Measured and modelled effect of land-use change from temperate grassland to *Miscanthus* on soil carbon stocks after 12 years

Amanda J. Holder1 | John Clifton-Brown1 | Rebecca Rowe2 | Paul Robson1 | Dafydd Elias2 | Marta Dondini3 | Niall P. McNamara2 | Iain S. Donnison1 | Jon P. McCalmont4

1Institute of Biological, Environmental and Rural Sciences (IBERS), Aberystwyth University, Aberystwyth, United Kingdom
2Centre for Ecology & Hydrology, Lancaster Environment Centre, Bailrigg, Lancaster, United Kingdom
3Institute of Biological and Environmental Sciences, University of Aberdeen, Aberdeen, United Kingdom
4College of Life and Environmental Sciences, University of Exeter, Exeter, United Kingdom

Correspondence
Amanda J. Holder, Institute of Biological, Environmental and Rural Sciences (IBERS), Aberystwyth University, Gogerddan, Aberystwyth, Wales SY23 3EQ, UK.
Email: amh21@aber.ac.uk

Funding information
Aberystwyth University: Biotechnology and Biological Sciences Research Council, Grant/Award Number: BB/CSP1730/1 and BBS/E/W/10963/A01B; European Commission FACCE SURPLUS ERA-NET MISCOMAR, Grant/Award Number: 652615; Engineering and Physical Sciences Research Council, Grant/Award Number: EP/M013200/1; Department for Environment, Food and Rural Affairs, Grant/Award Number: NF0426

Abstract
Soil organic carbon (SOC) is an important carbon pool susceptible to land-use change (LUC). There are concerns that converting grasslands into the C4 bioenergy crop *Miscanthus* (to meet demands for renewable energy) could negatively impact SOC, resulting in reductions of greenhouse gas mitigation benefits gained from using *Miscanthus* as a fuel. This work addresses these concerns by sampling soils (0–30 cm) from a site 12 years (T12) after conversion from marginal agricultural grassland into *Miscanthus x giganteus* and four other novel *Miscanthus* hybrids. Soil samples were analysed for changes in below-ground biomass, SOC and *Miscanthus* contribution to SOC (using a 13C natural abundance approach). Findings are compared to ECOSSE soil carbon model results (run for a LUC from grassland to *Miscanthus* scenario and continued grassland counterfactual), and wider implications are considered in the context of life cycle assessments based on the heating value of the dry matter (DM) feedstock. The mean T12 SOC stock at the site was 8 (±1 standard error) Mg C/ha lower than baseline time zero stocks (T0), with assessment of the five individual hybrids showing that while all had lower SOC stock than at T0 the difference was only significant for a single hybrid. Over the longer term, new *Miscanthus* C4 carbon replaces pre-existing C3 carbon, though not at a high enough rate to completely offset losses by the end of year 12. At the end of simulated crop lifetime (15 years), the difference in SOC stocks between the two scenarios was 4 Mg C/ha (5 g CO2-eq/MJ). Including modelled LUC-induced SOC loss, along with carbon costs relating to soil nitrous oxide emissions, doubled the greenhouse gas intensity of *Miscanthus* to give a total global warming potential of 10 g CO2-eq/MJ (180 kg CO2-eq/Mg DM).

KEYWORDS
bioenergy, land-use change, life cycle assessment, *Miscanthus*, pasture, soil organic carbon
Energy generation from fossil fuels (e.g. coal and gas) must be phased out as part of world-wide efforts to combat the impacts of climate change (IPCC, 2014). The European Union has set a target for renewable energy (wind, solar, hydro and bioenergy) to reach a minimum of a 27% share of the energy generation mix by 2030 (from the current share of ~17% (European Commission, 2017). In the United Kingdom, renewable energy other than wind, solar and hydro accounted for 9.4% of the total energy produced in 2017 and there is scope for bioenergy generation (e.g. from biomass crops, landfill, and sewage gas and anaerobic digestion) to increase (BEIS, 2018).

Agricultural grasslands represent a third of the utilized agricultural area across Europe (Eurostat, 2018) and due to changes in farming subsidies and temperate grassland agricultural management across Europe, areas of lower grade agricultural grassland may become available for biomass crops (Donnison & Fraser, 2016; Taube, Gierus, Hermann, Loges, & Schönbach, 2014). In the United Kingdom, Welsh agricultural land is primarily grass based (Welsh Government, 2018) and spatial modelling has suggested that there may be 0.5 M ha suitable for the planting of perennial bioenergy crops (such as Miscanthus and short rotation coppice; Lovett, Süninnenberg, & Dockerty, 2014). However, there are concerns that losses of soil carbon (C) caused by soil disturbance (Balesdent, Chenu, & Balaban, 2000; Conant, Easter, Paustian, Swan, & Williams, 2007) could reduce the C mitigation benefits gained from the conversion of grasslands into the production of bioenergy crops (McCalmont, Hastings, et al., 2017; Whitaker et al., 2018).

The biomass crop Miscanthus x giganteus (Mxg; Greef & Deuter, 1993) is a commercially available hybrid that is a fast-growing, tall perennial grass, with an efficient C₄ photosynthetic pathway. It is a low-input crop with the potential to be grown on agriculturally marginal land (Clifton-Brown, Schwarz, & Hastings, 2015; Lewandowski, Clifton-Brown, Scurlock, & Huisman, 2000). Compared to annual crops, Miscanthus has the potential to sequester C due to reduced soil disturbance (tillage is only required as part of the initial cultivation; Post & Kwon, 2000), the translocation of C from above-ground biomass to roots and rhizomes (Kuzyakov & Domanski, 2000), and the provision of soil C inputs from leaf litter (Amougou, Bertrand, Machet, & Recous, 2011). New, commercially relevant Miscanthus hybrids are being developed with different morphologies and traits (Lewandowski et al., 2016; Nunn et al., 2017) which may impact soil organic carbon (SOC), for example though variations in leaf litter and carbon allocation between above- and below-ground biomass (Clifton-Brown & Lewandowski, 2000; Richter, Agostini, Redmile-Gordon, White, & Goulding, 2015).

Land-use change from arable crop production to Miscanthus generally shows an increase or no change in SOC, whereas, in contrast, it has been found that Miscanthus plantations have lower or similar SOC when compared to grassland controls (Qin, Dunn, Kwon, Mueller, & Wander, 2016). However, to date, most studies have taken grassland sites adjacent to Miscanthus plantations as representative of pre-cultivation conditions (Clifton-Brown, Breuer, & Jones, 2007; Foereid, Neergaard, & Høgh-Jensen, 2004; Rowe et al., 2016; Schneckenberger & Kuzyakov, 2007; Zang et al., 2018; Zimmermann, Dauber, & Jones, 2012), and while the use of such sites where soil and climate conditions are similar can provide a reasonable indication they may not accurately replicate baseline SOC stocks (McCalmont, Hastings, et al., 2017; Richter et al., 2015). Therefore, there is a need to reduce some of the uncertainty around the impact of this LUC from grassland to Miscanthus on SOC (Whitaker et al., 2018), especially over the longer term.

Any carbon losses or gains from LUC should be considered over the expected lifespan of the Miscanthus crop, currently estimated to be between 10 and 15 years (Clifton-Brown et al., 2015). Clifton-Brown et al. (2007) found an increase in SOC under 15 year old Miscanthus compared to an adjacent grassland, whereas Zang et al. (2018) found that although SOC increased between samples taken at the same site 9 and 21 years after conversion, SOC was similar to samples taken from a neighbouring grassland (used to represent pre-conversion conditions). Reducing the uncertainty around the long-term impact of SOC using pre-cultivation data from the same site is needed to inform soil carbon model predictions and life cycle analyses (LCA).

Due to the limited number of long-term empirical studies of land use conversion into energy crops, a number of models have been used to estimate changes in SOC (Robertson, Davies, Smith, Dondini, & McNamara, 2015). ECOSSE (Estimation of Carbon in Organic Soils: Sequestration and Emissions) is a process-based model that has been successfully tested and used for simulating SOC under perennial energy crops including grassland and Miscanthus in this UK region (Dondini et al., 2015; Dondini, Richards, Pogson, Jones, et al., 2016a; Dondini, Richards, Pogson, McCalmon, et al., 2016b). However, empirical baseline data of SOC stocks in LUC from grassland to Miscanthus, coupled with data of SOC stocks under the mature crop (over 10 years old) would provide further model validation. ECOSSE can be used at the site or regional scale and represents an improvement on a previous model, RothC, due to a new approach to mineral and organic soils whereby the extent of processes occurring are adjusted according to soil conditions and not differentiated solely by soil type (Robertson et al., 2015; Smith et al., 2010).

LCA is a tool that can provide an indication of the environmental costs or benefits of producing energy from different methods and by enabling comparisons which help to inform
policy decisions relating to proposed LUCs (McManus & Taylor, 2015). LCA’s relating to LUC from grassland to Miscanthus have not included changes in soil carbon due to a lack of reliable data, and have tended to assume no change or an increase in SOC stocks (Hastings et al., 2017; Hillier et al., 2009). LCA estimates involving LUC are sensitive to the initial land use and condition (McManus & Taylor, 2015). For example, Robertson et al. (2017) investigated SOC as part of their LCA involving LUC to Miscanthus but this was from a previous arable land use with annual cultivation; potential losses at grassland sites, with less regular soil disturbance, could have a significant impact on LCA results (Hillier et al., 2009). Changes in SOC over the lifetime of the crop also have the potential to impact on greenhouse gas balances to a greater extent than other LUC associated costs such as increased soil nitrous oxide (N₂O) emissions (Whitaker et al., 2018).

Therefore, in this study, we aimed to (a) measure the change in SOC stock, and Miscanthus contribution to SOC, from a mature (>10 years old) Miscanthus crop following land use conversion from an agricultural grassland compared to baseline data of initial SOC stocks; (b) use the empirical data obtained to provide validation for ECOSSE model predictions; (c) use the ECOSSE model to predict SOC stocks following LUC from grassland for an estimated Miscanthus crop commercial lifetime of 15 years along with a continued grassland counterfactual, in order to establish the difference in SOC between the two scenarios; and (d) provide context for the predicted difference in SOC between the Miscanthus and grassland scenarios at the end of the 15 years by converting the difference in a global warming potential (GWP) for inclusion in an LCA comparison per unit of energy based on the heating value of the Miscanthus biomass.

In order to achieve this, we built on previous experimental work reported in Zatta, Clifton-Brown, Robson, Hastings, and Monti (2014) which although from a single site includes baseline SOC data (T₀) and data taken from the same site 6 years (T₆) after land use conversion from grassland into Mxg and four novel Miscanthus hybrids. Taking advantage of the difference in δ¹³C natural abundance values arising from the contrasting C₃ photosynthetic pathway of temperate grassland species compared to the C₄ pathway of Miscanthus (Kuzyakov & Domanski, 2000), we assessed changes in the contribution of Miscanthus to SOC between T₆ and T₁₂.

2 | MATERIALS AND METHODS

Sampling was conducted at a replicated plot trial situated at Aberystwyth, Wales, UK (52°26’ N, 4°01’W) on agriculturally marginal shallow dystric cambisol and dystric gleyso classified soil (up to 0.6 m soil depth in places but mainly with a gravel layer at depths >0.3 m). Prior to conversion, the site was a mature established perennial ryegrass sward. Historically, the site has predominantly been used for grass pasture and silage trials (resown ~5 yearly) with occasional oat crops (Zatta et al., 2014). The sample area consisted of four blocks of five randomized 25 m² plots; each plot contained one of five different Miscanthus hybrids. In September 2004, prior to planting, the existing mature perennial ryegrass sward was sprayed with glyphosate (3 L/ha) and inversion tillled with mouldboard plough and power harrow before a ryegrass cover crop was sown in October 2004. The cover crop was sprayed with atrazine (3 L/ha) on 5 April 2005 with the Miscanthus planted on 24 May 2005.

2.1 | Miscanthus hybrids

Bare root transplants of four novel hybrids (M. sacchariflorus x M. sinensis) cloned via in vitro tillering (hereafter Hyb 1, Hyb 2, Hyb 3 and Hyb 4), and rhizome segments of the commercially available Mxg were slot planted at a density of two plants per square meter. Compared to Mxg, after 3 years growth, Hyb 1–4 had a higher stem density (~39 vs. 30 stems m⁻²), lower canopy height (~2.05 vs. ~2.50 m) and lower above-ground biomass lignin (~10% vs. ~30%; P. Robson & J. Clifton-Brown, unpublished data).

The hybrids formed part of an ongoing yield trial with data recorded each year. Percentage differences between the above-ground autumn peak harvest and spring harvest (ripening loss) for each hybrid were calculated from the oven-dried weights of 10 stems taken from each plot in November 2007 and February 2008.

2.2 | Soil cores

Detailed methods regarding the pre-planting (6 May 2005, T₀) soil cores and those taken after 6 years of crop growth (5 May 2011, T₆) can be found in Zatta et al. (2014). Briefly, at T₀, five core samples (to 30 cm depth) were taken from two plots in each block, and at T₆, three core samples were taken from each plot. Each of the three T₆ core locations was taken to represent a portion of the overall field area covered by plant centre (8.1%), plant edge (24.5%) and inter-row (67.4%).

On 4 and 5 May 2017, 12 years since the plots were planted (T₁₂), three cores were again taken in each plot following the methods at T₆. The same 8.5 cm diameter cylinder auger (Eijkelkamp, Giesbeek, The Netherlands) was used with a Cobra TT jackhammer (Atlas Copco, Hemel Hempstead, UK) to take intact and uncompressed cores at three locations in each plot taken to represent a percentage of the overall field area. The soil core locations, individual plot heterogeneity and details of the field cover survey used to calculate the percentage area represented by each core are given in the Supplementary Information (S1 & S2). At T₁₂, the area represented by the plant centre (C_c) was determined to be 9.82%, the plant edge (C_e) 53.39% and the inter-row (C_i) 36.79%. Soil cores were
taken to a depth of 30 cm at position C₁, 31 cm at C₂ and 32 cm at C₃ to allow for soil displacement by rhizome growth (Zatta et al., 2014) and were subsequently split at 15 cm, 16 cm and 17 cm, respectively, before air drying to a constant weight. Soils were sieved (2 mm) to separate soil, stone and below-ground biomass (roots and rhizome). Soil was then ball milled (Planetary Mill, Fritsch GmbH, Idar-Oberstein, Germany). Air-dried below-ground biomass (roots and rhizome) were premilled (SM100, Retsch GmbH, Haan, Germany) before being finely cryomilled (6,870 Cryomill, SPEX, Stan-hope, UK) in liquid nitrogen. Bulk density was calculated using the same method as described in Zatta et al. (2014).

2.3 | Carbon analysis

Inorganic carbon was removed from a 3 g portion of each milled soil sample by adding 30 ml 1 M HCl, rinsing and oven drying to constant weight at 40°C (Clifton-Brown et al., 2007). A quantity of 200 mg of the acid-treated soil was analysed for percentage carbon content by combustion using a Vario Macro Cube (Elementar Analysensysteme GmbH, Langenselbold, Germany). Total organic carbon was calculated using Equation (1):

\[
\text{SOC} = \text{POC} \times \left( \frac{\text{ODW}_{\text{ac}}}{\text{ODW}_{\text{init}}} \right)
\]

where SOC (%) is the total soil organic carbon, POC is the percentage organic carbon in the acid-washed sample, ODW_{ac} is the oven-dried weight of the sample after acid washing and ODW_{init} is the oven-dried weight of the sample before acid washing.

SOC mass was calculated in two ways: to a fixed soil depth (using the soil bulk density) and to an equivalent soil mass (ESM; Ellert & Bettany, 1995; Wendt & Hauser, 2013). For the ESM approach, Equations (2) and (3) were used with a fitted cubic spline curve (Wendt & Hauser, 2013) to provide estimates of the cumulative ESM for a layer of soil mass 0–3,000 Mg/ha (SOC_ESM). The SOC mass for both methods was then scaled up to Mg/ha using the percentages relating to the representative area covered by each core location.

\[
\text{M}_{\text{soil(DL)}} = \left( \frac{\text{M}_{\text{sample}}}{\text{A}_{\text{sample}}} \right) \times 10^4
\]

where \(M_{\text{soil(DL)}}\) is the mass of soil in the depth layer (Mg/ha), \(M_{\text{sample}}\) is the dried mass of the soil core sample (g), \(A_{\text{sample}}\) is the area of the core sample (mm²) and 10⁴ is the conversion factor from g/mm² to Mg/ha.

\[
\text{SOC}_{\text{ESM}} = \left( \frac{\text{M}_{\text{soil(DL)}} \times \text{SOC}_{\text{cont}}}{10^4} \right) / 1000
\]

where \(\text{SOC}_{\text{ESM}}\) is the SOC mass in the sample soil mass layer (Mg/ha), \(\text{M}_{\text{soil(DL)}}\) is the mass of soil in the depth layer (Mg/ha; Equation 2), \(\text{SOC}_{\text{cont}}\) is the concentration of organic C (kg/Mg) from Equation (1) and 1,000 is the conversion factor from kg/ha to Mg/ha.

The carbon content of 5 mg of untreated milled soil and 2 mg of below-ground biomass was measured using an ECS 4010 (Costech Analytical Technologies Inc., CA) elemental analyser. Soil and below-ground biomass \(\delta^{13}\text{C}\) was measured using a Picarro Cavity Ringdown Spectrometer G2131-i (Picarro Inc., CA) coupled to the ECS 4010 using a Picarro Caddy split-flow interface (Balslev-Clausen, Dahl, Saad, & Rosing, 2013), and cane (−11.64‰) and beet sugar (−26.03‰; Iso-Analytical, Crewe, UK) were used as isotopic reference standards. \(\delta^{13}\text{C}\) was defined by Equation (4):

\[
\delta^{13}\text{C} = \left( \left( \frac{^{13}\text{C}}{^{12}\text{C}} \right) / \left( \frac{^{13}\text{C}_{\text{PDB}}}{^{12}\text{C}_{\text{PDB}}} \right) \right) - 1 \times 1000
\]

where \(\frac{^{13}\text{C}}{^{12}\text{C}_{\text{PDB}}}\) is the isotopic ratio of the Pee Dee Belemnite standard material (0.0112372) and \(^{13}\text{C}_{\text{PDB}}\) is the isotopic ratio of the measured below-ground biomass or soil sample.

Miscanthus contribution to soil carbon (\(C_{\text{mis}}\)) at T₆ and T₁₂ was calculated using Equation (5):

\[
C_{\text{mis}} = \left( \delta_n - \delta_0 / \delta_r - \delta_0 \right)
\]

where \(\delta_0\) is the soil carbon isotope abundance at T₀, \(\delta_n\) is the abundance at T₆ or T₁₂ and \(\delta_r\) is the abundance of the below-ground biomass at T₆ or T₁₂ (Balesdent, Mariotti, & Guillet, 1987).

2.4 | Modelling

The ECOSSE model (Smith et al., 2010) was run from the conversion year in 2005 and projected to 2020 using the ‘limited data site simulation’ mode for a continued grassland scenario and a LUC from grassland to \(M_{\text{xg}}\) scenario.

A default water table depth of 3 m with drainage class 2 was used. Soil texture percentages were sand 58%, silt 24% and clay 18% with a soil pH of 6 (Zatta et al., 2014). Long-term monthly averages for precipitation and air temperature as well as monthly 2005–2011 data were taken from the nearby (~0.7 km) Gogerddan weather station (Gogerddan weather station). As data were not available from this station for the years 2012–2016, meteorological data to cover this period were taken from another station approximately ~3.5 km distance (McCalmont, McNamara, Donnison, Farrar, & Clifton-Brown, 2017). Monthly potential evapotranspiration from 2005 to 2016 was calculated using data from both weather stations using the R (R Core Team, 2015) package ‘Evapotranspiration’ (Guo & Westra, 2016). Meteorological conditions from 2016 to 2020 were predicted by ECOSSE using the long-term monthly averages. For the continued grassland land use scenario, the values for initial carbon content (77 Mg C/ha) and bulk density (1.14 g/cm³ and 1.11 g/cm³ for the 0–15 and 15–30 cm depths, respectively) were taken from Zatta et al. (2014), along with a yearly plant yield of 8 Mg dry matter (DM)/ha.
based on average values for this area given in Smit, Metzger, and Ewert (2008).

For the grassland to *M. xg* LUC scenario, the initial carbon content (78.8 Mg C/ha) was based on the value in Zatta et al. (2014) which included inputs from the herbicide-killed pasture. All other initial details for the grassland and *M. xg* land use remained the same with the exception of the bulk density under *M. xg* which was taken from T6 data (1.08 g/cm³ and 1.13 g/cm³ for the 0–15 and 15–30 cm depths respectively; Zatta et al., 2014).

Input of C to the soil from crop residue and below-ground biomass is calculated by Ecosse as a function of net primary production (NPP) modified by empirical parameters within the model relating to each plant type (e.g. to account for harvest offtake). Further details can be found in Smith et al. (2010) and Dondini, Richards, Pogson, McCalmont et al. (2016b). Briefly, plant inputs enter the soil as a resistant plant material (RPM) and as a decomposable plant material (DPM) with a DPM:RPM ratio set depending on land use category (e.g. grassland or *Miscanthus*). There are five pools of soil organic matter (SOM) that each decompose at a specific rate constant and are sensitive to soil and climate data. There are specific C and N cycles within the model for grassland and *Miscanthus*. Decomposition is simulated by a number of equations into either BIO (‘biomass’ or active organic matter) or HUM (‘humus’ or more slowly turning over soil organic matter) pools, with inert organic matter (IOM) not contributing to the decomposition processes. In LUC scenarios, protected SOM (soil organic matter) is released from HUM to DPM and RPM. For the LUC to *M. xg* scenario, NPP (Table 1) was calculated from the spring-harvested yield (P. Robson & J. Clifton-Brown, unpublished data) plus 33% to account for over-winter ripening loss (primarily leaf litter drop, based on the relationship outlined in Clifton-Brown et al., 2007) and 20% to account for below-ground biomass gain (estimated from the weight of oven-dried coarse roots and rhizomes sampled over a 4 year period from a nearby established *M. xg* plantation (J.P. McCallmont, unpublished data). As in Zatta et al. (2014), for the conversion year, 1.5 Mg DM/ha was added to account for the input from the herbicide-sprayed pasture and an estimated NPP of 16 Mg DM/ha (approximate mean NPP for years 11 and 12) was used for the projected growing seasons (2017–2020), when yields are expected to reduce towards the end of the commercial crop lifespan (Clifton-Brown et al., 2015; Larsen, Jørgensen, Kjeldsen, & Lærke, 2014).

The root mean square error (RMSE) and relative error (RE) were used to evaluate the accuracy of the model outcomes compared to estimates of SOC derived from soil cores at T6 and T12.

### 2.5 Global warming potential

The difference between the Ecosse-predicted grassland and *M. xg* SOC at the end of 2020 (15 years after LUC) was converted from Mg C/ha to Mg CO₂-ep/ha using the molecular weight (IPCC, 2007). This was converted to a GWP (g CO₂-ep/MJ) using an estimated cumulative yield for a 15 year period of 180 Mg DM/ha (Larsen et al., 2014) and an energy content of 17.95 GJ/Mg DM (Felten, Fröba, Fries, & Emmerling, 2013). This GWP, relating to the difference in SOC, is compared and added to a previously published LCA value for *Miscanthus* cultivation, 4.4 g CO₂-ep/MJ (Hastings et al., 2017), that excluded changes in SOC stocks but included the entire supply chain (propagation, harvest, pelleting and transport) with a *Miscanthus* higher heating value of 18 GJ/Mg DM (Collura, Azambre, Finqueneisel, Zamny, & Weber, 2006; Hastings et al., 2017).

To consider the inclusion of other GHG costs relating to the LUC, the carbon cost of increased soil N₂O emissions over the establishment to *Miscanthus* (4.13 Mg CO₂-ep/ha [8.83 kg N₂O-N/ha], Holder et al., 2019), and reversion process back to grassland (3.41 Mg CO₂-ep/ha [7.29 kg N₂O-N/ha], McCallmont et al., 2018), were converted to g CO₂-ep/MJ using the cumulative 15 year yield. In both N₂O studies, no fertilizer was used during the *Miscanthus* management or LUC, and emissions were estimated from weekly (over a 20 month period, McCallmont et al., 2018) or biweekly (over an 18 month period, Holder et al., 2019) static chamber sampling.

### 2.6 Data analysis

Data analysis was performed in R version 3.5.1 (R Core Team, 2015), and model assumptions were tested using the

| Growing season | NPP (Mg/ha) |
|----------------|-------------|
| 2005           | 1.9         |
| 2006           | 2.2         |
| 2007           | 16.7        |
| 2008           | 23.2        |
| 2009           | 21.2        |
| 2010           | 22.0        |
| 2011           | 26.3        |
| 2012           | 22.9        |
| 2013           | 21.7        |
| 2014           | 18.3        |
| 2015           | 14.3        |
| 2016           | 19.3        |
| 2017–2020      | 16.0        |
Levene's and Shapiro–Wilk tests. At T₀, the mean of the five soil core samples per plot was used to provide one value for each plot sampled. At T₆ and T₁₂, the three cores samples per plot were scaled (as detailed in the methods) and added together to give one value per plot.

To assess the effect of LUC on soil carbon stock, mean block level T₀ SOC was compared to mean block level T₆ and T₁₂ SOC using a linear mixed-effect model from package ‘nlme’ (Pinheiro et al., 2017) with time point as the fixed factor (T₀, T₆, T₁₂), the random effect of block and an autocorrelation structure (AR1). Data were subsequently split into two groups (T₀ with T₆, and T₀ with T₁₂) to allow the influence of hybrid on changes in total scaled SOC stock compared to pre-conversion values (T₀). Land use (Mxg, Hyb 1–4 and pre-conversion grassland) was used as a fixed factor with the random effect of block. Model results were summarized using type III ANOVA (package ‘car’, Fox & Weisberg, 2011) and Tukey HSD (package ‘multcomp’, Hothorn, Bretz, & Westfall, 2008) post hoc tests.

Miscanthus C percentage contribution (C mis) data were split into 0–15 cm and 15–30 cm depths. Data for the 15–30 cm depth were log transformed to improve residuals. The contribution of C mis to the total SOC stock was then explored with the hybrid, time points (T₆, T₁₂) and sampling positions (Ci, Cc, Ce) included as fixed factors with the random effect of block.

Below-ground biomass for each depth and sample position was analysed separately using non-parametric paired Wilcoxon tests as residuals were not significantly improved using transformations. Correlations between SOC and C mis versus below-ground biomass, and SOC versus ripening loss were completed using the linear model function.

3 | RESULTS

3.1 | Soil organic carbon

Mean SOC (0–30 cm depth) at T₁₂ was 71 ± 1 (standard error [SE]) Mg/ha, (SOCₑ₆ 67 ± 1 (SE) Mg/ha, for a reference soil mass layer of 0–3,000 Mg/ha). Soil bulk density results for each time point are summarized in Table 2. SOC was effected by year (χ² (2) = 16.52, p < 0.001) with post hoc testing showing that both T₆ and T₁₂ were significantly lower than T₀ (79 ± 1 (SE) Mg/ha), but that T₁₂ was not significantly different to T₆ (71 ± 1 (SE) Mg/ha). However, in subsequent analysis by hybrid, the difference to T₀ is only significant (p < 0.05) for Hyb 2 (Figure 1). Between T₆ and T₁₂ SOC, both reduced in 0–15 cm layer and increased in the 15–30 cm layer by 4 Mg/ha (χ² (1) = 18.08, p < 0.0001).

Miscanthus contribution (C mis) to SOC in the 0–15 cm layer (Figure 2a) was effected by sample position (χ² (2) = 19.78, p < 0.001) decreasing with distance from the plant centre. However, at T₁₂, C mis was spread out more evenly across the three sampling positions than at T₆ (χ² (2) = 8.08, p = 0.02). In contrast, in the 15–30 cm layer, C mis was similar in all positions (Figure 2b), although it decreased with Hyb 2 and Hyb 4 by 2% (χ² (4) = 22.36, p < 0.001).

3.2 | Biomass

The distribution of below-ground biomass (roots and rhizome) also changed from T₆ to T₁₂ with outward spread from the original planting position towards the inter-row in the upper soil depth (Figure 3).

At the 0–15 cm depth, below-ground biomass was only reduced at position Cc (p = 0.02) between time points T₆ and T₁₂ (by 37 ± 10 (SE) Mg/ha), whereas there was a reduction in all positions in the lower 15–30 cm layer (p < 0.05; Figure 3).

No correlation was found between below-ground biomass and SOC at T₁₂ as was found in T₆ (Zatta et al., 2014). However, C mis was positively and significantly correlated with below-ground biomass at both time points (r = 0.67 at

| TABLE 2 | Soil bulk density for the two soil depths at each sampling occasion (T₀ and T₆ from Zatta et al., 2014) |
|---------------|-------------|-------------|-------------|
| Depth (cm)    | T₀         | T₆         | T₁₂         |
| 0–15          | 1.14       | 1.08       | 1.04        |
| 15–30         | 1.11       | 1.13       | 1.21        |

FIGURE 1 Soil organic carbon (SOC) in 0–15 and 15–30 cm depths, pre-conversion (T₀) from grassland to Miscanthus x giganteus (Mxg) and four Miscanthus hybrids (Hyb 1–4), 6 years after conversion (T₆) and 12 years after conversion (T₁₂). Error bars show the standard error of the mean for the total 0–30 cm values, and the same letter indicates non-significant difference (p > 0.05).
and $r = 0.65$, $p < 0.0001$ at T12) in the upper 0–15 cm soil depth (Figure 4). Roots were not separated from rhizome in T6 or T12 but only small fragments of rhizome were found in samples from the lower depth at both time points.

**FIGURE 4** Miscanthus-derived soil carbon as a percentage of total soil organic carbon (SOC) against below-ground biomass for hybrids Miscanthus x giganteus (Mxg) and Hyb 1–4. Data includes all sample positions in the 0–15 cm soil layer at 12 years after planting T6; and $r = 0.65$, $p < 0.0001$ at T12) in the upper 0–15 cm soil depth (Figure 4). Roots were not separated from rhizome in T6 or T12 but only small fragments of rhizome were found in samples from the lower depth at both time points.

Hyb 4 had the greatest reduction in below-ground biomass in the 0–15 cm soil depth between time points (−14 ± 12 mg/cm³, T6 to T12) and also had the highest percentage inputs from ripening losses (leaf/litter drop; 36%, Table 3).

Hyb 2 had the lowest over-winter ripening loss although no significant difference was found between ripening loss for the different Miscanthus hybrids. Ripening loss was positively, but not significantly, correlated with change in SOC (between T0 and T12) in the 0–15 cm depth layer ($r = 0.77$, $p = 0.13$, Figure 5).

### 3.3 | Modelling

Measured SOC at T6 and T12 was within the 95% confidence interval (CI) of the ECOSSE model predictions for all the hybrids. For the LUC from grassland to Mxg scenario, the model RMSE of 5.49% was within the RMSE 95% CI of 9.67%, and the RE of 5.41% was within the RE 95% CI of 9.62% (based on soil core results from T6 and T12).

At the beginning of the 15 year simulation, the LUC to Mxg scenario shows slightly higher SOC than the continued grassland scenario (reflecting the higher initial C value used). After this, there is a clear drop in levels of SOC under Miscanthus before they begin to level out. After 15 years, the predicted loss compared to T0 was 12 Mg/ha; however, the model suggests there is also a slow decline in the SOC under the continued grassland scenario which shows a loss of 7 Mg/ha after 15 years (Figure 6). At the end of 2020, the difference in SOC stocks between the continued grassland scenario, and LUC to Mxg scenario is 4 Mg C/ha.
Life cycle analysis

The carbon cost relating to the difference in predicted SOC stocks between the continued grassland and LUC to Mxg scenarios of 4 Mg C/ha (or 15 Mg CO₂‐eq/ha) equates to 5 g CO₂‐eq/MJ based on the energy content of the estimated 15 year yield. This represents a 125% increase when added to a previous LCA that excluded soil carbon changes (Table 4).

### 4 DISCUSSION

#### 4.1 Total soil organic carbon

In the light of concerns over the impact on soil carbon when planting bioenergy crops into grassland (McCalmont, Hastings, et al., 2017; Whitaker et al., 2018), this study has shown a 10% loss in SOC after 12 years of LUC change from this temperate marginal grassland to Miscanthus at this site. In this new analysis, unlike Zatta et al. (2014), we did find a reduction in soil carbon stock at T₆ compared to T₀, but the breakdown by hybrid confirmed that the difference was only significant for a single hybrid (at T₆ and T₁₂; Figure 1). The overall reduction in carbon from T₀ to T₁₂ of 8 Mg/ha, is within the range +4 to −9 Mg/ha reported in other grassland

### TABLE 3 Change in below‐ground (BG) biomass and Miscanthus‐derived soil carbon (as a percentage of total soil organic carbon [SOC]) at 0–15 cm depth after 6 (T₆) and 12 (T₁₂) years of land conversion from grassland to Miscanthus. Biomass and Cₘₛₐ differences are taken from mean values across all three sampling positions (Cc, Ce, Cᵢ). Above‐ground ripening loss is the difference between autumn peak and spring harvest yields. The standard error is shown in brackets.

| Hybrid | BG biomass (mg/cm³): Difference T₆ to T₁₂ | Cₘₛₐ (% of SOC): Difference T₆ to T₁₂ | Above‐ground ripening loss (%) |
|--------|------------------------------------------|--------------------------------------|-------------------------------|
| Mxg    | +4 (±10)                                  | +10 (±2)                             | 26 (±9)                       |
| Hyb 1  | +6 (±6)                                   | +10 (±3)                             | 31 (±4)                       |
| Hyb 2  | −4 (±9)                                   | +5 (±1)                              | 19 (±1)                       |
| Hyb 3  | +4 (±12)                                  | +8 (±3)                              | 25 (±8)                       |
| Hyb 4  | −14 (±12)                                 | +7 (±3)                              | 36 (±4)                       |

### FIGURE 5 Correlation between change in T₀ and T₁₂ mean soil organic carbon (SOC) and estimated ripening loss at the 0–15 cm depth for hybrids Miscanthus x giganteus (Mxg) and Hyb 1–4.

### FIGURE 6 Results of the 15 year (2005–2020) ECOSSE simulation of soil organic carbon (SOC) under a continued grassland scenario (grassland) and a land-use change (LUC) from grassland to Miscanthus x giganteus (Mxg) scenario. Mean SOC from soil cores taken immediately pre-conversion (T₀) and from under Mxg in 2011 and 2017 are shown with error bars indicating the 95% confidence intervals.

### TABLE 4 Global warming potential (GWP) over a 15 year crop lifetime of the estimated carbon costs associated with the Miscanthus production chain, predicted difference in soil organic carbon (SOC) stocks (compared to a grassland counterfactual), and estimated increases in soil nitrous oxide (N₂O) emissions related to the land conversion and reversion.

| Cost association                  | GWP (g CO₂‐eq) | GWP (sum, g CO₂‐eq) |
|-----------------------------------|----------------|---------------------|
| Production chain (Hastings et al., 2017) | 4              |                      |
| Difference in SOC                 | 5              | 9                   |
| Establishment N₂O (Holder et al., 2019) | 1              | 10                  |
| Reversion N₂O (McCalmont et al., 2018) | 1              | 11                  |
to *Miscanthus* field-based studies (Clifton-Brown et al., 2007; Schneckenberger & Kuzyakov, 2007; Zang et al., 2018; Zimmermann et al., 2012). There was also no difference between carbon stocks at the two sampling points (T$_6$ and T$_{12}$) suggesting a reasonably stable carbon state. However, this is in contrast to Zang et al. (2018) where soil organic matter increased between sampling occasions (9 and 21 years after *Miscanthus* planting). This difference may be as a result of different soil pH and nutrient levels, or the slightly cooler (annual average air temperature 6.7°C vs. 10.4°C) and wetter (annual average precipitation 1,074 mm vs. 654 mm) climate in this study, which could all influence *Miscanthus*-derived carbon (Zimmermann et al., 2012).

The initial tillage and planting of the cover crop in this study occurred in the autumn (October 2004) before the T$_0$ samples were taken in early May 2005 (prior to *Miscanthus* planting). It is therefore possible that if the original sampling had taken place in the autumn, estimated SOC stock may have been higher. Tillage results in releases of SOC due to the change in conditions that are created in the soil matrix and the creation of newly available substrate that can stimulate soil bacteria/microbial activity and decomposition rates. However, initial increases in CO$_2$ immediately following autumn ploughing have mainly been attributed to the release of soil CO$_2$ from large soil pores and from the release of dissolved CO$_2$ from soil water, and there is generally a lag time before CO$_2$ from bacterial decomposition of soil organic matter and SOC is released (Reicosky & Lindstrom, 1995). Turnover times for light fraction SOC are generally in terms of months to years (Post & Kwon, 2000) and are connected to soil moisture and temperature, with temperature increases stimulating turnover (La Scala Jr. et al., 2008). During the winter months following tillage at this experimental site, microbial activity and decomposition could be expected to be slow, due to low air and soil temperatures (mean air temperature at the site October 2004–April 2005 was 8°C) and therefore changes in SOC from October to April minimal. Baseline soil carbon stocks at our site were also remarkably similar to another nearby periodically re-seeded grassland site used for a land use transition experiment (see McCalmont, McNamara, et al., 2017), which contained 79 Mg C/ha in the top 30 cm. Results presented here might, therefore, be assumed to be reasonably representative of land use transitions on these typical improved marginal grassland systems in the United Kingdom. Grasslands with deeper soils have shown contrasting changes to SOC following LUC to *Miscanthus*. In empirical studies that sampled soils to a depth of 1 m across a range of soil types, Rowe et al. (2016) found that significant SOC losses were only found in the top 30 cm, whereas Qin et al. (2016) found SOC was generally increased in the top 30 cm. However, both studies conclude that taken over the whole 1 m soil profile SOC was not significantly lost. In some cases, surface losses were offset by increases lower in the profile and in others changes were limited to the surface and therefore impacts were diluted when considered over the whole depth. Impacts may also be different for longer term, semi-natural grassland sites where initial carbon stocks may be higher (Guo & Gifford, 2002).

### 4.2 Miscanthus derived carbon and spatial distribution

C$_{\text{mis}}$ mirrored the ground cover survey and below-ground biomass found (with the spreading of *Miscanthus* into the outer C$_1$ and C$_2$ sampling positions) supporting the use of multiple core sampling when scaling up from small samples to Mg/ha (Neukirchen, Himken, Lammel, Czyziona-Krause, & Olfs, 1999). The land-use change is clearly seen in the increase of C$_{\text{mis}}$ between T$_0$ and T$_6$. Although new *Miscanthus* C$_4$ carbon replaced pre-existing C$_3$ carbon, this was not at a high enough rate to completely offset losses by the end of year 12. The impact of LUC on SOC generally differs with soil depth (Poeplau & Don, 2014; Rowe et al., 2016; Zang et al., 2018). In this study, it was found that between T$_6$ and T$_{12}$ C$_{\text{mis}}$ increased in the top layer, although SOC also declined (Figure 2). A higher percentage of C$_{\text{mis}}$ in the topsoil (0–10 cm) compared to deeper soil layers is in accordance with findings by Poeplau and Don, (2014) and Hu, Schäfer, Duplay, and Kuhn (2018). This is likely to be attributed to the distribution of the main *Miscanthus* root and rhizome biomass, which are concentrated in the upper layer (Figure 3) and positively correlated to C$_{\text{mis}}$ at T$_6$ and T$_{12}$ (Figure 4). However, SOC also declined in this upper layer, which may be in part attributed to the 'priming effect' where increased microbial activity (stimulated by ploughing and an increase in accessible C generated from higher plant biomass, root exudates and litter) leads to the use of more stable soil carbon (Cheng, 2009; Hopkins et al., 2013; Kuzyakov, 2010). In contrast, between T$_6$ and T$_{12}$, SOC in the lower 15–30 cm depth increased despite C$_{\text{mis}}$ remaining at a similar level (Figure 2). The reason for this difference is unclear, but it may be a legacy of the cultivation where although ploughing could be expected to add C$_3$ inputs from dead roots/residues in both soil depths there are slower turnover rates at the lower 15–30 cm layer due to the higher bulk density (Table 2) resulting in less aeration for microbial activity. The increase in SOC in this lower layer was only seen at the plant edge and inter-row positions where there is also the increased possibility of weeds providing C$_3$ inputs to the soil, but further research would be needed to confirm these possibilities.

### 4.3 Influence of hybrid

Despite the novel hybrids (Hyb 1–4) having lower lignin content than *Mxg*, and three out of the four novel hybrids having
a lower C:N ratio, the influence of hybrid was small. This is in contrast to the suggestion made in Zatta et al. (2014) that after a longer time period differences in the SOC levels for the hybrids would reflect differences in carbon partitioning. All five hybrids sequestered similar amounts of Cmis and only Hyb 2 had lower overall SOC compared to the baseline (at T6 and T12). Therefore, this study suggests that for this type of interspecies hybrid (M. sacchariflorus x M. sinensis) the potential of yield improvements are not generally at the cost of soil carbon losses compared to the commercial standard Mxg. However, investigation into differences in the chemical and physical properties of the root biomass of Hyb 2 may provide more insights.

Leaf litter inputs to the soil are an important part of carbon cycling (Amougou et al., 2011) and we found that Hyb 4, which lost the most below-ground biomass between T6 and T12, also had the highest ripening loss which may have acted as compensation. Hyb 2, the only hybrid with significantly lower SOC than at T6, also had low ripening loss inputs (Table 3). The correlation between ripening loss and change in SOC found in this study after 12 years (Figure 5), although not significant is in line with the prediction from the RothC model in Zatta et al. (2014) where ripening loss for each hybrid was correlated to projected SOC in 2025.

### 4.4 Modelling

The ECOSSE model predicted SOC under Miscanthus within the statistical error of the field measurements and no bias was found. However, SOC under Mxg projected to 2020 with ECOSSE (66 Mg C/ha) is less than was predicted using the RothC model (72 Mg C/ha, Zatta et al., 2014). The initial drop in SOC following land use conversion to Mxg is greater with ECOSSE, which may be attributed to the LUC routine within ECOSSE which aims to simulate carbon loss from cultivation. Differences in predictions may also be as a result of differences in weather data used in the two models after 2011. However, both models predicted the SOC to within the 95% CIs at T6 and T12 when soil core samples were taken. Although the model can be run using different yield results for the novel hybrids, differences in decomposition rates for above- and below-ground biomass would allow for greater accuracy in comparisons of genotypic differences.

In this work, it was not possible to compare samples from maintained grassland at the same site or within an acceptable distance but the ECOSSE model suggests SOC under continued grassland also has a steady decline of 7 Mg/ha over 15 years (Figure 6). It should not therefore be assumed that even without any cultivation (whether to Miscanthus or a new grass ley) SOC would remain the same as baseline levels over time. UK wide surveys recording trends in soil carbon over time (at the 0–15 cm depth) have also reported significant reductions (~6%) in soil carbon under managed fertile grasslands between 1998 and 2007 (Bellamy, Loveland, Bradley, Lark, & Kirk, 2005; Emmett et al., 2010). These losses may be attributable to a number of factors including climate change and changes in management methods resulting in more efficient harvesting and a reduced use of organic manures (Bellamy et al., 2005; Smith et al., 2007). The grassland scenario is run with the same yearly biomass yield input, whereas changes in weather and management would in reality impact on yields, and hence carbon inputs, resulting in differences in SOC.

The difference in predicted SOC between the LUC change and continued grassland scenarios (~6%, at 2020, the end of the estimated Mxg crop lifetime) was within the range of −48% to +15% found for eight established (>5 years) Miscanthus plantations compared to neighbouring grassland sites (Rowe et al., 2016). The contrasting results for the different sites within Rowe et al. (2016), along with the results of this study, show that significant losses in SOC can occur, and while Qin et al., (2016) found no overall change in SOC in relation to grassland to Miscanthus conversions, CIs ranged from −9% to +21% (for the mean of five datasets reflecting the change in SOC in Miscanthus crops >10 years).

### 4.5 Global warming potential impacts

Soil sustainability is an important consideration when assessing the impacts of potential LUC scenarios (; Hillier et al., 2009). In this long-term LUC study where initial SOC stocks are similar to that expected for temperate grasslands in this climate (; Kiely et al., 2009; McCalmont, McNamara, et al., 2017), we have seen decreases in SOC (compared to baseline levels, and between modelled predictions of grassland and Miscanthus), which more than doubled a production cost LCA result (Table 4). Similarly soil N2O emissions during crop establishment and reversion to the next crop have recently been shown to represent a significant portion of the greenhouse gas balance (Holder et al., 2019; McCalmont et al., 2018).

The starting Miscanthus production GWP figure used of 4 g CO2-eq/MJ from Hastings et al. (2017) does not include changes in soil carbon stocks or soil greenhouse gas fluxes, based on the premise that on average C would be sequestered or at worst maintained. However, when the cost of change in soil carbon (4 Mg C/ha, 5 g CO2-eq/MJ, compared to a continued grassland counterfactual), along with the cost of soil N2O emissions from land conversion (1 g CO2-eq/MJ, Holder et al., 2019) were added to the original GWP, the resulting
cost of producing a *Miscanthus* crop over a 15 year period (10 g CO₂-eq/MJ or 180 kg CO₂-eq/Mg DM) still remained far lower than estimates for producing energy from natural gas (59 g CO₂-eq/MJ), currently the highest consumed fossil fuel energy source (BEIS, 2018), and coal (121 g CO₂-eq/MJ; Hastings et al., 2017).

Whether the bioenergy crop itself should bear the greenhouse gas cost of land conversion at the beginning of the cropping cycle (Holder et al., 2019) or reversion at the end (McCalmont et al., 2018), or indeed both, is open to debate. It may also be the case that any losses in SOC are temporary depending on the LUC after *Miscanthus*, if for example the land is re-converted to a permanent pasture. As shown in McCalmont et al. (2018) soil N₂O emissions connected to cultivation disturbances are strongly driven by the legacy of the previous crop species, and losses or gains in soil carbon are also sensitive to the initial land condition (Qin et al., 2016; Richards et al., 2017) suggesting a case for LCA studies to attribute conversion period greenhouse gas emissions to the previous crop and incorporate projected reversion costs into the GWP balance of the current one.

**ACKNOWLEDGEMENTS**

The trial used was planted with support from the Department for Environment Food and Rural Affairs (DEFRA, NF0426). This work was funded through the Institute of Biology, Environment and Rural Sciences (IBERS), Aberystwyth University and Biotechnology and Biological Sciences Research Council (BBSRC) strategic funding for work on *Miscanthus* (grant numbers BB/CSP1730/1 and BBS/E/ W10963A01B), and Engineering and Physical Sciences Research Council-funded MAGLUE project (EPSRC EP/M013200/1). The funding was also received from the European Commission under grant agreement 652615 and implemented under the FACCE SURPLUS ERA-Net as a bioenergy crop: From small beginnings to potential realisation.

**REFERENCES**

Amougou, N., Bertrand, I., Machet, J. M., & Recous, S. (2011). Quality and decomposition in soil of rhizome, root and senescent leaf of *Miscanthus x giganteus*, as affected by harvest date and N fertilization. *Plant and Soil*, 338(1), 83–97. https://doi.org/10.1007/s11104-010-0443-x

Balesdent, J., Chenu, C., & Balabané, M. (2000). Relationship of soil organic matter dynamics to physical protection and tillage. *Soil & Tillage Research*, 53(3–4), 215–230. https://doi.org/10.1016/S0167-1987(99)00107-5

Balesdent, J., Mariotti, A., & Guillet, B. (1987). Natural ¹³C abundance as a tracer for studies of soil organic matter dynamics. *Soil Biology and Biochemistry*, 19(1), 25–30.

Balslev-Clausen, D., Dahl, T. W., Saad, N., & Rosing, M. T. (2013). Precise and accurate ¹³C analysis of rock samples using flash combustion-cavity ring down laser spectroscopy. *Journal of Analytical Atomic Spectrometry*, 28(4), 516–523. https://doi.org/10.1039/c2ja30240c

BEIS (Department for Business Energy & Industrial Strategy). (2018). UK energy in brief. Retrieved from https://www.gov.uk/government/statistics/uk-energy-in-brief-2018

Bellamy, P. H., Loveland, P. J., Bradley, R. I., Lark, R. M., & Kirk, G. J. D. (2005). Carbon losses from all soils across England and Wales 1978–2003. *Nature*, 437(7056), 245–248. https://doi.org/10.1038/nature04038

Cheng, W. (2009). Rhizosphere priming effect: Its functional relationships with microbial turnover, evapotranspiration, and C-N budgets. *Soil Biology and Biochemistry*, 41(9), 1795–1801. https://doi.org/10.1016/j.soilbio.2008.04.018

Clifton-Brown, J. C., Breuer, J., & Jones, M. B. (2007). Carbon mitigation by the energy crop, *Miscanthus*: Global Change Biology, 13(11), 2296–2307. https://doi.org/10.1111/j.1365-2486.2007.01438.x

Clifton-Brown, J. C., & Lewandowski, I. (2000). Water use efficiency and biomass partitioning of three different *Miscanthus* genotypes with limited and unlimited water supply. *Annals of Botany*, 86(1), 191–200. https://doi.org/10.1006/anbo.2000.1183

Clifton-Brown, J., Schwarz, K. U., & Hastings, A. (2015). History of the development of *Miscanthus* as a bioenergy crop. From small beginnings to potential realisation. *Biology and Environment*, 115B(1), 1–13. https://doi.org/10.3318/BIOE.2015.05

Collura, S., Azambre, B., Finqueneisel, G., Zimny, T., & Weber, J. V. (2006). *Miscanthus x giganteus* straw and pellets as sustainable fuels. Combustion and emission tests. *Environmental Chemistry Letters*, 4, 75–78. https://doi.org/10.1007/s10311-006-0036-3

Conant, R. T., Easter, M., Paustian, K., Swan, A., & Williams, S. (2007). Impacts of periodic tillage on soil C stocks: A synthesis. *Soil and Tillage Research*, 95(1–2), 1–10. https://doi.org/10.1016/j.still.2006.12.006

Dondini, M., Jones, E. O., Richards, M., Pogson, M., Rowe, R. L., Keith, A. M., … Smith, P. (2015). Evaluation of the ECOSSE model for simulating soil carbon under short rotation forestry energy crops in Britain. *GCB Bioenergy*, 7(3), 527–540. https://doi.org/10.1111/gcbb.12154

Dondini, M., Richards, M., Pogson, M., Jones, E. O., Rowe, R. L., Keith, A. M., … Smith, P. (2016a). Evaluation of the ECOSSE model for simulating soil organic carbon under *Miscanthus* and short rotation coppice-willow crops in Britain. *GCB Bioenergy*, 8(4), 790–804. https://doi.org/10.1111/gcbb.12286

**ORCID**

Amanda J. Holder https://orcid.org/0000-0002-5355-2525

John Clifton-Brown https://orcid.org/0000-0001-6477-5452

Paul Robson https://orcid.org/0000-0003-1841-3594

Jon P. McCalmont https://orcid.org/0000-0002-5978-9574
Dondini, M., Richards, M. I. A., Pogson, M., McCalmont, J., Drewer, J., Marshall, R., … Smith, P. (2016). Simulation of greenhouse gases following land-use change to bioenergy crops using the ECOSSE model: A comparison between site measurements and model predictions. *GCB Bioenergy*, 8(5), 925–940. https://doi.org/10.1111/gcb.12298

Donnison, I. S., & De Neergaard, A. (2018). Agri-environmental indicator – cropping patterns. Second report on the state of the environment in the UK. Retrieved from https://ec.europa.eu/commmission/sites/energyunion/files/2nd-report-state-energy-union_en.pdf

Ellert, B. H., & Bettany, J. R. (1995). Calculation of organic matter and nutrients stored in soils under contrasting management regimes. *Canadian Journal of Soil Science*, 75, 529–538. https://doi.org/10.4141/cjss95-075

Emmett, B. A., Reynolds, B., Chamberlain, P. M., Rowe, E., Spurgeon, D., & Brittain, S. A. … Woods, C. (2010). *Countryside survey: Soils report from 2007*. Technical report no. 9/07.

European Commission. (n.d.). 2030 climate & energy framework. Retrieved from https://ec.europa.eu/energy/policies/strategies/2030_en

European Commission. (2017). *Second report on the state of the energy union*. Retrieved from https://ec.europa.eu/energy/policies/files/2nd-report-state-energy-union_en.pdf

European Commission Joint Research Centre. (n.d.). ESDAC European soil database maps. Retrieved from https://esdac.jrc.ec.europa.eu/resource-type/european-soil-database-maps#

Eurostat. (2018). Agri-environmental indicator - cropping patterns. Retrieved from http://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_cropping_patterns

Felt, D., Fröba, N., Fries, J., & Emmerling, C. (2013). Energy balances and greenhouse gas-mitigation potentials of bioenergy cropping systems (*Miscanthus*, rapeseed, and maize) based on farming conditions in Western Germany. *Renewable Energy*, 55, 160–174. https://doi.org/10.1016/j.renene.2012.12.004

Foeried, B., De Neergaard, A., & Høgh-Jensen, H. (2004). Turnover of biomass in the UK. Retrieved from https://doi.org/10.3389/fpls.2017.01058

Holden, A. J., McCalmont, J. P., Rowe, R., McNamara, N. P., Elias, D., & Donnison, I. S. (2019). Soil N2O emissions with different reduced tillage methods during the establishment of *Miscanthus* in temperate grassland. *GCB Bioenergy*, 11(3), 539–549. https://doi.org/10.1111/gcbb.12570

Hopkins, F., Gonzalez-Meler, M. A., Flower, C. E., Lynch, D. J., Czimczik, C., Tang, J., & Subke, J.-A. (2013). Ecosystem-level controls on root-rhizosphere respiration. *New Phytologist*, 199(2), 339–351. https://doi.org/10.1111/nph.12271

Hothorn, T., Bretz, F., & Westfall, P. (2008). Simultaneous inference in general parametric models. *Biometrical Journal*, 50(3), 346–363.

Hu, Y., Schäfer, G., Duplay, J., & Kuhn, N. J. (2018). Bioenergy crop induced changes in soil properties: A case study on *Miscanthus* fields in the Upper Rhine Region. *PLoS ONE*, 1–15, https://doi.org/10.1371/journal.pone.0200901

IPCC. (2007). *Climate change 2007: Synthesis report*. Contribution of working groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change. R. K. Pachauri, & A. Reisinger (Eds.). Geneva, Switzerland. Author.

IPCC. (2014). Summary for policymakers. In O. Edenhofer, R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel, & J. C. Minx (Eds.), *Climate change 2014: Mitigation of climate change*. Contribution of working group III to the fifth assessment report of the intergovernmental panel on climate change. Cambridge, UK and New York, NY: Cambridge University Press. https://doi.org/10.1017/CBO9781107415324

Kiely, G., McGoff, N. M., Eaton, J. M., Xu, X., Leahy, P., & Carton, O. (2009). SoilC – Measuring and modelling of soil carbon stocks and stock changes in Irish soils. STRIVE Report Series No. 35.

Kuzmanyakov, Y. (2010). Priming effects: Interactions between living and dead organic matter. *Soil Biology and Biochemistry*, 42(9), 1363–1371. https://doi.org/10.1016/j.soilbio.2010.04.003

Kuzmanyakov, Y., & Domanski, G. (2000). Carbon inputs by plants into the soil. *Review. Journal of Plant Nutrition and Soil Science*, 163(4), 421–431. https://doi.org/10.1002/1522-2624(200008)163

La Scala Jr., N., Lopes, A., Spokas, K., Bolonhezi, D., Archer, D. W., & Reicosky, D. C. (2008). Short-term temporal changes of soil carbon losses after tillage described by a first-order decay model. *Soil and Tillage Research*, 99(1), 108–118. https://doi.org/10.1016/j.still.2008.01.006

Larsen, S. U., Jørgensen, U., Kjeldsen, J. B., & Lærke, P. E. (2014). Long-term *Miscanthus* yields influenced by location, genotype, row distance, fertilization and harvest season. *Bioenergy Research*, 7, 620–635. https://doi.org/10.1007/s12155-013-9389-1

Lewandowski, I., Clifton-Brown, J. C., Scurlock, J. M. O., & Huisman, W. (2000). *Miscanthus*: European experience with a novel energy crop. *Biomass and Bioenergy*, 19, 209–227.

Lewandowski, I., Clifton-Brown, J., Trindade, L. M., van der Linden, G. C., Schwarz, K.-U., Müller-Sámann, K., … Kalinina, O. (2016). Progress on optimizing *Miscanthus* biomass production for the European bioeconomy: Results of the EU FP7 project OPTIMISC.
Richter, G. M., Agostini, F., Redmile-Gordon, M., White, R., & Goulding, K. W. T. (2015). Sequestration of C in soils under Miscanthus can be marginal and is affected by genotype-specific root distribution. *Agriculture, Ecosystems and Environment*, 200, 169–177. https://doi.org/10.1016/j.agee.2014.11.011

Robertson, A. D., Davies, C. A., Smith, P., Dondini, M., & McNamara, N. P. (2015). Modelling the carbon cycle of Miscanthus plantations: Existing models and the potential for their improvement. *GCB Bioenergy*, 7(3), 405–421. https://doi.org/10.1111/gcbb.12144

Robertson, A. D., Whitaker, J., Morrison, R., Davies, C. A., Smith, P., & McNamara, N. P. (2017). A Miscanthus plantation can be carbon neutral without increasing soil carbon stocks. *GCB Bioenergy*, 9(3), 645–661. https://doi.org/10.1111/gcbb.12397

Rowe, R. L., Keith, A. M., Elias, D., Dondini, M., Smith, P., Oxley, J., & McNamara, N. P. (2016). Initial soil C and land-use history determine soil C sequestration under perennial bioenergy crops. *GCB Bioenergy*, 8(6), 1046–1060. https://doi.org/10.1111/gcbb.12311

Schneckenberger, K., & Kuzyakov, Y. (2007). Carbon sequestration under Miscanthus in sandy and loamy soils estimated by natural 13C abundance. *Journal of Plant Nutrition and Soil Science*, 170(4), 538–542. https://doi.org/10.1002/jpln.200625111

Smit, H. J., Metzger, M. J., & Ewert, F. (2008). Spatial distribution of grassland productivity and land use in Europe. *Agricultural Systems*, 98(3), 208–219. https://doi.org/10.1016/j.agsy.2008.07.004

Smith, J., Gottschalk, P., Bellarby, J., Chapman, S., Lilly, A., Towers, W., … Smith, P. (2010). Estimating changes in Scottish soil carbon stocks using ECOSSE. I. Model description and uncertainties. *Climate Research*, 45(1), 179–192. https://doi.org/10.3354/cr010902

Smith, P., Chapman, S. J., Scott, W. A., Black, H. I. J., Wattenbach, M., Milne, R., … Smith, J. U. (2007). Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978–2003. *Global Change Biology*, 13(12), 2605–2609. https://doi.org/10.1111/j.1365-2486.2007.01458.x

Taube, F., Gierus, M., Hermann, A., Loges, R., & Schinbach, P. (2014). Grassland and globalization – challenges for north-west European grass and forage research. *Grass and Forage Science*, 69(1), 2–16. https://doi.org/10.1111/jgs.12043

Welsh Government. (2018). *Farming facts and figures, wales 2017*. Retrieved from https://gov.wales/docs/stats/2017/170620-farming-facts-figures-2017-en.pdf

Wendt, J. W., & Hauser, S. (2013). An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. *European Journal of Soil Science*, 64(1), 58–65. https://doi.org/10.1111/ejss.12002

Whitaker, J., Field, J. L., Bernacchi, C. J., Cerri, C. E. P., Ceulemans, R., Davies, C. A., … McNamara, N. P. (2018). Consensus, uncertainties and challenges for perennial bioenergy crops and land use. *GCB Bioenergy*, 10(3), 150–164. https://doi.org/10.1111/gcbb.12488

Zang, H., Blagodatskaya, E., Wen, Y., Xu, X., Dyckmans, J., & Kuzyakov, Y. (2018). Carbon sequestration and turnover in soil under the energy crop Miscanthus: Repeated 13C natural abundance approach and literature synthesis. *GCB Bioenergy*, 10(4), 262–271. https://doi.org/10.1111/gcbb.12485

Zatta, A., Clifton-Brown, J., Robson, P., Hastings, A., & Monti, A. (2014). Land-use change from C3 grassland to C4 Miscanthus: Effects on soil carbon content and estimated mitigation benefit after six years. *GCB Bioenergy*, 6(4), 360–370. https://doi.org/10.1111/gcbb.12054
Zimmermann, J., Dauber, J., & Jones, M. B. (2012). Soil carbon sequestration during the establishment phase of Miscanthus × giganteus: A regional-scale study on commercial farms using 13C natural abundance. *GCB Bioenergy*, 4(4), 453–461. https://doi.org/10.1111/j.1757-1707.2011.01117.x

**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of the article.