finally, standing biomass was partially harvested in October 2012 and March 2013 to assess C and N concentrations at the beginning and end of crop senescence. Nine stems were selected at random from different plants. Stems and leaves were separated, weighed and dried at 105 °C until constant weight (~24 h) to calculate moisture content. Dry biomass was then freeze-milled and analysed for C and N concentrations. All C and N concentrations were determined using the same elemental combustion analyser (Costech ECS 4010).

Site-specific life cycle GHG balance

To assess the contribution of site GHG emissions and changes in C stocks to the life cycle GHG balance of *Miscanthus*, an annual budget was calculated taking into account soil GHG fluxes, NEE and topsoil C stocks (0–30 cm). The mean annual NEE was used for net CO₂ emissions and cumulative annual CH₄ and N₂O emissions were derived from chamber fluxes using monthly data from the four years. CO₂ chamber data refer to soil emissions only and were not used in life cycle estimates. CH₄ and N₂O cumulative annual emissions were transformed using 100-year global warming potentials (GWPs), calculated as CO₂ equivalents (CO₂-eq) according to Myhre *et al.* (2013) (CH₄ = 34; N₂O = 298). The cradle-to-farm gate GHG balance was presented as an annual GHG balance per unit area extrapolated to the end of the plantation lifetime. This assumed an 18 year lifecycle of the plantation (DEFRA, 2007) and followed conventional cultivation routines (Table S4) including ploughing before planting as well as at the end of the plantation lifetime to prepare the site for the next crop (Styles & Jones, 2007; Thornley *et al.*, 2009). Direct and indirect emissions associated with other site operations were considered according to *Miscanthus*-specific estimates of diesel requirements reported by Lewandowski *et al.* (1995), Smeets *et al.* (2009) and Thornley *et al.* (2009).

Applying an assumed 20% moisture content of *Miscanthus* biomass when harvested and combusted (Lewandowski *et al.*, 2000) a realized calorific value (lower heating value (LHV)) of 14 MJ kg⁻¹ (ECN, 2015) was used to estimate GHG intensity. Additionally, a lifetime harvested yield from the plantation was estimated to be 129.2 tonnes dry biomass ha⁻¹ (Table S5).

In accordance with the common observation that productivity declines as a *Miscanthus* stand ages (Clifton-Brown *et al.*, 2007; Angelini *et al.*, 2009; Arundale *et al.*, 2014), the findings of Lesur *et al.* (2013) were applied to decrease yields proportional to stand age. Lesur *et al.* (2013) observed a maximum yield of 16.8 dry t ha⁻¹ in year 8 and a decrease of 0.647 dry t ha⁻¹ in each subsequent year. This reported maximum yield seems unrealistic at our site; therefore, the highest observed yield (10.28 dry t ha⁻¹ in 2010) was assumed to be the site-specific maximum. Consequently, this is approximately 49% of that reported by Lesur *et al.* (2013) and so the rate of yield decline is scaled accordingly (0.396 dry t ha⁻¹ yr⁻¹). The resulting lifetime plantation yield (129.2 dry t ha⁻¹) compares well with the alternative approach (121.3 dry t ha⁻¹) to average measured yields of the first seven years and assume that average is stable over the plantation's lifetime (Table S5). It is important to note that in other areas of the world the harvested biomass may have a lower moisture content (Heaton, 2006), thereby incurring an increased LHV.

The final cradle-to-grave GHG balance was estimated for the *Miscanthus* plantation and reported using the standard notation of emissions per unit of energy generated (GHG intensity; g CO_{2-eq} kWh⁻¹). This calculation was divided into three procedures (combustion, transportation and production) and was based on a number of informed assumptions. The *Miscanthus* biomass was assumed to be cofired for electricity generation in conventional steam turbine power stations where conversion efficiency of this solid biomass was 30% (1 MJ biomass = 0.30 MJ electricity) (Howes *et al.*, 2002). Although the conversion rate efficiency of biomass to energy can be considerably higher in combined heat and power (CHP) plants (~70%; Cannell, 2003), conventional electricity generation was employed to estimate the most realistic current scenario when comparing with traditional fossil fuels. This resulted in a GHG intensity associated with combustion as defined by Eqn (1).

$$\text{GHG}_{\text{com}} = \frac{Y \times C_{\text{conc}} \times \frac{\text{CO}_{2\text{mol}}}{C_{\text{mol}}}}{\left(\frac{Y \times \text{Cal} \times \text{Eff}}{E_{\text{conv}}}\right)} \quad (1)$$

where GHG_{com} is the GHG intensity of *Miscanthus* combustion for electricity generation in g CO_{2-eq} kWh⁻¹; *Y* is the harvested yield in g biomass ha⁻¹ at an assumed 20% moisture content (i.e. 129200000 over this plantation's lifetime); *C*_{conc} is the carbon concentration of the harvested biomass as a fraction (0 to 1); *CO*_{2 mol} is the molecular mass of CO₂; *C*_{mol} is the molecular mass of carbon; *Cal* is the calorific content of *Miscanthus* in MJ g biomass⁻¹ (i.e. a LHV of 0.014 given an assumed 20% moisture content); *Eff* is the conversion rate efficiency in power stations as a fraction (i.e. 0.30); and *E*_{conv} is the energy conversion from MJ to kWh (i.e. 0.278 as 1 kWh = 3.6 MJ; Thompson & Taylor, 2008). The GHG intensity associated with transporting the biomass to a power plant assumed a 160-km round trip (based on the location of local power plants) using a vehicle averaging 2.44 km per litre of diesel while carrying the equivalent of 25 tonnes biomass (NAP, 2010). Total GHG emissions of using 1 l of diesel to transport over land were assumed to be 3644 g CO_{2-eq} (Smeets *et al.*, 2009). This resulted in Eqn (2).

$$\text{GHG}_{\text{trans}} = \frac{\left(\frac{\text{PP}_{\text{dist}}}{F_{\text{eff}}} \times F_{\text{emi}}\right)}{\left(\frac{L \times \text{Cal} \times \text{Eff}}{E_{\text{conv}}}\right)} \quad (2)$$

where GHG_{trans} is the GHG intensity of *Miscanthus* biomass being transported between the plantation and a power station in g CO_{2-eq} kWh⁻¹; *PP*_{dist} is the round trip distance to the power station (i.e. 160 km); *F*_{eff} is the fuel efficiency of the truck used in transportation (i.e. 0.41 l km⁻¹); *F*_{emi} is the truck emissions associated with 1 l of fuel used during transportation (i.e. 3644 g CO_{2-eq} l⁻¹); and *L* is the truck load of biomass (i.e. 250 000 000 g). Finally, the GHG intensity of cradle-to-farm gate production was calculated using Eqn (3).

$$\text{GHG}_{\text{prod}} = \frac{(\text{GHG}_{\text{site}} \times P_{\text{life}})}{\left(\frac{Y \times \text{Cal} \times \text{Eff}}{E_{\text{conv}}}\right)} \quad (3)$$

where GHG_{prod} is the GHG intensity in g CO_{2-eq} kWh⁻¹ of *Miscanthus* biomass being grown and harvested including; GHG_{site} is the GHG balance in g CO_{2-eq} ha⁻¹ yr⁻¹ of all direct and indirect emissions, using NEE to estimate CO₂ exchange as well as CH₄ and N₂O measurements at the soil surface; and *P*_{life} is the plantation lifetime in years (i.e. 18). Ultimately, the sum of these three procedures were compared to full life cycle GHG budgets for coal and natural gas when used for electricity generation, as derived from MacKay & Stone (2013).

Statistical analysis

Outliers of GHG measurements were excluded when outside 2× standard deviation, as per Altman & Bland (1995), assuming normal distribution between all measurements of each gas at each time point, thereby retaining 95% of the data. All statistical analyses were performed with R version 3.0.2 (R Core Team, 2014). A global model was formed to define relationships between GHG fluxes and environmental variables (soil temperature, soil moisture, crop phase and a soil temperature * soil moisture interaction). User-defined growth phases of the crops were used to specify whether the crop was dormant (D), emerging (E) or growing (G). These each referred to four months of the year (November to February, March to June and July to October, respectively); the phases were found to be a significantly better predictor of CO₂ efflux than the traditional spring–summer–autumn–winter divisions.

Regression analysis was used to quantify the variance in GHG emissions explained by each of the environmental variables through use of the lme function as part of the nlme package (Pinheiro *et al.*, 2013) and the r.squaredGLMM function, part of the MUMIN package (Bartoń, 2012). To meet the assumptions of linear mixed effects (LME) models, log transformations to the flux data were required for soil CO₂ emissions and residuals were transformed using the varPower function (in nlme) for CH₄ and N₂O fluxes. Each chamber was used as the random effect to account for repeated sampling from the same location. This allowed estimates of how much variation in the measurements was explained by the different environmental factors.

Relationships of soil GHG emissions with soil temperature and soil moisture were explored in detail. The temperature sensitivity of CO₂ fluxes was determined as per Raich & Potter (1995) and Luo *et al.* (2001) to estimate a Q₁₀ value associated with the relationship, defined as the relative change in CO₂ flux given a 10 °C rise in temperature. This followed a nonlinear (exponential) relationship and applied the nls function as part of the base stats package within R, reporting an associated *P* value to describe the closeness of the defined relationship and data points. Further, because the goodness-of-fit *r*² metric is not as statistically robust for nonlinear relationships (Spiess & Neumeier, 2010), these are not reported and instead a Q₁₀ value was calculated for each chamber individually, and therefore, a standard error could be applied to the average. These relationships were defined for both monthly averages and the full data set. This was done for two reasons: (i) to reduce bias where more measurements were taken in some certain crop phases and (ii) to assess how a few measurements at extreme temperatures influenced Q₁₀ values. To test which relationship (monthly vs. all data) best described the temperature sensitivity a generalized additive model (GAM) approach was applied using the gam formula in the MGCV package (Wood, 2011). The resulting nls relationships were compared using the ANOVA function as part of the base stats package within R.

To compare the difference in chamber GHG measurements between temporal groups (days, months, phases or years), repeated-measures analysis of variance (ANOVA) was used applying the aov function as part of the base stats package in R. Where the assumptions of ANOVAs could not be met, residuals were transformed using either the varPower or varExp function as described earlier. The transformed (modelled) data were then analysed using the lme function with chamber as the random effect. This provided significance levels (i.e. *P*-values) to the tests performed.

Results

Climatic conditions and net ecosystem exchange

Continuous half-hourly measurements of air and soil temperature showed clear seasonal trends with annual means (9.60 and 9.55 °C, respectively) in line with 30-year averages (Fig. S1). While precipitation was distributed relatively evenly over the whole measurement period, on average March had the least rainfall (16.68 mm; 0.54 mm day⁻¹) and November had the most (70.60 mm; 2.35 mm day⁻¹). Both soil temperature and precipitation saw notable interannual variation with highs and lows in growth years 6 (9.86 °C) and 5 (8.91 °C) and in years 7 (818 mm) and 6 (405 mm), respectively (Table S2; Fig. S2). Mean NEE over the four full growing seasons was -678.08 ± 110.70 g CO₂-C m⁻² yr⁻¹ with more days between frosts in 2010 leading to the greatest uptake during this year. The large standard deviation reflects the notable interannual variation.

Soil GHG emissions and environmental controls on soil respiration

Soil fluxes of CH₄ and N₂O were largely negligible, with no discernible temporal trends and no clear relationships to environmental variables (Fig. 1). Using linear integration to cumulate average monthly fluxes to annual totals, CH₄ and N₂O emissions were found to be the same weight, totalling 0.38 kg CH₄-C ha⁻¹ yr⁻¹ and 0.38 kg N₂O-N ha⁻¹ yr⁻¹, respectively. In the case of N₂O emissions, only the fluxes in June 2010 were significantly different from zero and therefore contributed largely to the cumulative annual average.

Soil CO₂ emissions were significantly higher than those of CH₄ and N₂O, contributing 3.00 ± 0.22 t CO₂-C ha⁻¹ yr⁻¹. Emissions throughout the year followed a clear seasonal trend with highest emissions during the crops growth phase when soil temperatures were warmer; the lowest emissions were seen during the dormant crop phase when temperatures were cooler (Table 1). The climatic variables of temperature and precipitation explained the differences between years, with particularly warm and dry periods during measurements taken in June and September 2009 responsible for high cumulative totals in growth year 4. The highest single measurement (283 mg CO₂-C m⁻² h⁻¹) was observed in September 2009 and the lowest (0.83 mg CO₂-C m⁻² h⁻¹) in January 2013 (Fig. 2).

Using either all available data points or monthly averages, soil respiration correlated well with both soil temperature and soil moisture (GAM results for all correlations *P* < 0.01) (Fig. 3). Using nonlinear regressions for each block of chambers, mean Q₁₀ values and standard errors were derived using both monthly average data and the full data set (Table 2). In all cases soil respiration was most sensitive to temperature during the crop growth phase and least sensitive during the dormant crop phase, when average temperatures were highest and lowest, respectively. ANOVA results showed the uncertainty of these Q₁₀ values was lower (*P* = 0.009) when monthly averages were used in place of the full data sets.

Less than 5% of the variance observed for CH₄ or N₂O fluxes was explained by any of the environmental variables studied (Table 3). However, the same variables explained far more variation in chamber CO₂ fluxes; soil temperature explained more than half of the variance seen in soil respiration throughout the 4-year measurement period.

Carbon and nitrogen stocks

The paired-site proxy used as a 'time-zero' indicated that there was no temporal difference (*P* > 0.05) in soil

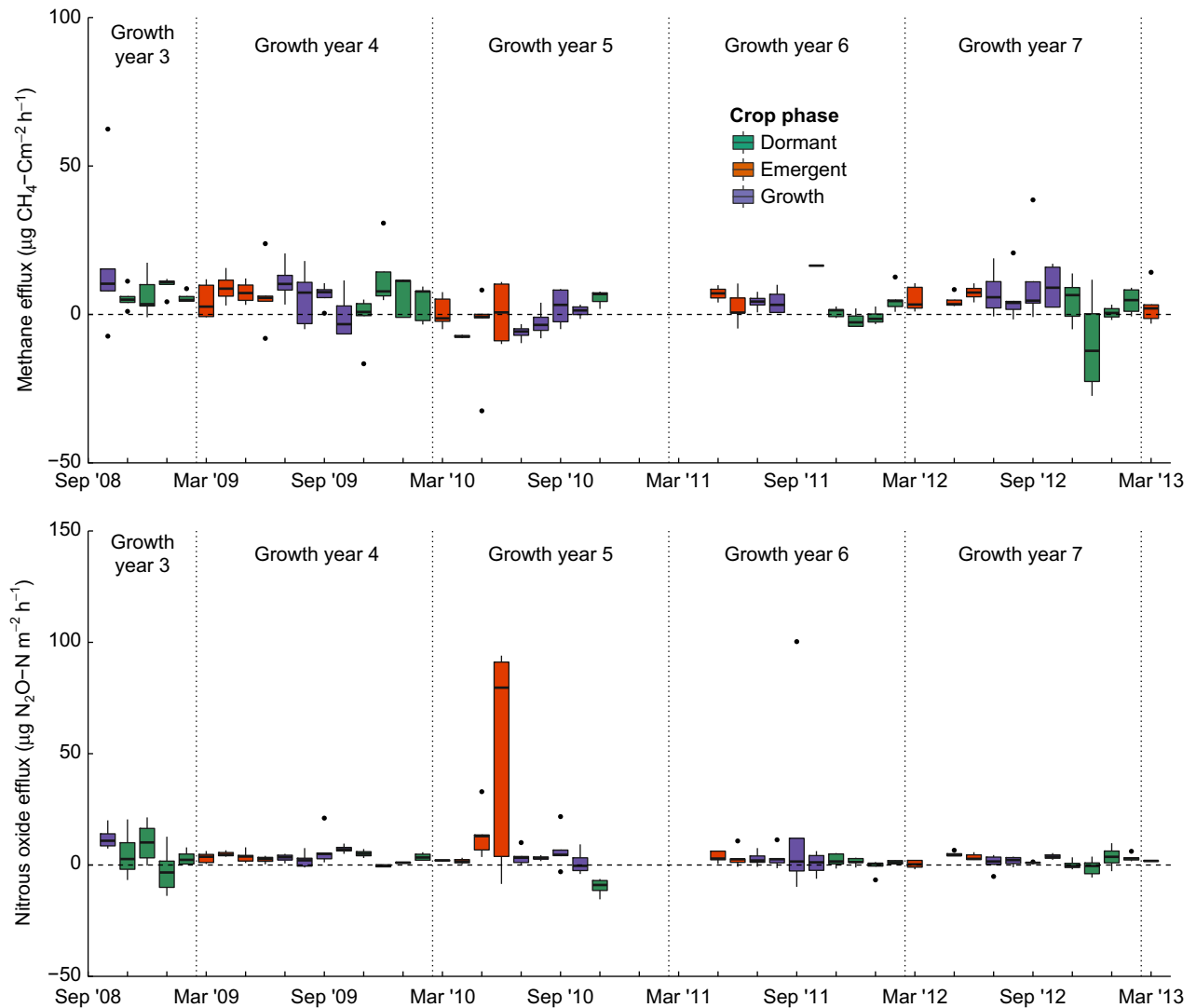


Fig. 1 Soil methane (CH_4) and nitrous oxide (N_2O) emissions in $\mu\text{g CH}_4\text{-C m}^{-2} \text{h}^{-1}$ and $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$ calculated from static chambers ($n = 5$) within a *Miscanthus* plantation in Lincolnshire, UK. Measurements are grouped and coloured by crop phase (dormant, green; emergent, orange; growth, purple). The boxes represent the interquartile range (25% to 75%) and the line within is the median value; whiskers describe the highest and lowest data points still within $1.5\times$ the interquartile range. Outliers of this $1.5\times$ the interquartile range are shown by filled circles.

C or N stocks between 0- to 15-cm and 15- to 30-cm layers (Fig. 4). Soil C stocks were estimated to be 81.3 t ha^{-1} in the top 30 cm in March 2006 and, 7 years later, in March 2013, measured as 81.9 t ha^{-1} in the same soil layer. Similarly unchanging soil N stocks were observed with 8.2 t ha^{-1} in the top 30 cm in March 2006 and 8.1 t ha^{-1} in March 2013.

Annual inputs to the litter layer through crop senescence (not including harvesting inefficiency) decreased over time from $2.59 \text{ t dry biomass ha}^{-1}$ in growth year 3 to $1.75 \text{ t dry biomass ha}^{-1}$ in growth year 7. After heavily stunted growth during the first two years, all standing biomass was cut and left on the site in April 2008,

estimated to be $3 \text{ t biomass ha}^{-1}$. From this point, litter inputs comprised both senesced leaves (green bars; Fig. 5) and residues from harvesting inefficiency (grey bars; Fig. 5). Considerable litter accumulation was observed between 2009 and 2013 (blue points; Fig. 5), suggesting a decomposition rate (k) slower than the rate of inputs. Using our measurements of the litter layer, we estimated a decomposition rate between those reported by Amougou *et al.* (2012) and Yamane & Sato (1975): $k \sim 0.63$.

Both senesced and living *Miscanthus* biomass had similar C concentrations (Table 4). In contrast, N concentration in standing biomass almost halved between

Table 1 Soil respiration from four years of static chamber measurements under a *Miscanthus* plantation in Lincolnshire, UK. Measurements averaged and cumulated by crop phase (dormant, emergent, growth) within each growth year (March–February) between March 2009 and February 2013 (± 1 SE)

Growth year	Crop phase	CO ₂ efflux (mg CO ₂ -C m ⁻² h ⁻¹)	Cumulative CO ₂ efflux (t CO ₂ -C ha ⁻¹)
4	Dormant	12.77 \pm 1.86	0.64 \pm 0.09
	Emergent	27.60 \pm 2.26	0.75 \pm 0.08
	Growth	106.89 \pm 17.84	2.83 \pm 0.45
	All	47.09 \pm 7.70	4.22 \pm 0.50
5*	Dormant	–	–
	Emergent	17.17 \pm 2.39	0.33 \pm 0.01
	Growth	56.86 \pm 9.76	1.59 \pm 0.39
	All	–	–
6	Dormant	16.12 \pm 1.45	0.66 \pm 0.11
	Emergent	30.47 \pm 6.26	0.49 \pm 0.05
	Growth	55.45 \pm 5.41	1.33 \pm 0.14
	All	34.30 \pm 3.55	2.67 \pm 0.19
7	Dormant	9.83 \pm 1.75	0.31 \pm 0.08
	Emergent	26.77 \pm 2.60	0.61 \pm 0.03
	Growth	42.07 \pm 3.94	1.13 \pm 0.07
	All	26.86 \pm 2.58	2.24 \pm 0.15

*denotes that the sensors were removed for too long to calculate average or cumulated emissions.

October (when senescence and nutrient translocation began) and March, and was reduced by a further 40 % in the litter layer (Table 4). Relatively little difference was seen in C concentration between stems and leaves, whereas N concentration was significantly different, resulting in C : N ratios of 206 and 56 for stems and leaves, respectively, in harvested biomass (Table 4). The mean oven-dried (0% moisture content) harvested yield was 6.07 t ha⁻¹ yr⁻¹ over the 5-year measurement period, equating to 2.85 t C ha⁻¹ yr⁻¹ (assuming 47% C concentration; Table 4); litter inputs were estimated as 2.69 t ha⁻¹ yr⁻¹ on average, equivalent to 1.24 t C ha⁻¹ yr⁻¹ (assuming 47% C concentration).

Life cycle GHG balance of *Miscanthus* vs. fossil fuels

When calculated over the predicted crop life cycle of 18 years, the total GHG balance from cradle to farm gate was a net removal of 441 t CO₂-eq ha⁻¹ (Table 5). Soil C stocks were assumed to remain constant (as this creates the most cautious scenario and no empirical data at the site suggest otherwise) and the litter layer unchanged for the remainder of the crop's lifetime following the measurement period. Both CH₄ and N₂O emissions contributed very little to offsetting the net sequestration observed through NEE measurements.

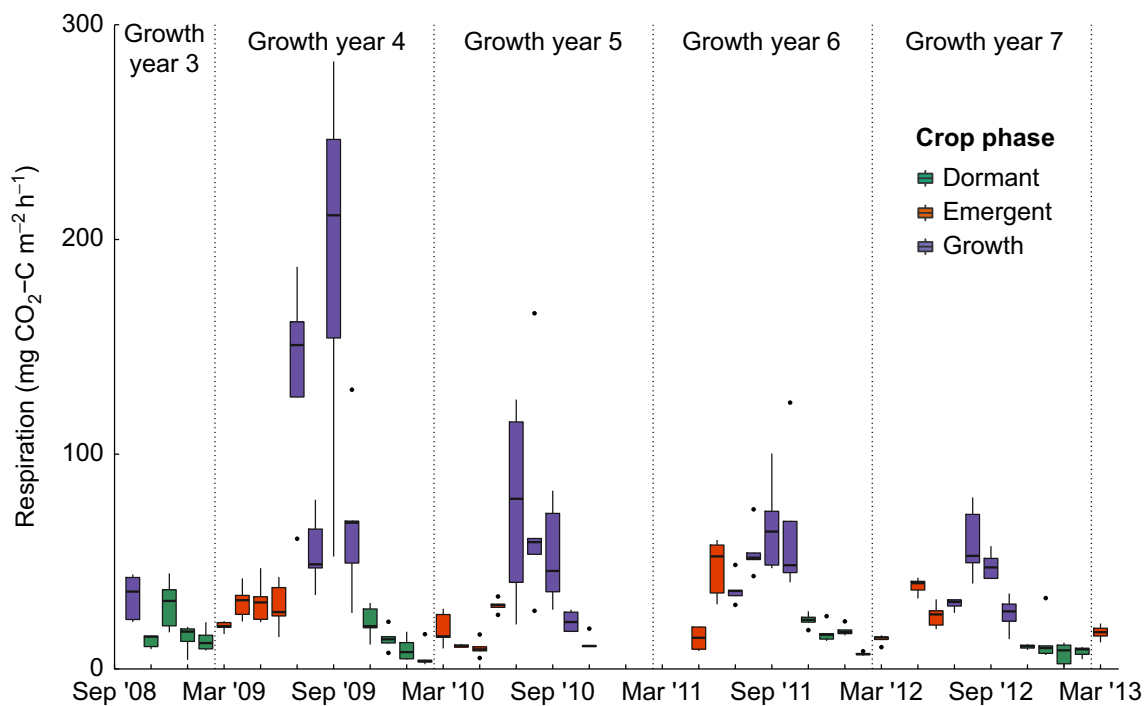


Fig. 2 Soil respiration (CO₂ emissions) in mg CO₂-C m⁻² h⁻¹ calculated from static chambers ($n = 5$) within a *Miscanthus* plantation in Lincolnshire, UK. Measurements are grouped and coloured by crop phase (dormant, green; emergent, orange; growth, purple). The boxes represent the interquartile range (25–75%) and the line within is the median value; whiskers describe the highest and lowest data points still within 1.5 \times the interquartile range. Outliers of this 1.5 \times the interquartile range are shown by filled circles.

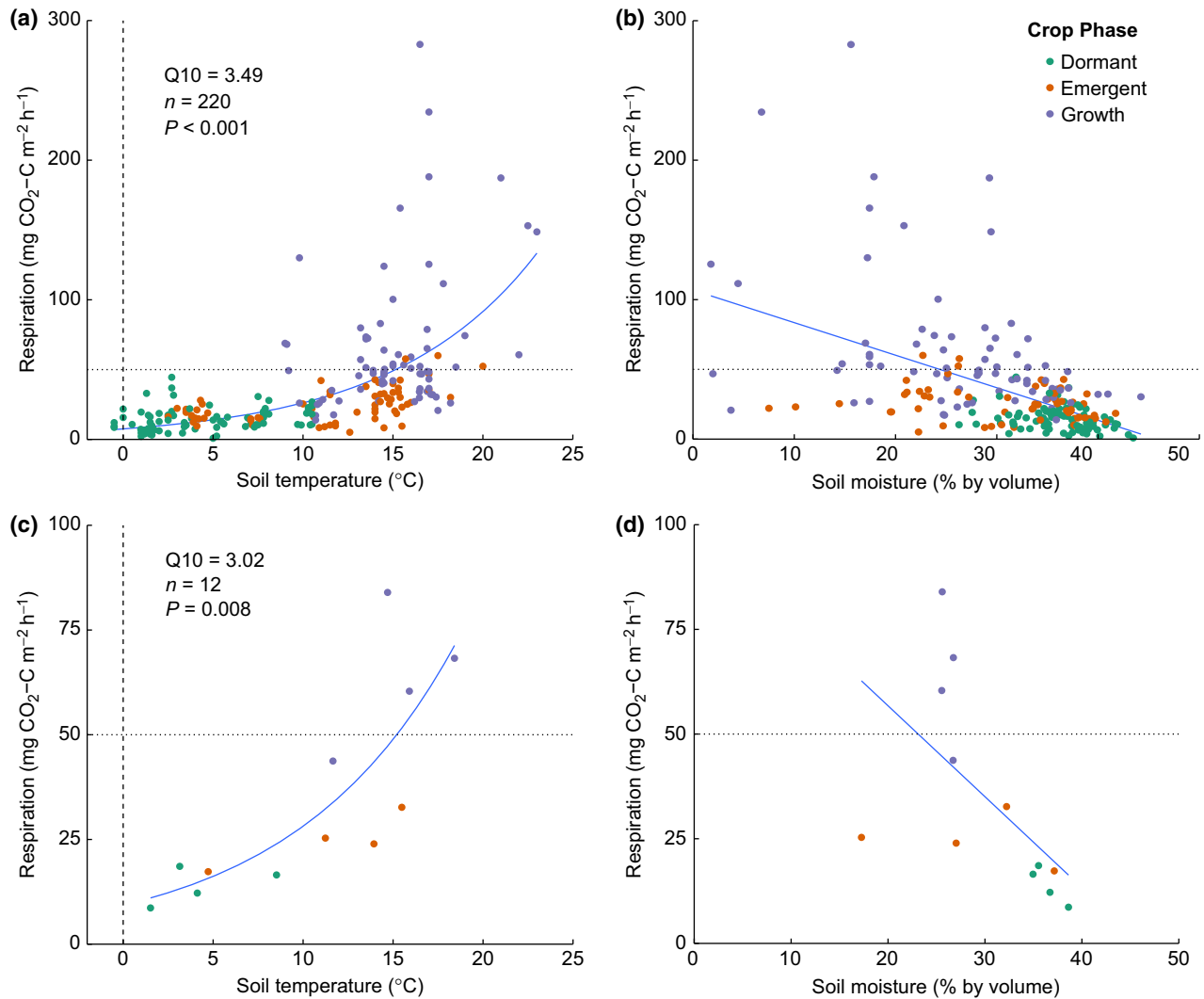


Fig. 3 Relationships between soil respiration and soil temperature (a and c) and soil moisture (b and d) using all available data points (a and b) and monthly average data (c and d) from measurements beneath a *Miscanthus* plantation in Lincolnshire, UK. Colours refer to crop phase: dormant (green), emergent (orange) and growth (purple). Regression analysis was used to fit an exponential relationship for soil temperature, reporting the associated *P*-values of how well the suggested relationship fit the measured data. Dashed vertical lines indicate 0 on plots where negative values were measured. Dotted horizontal lines are applied to aid comparison between top and bottom panels given that the scales differ.

Table 2 Temperature sensitivity of soil respiration calculated from monthly average data (± 1 SE) and the full data set of soil GHG emissions from a *Miscanthus* plantation in Lincolnshire, UK, between October 2008 and March 2013

Data set	Crop phase	Q_{10}	Mean soil temperature (range) (°C)
All chamber	All	4.39 ± 1.27	10.07 (−0.50 to 23.00)
All chamber	Dormant	1.64 ± 0.25	4.57 (−0.50 to 10.50)
All chamber	Emergent	2.03 ± 0.22	11.36 (2.50 to 20.00)
All chamber	Growth	3.18 ± 1.21	15.17 (9.00 to 23.00)
Monthly average chambers	All	3.03 ± 0.34	10.28 (1.52 to 18.41)

Table 3 Variance explained through regression analysis using linear mixed effects models on soil GHG emissions using all static chambers measurements between October 2008 and March 2013 under a *Miscanthus* plantation in Lincolnshire, UK

Factor	CO ₂ (%)	CH ₄ (%)	N ₂ O (%)
Soil temperature	48.48	1.62	0.90
Soil moisture	29.75	0.22	3.72
Crop phase	51.76	1.33	1.37
Temp × Moisture interaction	54.35	1.74	4.78

Cutting and baling the harvested biomass contributed the most to direct emissions but these were orders of magnitude lower than NEE measurements.

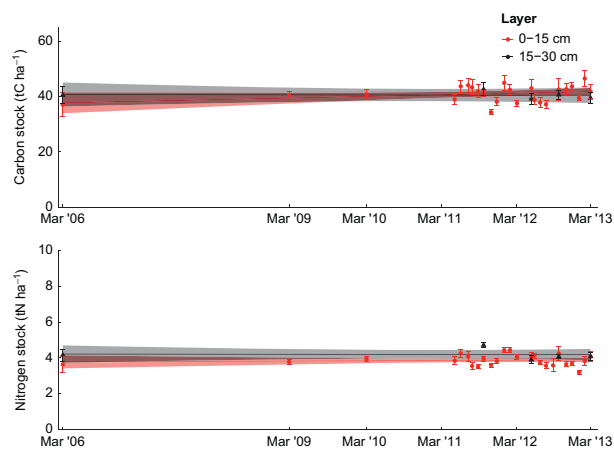


Fig. 4 Soil carbon and nitrogen stocks in tonnes per hectare measured from 2009 to 2013 at two depth intervals of the topsoil (red, 0–15 cm; grey, 15–30 cm) under a *Miscanthus* plantation in Lincolnshire, UK. Paired-site proxy measurements were used for 2006 data. Linear regression provided a relationship to time with colour-consistent shaded 95% confidence intervals.

Compared to the life cycles of coal and natural gas, *Miscanthus* had a substantially lower GHG intensity (Table 6). Further, the life cycle estimate of -1401 g $\text{CO}_2\text{-eq kWh}^{-1}$ suggests noteworthy sequestration beyond offsetting the known emissions. Any GHG intensity associated with cradle-to-farm gate ‘production’ below -1525.03 g $\text{CO}_2\text{-eq kWh}^{-1}$ would completely offset the emissions from transportation and combustion when using conventional power plants with conversion efficiency of 30% (Table 6). However, an important consideration in using GHG intensity as a comparison metric is that it does not account for the land area required to generate each unit of energy (kWh ha^{-1}). Consequently, a higher yield at this site, or an improved conversion efficiency (e.g. 70% achieved by CHP generators), would lead to lower emissions per kWh but would not necessarily increase net sequestration per kWh (Table 6). For reference, using 1 t of *Miscanthus* biomass (at 20% moisture content; LHV = 14 MJ kg^{-1}) for electricity generation produces 1167 kWh at 30% efficiency and 2722 kWh at 70% efficiency, while both emit 1722 kg $\text{CO}_2\text{-eq}$ through combustion (assuming 47% C concentration) (Eqn 3).

Discussion

This study addressed three main objectives: i) to quantify GHG emissions from a *Miscanthus* plantation and examine the influence of soil temperature and moisture on these emissions, ii) to examine the dynamics of litter and soil C stocks that define long-term sequestration and iii) to estimate the life cycle GHG intensity of

electricity generation using *Miscanthus* harvested from this site, ultimately comparing this with conventional fossil fuels.

Net ecosystem exchange and soil GHG emissions

The annual net CO_2 flux, reported as NEE, was on average -24.85 t CO_2 ha^{-1} yr^{-1} (Table 5), despite low yields compared to other studies in similar climatic regions (Lewandowski *et al.*, 2000; Christian *et al.*, 2008). A trial in Illinois, USA, comparing *Miscanthus* with switchgrass (*Panicum virgatum*) and prairie grasslands reported a GHG balance of -20.31 t CO_2 ha^{-1} yr^{-1} for *Miscanthus* in its third year after establishment (Zeri *et al.*, 2011), 14% lower than switchgrass (-17.78), 88% lower than prairie (-10.82) and 18% higher than our reported NEE. This Illinois *Miscanthus* plantation produced approximately 16 t dry biomass ha^{-1} in October of the third growth season, more than double the spring yield at our Lincolnshire site. Both studies emphasize the large sequestration potential of *Miscanthus*, despite annual harvests removing all aboveground biomass. While the negative NEE at our site implied considerable sequestration, soil respiration (10.99 t CO_2 ha^{-1} yr^{-1}) offset a large portion and dominated the GHG flux at the soil surface. This value was within the same range as other *Miscanthus* plantations (Wanga *et al.*, 2005; Behnke *et al.*, 2012; Case *et al.*, 2014), as well as other bioenergy crops: switchgrass (*Panicum virgatum*) (Frank *et al.*, 2004; Lee *et al.*, 2012), maize (*Zea mays*) (Rochette *et al.*, 1999; Ding *et al.*, 2007) and short rotation coppice (SRC) poplar (*Populus* spp.) (Verlinden *et al.*, 2013).

In contrast to the CO_2 fluxes, both CH_4 and N_2O made a negligible contribution to the GHG budget of the plantation over 4 years. That said, in June 2010 N_2O emissions were an order of magnitude larger than all other months (Fig. 1). Soil N_2O efflux is often very sporadic (Parkin, 1987; Dalal *et al.*, 2003) and most commonly associated with rainfall events and rapid changes in water filled pore space (Dobbie *et al.*, 1999). Consequently, rainfall events that occurred prior to measuring are likely to have influenced the high flux measured in June 2010, although this is unlikely to be the sole cause. To elucidate the drivers of this lone peak, more regular flux measurements are required to gauge the influence of explanatory variables. If these events are short bursts and occur more often than detected by our measurement schedule, the contribution of N_2O to the overall GHG budget would be much larger due to the high GWP of N_2O .

The *Miscanthus* plantation was shown to be a small source of CH_4 contradicting two previous studies at other sites (Toma *et al.*, 2011; Gauder *et al.*, 2012); however, spatial heterogeneity in soils is likely to

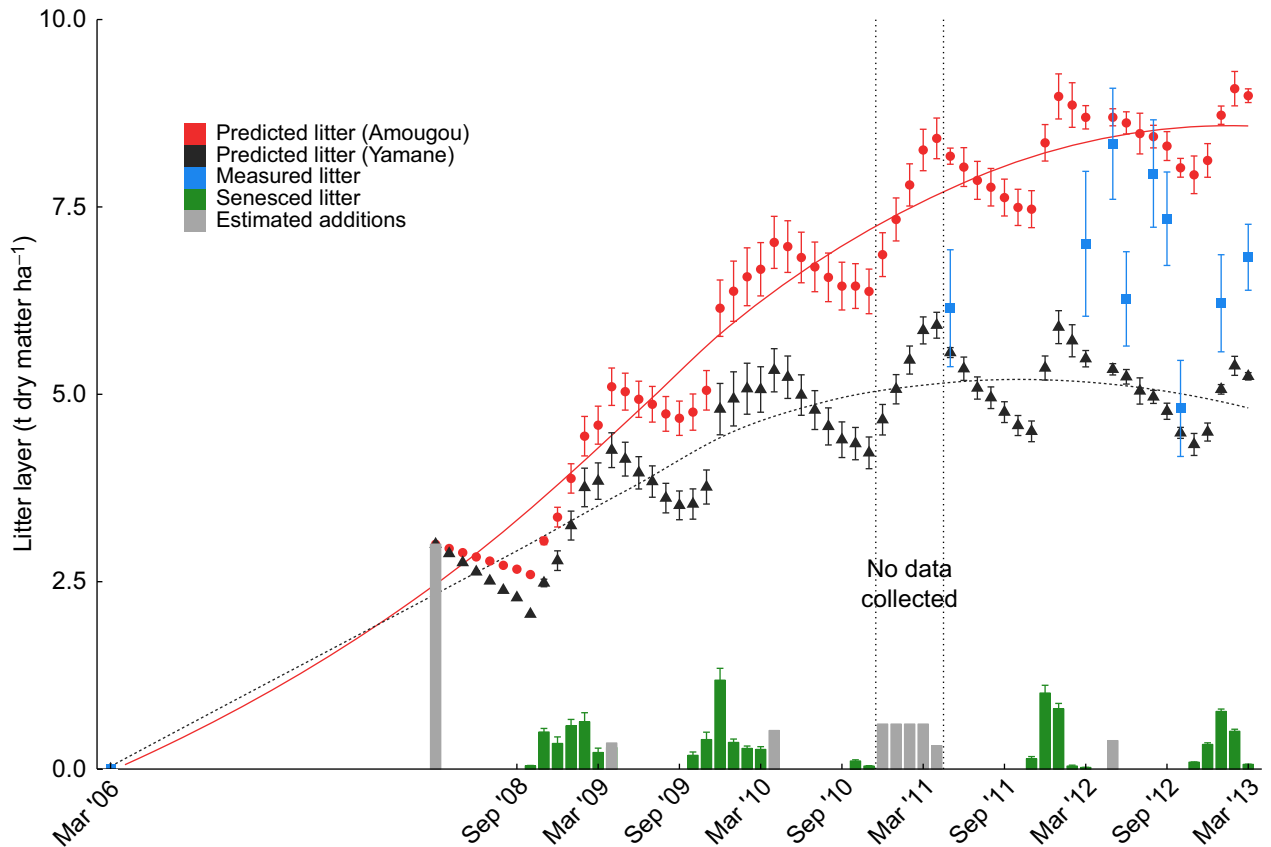


Fig. 5 Measured and modelled accumulation of *Miscanthus* plant litter on the soil surface over 7 years of growth. Two decomposition rates used (Amougou, $k = 0.776$, red points and line; Yamane, $k = 0.511$, black points and line) and smoothed loess regressions fitted through simulated data points. Senesced aboveground biomass (green bars) was measured through all months after September 2008 with the exception of December 2010 to April 2011 where senescence was estimated using an average from other years. Additions through harvesting inefficiency were estimated as 5% of total harvested biomass and occurred in April or May of each year (grey bars after September 2008).

Table 4 Average (± 1 SE) carbon and nitrogen concentrations of *Miscanthus* biomass from a plantation in Lincolnshire, UK. Sampling occurred during the 7th growth year of the perennial crop; litter layer values refer to an average of all samples collected between November 2011 and March 2013

		Standing biomass		Litter layer*
		October 2012	March 2013	
Stems	C concentration (%)	46.25 \pm 0.20	47.72 \pm 0.22	–
	N concentration (%)	0.40 \pm 0.03	0.24 \pm 0.04	–
	C : N	118.16 \pm 6.17	205.93 \pm 30.49	–
Leaves	C concentration (%)	44.98 \pm 0.19	45.85 \pm 0.56	44.15 \pm 0.30
	N concentration (%)	1.77 \pm 0.07	0.98 \pm 0.25	0.58 \pm 0.06
	C : N	25.70 \pm 1.06	55.78 \pm 17.83	85.94 \pm 7.41

*The litter layer consisted primarily of leaf litter but some stems were likely to be included. Standing biomass measurements of October represent the end of the growing season and March the end of senescence.

cause variation between sites (Smith *et al.*, 2000). While there are a number of factors which influence the processes that govern CH₄ and N₂O efflux (e.g. disturbance, Hütsch (2001); fertilizer, Mosier *et al.*

(1991); C : N of biomass, Gundersen *et al.* (2012)), the management intensity of *Miscanthus* plantations is typically low (no tillage, low fertilizer application) reducing the likelihood of high emissions. This may explain

Table 5 Life cycle greenhouse gas balance of *Miscanthus* cultivation from cradle-to-farm gate based on an 18-year life cycle and cultivation conditions of a *Miscanthus* plantation in Lincolnshire, UK

Process step	GER* (MJ diesel ha ⁻¹)	GCR† diesel (kg CO ₂ -eq. ha ⁻¹)	Times applied over life cycle	GHG balance (kg CO ₂ -eq. ha ⁻¹ yr ⁻¹)
Direct emissions				
Soil preparation				
Ploughing‡	744.0	63.77	2	7.09
Harrowing‡	310.3	26.60	1	1.48
Herbicide application‡	51.0	4.37	1	0.24
Planting‡	170.1	14.58	1.3	1.05
Rolling‡	340.1	29.15	1	1.62
Crop maintenance				
Fertilizer application‡	416.6	35.71	8	15.87
Harvesting				
Cutting¶	661.9	56.74	2	6.30
Cutting/baling¶	1486.3	127.39	16	113.24
Crop removal				
Herbicide application‡	51.0	4.37	1	0.24
Indirect emissions				
Rhizome propagation§ (10 000 ha ⁻¹)	2000	171.43	1	9.52
Herbicide production§		16.0	2	1.78
Measured field data				
Annual N ₂ O fluxes			18	176.53
Annual CH ₄ fluxes			18	17.34
Annual NEE			18	-24 847.88
Annual total				-24 495.81
Life cycle total				-440 925.66

*GER: gross energy requirement conversion factor of 42.51 MJ l⁻¹ diesel (Elsayed and Mortimer, 2001; DEFRA R-AEA, 2015).

†GCR: gross C requirement of diesel conversion factor 0.0857 kg CO₂ MJ⁻¹ diesel (Smeets *et al.*, 2009).

‡Thornley *et al.* (2009).

§Smeets *et al.* (2009).

¶Styles & Jones (2007).

||Data from site GHG budget.

Bold values refer to summed totals - both annually and over the full plantation lifetime.

Table 6 Greenhouse gas footprints of *Miscanthus* biomass used for electricity generation under two efficiency scenarios (30% and 70%) compared against coal and natural gas

Process step	<i>Miscanthus</i> (g CO ₂ -eq kWh ⁻¹)*		Coal (g CO ₂ -eq kWh ⁻¹) 38% efficiency	Natural gas (g CO ₂ -eq kWh ⁻¹) 47% efficiency
	30% efficiency	70% efficiency		
Production†	-2925.80	-1253.91		
Transportation‡	48.78	20.90		
Combustion§	1476.25	632.68		
Total	-1400.77	-600.33	837-1130¶	423-535¶

*1 kWh = 3.6 MJ.

†Data from site GHG budget (see method for details).

‡Smeets *et al.* (2009) (see method for details).

§Cannell (2003) (see method for details).

¶MacKay & Stone (2013).

Bold values refer to summed totals.

the low trace GHG emissions seen in this study and reported elsewhere (e.g. Toma *et al.*, 2011; Drewer *et al.*, 2012; Gauder *et al.*, 2012). It is worth noting that land use change to intensive management

practices after *Miscanthus* propagation may stimulate rapid mineralization of labile nutrients (particularly C and N) that accumulated during the plantation's lifetime.

Environmental drivers of soil respiration

Due to very low CH₄ and N₂O fluxes, it is not possible to draw conclusions regarding the weak relationships observed between climatic variables and emissions. In contrast, soil respiration did vary significantly with season, closely following changes in soil temperature and crop phenology (Table 1; Fig. 2). This confirms results from other studies where largest CO₂ emissions were observed when temperatures and photosynthetic activity were greatest (Yazaki *et al.*, 2004; Wanga *et al.*, 2005; Gauder *et al.*, 2012) and follows conventional understanding of both heterotrophic and autotrophic soil respiration (Ryan & Law, 2005; Tang *et al.*, 2005). Soil respiration also varied interannually (4.22 to 2.24 t C ha⁻¹ for growth years 4 and 7, respectively; Table 1) despite similar climatic conditions between years (Table S6). Yazaki *et al.* (2004) took similar measurements from a *Miscanthus sinensis* plantation in Japan, estimating much more consistent emissions between two years. While the average aboveground biomass was similar, annual soil respiration from the Japanese plantation was more than three times higher than ours (~14 t C ha⁻¹). Additionally, in the same study the temperature sensitivity (Q₁₀) of total soil respiration varied between 2.7 and 3.1. This agrees well with the average Q₁₀ values calculated for our site (Table 2), despite the Japanese site having higher soil temperatures and not including Q₁₀ estimates between December and April (when they are likely to be lowest). The relatively low soil temperatures at our site, and their impact on soil respiration, may explain why the low productivity still creates a lower NEE than that of the higher yielding site in Illinois (Zeri *et al.*, 2011); while C assimilation through photosynthesis in Illinois is considerably higher than in Lincolnshire, so is the annual mean air temperature (11.1 vs. 9.6 °C) and, in particular, temperatures during the growing season. Consequently, soil respiration is likely to greatly offset the increased C sequestration through photosynthesis; while biomass production in Illinois is larger than that in Lincolnshire, the overall GHG balance of the *Miscanthus* plantation may be more favourable in the cooler climate.

Carbon and nitrogen stocks

Soil C and N stocks did not change over 4 years and when compared with a proxy for before *Miscanthus* was planted, stocks were still unchanged (Fig. 4). While this is consistent with some studies of *Miscanthus* (Zatta *et al.*, 2014; Rowe *et al.*, 2015), many others report increases in topsoil (0–30 cm) C stocks of more than 1 t C ha⁻¹ yr⁻¹ with prior land use and management practices playing a key role in the direction of change (Kahle

et al., 2001; Dondini *et al.*, 2009; Zimmermann *et al.*, 2012; Poeplau & Don, 2014). There is a reasonable chance that topsoil C stocks were negatively impacted through disturbance of ploughing and planting, but were also enhanced by the addition of rhizomes and rapid fine root turnover as the plantation established itself. Indeed, Amougou *et al.* (2011) reported combined rhizome and root C input rates of 2.91 t ha⁻¹ for the top 30 cm over the first three years after planting. These input rates are then expected to decline as the plantation ages; Richter *et al.* (2015) noted a combined C input rate of 1.43 t ha⁻¹ for the top 100 cm over the first 14 years after planting (see Agostini *et al.* 2015 for a review of existing data on this topic). Aside from the lower yields noted at this Lincolnshire site, and therefore likely smaller belowground biomass pools, there is no clear reason why soil C stocks are not increasing over time. We hypothesize that at this site fresh C inputs may be stimulating (priming) the decomposition of existing soil C, therefore negating any C sequestration (Zatta *et al.*, 2014). Testing this hypothesis would require the use of stable isotopes to trace the fate of native soil C and fresh C inputs in these crops.

N deficiency in the soil may also explain low C sequestration rates through limitation of decomposition and microbial activity (Hu *et al.*, 2001; Craine *et al.*, 2007). The C : N ratio of senesced *Miscanthus* biomass was between 70 and 120, and soil C : N was around 10 (Table 4; Fig. 4). These are high values for an arable crop, and therefore, a lack of N fertilizer may be a limiting factor in microbial decomposition (Anderson & Domsch, 1989). That said, these C : N ratios are within a normal range for *Miscanthus* plantations where soil C sequestration has been noted (Dondini *et al.*, 2009; Amougou *et al.*, 2011) and therefore cannot alone explain the lack of sequestration at this site. Additionally, other studies have observed similar accumulation rates of senesced biomass (2 t ha⁻¹ yr⁻¹; *k* ~ 0.63) while also reporting increased soil C stocks (Yamane & Sato, 1975; Amougou *et al.*, 2011, 2012).

In the absence of soil C sequestration at this site, the measured NEE of -6.78 t C ha⁻¹ yr⁻¹ is very low and requires an explanation for where C is being sequestered. Following biomass removal at harvest, C pools may remain in live belowground biomass, an increased O-horizon and in the soil organic matter (SOM) that was removed before calculating soil C stocks. When these additional pools are considered, -6.78 t C ha⁻¹ yr⁻¹ is not unrealistic: 2.85 t C ha⁻¹ yr⁻¹ was present in harvested biomass and 1.24 t C ha⁻¹ yr⁻¹ was added to the O-horizon through senescence and harvesting inefficiency (Fig. 5). This leaves 2.69 t C ha⁻¹ yr⁻¹ to be allocated to live belowground biomass, to soils below the measured topsoil (30 cm) and to SOM

fractions, a realistic possibility given the recalcitrant nature of *Miscanthus* biomass (Amougou *et al.*, 2011) and its characteristic deep-rooting (Neukirchen *et al.*, 1999). Indeed, live and dead root biomass was estimated to be 4.46 t dry mass ha⁻¹ in the top 30 cm of soils at this site and annual C inputs under *Miscanthus* can be substantial (Agostini *et al.*, 2015). It is also important to note that dissolved organic carbon and carbon lost through root exudation may contribute to this unquantified sink of soil carbon (Hromadko *et al.*, 2010).

Comparative life cycle GHG budgets of *Miscanthus*

Miscanthus was calculated to remove 441 t CO₂-eq ha⁻¹ (over 18 years) from the atmosphere using a 'cradle-to-farm gate' analysis (Table 5). This compares well against a SRC willow plantation, grown for 23 years, removing 496 t CO₂-eq ha⁻¹ without consideration of soil GHG emissions (Heller *et al.*, 2003). It is worth noting that while our method of linear integration to cumulate soil CO₂ emissions is robust, it may be less appropriate for N₂O. Soil N₂O emissions are spatially and temporally heterogeneous and as a result chamber measurements may not capture the true site-scale emission rates (Williams *et al.*, 1992; Bouwman *et al.*, 2002; Stehfest & Bouwman, 2006). This may have contributed towards the favourable cradle-to-farm gate GHG balance in comparison with other studies, where soil GHG emissions were modelled rather than measured (Brandão *et al.*, 2011; Hamelin *et al.*, 2012). While we acknowledge that the low temporal resolution of measurements may limit our ability to accurately quantify the contribution of N₂O to the life cycle GHG budget, both this study and those previously published report low N₂O emissions under *Miscanthus* (Toma *et al.*, 2011; Drewer *et al.*, 2012; Gauder *et al.*, 2012). Higher resolution (both temporally and spatially) N₂O measurements would reduce uncertainty and are needed to underpin the refinement of emission factors for use in LCAs. With respect to NEE, limiting gaps in NEE measurements would also improve the accuracy of field GHG emissions data for LCAs. The measurement gaps reported here were assumed to cause limited error because they occurred in winter when photosynthesis and GHG fluxes were low. Further, average annual values were derived from a full 48-month period. Ultimately, gaps during winter months are likely to have far smaller impact on annual NEE estimates than other factors such as interannual climatic variation (Massman & Lee, 2002; Baldocchi, 2014).

The life cycle GHG intensity of electricity generation using *Miscanthus* from this site is very low compared to that of electricity generated from coal or natural gas. While both fossil fuels are a net source of GHGs, the *Miscanthus* plantation was a noteworthy GHG sink,

offsetting between 0.6 and 1.4 kg CO₂-eq per kWh (Table 6). This range is very low compared to a similar study of *Miscanthus* grown in Canada (Sanscartier *et al.*, 2014) where between 0.02 and 0.19 kg CO₂-eq was offset per kWh, including soil C sequestration. However, GHG intensity (emissions per unit energy generated) does not account for the land area required to generate each kWh – a major concern when determining the sustainability of bioenergy crops (Dornburg *et al.*, 2003; Rowe *et al.*, 2009). At this site, each hectare is capable of producing 8372 kWh of electricity, assuming a combustion efficiency of 30% and an average annual yield of 7.18 ha⁻¹ (20% moisture content). A higher yielding site with similar environmental characteristics may increase C sequestration through NEE but not necessarily enough to improve the GHG balance per kWh produced, especially if these higher yields come at a cost of increased emissions during production and growth through intensive management (e.g. fertilizer application or precision planting). A recent study comparing *Miscanthus* with maize and switchgrass in North America (Qin *et al.*, 2015) drew similar conclusions to those described here: *Miscanthus* has the potential to produce energy at low, or even C-negative, GHG intensities. It is also important to recall that soil C sequestration can offset a significant portion of the emissions derived from generating electricity. Given a 30% combustion efficiency and 129.2 t ha⁻¹ yield (18 years at Lincolnshire), an increase of 1 t C ha⁻¹ yr⁻¹ in soils would offset 438 g CO₂-eq kWh⁻¹ on a life cycle basis (Eqn 3, GHG_{site} fixed at -3.66 t CO₂-eq ha⁻¹ yr⁻¹). An increase of 1 t C ha⁻¹ yr⁻¹ in the top 30 cm is not unrealistic; at this site, *Miscanthus* inputs were previously shown to add 0.86 t C ha⁻¹ yr⁻¹ to the top 30 cm (Robertson *et al.*, 2016) and Poeplau & Don (2014) saw an average increase of 1.68 ± 0.7 t C ha⁻¹ yr⁻¹ from a range of *Miscanthus* crops across Europe. The unchanged topsoil C stocks reported here, therefore, have important consequences for whether it is deemed a preferable alternative to conventional fossil fuels.

Due to minimal land management and fertilizer requirements (Cadoux *et al.*, 2012), *Miscanthus* is often seen as an attractive option when land is unsuitable for conventional arable crops. However, policymakers still require more data to reliably assess its sustainability when used for bioenergy by combustion. As hypothesized, this study found CO₂ to dominate site GHG fluxes but noted substantially more sequestered than emitted over each year. Furthermore, despite relatively low yields and a lack of soil C sequestration, the crop studied here had a considerably lower GHG intensity than coal or natural gas when used for electricity generation. Additional research is required to elucidate why soil C stocks are not changing under this plantation

(Zatta *et al.*, 2014; Robertson *et al.*, 2016) and future bioenergy sustainability studies should prioritize land use efficiency over GHG intensity comparisons. Nevertheless, this study demonstrates that even when yields are lower than many other sites due to climate or establishment issues, GHG benefits can still outweigh costs and contribute to climate change mitigation through the provision of low C renewable energy.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Figure S1 Daily soil and air temperature (°C) measured at the *Miscanthus* plantation in Lincolnshire, UK, between growth years 3 and 7. Half-hourly data were averaged to give daily points between 1 May 2008 and 10 March 2013.

Figure S2 Daily soil temperature (°C) and precipitation (mm) measured at the *Miscanthus* plantation in Lincolnshire, UK, between growth years 3 and 7. Half-hourly data were averaged or summed for temperature and precipitation, respectively, between 20 August 2008 and 10 March 2013.

Table S1 Soil characteristics at the Lincolnshire *Miscanthus* plantation estimated using 5 reps taken from each month between February 2009 and November 2010 (inclusive).

Table S2 Micrometeorological mast installation and removal dates for different growth years of a *Miscanthus* plantation in Lincolnshire, UK.

Table S3 Instrumentation used by the meteorological tower in a *Miscanthus* plantation, Lincolnshire, UK.

Table S4 Cultivation and management operations during the life-cycle for *Miscanthus* at the Lincolnshire field site.

Table S5 Observed (2007–2013) and predicted (2014–2024) *Miscanthus* yields (assumed 20% moisture content throughout) at the Lincolnshire site estimated using one stable and one declining approach.

Table S6 Average temperature (± 1 SE) and cumulative precipitation and radiation measurements from continuous (half-hourly) data collected between growth years (March–February) 3 and 7 of a *Miscanthus* plantation in Lincolnshire, UK.