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Soil & Water Assessment Tool (SWAT) simulated hydrological impacts of land use change from temperate grassland to energy crops: A case study in western UK

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Abstract
When considering the large-scale deployment of bioenergy crops, it is important to understand the implication for ecosystem hydrological processes and the influences of crop type and location. Based on the potential for future land use change (LUC), the 10,280 km² West Wales Water Framework Directive River Basin District (UK) was selected as a typical grassland dominated district, and the Soil & Water Assessment Tool (SWAT) hydrology model with a geographic information systems interface was used to investigate implications for different bioenergy deployment scenarios. The study area was delineated into 855 sub-basins and 7,108 hydrological response units based on rivers, soil type, land use, and slope. Changes in hydrological components for two bioenergy crops (Miscanthus and short rotation coppice, SRC) planted on 50% (2,192 km²) or 25% (1,096 km²) of existing improved pasture are quantified. Across the study area as a whole, only surface run-off with SRC planted at the 50% level was significantly impacted, where it was reduced by up to 23% (during April). However, results varied spatially and a comparison of annual means for each sub-basin and scenario revealed surface run-off was significantly decreased and baseflow significantly increased (by a maximum of 40%) with both Miscanthus and SRC. Evapotranspiration was significantly increased with SRC (at both planting levels) and water yield was significantly reduced with SRC (at the 50% level) by up to 5%. Effects on streamflow were limited, varying between −5% and +5% change (compared to baseline) in the majority of sub-basins. The results suggest that for mesic temperate grasslands, adverse effects from the drying of soil and alterations to streamflow may not arise, and with surface run-off reduced and baseflow increased, there could, depending on crop location, be potential benefits for flood and erosion mitigation.

KEYWORDS
bioenergy, evapotranspiration, flooding, hydrology, Miscanthus, short rotation coppice, streamflow
1 | INTRODUCTION

Land use change (LUC) involving different crop types or management can influence ecosystem level hydrological processes. Quantification of these impacts is necessary to inform policy decisions based on trade-offs between a range of potential positive and negative environmental impacts (DeFries & Eshleman, 2004; Foley et al., 2005; Mohr & Raman, 2013). The use of bioenergy crops for renewable energy generation can help to reduce reliance on fossil fuels and attain climate change objectives (Chum et al., 2011; CCC, 2018a). Although large-scale uptake of dedicated energy crops in Europe has been slow to date (Lindegaard et al., 2016), their use as part of the energy generation mix is increasing (BEIS, 2018a) and renewable energy from biomass remains part of international and European climate mitigation policies (CCC, 2018b; IPCC, 2014). In Europe, as part of the long-term strategy and vision for a ‘Climate neutral Europe by 2050’, sustainable expansion of bioenergy crops is likely to target economically marginal lands, avoiding any perceived competition with food crops whilst maximizing returns for land owners (CCC, 2018b; European Commission, 2018). However, the implication of this LUC for ecosystem hydrological processing is not fully understood, particularly for second-generation (non-food) bioenergy crops such as short rotation coppice (SRC; e.g. willow, Salix spp. and poplar, Populus spp.) and perennial grasses (e.g. switchgrass, Panicum virgatum L. and Miscanthus, M. x giganteus).

Temperate grasslands comprise a third of the utilized agricultural area across Europe and present a large potential area for the deployment of energy crops (Eurostat, 2018a). Changes in grazing management and reductions in agricultural subsidies, combined with typically poorer quality soils, are resulting in large areas of grassland becoming economically unprofitable (Donnison & Fraser, 2016; Eurostat, 2018b; Taube, Gierus, Hermann, Loges, & Schönbach, 2014). This is particularly noticeable for European regions such as Wales (UK) with a grass-dominated agricultural landscape and a high proportion of land (80%) designated by the European Commission as ‘Less Favoured Areas’ (LFAs, agriculturally disadvantaged land in terms of soils, relief, aspect or climate, and receiving funding under the European Agricultural Fund for Rural Development, European Commission, n.d.).

Land suitability modelling suggests large areas (2,093 km², 36% of west Wales) are suitable for bioenergy crops Miscanthus and SRC (Lovett, Sün nenberg, & Dockerty, 2014). Ambitious planting rates of up 50 km²/year have also been proposed as attainable with the potential for rural employment and diversification highlighted (ADAS UK Ltd [ADAS] & Energy Technologies Institute [ETI], 2016), which is especially relevant in the light of the uncertain future of UK (and indeed European) agricultural subsidies.

In comparison with grazed grassland, Miscanthus and SRC have the potential to impact on soil hydrological balance through an increased demand for water (Clifton-Brown, Lewandowski, Bangert, & Jones, 2002; Weih & Nordh, 2002), changes in root morphologies impacting water access through the soil profile (Crow & Houston, 2004; Neukirchen, Himken, Lammel, Czyponia-Krause, & Olfs, 1999), differences in leaf development and morphology influencing evapotranspiration and precipitation interception (Finch & Riche, 2010; Holder, Mccalmont, Mccnamara, Rowe, & Donnison, 2018; Stephens, Hess, & Knox, 2001), and taller, stronger stems changing hydraulic resistance to overland flows (Kort, Collins, & Ditsch, 1998; Marshall et al., 2009). As a result, there is generally an increase in evapotranspiration and a reduction in soil water recharge and surface run-off, compared to existing land uses (Holder et al., 2018; Mccalmont et al., 2017; Rowe, Street, & Taylor, 2009). These traits could be of benefit in landscape flood mitigation schemes (Environment Agency, 2015; Stephens et al., 2001) but can alter river flows and environments for aquatic and riparian species (Arthington, Naiman, McClain, & Nilsson, 2010; Poff & Zimmerman, 2010) and adversely affect dry-land areas (Langeveld et al., 2012).

Resulting impacts of LUC to energy crops will be dependent on the extent of the area planted within river catchments and on regional climate, soil type, slope and altitude and stage of crop maturity (Hastings et al., 2014; Stephens et al., 2001; Vanloocke, Bernacchi, & Twine, 2010). This is reflected in previous studies of the impacts of land use conversions involving grassland to Miscanthus and SRC. For example, in modelled conversions from mixed land uses (grassland, corn and soybean) to Miscanthus in different regions of the American Midwest, Cibin, Trybula, Chaubey, and Brouder (2015) found that streamflow was reduced by around 8%, whereas Feng et al. (2018) found a mean reduction in streamflow of 23% (reflecting differing percentages of each land use type and varying topography). For SRC compared to conventional pasture, Hartwich et al. (2016) found that decreases in modelled surface run-off varied from 20% to 78% in their study of the Northern German Plain with regional differences in climate and soils. These differences highlight the need for location-specific modelling for the quantification of the potential impacts, positive or negative, of large-scale bioenergy cultivation.

Hydrology simulation models linked to geographic information systems (GIS) can be used to gauge the effects of different LUC scenarios over varying spatial and temporal scales for specific locations, and a number of different models have been used in connection with biofuel scenarios (Engel et al., 2010; Finch et al., 2004; Vanloocke et al., 2010). The Soil & Water Assessment Tool (SWAT)
is a physically based (i.e. representation of hydrological processes based on known principles of energy and water flux) hydrology model (Arnold, Srinivasan, Muttiah, & Williams, 1998) that can be incorporated into GIS software (Dile, Daggupati, George, Srinivasan, & Arnold, 2016). SWAT has been widely used to assess the impacts on hydrology and water quality of different land use management strategies (Engel et al., 2010) and has been successfully improved and used to represent Miscanthus and SRC crops (Hartwich et al., 2016; Trybula et al., 2015) enabling the use of the model for grassland LUC scenarios in Europe where the implications are unclear.

In this study, we aim to utilize the SWAT model with a GIS interface to quantify how water yield (amount of water leaving the catchment), soil water storage, evapotranspiration, surface run-off, baseflow (groundwater flow) and stream-flow respond to LUC from grassland to Miscanthus and SRC in a typical temperate agricultural grassland region at two planting levels: an ambitious ‘maximum’ (50% of available improved pasture) and more ‘limited’ (25% of improved pasture) level. Differences in responses between planting levels and bioenergy crop are also considered.

### 2 | MATERIALS AND METHODS

#### 2.1 | West Wales River Basin and model description

The West Wales Water Framework Directive River Basin District (area 10,280 km²), hereafter referred to as the watershed (Figure 1; Environment Agency, 2014), is located in the western part of the UK and was chosen as a temperate region of Europe dominated by grass-based agriculture and classed agriculturally as an ‘LFA’.

Hydrology for the watershed was modelled using the QSWAT v1.5 (rev. 664) extension with QGIS software (QGIS, 2014) and SWAT 2012 Editor interface (Arnold et al., 1998; Dile et al., 2016). A physical description of the watershed within the model (representing the baseline scenario of existing land use and conditions) was built up using the GIS layers detailed in Table 1.

The United Nations Food and Agriculture Organisation’s map (UNFAO, 2003), showing dominant soil types, was matched to the soil types given in the British Geological Survey soils map (British Geological Survey Materials, 2018) and the SWAT database soil codes. The watershed consists of mainly loamy soils with varying amounts of clay, silt and sand. Dystric Cambisols account for 50% of the area, Dystric Gleysols 23% and Gleyic Cambisols 19%. The remainder consists of small areas of Podzol (5%) and Humic Gleysols (2%). The watershed is predominately made up of low quality agricultural land (Welsh Government, n.d.), 40% of the watershed is >15% slope and 42% is >200 m a.s.l. (Ordnance Survey, 2018). The dominant agricultural land is improved grass pasture (52%), with only 4% of the area designated as arable or horticulture. Urban areas account for 3% of the watershed with the remainder of the land cover made up of natural grasslands (19%), woodlands (18%), and small pockets of heath and marsh (4%; Rowland et al., 2017).

The watershed was delineated into 855 sub-basins based on the digital elevation model and river data. Hydrological response units (HRUs) within each sub-basin were divided based on soil type, land use and slope (divided into two bands, above and below 15%). Insignificant HRUs were excluded using the following threshold filters to ignore areas of less than: 10% land use; 20% soil class; and 10% slope band; and redistributed proportionally among those remaining (Dile, Srinivasan, & George, 2018).

Climate data were obtained for 15 years from 1999 to 2013, the most recent period with all required data available (Table 1). The SWAT model was run on a monthly time step for the full duration using 1999 to 2003 as a 5 year warm up period (no results from the warm up period are used in the analysis). Climate data (precipitation, wind, relative humidity and solar radiation) obtained from the National Centers for Environmental Prediction (NCEP, n.d.) were checked...
for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location for accuracy with long-term weather data ranges using four UK Met Office climate stations (Met Office, 2014) located within the watershed (Figure 2). Mean annual precipitation in the watershed from 2004 to 2013 was 1,532 mm (Met Office, n.d.). Potential evapotranspiration (PET) was calculated using data from the circled climate location.
The curve number (CN) method (USDA, 1986) was used in relation to simulation of surface run‐off within the model with adjustments allowed based on the steepness of the slope.

2.2 | Plant growth simulation and management

In order to reflect expected growth rates for the region, plant inputs for the different land cover types were adjusted from the SWAT default values using values from the literature and, in the case of Miscanthus, some data was also obtained from measurements taken at a field-scale trial site within the watershed. The main plant inputs used for the LUC crops and other land use cover plant types are shown in Tables 2 and 3, respectively. Arable agriculture in the watershed was based on typical crops grown in the region: wheat, barley, oats and oilseed rape (Welsh Government, 2018). Woodland biomass at the start of the simulations was input as 153 Mg DM/ha for evergreen forests and 136 Mg DM/ha for deciduous woodland (Forestry Commission, 2011, 2017).

2.3 | Miscanthus field measurements

A number of plant growth input values available in the literature for Miscanthus are based on measurements made in the American Midwest region from fertilized crops. Therefore, to check the suitability for their use in the region simulated in this project, the main Miscanthus growth values were checked using data obtained from an established Miscanthus plantation (~6 ha) located within the watershed. A full description of the field site (planted in 2012) and methods used for biomass sampling are given in McCalmont et al. (2017).

Mean annual harvest yields simulated by the model (14.74 Mg/ha, 2004–2013) were checked against the mean peak autumn yield (14.95 Mg/ha, 2014–2016, J. P. McCalmont, unpublished data) recorded at the site. The value used for radiation use efficiency (BIO_E: 41, Trybula et al., 2015) was found to be similar to an estimate of 42 made using measurements of photosynthetically active radiation and gains in Miscanthus above and belowground biomass between May 2015 and November 2016 (J. P. McCalmont, unpublished data).

### TABLE 2

| Input description                              | Pasture (CRDY) | Miscanthus       | Short rotation coppice |
|------------------------------------------------|----------------|------------------|------------------------|
| Radiation use efficiency \(\text{kg ha}^{-1}/\text{MJ m}^{-2}\) | 10 (Belanger, Gastal, & Warembourg, 1994; Cristiano, Posse, & Bella, 2015) | 42 (Trybula et al., 2015) Measurements | 28 (Bullard, Mustill, Carver, & Nixon, 2002; Linderson, Iritz, & Lindroth, 2007; Verlinden, Broeckx, Bulcke, Acker, & Ceulemans, 2013) |
| Max. stomatal conductance \(\text{m/s}\)       | 0.005 (SWAT: tall fescue) | 0.005 (Beale, Bint, & Long, 1996; Clifton‐Brown & Lewandowski, 2000) | 0.004 (SWAT: poplar) |
| Light extinction coefficient                   | 0 (SWAT: tall fescue) | 0.68 (Clifton‐Brown & Lewandowski, 2000) | 0.5 (Linderson et al., 2007) |
| Max. leaf area index                           | 4 (Asner, Scurlock, & Hicke, 2003) | 11 (Trybula et al., 2015) | 9 (Hartwich et al., 2016; Pellis, Laureysens, & Ceulemans, 2004; Schmidt‐Walter & Lammersdorf, 2012) |
| Min. leaf area index during dormancy           | 0.8 | 0 (Guo et al., 2018; Trybula et al., 2015) | 0.75 (SWAT: poplar) |
| Max. canopy storage (mm)                       | 0 | 2.2 (Stephens et al., 2001) | 2.2 (Schmidt‐Walter & Lammersdorf, 2012; Stephens et al., 2001) |
| Max. canopy height \(\text{m}\)                | 0.75 | 3 Measurements | 8 (Hartwich et al., 2016) |
| Max. root depth \(\text{m}\)                  | 2 (SWAT: tall fescue) | 2.5 (Neukirchen et al., 1999) | 2 (Hartwich et al., 2016) |
| Optimum temperature \(\text{°C}\)              | 15 (SWAT: tall fescue) | 20 | 15 |
| Base temperature \(\text{°C}\)                 | 0 (SWAT: tall fescue; Hurtado‐Uri, Hennessy, Shalloo, O’Connor, & Delaby, 2013) | 8 (Hastings, Clifton‐Brown, Wattenbach, Mitchell, & Smith, 2009) | 5 (Hartwich et al., 2016) |

Abbreviation: SWAT, Soil & Water Assessment Tool.
### Table 3

Main plant growth values used in the simulations for the land use types of arable (AGRL), lawn grass (BERM), natural grassland (FESC), evergreen forest (FRSE), heather/shrub grassland (MIGS), deciduous woodland (OAK), heather (SHRB) and fen/marsh/bog/saltmarsh (WETL). The model input variable name (Code) and references are shown where used (SWAT denotes the SWAT database).

| Description                                      | Code     | AGRL | BERM | FESC | FRSE | MIGS | OAK  | SHRB | WETL |
|--------------------------------------------------|----------|------|------|------|------|------|------|------|------|
| Radiation use efficiency (kg ha⁻¹/MJ m⁻²)        | BIO_E    | 33.5 (SWAT) | 10 (Belanger et al., 1994) | 15 (Belanger et al., 1994; Cristiano et al., 2015) | 15 (SWAT) | 2 (Garbulsky et al., 2010) | 2 (Garbulsky et al., 2010) | 2 (Garbulsky et al., 2010) | 5 (Garbulsky et al., 2010) |
| Max. leaf area index (BLAI)                      | BLAI     | 5 (Asner et al., 2003; AHDB 2018) | 4 (SWAT) | 4 (SWAT) | 6 (Asner et al., 2003) | 4 (Asner et al., 2003) | 6.5 (Asner et al., 2003; ORNL DAAC, n.d.) | 3.5 (Asner et al., 2003; Gonzalez et al., 2013) | 5 (Asner et al., 2003) |
| Max. canopy storage (CANMX) (mm)                 | CANMX    | 0.8 (Wang, Li, & Rao, 2006) | — | 1.2 (Burgy & Pomeroy, 1958) | 3.7 (Hörmann et al., 1996) | 1.5 (Dunkerley, 2000) | 2.3 (Hörmann et al., 1996) | 1.5 (Dunkerley, 2000) | 1.2 (Burgy & Pomeroy, 1958) |
| Optimum temperature (TOPT) (°C)                  | TOPT     | 20 (Finch, Samuel, & Lane, 2002) | 15 (SWAT: FESC) | 15 (SWAT) | 20 | 15 (SWAT: FESC) | 15 (Bequet et al., 2011) | 15 | 15 |
| Base temperature (TBASE) (°C)                    | TBASE    | 5 (Finch et al., 2002) | 0 (SWAT: FESC) | 0 (SWAT) | 0 (SWAT) | 0 (SWAT: FESC) | 5 (Bequet et al., 2011) | 0 | 5 |
| Fraction of tree biomass converted to residue    | BIO_LEAF | — | — | — | 0.0045 (Yang & Zhang, 2016) | — | 0.003 (Yang & Zhang, 2016) | — | — |
| No. years to tree maturity                       | MAT_YRS | — | — | — | 30 (SWAT) | — | 100 | — | — |

Abbreviation: SWAT, Soil & Water Assessment Tool.
Canopy height was recorded weekly during the 2017 growing season at eight randomly located measuring points within the crop (locations as shown in Holder et al., 2018) and reached a maximum of 3 m. Above ground biomass samples taken in February, June and August 2017 (from locations close to the eight measuring points) were freeze dried and subsequently ground to <2 mm using a Retsch mill (SM100; Retsch, Haan, Germany) before being further cryomilled in liquid nitrogen to a fine powder (6870 Cryomill; SPEX, Stanhope, UK). Samples were then analysed for total nitrogen (N) using a Vario Macro Cube Elementar (Analysensysteme GmbH, Langenselbold, Germany). Analysis of total phosphorus (P) was carried out by IBERS Analytical Chemistry (Aberystwyth, UK). This provided estimates of N and P at three seasonal time points (Table 4).

### Management operations

The following management operations were employed within the model depending on the land use/scenario for each HRU.

#### 2.4.1 Improved grassland

Sheep grazing at a stocking density of two livestock units starting in April for a duration of 212 days (to a minimum biomass of 1.5 Mg DM/ha; Genever & Buckingham, 2016). The daily dry weight of biomass eaten and trampled was set to 18 kg/ha (each), and fresh manure inputs to 60% of biomass consumed. Nitrogen fertilizer was added in March, April and July (40, 50, 20 kg N/ha respectively) and phosphorus was added in March, April and September (25, 15, 10 kg P/ha respectively; DEFRA, 2017). Pesticides were applied on a 2 year rotation: Year 1, Fluroxypyr MHE, Clopyralid and Triclopyr amine (0.32, 0.23, 0.42 kg/ha) were added in mid-April based on the contents of Pastor®; Year 2, Glyphosate amine (0.54 kg/ha) was added at the beginning of October based on Roundup 360® (Ballingall, 2014; Fera Science Ltd, 2018).

#### 2.4.2 Miscanthus

Fertilizer was automatically added by SWAT (according to crop N stress levels) to a maximum of 60 kg N ha⁻¹ year⁻¹ (amount required to obtain realistic yields within the model) and the above ground biomass was harvested annually in November at a 90% efficiency (based on field observations).

#### 2.4.3 Short rotation coppice

Fertilizer was automatically added by SWAT (according to crop N stress levels) to a maximum of 5 kg N ha⁻¹ year⁻¹ (being the amount required to obtain realistic yields within the model) and above ground biomass harvested in November on a 3 year rotation with a 70% efficiency (based on the SWAT database and Guo et al., 2015).

#### 2.4.4 Lawn grass

Fertilizer was automatically added to a maximum of 40 kg N ha⁻¹ year⁻¹. Grass was cut from April to August every 2 weeks, and then once a month during September and October.
2.4.5 | Arable

Fertilizer was automatically added to a maximum of 26 kg P ha\(^{-1}\) year\(^{-1}\) and 111 kg N ha\(^{-1}\) year\(^{-1}\) (DEFRA, 2017). All above ground biomass harvested (and plant growth killed) annually on 1 August (AHDB, 2018).

2.4.6 | Natural grassland

Light cattle grazing at a stocking density of 1.2 livestock units from mid-May for a duration of 90 days (to a minimum biomass of 3 Mg DM/ha; Genever & Buckingham, 2016). The daily dry weight of biomass eaten and trampled was set as 22.5 kg/ha (each), and fresh manure inputs were 60% of biomass consumed. Beef fresh manure was also automatically added to a maximum of 25 kg ha\(^{-1}\) year\(^{-1}\) (DEFRA, 2017; Welsh Government, 2018).

2.5 | Calibration

The initial model (representing existing land use) was calibrated for streamflow using the SWAT-CUP 2012 v.5.1.6 Sequential Uncertainty Fitting (SUFI2) procedure (Abbaspour, 2015) and the protocol outlined in Abbaspour et al. (2015). Water flow calibration and validation stations were selected from the National River Flow Archive (NERC & CEH, n.d.), discarding those with outside factors that may influence flow (e.g. private ground water extraction). To achieve calibration, only watershed level parameters were amended (Table S2.1). Observed streamflow from gauging stations C1 to C4 (Figure 2) was compared to modelled streamflow from the relevant sub-basin outlet and accuracy was assessed using \(R^2\) and Nash–Sutcliffe efficiency results. Gauging stations located at V1–V3 (Figure 2) were used to validate the modelled streamflow data.

2.6 | Scenarios

The baseline scenario is the calibrated model with existing land use. Four further simulations were run by splitting and changing the existing improved pasture land use and management to include the relevant percentage of energy crop (restricted to <15% slope, DEFRA, 2002; Lovett et al., 2014). Miscanthus planted on 50% (M50) and 25% (M25) and SRC planted on 50% (SRC50) and 25% (SRC25) of existing improved grass pasture within each sub-basin. The maximum LUC scenario using 50% of existing pasture (2,192 km\(^2\)) is based on the potentially suitable land in the district suggested in Lovett et al. (2014). The reduced, limited, level of LUC at 25% (1,096 km\(^2\)) reflects a level that could be reached in ~20 years if potential ambitious planting schemes (ADAS & ETI, 2016) were taken up.

2.7 | Analysis of results

Data analysis was performed in R version 3.5.1 (R Core Team, 2015) using linear models and linear mixed models (package ‘nlme’, Pinheiro, Bates, DebRoy, & Sarkar, 2017), with Tukey HSD (package ‘multcomp’, Hothorn, Bretz, & Westfall, 2008) post-hoc tests for significant results. Model residual plots were checked for the appropriateness of each model. Linear mixed model results were summarized using type III ANOVA (package ‘car’, Fox & Weisberg, 2011) which performs a Wald chi-square test.

For each level of planting, maximum (50%) or limited (25%), impacts of the crop type (baseline, Miscanthus and SRC) and season on the hydrological components of surface run-off, baseflow, soil water content, evapotranspiration and water yield were explored using whole watershed means calculated for each month (2004–2013). For surface run-off, baseflow and water yield transformations were used to improve model residuals (cube root with surface run-off and square root with baseflow and water yield). Analysis was conducted separately for each planting level with models including crop type and month (and their interactions) as fixed factors and year as a random effect, with an auto correlation structure (AR1).

In addition, to compare between planting levels and bioenergy crop type, differences to the baseline (mm change in monthly means) were used. Linear mixed models included the fixed factors of LUC level (25% and 50%), crop type (Miscanthus and SRC), month, and the random effect of year and an auto correlation structure (AR1). Surface run-off and baseflow data were transformed before testing (cube root and natural logarithm transformations respectively).

To allow for spatial effects to be examined, mean annual values (2004–2013) for all sub-basins were produced and impacts on surface run-off, baseflow, soil water content, evapotranspiration, water yield and streamflow were examined separately for each level of planting (50% or 25%) using linear models with crop type (SRC, Miscanthus, baseline) as a fixed factor. Streamflow data were transformed using the natural logarithm to improve residuals.

3 | RESULTS

3.1 | Model calibration

The watershed area was delineated into 855 sub-basins (Figure 3) and 7,108 HRUs. Satisfactory calibration between observed and modelled streamflow was achieved with Nash–Sutcliffe efficiency coefficient values of >0.50 for the baseline scenario representing existing land cover (Table 5; Figure S2.1.1–S2.1.7). The CNs were increased from starting values for land in good hydrological condition in order to improve the
correlation between observed and modelled streamflow. The final values used are shown in Table 6. Following amendments to plant growth parameters, simulated yields were checked against published data (Table 7; Figure S2.2.1–S2.2.4).

3.2 | Effects at the West Wales River Basin watershed level

Impacts for the whole 10,280 km² watershed varied across the months with the greatest differences occurring during the growing season (May–September, Figure 4). However, of the hydrological components tested (surface run-off, baseflow, soil water content, evapotranspiration and water yield), only surface run-off was significantly different compared to the baseline, where planting SRC at the 50% level resulted in significant reductions ($p = 0.03$) ranging from 17% (8 mm, January) to 23% (3 mm, April; Figure 4a).

Using the percentage change (compared to the baseline) to assess impacts of planting levels and bioenergy crop types, the 50% planting level (with both Miscanthus and SRC) led to greater reductions in overall surface run-off than at the 25% level ($\chi^2(1) = 4.56$, $p = 0.03$). In contrast, although the 50% planting level resulted in greater increases in baseflow than the 25% level ($\chi^2(1) = 49.94$, $p < 0.001$), impacts were significantly different between the bioenergy crop types, where baseflow was increased more during the spring with Miscanthus than with SRC ($\chi^2(1) = 10.21$, $p = 0.001$; Figure 4b).

The direction of change for evapotranspiration following LUC differed with bioenergy crop type, where it was increased with SRC during the early part of the year (January–May), but decreased with Miscanthus during the same period ($\chi^2(11) = 118.42$, $p < 0.001$; Figure 4c). From October to December, both crop types showed a decrease following higher evapotranspiration over the growing season. Greater impacts generally resulted from the 50% planting level compared to the 25% level, although this also depended on crop species with greater differences found with Miscanthus than with SRC ($\chi^2(1) = 10.86$, $p = 0.001$).

Water yield showed a decrease during the growing season with both bioenergy crops; however, during the early part of the year, the Miscanthus crop resulted in an increase, which was in contrast to the decreasing trend with SRC ($\chi^2(11) = 27.85$, $p = 0.003$). Impacts were again greater at the 50% planting level compared to the 25% but differences between crop types and planting levels were low from October to December ($\chi^2(1) = 10.92$, $p = 0.001$).

### TABLE 5

| Location | $R^2$ | NS  |
|----------|-------|-----|
| C1       | 0.65  | 0.50|
| C2       | 0.73  | 0.67|
| C3       | 0.84  | 0.67|
| C4       | 0.83  | 0.81|
| V1       | 0.87  | 0.56|
| V2       | 0.76  | 0.59|
| V3       | 0.88  | 0.76|

FIGURE 3  The West Wales River Basin District watershed delineated into 855 sub-basins. The spread of the (a) maximum and (b) limited land use change scenarios (50% and 25%, respectively, of improved pasture in each sub-basin) is represented.
3.3 | Sub-basin variation

Land use change was simulated in 726 of the 855 sub-basins (Figure 3), although it is also possible for non-LUC sub-basins to be impacted if, for example, they are downstream of the change. As changes in streamflow were limited in the majority of sub-basins (Figure 5) and maximum changes in soil water content ranged from −3% to +2% across all the sub-basins, these components were not found to significantly vary spatially (soil water content $F_{2,2562} = 0.46$, $p = 0.63$; $F_{2,2562} = 1.83$, $p = 0.16$; streamflow $F_{2,2562} = 0.30$, $p = 0.74$; $F_{2,2562} = 0.38$, $p = 0.68$; at the 25% and 50% levels respectively). However, reductions in streamflow of more than 50% were found in the same 10 sub-basins for each LUC scenario. Streamflow in these 10 sub-basins ranged from 0.5 to 1.6 m$^3$/s (daily mean) in the baseline (existing land use) scenario.

The different LUC levels and crops had varying impacts on the other hydrological components (Figure 6; Table 8). Surface run-off was significantly lower than the baseline scenario for Miscanthus and SRC in both the 25% ($F_{2,2562} = 32.77$, $p < 0.001$) and 50% ($F_{2,2562} = 156.8$, $p < 0.001$) scenarios, with differences ranging from 0 to −182 mm (0% to −40%, Figure 6a). No significant differences in surface run-off were found between Miscanthus and SRC.

Baseflow results also showed greater differences in Miscanthus compared to SRC in the 50% LUC scenario where a significant difference ($p = 0.02$) was found between the two crops (Figure 6b). Eighty-four sub-basins in the M50 scenario increased baseflow by more than 30%, compared to 11 sub-basins in the SRC50 scenario. The maximum amount of the increase was 39% (136 mm) for M50 and 36% (127 mm) for SRC50. Baseflow was significantly higher than the baseline scenario for both Miscanthus and SRC in the 25% ($F_{2,2562} = 70.29$, $p < 0.001$) and 50% ($F_{2,2562} = 233.6$, $p < 0.001$) LUC scenarios.

Changes in evapotranspiration with Miscanthus and SRC compared to the pasture baseline ranged from −2% (−15 mm, M50) to 5% (+32 mm, SRC50) and whilst the difference was only significant for SRC ($p < 0.001$), a distinct difference was seen between the two crops ($p < 0.001$). Where changes in evapotranspiration relating to the Miscanthus scenarios occurred, the result was a small reduction; however, with SRC increases were produced (Figure 6c). The same trend was identified in the 25% LUC scenarios. It was also found that some of the sub-basins with the highest increase in evapotranspiration also had the highest reductions in water yield (Figure 6c,d).

Changes in water yield compared to the baseline scenario were not significant at the 25% LUC level. However, for the 50% LUC scenarios, SRC was significantly lower than both the Miscanthus ($p = 0.001$) and baseline ($p = 0.01$) scenarios (Figure 6d). Differences in water yield ranged from a reduction of 4% (−30 mm, SRC50) to an increase of 2% (+16 mm, M50).
This study has shown that large-scale planting of *Miscanthus* or SRC crops does have a significant impact on the hydrological cycle for the West Wales River Basin. The simulated reductions in surface run-off and increases in baseflow for *Miscanthus* and SRC (at the limited and maximum LUC levels) correspond with previous predictions relating to LUC to *Miscanthus* and SRC (Environment Agency, 2015; Stephens et al., 2001) where changes to these hydrological

| Land use                  | Code   | Simulated (SD) | Reference                                                                 |
|---------------------------|--------|----------------|---------------------------------------------------------------------------|
| Cereals/oil seed rape     | AGRL   | Y August: 4 (2.5) | 7 Cereals, 3 oil seed rape (DEFRA, 2017)                                  |
| Urban grass (mowed)       | BERM   | 1.5 (0.4)       | ~4 cm sward height                                                        |
| Improved pasture (grazed) | CRDY   | 2.86 (2.6)      | ~2 depending on grazing strategy (Genever & Buckingham, 2016)             |
| Natural grassland (light grazing) | FESC | 3.5 (0.3) | 3–7 (Mills, 2016); 1–3 (Milne, Pakeman, Kirkham, Jones, & Hossell, 2002) |
| Heather/shrub grassland   | MIGS   | 9.75 (2.78)     | 6–27 (Mills, 2016); 5–10 (Milne et al., 2002)                            |
| Heather                   | SHRB   | 9.10 (2.26)     | 6–10 (Mills, 2016); 5–10 (Milne et al., 2002)                            |
| Fen/marsh/bog/saltmarsh   | WETL   | 14.78 (10.74)   | 1–22 (Mills, 2016)                                                       |
| Short rotation coppice    | WSRC   | Y November:      | 5–16 (Aylott et al., 2008); 10–15 (Cunniff et al., 2015)                 |
| *M. x giganteus*          | MSXG   | Y November:      | 14 (Larsen et al., 2014); 15 measurements                                |

Abbreviation: SWAT, Soil & Water Assessment Tool.
components followed a similar trend. The maximum monthly reduction (in mm) across the watershed for surface run‐off with Miscanthus, 17 mm (in November, a 17% reduction compared to the baseline scenario), was similar to the 18 mm maximum reduction simulated by Cibin et al. (2016) in modelled LUC from grassland to Miscanthus within a U.S. catchment. The 20%–30% reduction in surface run‐off found for the majority of the sub‐basins is also within the range of 20%–78% predicted by Hartwich et al. (2016) in modelled LUC from grassland to SRC (in different regions of the Northern German Plain).

It should be noted that the surface run‐off calculations used in the model simulations are based on the CN method (Soil Conservation Service, 1976) and Manning's roughness coefficients (e.g. Chow, 1959). These are well established for traditional crops, grassland and woodland but empirical measurements (to act as a basis for coefficient values) are lacking for Miscanthus and SRC (Environment Agency, 2015). The values we adopted for Miscanthus were previously used by Cibin et al. (2016) and are based on values for Alamo switchgrass (P. virgatum L.). Switchgrass is a similar perennial grass to Miscanthus but may exhibit morphological differences, for example an increased stem density compared to Miscanthus (Cassida, Muir, Hussey, & Read, 2005) that could result in differences in hydraulic resistance and hence surface run‐off rates. Similarly, new Miscanthus varieties (currently in pre‐commercial trials, Lewandowski et al., 2016) can have significantly different morphologies. SRC CNs used were based on existing values for trees, but an SRC plantation differs in stand layout and density compared to natural woodland and therefore (for both SRC and Miscanthus) empirical measurements would improve model inputs. However, whilst accuracy of the model could be improved in this respect, replacing grassland in comparison with grassland with the more rigid stems and greater height of both Miscanthus and SRC means that these crops would be expected to reduce run‐off and sediment flow.

Due to both physiological and physical factors (e.g. higher water use and greater leaf area index [LAI]), energy crops are generally associated with higher evapotranspiration than grassland, especially during the growing season (Cibin et al., 2016; Guo et al., 2018; Hartwich et al., 2016), something also found in this study. Differences in SRC compared to Miscanthus in evapotranspiration and water yield are slightly more complex. Whilst the longer SRC growing season can, in part, account for the greater impact of SRC than Miscanthus, modelled differences are also likely to be linked to specific parameters used for the LAI value during plant dormancy. In the Miscanthus scenarios this was set to zero (as in Trybula et al., 2015), whereas the LAI for the SRC scenarios during dormancy was set to 0.75 (as per the SWAT database for willow and poplar species). Although SRC and Miscanthus are not transpiring during winter months, LAI influences calculations of canopy storage and hence the evaporation of intercepted precipitation.

Whilst changes in water quality were not modelled, measured soil N losses following the establishment of Miscanthus and SRC have been found to reduce in comparison with annual crops and grassland due to lower fertilizer use and differences in N use efficiency (Christian & Riche, 1998; Schmidt‐Walter & Lamersdorf, 2012). Therefore, the reduction in fertilizer use with both Miscanthus and SRC (110, 60 and 5 kg N ha⁻¹ year⁻¹ for pasture, Miscanthus and SRC respectively) could be expected to reduce nitrate leaching. In addition, whilst the model required the addition of fertilizer to obtain expected crop growth based on published data (Aylott et al., 2008; Cunniff et al., 2015; Larsen, Jørgensen, Kjeldsen, & Lærke, 2014), fertilizer use is not routine in UK commercial production of these crops, particularly when cultivating on previously fertilized pasture land (Aylott et al., 2008; Terravesta Ltd, 2018). Fertilizer applications have been used in other SWAT‐based studies (e.g. 122 kg urea ha⁻¹ year⁻¹ with Miscanthus, Cibin et al., 2016, and 50 kg N ha⁻¹ year⁻¹ with willow, Wang, Jager, Baskaran, & Brandt, 2018) and although the best yield responses to N fertilization are generally achieved at around 60–100 kg N/ha, Miscanthus and SRC do not always show a response to fertilization (Aronsson, Rosenqvist, & Dimitriou, 2014; Cadoux, Riche, Yates, & Machet, 2012; Quaye & Volk, 2013).
The different rooting structures and water requirements of SRC and Miscanthus have the potential to cause drying of the soil profile under rain-limited conditions (Donnelly, Styles, Fitzgerald, & Finnan, 2011; Stephens et al., 2001). Such drying could have negative impacts such as reductions in yields (Knapp, Briggs, & Koelliker, 2001; Richter, Riche, Dailey, Gezan, & Powlson, 2008) and changes in microbial processes and associated nutrient availability with implications for soil carbon stocks and greenhouse gas emissions (Jensen, Beier, Michelsen, & Emmett, 2003; Smith et al., 2008). However, such drying did not occur in either scenario modelled in this study with soil moisture levels remaining similar to the pasture baseline. This is in contrast to Hartwich et al. (2016) where soil water content was reduced in simulated LUC from pasture to SRC crops in the drier Northern German Plain, where soils are likely to have a higher sand content. Rainfall levels in west Wales (1,532 mm/year) are also towards the top end of the range (of between 1,000 and 1,600 mm/year).
TABLE 8  Mean annual sub-basin surface run-off (SURQ), baseflow (GWQ), soil water content (SW), evapotranspiration (ET) and water yield (WY) in mm, and streamflow (daily mean, m³/s) for each of the scenarios (SE shown in brackets). The scenarios reflect planting Miscanthus (M) or short rotation coppice (SRC) on approximately 50% (2,192 km²) and 25% (1,096 km²) of existing improved pasture areas compared to the baseline (Base) of no land use change. Significance (p < 0.001) is shown for Base versus M/SRC.

|        | Base (mm) | 25% M | 25% SRC | 50% M | 50% SRC |
|--------|-----------|-------|---------|-------|---------|
| SURQ   | 344 (4)   | 314 (3)*** | 311 (3)*** | 284 (3)*** | 278 (3)*** |
| GWQ    | 387 (2)   | 417 (2)*** | 413 (2)*** | 477 (2)*** | 439 (2)*** |
| SW     | 166 (0.3) | 166 (0.3) | 166 (0.3) | 167 (0.3) | 167 (0.3) |
| ET     | 678 (1)   | 677 (1)   | 684 (1)*** | 676 (1) | 691 (1)*** |
| WY     | 851 (3)   | 852 (3)   | 845 (3)   | 853 (3) | 838 (3)*** |
| Flow out | 1.27 (0.13) | 1.25 (0.14) | 1.24 (0.13) | 1.25 (0.14) | 1.24 (0.13) |

Note: Significance denoted by ‘***’.

for areas including Ireland, western Great Britain, northern Italy, Switzerland, Austria and northern Spain (European Environment Agency, 2012). The soils in this study also have a high clay and silt content, factors that are likely to limit drying impacts compared to drier locations or free-draining, lighter soils (Balogh et al., 2011; Marshall, Holmes, & Rose, 1996). Therefore, in assessing the land suitability for the cultivation of energy crops, local conditions should be considered to ensure rainfall rates are sufficient to meet crop demand (Richter et al., 2008). The fact that the majority of grasslands in Europe (as a fraction of total agricultural land area) tend to be located in wetter areas (Smit, Metzger, & Ewert, 2008) confirms that these locations should perhaps be targeted for this kind of agricultural diversification.

Reductions in the amount of water leaving the sub-basins (water yield) were only significant for the maximum SRC LUC scenario, and changes in streamflow were not significant for any of the LUC scenarios. This indicates that changes in aquatic environments are likely to be limited across the whole watershed. However, some sub-basins did show reductions in streamflow of over 50% which, when coupled with the difficulties in understanding and predicting biotic responses to altered flow rates (Bunn & Arthington, 2002; Shafroth et al., 2010), demonstrates the importance of local environmental flow assessments in proposed large-scale energy crop planting (Poff et al., 2010). The significant reduction in surface run-off and increase in baseflow found for both LUC levels and crop types could also impact on aquatic and riparian species (Gurnell, Bertoldi, & Corenblit, 2012), which should be considered when selecting suitable locations for energy crop deployment.

However, improvements in soil water infiltration seen in this study may also benefit flood mitigation by increasing soil water capacity during periods of high rainfall, as has been found with the use of young trees (<7 years old) in shelterbelts (Marshall et al., 2009). Although increases in baseflow were higher with Miscanthus than with SRC during the spring (possibly as a result of increased soil infiltration with Miscanthus due to the later leaf development), overall SRC in our modelling performed better than Miscanthus in terms of potential flood mitigation benefits. This is largely due to overall reductions in water yield (at the 50% LUC scenario) and increases in evapotranspiration (at both LUC levels). The annual Miscanthus harvest is also in contrast to SRC where the 3 year harvest cycle results in more overwinter standing plant material for 2 out of 3 years. However, the timing of the harvest for Miscanthus in the model was simulated as occurring in November, but Miscanthus can be (and often is in the UK) harvested as late as early spring where the presence of the senesced biomass continues to intercept precipitation (Holder et al., 2018), and tall stalks would provide further resistance to overland flows and may reduce some of the differences between the two crops.

Reductions in surface run-off and increases in baseflow brought about by LUC can also act to slow and buffer high overland flows (Bronstert, Niehoff, & Brger, 2002; Marshall et al., 2009; OECD, 2016) with the predicted impact of slowing the flow rate across floodplains. This factor could therefore potentially release currently excluded land in flood zone areas for the planting of biomass crops (Environment Agency, 2015). In the scenarios we tested, slope was restricted to below 15% in order to allow for crop management and harvest, but if the crops were planted with the main aim of flood mitigation or nutrient buffering (e.g. as land margin buffer strips, Ferrarini et al., 2017) with less demand for commercial return, this assumption could be relaxed somewhat with the acknowledgment that annual harvest may sometimes be lost due to prevailing conditions preventing land access.

The large-scale planting areas considered in this study were chosen to highlight the maximum effects of the land conversion scenarios. To set the more limited LUC scenario (1,096 km²) in context, it has the potential to provide 12%, 1,639 GWh (assuming a yield of 12 Mg DM/ha, Larsen et al., 2014; an energy content of 17.95 GJ/Mg DM, Felten, Fröba, Fries, & Emmerling, 2013; with a conversion efficiency of 25%, Nguyen & Hermansen, 2015).
of the Welsh Government target for 70%, 13,431 GWh (BEIS, 2018b) of Welsh electricity consumed to come from renewables (National Assembly for Wales, 2017).

Specific locations for planting of energy crops within the watershed will ultimately be based on economic and social constraints and it is not likely that just Miscanthus or SRC would be grown but rather a mix chosen to suit local conditions and opportunities. Projections based on profitability (using existing farm scales and energy crop prices) have suggested a commercially viable planting area of 390 km² of energy crops in Wales (Alexander et al., 2014). However, there is scope for this to increase (by as much as 300 km²/year across the UK) due to improvements in agronomy, changes to climate resulting in greater yields, boosts in demand, and increases in prices paid for supply or if incentivized with subsidies (ADAS & ETI, 2016; Alexander et al., 2014; Hastings et al., 2017). Overall, whilst there is potential for negative impacts in a small number of sub-basins, this study shows that even with very ambitious levels of LUC the production of bioenergy crops within this catchment is unlikely to result in damaging impacts on basin-level hydrological processes. The impacts on other ecosystem services however were not addressed and would need to be considered in any policies that seek to support large-scale planting of energy crops.

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REFERENCES

Abbaspour, K. C. (2015). SWAT-CUP: SWAT calibration and uncertainty programs—A user manual. Duebendorf, Switzerland: Department of Systems Analysis, Integrated Assessment and Modelling (SIAM), Eawag, Swiss Federal Institute of Aquatic Science and Technology. Retrieved from https://swat.tamu.edu/media/114860/usermanual_swatcup.pdf
Abbaspour, K. C., Rouholahnejad, E., Vaghefi, S., Srinivasan, R., Yang, H., & Klove, B. (2015). A continental-scale hydrology and water quality model for Europe: Calibration and uncertainty of a high-resolution large-scale SWAT model. Journal of Hydrology, 524, 733–752. https://doi.org/10.1016/j.jhydrol.2015.03.027
ADAS UK Ltd, & Energy Technologies Institute. (2016). Job implications of bioenergies. Retrieved from https://www.eti.co.uk/library/adas-relb-job-implications-of-establishing-a-bioenergy-market
Agriculture and Horticulture Development Board (AHDB). (2018). Wheat growth guide. AHDB Cereals & Oilseeds, 1–44. https://cereals.ahdb.org.uk/media/185687/g66-wheat-growth-guide.pdf
Alexander, P., Moran, D., Smith, P., Hastings, A., Wang, S., Sünnerberg, G., … Cisowska, I. (2014). Estimating UK perennial energy crop supply using farm-scale models with spatially disaggregated data. Global Change Biology Bioenergy, 6(2), 142–155. https://doi.org/10.1111/gcbb.12121
Amougou, N., Bertrand, I., Cadoux, S., & Recous, S. (2012). Miscanthus × giganteus leaf senescence, decomposition and C and N inputs to soil. Global Change Biology Bioenergy, 4(6), 698–707. https://doi.org/10.1111/j.1757-1707.2012.01912.x
Arnold, J. G., Srinivasan, R., Mutthia, R. S., & Williams, J. R. (1998). Large area hydrologic modeling and assessment Part 1: Model development. Journal of the American Water Resources Association, 34(1), 73–89. https://doi.org/10.1111/j.1752-1688.1998.tb05961.x
Aronsson, P., Rosenqvist, H., & Dimitriou, I. (2014). Impact of nitrogen fertilization to short-rotation willow coppice plantations grown in Sweden on yield and economy. Bioenergy Research, 7(3), 993–1001. https://doi.org/10.1007/s12155-014-9435-7
Arthington, A. H., Naiman, R. J., McClain, M. E., & Nilsson, C. (2010). Freshwater Biology, 55, 1–16. https://doi.org/10.1111/j.1365-2427.2009.02340.x
Asner, G. P., Scurlock, J. M. O., & Hicke, J. A. (2003). Global synthesis of leaf area index observations: Implications for ecological and remote sensing studies. Global Ecology and Biogeography, 12, 191–205. https://doi.org/10.1046/j.1466-822x.2003.00262.x
Ayllott, M. J., Casella, E., Tubby, I., Street, N. R., Smith, P., & Taylor, G. (2008). Yield and spatial supply of bioenergy poplar and willow short-rotation coppice in the UK. New Phytologist, 178(2), 358–370. https://doi.org/10.1111/j.1469-8137.2008.02396.x
Ballingall, M. (2014). Weed management in grassland, Technical Note TN643 SRUC. Perth, Australia: SAC Consulting.
Balogh, J., Pintér, K., Fóti, S., Cserhalmi, D., Papp, M., & Nagy, Z. (2011). Dependence of soil respiration on soil moisture, clay content, soil organic matter, and CO₂ uptake in dry grasslands. Soil Biology and Biochemistry, 43, 1006–1013. https://doi.org/10.1016/j.soilbio.2011.01.017
Beale, C. V., Bint, D. A., & Long, S. P. (1996). Leaf photosynthesis in the C4-grass Miscanthus × giganteus, growing in the cool temperate climate of southern England. Journal of Experimental Botany, 47(2), 267–273. https://doi.org/10.1093/jxb/47.2.267
BEIS (Department for Business Energy & Industrial Strategy). (2018a). UK energy in brief. Retrieved from https://www.gov.uk/government/statistics/uk-energy-in-brief-2018
BEIS (Department for Business Energy & Industrial Strategy). (2018b). Electricity generation and supply figures for Scotland, Wales, Northern Ireland and England 2004 to 2017. Retrieved from https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/770767/Electricity Regional generation 2004-2017.xls
Belanger, G., Gastal, F., & Warembourg, F. R. (1994). Carbon balance of tall fescue (Festuca arundinacea Schreber): Effects of nitrogen
fertilization and the growing season. *Annals of Botany*, 74, 653–659. https://doi.org/10.1006/anbo.1994.1167

Bequet, R., Campioli, M., Kint, V., Vansteenkiste, D., Muys, B., & Ceulemans, R. (2011). Leaf area index development in temperate oak and beech forests is driven by stand characteristics and weather conditions. *Trees*, 25(5), 935–946. https://doi.org/10.1007/s00468-011-0568-4

British Geological Survey Materials. (2018). Soil parent material. Retrieved from https://www.bgs.ac.uk/products/onshore/soilPMM.html

Bronstert, A., Niehoff, D., & Brger, G. (2002). Effects of climate and land-use change on storm runoff generation: Present knowledge and modelling capabilities. *Hydrological Processes*, 16, 509–529. https://doi.org/10.1002/hyp.326

Bullard, M. J., Mustill, S. J., Carver, P., & Nixon, P. M. I. (2002). Yield improvements through modification of planting density and harvest frequency in short rotation coppice *Salix* spp.—2. Resource capture and use in two morphologically diverse varieties. *Biomass and Bioenergy*, 22(1), 27–39. https://doi.org/10.1016/S0961-9534(01)00055-1

Bunn, S. E., & Arthington, A. H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management*, 30(4), 492–507. https://doi.org/10.1007/s00267-002-2737-0

Burgy, R. H., & Pomeroy, C. R. (1958). Interception losses in grassy vegetation. *Transactions, American Geophysical Union*, 39(6), 1095–1100. https://doi.org/10.1029/TR039i006p01095

Cadoux, S., Riche, A. B., Yates, N. E., & Machet, J. (2012). Nutrient requirements of *Miscanthus × giganteus*: Conclusions from a review of published studies. *Biomass and Bioenergy*, 38, 14–22. https://doi.org/10.1016/j.biombioe.2011.01.015

Cassida, K. A., Muir, J. P., Hussey, M. A., & Read, J. C. (2005). Biomass yield and stand characteristics of switchgrass in south central US environments. *Crop Science*, 45, 673–681. https://doi.org/10.2135/cropsci2005.0673

Centre for Ecology & Hydrology (CEH). (n.d.). UK lakes portal. Retrieved from https://cief.ceh.ac.uk/apps/lakes

Chow, V. T. (1959). *Open-channel hydraulics*. New York, NY: McGraw-Hill.

Christian, D. G., & Riche, A. B. (1998). Nitrate leaching losses under *Miscanthus* grass planted on a silty clay loam soil. *Soil Use and Management*, 14(3), 131–135. https://doi.org/10.1111/j.1475-2743.1998.tb00136.x

Chum, H., Faaij, A., Moreira, J., Berndes, G., Dhamija, P., Dong, H., … Hong, D., … Pingoud, K. (2011). Chapter 2: Bioenergy. In O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, P. Eickemeier, G. Hansen, S. Schlömer, & C. vonSteinbauer (Eds.), IPCC special report on renewable energy sources and climate change mitigation. Cambridge, UK and New York, NY: Cambridge University Press. Retrieved from https://www.ipcc.ch/report/renewable-energy-sources-and-climate-change-mitigation/

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
Environment Agency. (2015). *Energy crops and floodplain flows*. Bristol, UK: Environment Agency.

European Commission. (n.d.). Agricultural and rural development. Less favoured areas scheme. Retrieved from https://ec.europa.eu/agriculture/rural-development-previous/2007-2013/less-favoured-areas-scheme_en

European Commission. (2018). A clean planet for all. A European strategic long-term vision for a prosperous, modern, competitive and climate neutral economy. Retrieved from https://ec.europa.eu/clima/sites/clima/files/docs/pages/com_2018_733_en.pdf

European Environment Agency. (2012). Average annual precipitation in the EEA area. Retrieved from https://www.eea.europa.eu/data-and-maps/figures/average-annual-precipitation

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

Eurostat. (2018b). Agricultural accounts and prices. Retrieved from http://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agricultural_accounts_and_prices

Felten, 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.rene.2012.12.004

Feng, Q., Chaubey, I., Cubin, R., Engel, B., Sudheer, K. P., Volenec, J., & Oman, N. (2018). Perennials biomass production from marginal land in the Upper Mississippi River Basin. *Land Degradation and Development*, 29(6), 1748–1755. https://doi.org/10.1002/ldr.2971

Fera Science Ltd. (2018). Pesticide usage surveys. Retrieved from https://secure.fera.defra.gov.uk/pusstats/

Ferrarini, A., Fornasier, F., Serra, P., Ferrari, F., Trevisan, M., & Amaducci, S. (2017). Impacts of willow and *Miscanthus* bioenergy buffers on biogeochemical N removal processes along the soil–groundwater continuum. *Global Change Biology Bioenergy*, 9, 246–261. https://doi.org/10.1016/j.gcbbe.2012.12.004

Finch, H. J. S., Samuel, A. M., & Lane, G. P. F. (2002). *Lockart and Wiseman's crop husbandry* (8th ed.). Cambridge, UK: Woodhead Publishing Ltd.

Finch, J. W., Hall, R. L., Rosier, P. T. W., Clark, D. B., Stratford, C., Davies, H. N., & Christian, D. (2004). The hydrological impacts of energy crop production in the UK. London, UK: Department of Trade and Industry.

Finch, J. W., & Riche, A. B. (2010). Intervention losses from *Miscanthus* at a site in south-east England—An application of the Gash model. *Hydrological Processes*, 24(18), 2594–2600. https://doi.org/10.1002/hyp.7673

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global consequences of land use. *American Association for the Advancement of Science*, 309(5734), 570–574. https://doi.org/10.1126/science.1111772

Food and Agriculture Organisation of the United Nations (UNFAO) (2003). The Digital Soil Map of the World v3.6. (c) FAO/UNESCO 1995 All rights reserved worldwide. Retrieved from http://www.waterbase.org/download_data.html

Forestry Commission. (2011). Biomass in live woodland trees in Britain: National Forest Inventory Report. Retrieved from http://data.gov.uk/dataset/bede6874-4fc1-4a8a-9baee22b2a110380/national-forest-inventory-biomass-in-live-woodland-trees-gb-2011

Forestry Commission. (2017). Woodland Areas and Planting. Forestry statistics 2017. Retrieved from http://www.forestry.gov.uk/statistics

Fox, J., & Weisberg, S. (2011). An R companion to applied regression (2nd ed.). Thousand Oaks, CA: Sage. Retrieved from http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Garbulsky, M. F., Peñuelas, J., Papale, D., Ardö, J., Goudlen, M. L., Kiely, G., … Fillella, I. (2010). Patterns and controls of the variability of radiation use efficiency and primary productivity across terrestrial ecosystems. *Global Ecology and Biogeography*, 19(2), 253–267. https://doi.org/10.1111/j.1466-8238.2009.00504.x

Genever, L., & Buckingham, S. (2016). Planning grazing strategies for better returns. In *Beef and Sheep Better Returns Programme Manual*, 26. Warwickshire, UK: AHDB Beef and Lamb.

Gonzalez, M., Augusto, L., Gallet-Budynek, A., Xue, J., Yauschew-Ragueneau, N., Guyon, D., … Bakker, M. R. (2013). Contribution of understory species to total ecosystem aboveground and belowground biomass in temperate *Pinus pinaster* Ait. forests. *Forest Ecology and Management*, 289, 38–47. https://doi.org/10.1016/j.foreco.2012.10.026

Guo, T., Engel, B. A., Shao, G., Arnold, J. G., Srinivasan, R., & Kiniry, J. R. (2015). Functional approach to simulating short-rotating woody crops in process-based models. *Bioenergy Research*, 8, 1598–1613. https://doi.org/10.1007/s12155-015-9615-0

Guo, D., & Westra, S. (2016). Evapotranspiration: Modelling actual, potential and reference crop evapotranspiration. R package. Retrieved from https://cran.r-project.org/package=Evapotranspiration

Guo, T., Cibin, R., Chanbey, I., Gitau, M., Arnold, J. G., Srinivasan, R., … Engel, B. A. (2018). Evaluation of bioenergy crop growth and the impacts of bioenergy crops on streamflow, tile drain flow and nutrient losses in an extensively tile-drained watershed using SWAT. *Science of the Total Environment*, 613–614, 724–735. https://doi.org/10.1016/j.scitotenv.2017.09.148

Gurnell, A., Bertoldi, W., & Corenblit, D. (2012). Changing river channels: The roles of hydrological processes, plants and pioneer fluvial landforms in humid temperate, mixed load, gravel bed rivers. *Earth-Science Reviews*, 111, 129–141. https://doi.org/10.1016/j.earscirev.2011.11.005

Hartwich, J., Schmidt, M., Bölscher, J., Reinhardt-Imjela, C., Murach, D., & Schulte, A. (2016). Hydrological modelling of changes in the water balance due to the impact of woody biomass production in the North German Plain. *Environmental Earth Sciences*, 75(14), 1071. https://doi.org/10.1007/s12665-016-5870-4

Hastings, A., Clifton-Brown, J., Wattenbach, M., Mitchell, C. P., & Smith, P. (2009). The development of MISCANFOR, a new *Miscanthus* crop growth model: Towards more robust yield predictions under different climatic and soil conditions. *Global Change Biology Bioenergy*, 1, 154–170. https://doi.org/10.1111/j.1757-1707.2009.00107.x

Hastings, A., Mos, M., Yesufu, J. A., McMclmont, J. P., Schwarz, K., Shafei, R., … Clifton-Brown, J. (2017). Economic and environmental assessment of seed and rhizome propagated *Miscanthus* in the UK. *Frontiers in Plant Science*, 8(June), 1–16. https://doi.org/10.3389/fpls.2017.01058

Hastings, A., Tallis, M. J., Casella, E., Matthews, R. W., Henshall, P. A., Milner, S., … Taylor, G. (2014). The technical potential of Great Britain to produce ligno-cellulosic biomass for bioenergy in current and future climates. *Global Change Biology Bioenergy*, 6(2), 108–122. https://doi.org/10.1111/gcbb.12103

Hess, T. M., Holman, I. P., Rose, S. C., Rosolova, Z., & Parrott, A. (2010). Estimating the impact of rural land management changes on...
Neukirchen, D., Himken, M., Lammel, J., Czypionka-Krause, U., & Olfs, H.-W. (1999). Spatial and temporal distribution of the root system and root nutrient content of an established Miscanthus crop. *European Journal of Agronomy*, 11, 301–309. https://doi.org/10.1016/s1161-0301(99)00031-3

Ng, T. L., Eheart, J. W., Cai, X., & Miguez, F. (2010). Modeling Miscanthus in the Soil and Water Assessment Tool (SWAT) to simulate its water quality effects as a bioenergy crop. *Environmental Science and Technology*, 44(18), 7138–7144. https://doi.org/10.1021/es1039677

Nguyen, T. L. T., & Hermansen, J. E. (2015). Life cycle environmental performance of Miscanthus gasification versus other technologies for electricity production. *Sustainable Energy Technologies and Assessments*, 9, 81–94. https://doi.org/10.1016/j.seta.2014.12.005

Nisbet, T. (2005). *Water use by trees—Forestry Commission information note FCIN065* (pp. 1–8). Edinburgh, UK: Forestry Commission. Retrieved from https://www.forestry.gov.uk/pdf/FCIN065.pdf

Oak Ridge National Laboratory Distributed Active Archive Center (ORNL DAAC). (n.d.). A global database of field-observed leaf area index in woody plant species, 1932–2011. Retrieved from https://daac.ornl.gov/VEGETATION/guides/LAI_Woody_Plants.html

Ordnance Survey. (2018). *OS Terrain 50 & OS Open Rivers* (c) Crown copyright and database right 2018. Retrieved from https://www.ordnancesurvey.co.uk/business-and-government/products/terrain-50.html

Organisation for Economic Co-operation and Development (OECD). (2016). *Mitigating droughts and floods in agriculture: Policy lessons and approaches*. Paris, France: OECD Publishing. https://doi.org/10.1787/9789264246744-en

Pellis, A., Laureysens, I., & Ceulemans, R. (2004). Growth and production of a short rotation coppice culture of poplar I. Clonal differences in leaf characteristics in relation to biomass production. *Biomass and Bioenergy*, 27(1), 9–19. https://doi.org/10.1016/j.biombioe.2003.11.001

Pinheiro, J., Bates, D., DebRoy, S., & Sarkar, D.; R Core Team. (2017). nlme: Linear and nonlinear mixed effects models. R package version 3.1-131. Retrieved from https://cran.r-project.org/package=nlme

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., … Warner, A. (2010). The ecological limits of hydrologic alteration (ELOHA): A new framework for developing regional environmental flow standards. *Freshwater Biology*, 55, 147–170. https://doi.org/10.1111/j.1365-2427.2009.02204.x

Poff, N. L., & Zimmerman, J. K. H. (2010). Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. *Freshwater Biology*, 55, 194–205. https://doi.org/10.1111/j.1365-2427.2009.02272.x

QGIS (Development Team). (2014). QGIS geographic information system. Open Source Geospatial Foundation Project. Retrieved from http://qgis.osgeo.org

Quaye, A. K., & Volk, T. A. (2013). Biomass production and soil nutrients in organic and inorganic fertilized willow biomass production systems. *Biomass and Bioenergy*, 57, 113–125. https://doi.org/10.1016/j.biombioe.2013.08.002

R Core Team. (2015). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Richter, G. M., Riche, A. B., Dailey, A. G., Gezan, S. A., & Powellson, D. S. (2008). Is UK biofuel supply from Miscanthus water-limited? *Soil Use and Management*, 24(3), 235–245. https://doi.org/10.1111/j.1475-2743.2008.00156.x

Rowe, R. L., Street, N. R., & Taylor, G. (2009). Identifying potential environmental impacts of large-scale deployment of dedicated bioenergy crops in the UK. *Renewable and Sustainable Energy Reviews*, 13(1), 271–290. https://doi.org/10.1016/j.rser.2007.07.008

Rowland, C. S., Morton, R. D., Carasco, L., McShane, G., O’Neill, A. W., & Wood, C. M. (2017). Land cover map 2015 (25m raster, GBD). NERC Environmental Information Data Centre. Retrieved from https://doi.org/10.5285/bb15e200-9349-403c-bda9-b430093807c7

Schmidt-Walter, P., & Lamersdorf, N. P. (2012). Biomass production with willow and poplar short rotation coppices on sensitive areas—the impact on nitrate leaching and groundwater recharge in a drinking water catchment near Hanover, Germany. *Bioenergy Research*, 5(3), 546–562. https://doi.org/10.1007/s12155-012-9237-8

Shafroth, P. B., Wilcox, A. C., Lytle, D. A., Hickey, J. T., Andersen, D. C., Beauchamp, V. B., … Warner, A. (2010). Ecosystem effects of environmental flows: Modelling and experimental floods in a dryland river. *Freshwater Biology*, 55, 68–85. https://doi.org/10.1111/j.1365-2427.2009.02271.x

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, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., … Smith, J. (2008). Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1492), 789–813. https://doi.org/10.1098/rstb.2007.2184

Soil Conservation Service. (1976). Section 4: Hydrology. In V. McKeever, W. Owen, & R. Rallison (Eds.), *National engineering handbook*. Washington, DC: US Department of Agriculture, SCS.

Stephens, W., Hess, T., & Knox, J. (2001). Review of the effects of energy crops on hydrology. Retrieved from https://dspace.lib.cranfield.ac.uk/handle/1826/3368

Taube, F., Giers, M., Hermann, A., Loges, R., & Schönbach, 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/gfs.12043

Terravesta Ltd. (2018). Growing miscanthus. Retrieved from https://www.terravesta.com

Trybula, E. M., Cibin, R., Burks, J. L., Chaubey, I., Brouder, S. M., & Volenc, J. J. (2015). Perennial rhizomatous grasses as bioenergy feedstock in SWAT: Parameter development and model improvement. *Global Change Biology Bioenergy*, 7(6), 1185–1202. https://doi.org/10.1111/gcbb.12210

United States Department of Agriculture (USDA). (1986). Urban hydrology for small watersheds—TR55. Technical release 55. Retrieved from https://doi.org/TechnicalRelease55

Vanloocke, A., Bernacchi, C. J., & Twine, T. E. (2010). The impacts of Miscanthus × giganteus production on the Midwest US hydrologic cycle. *Global Change Biology Bioenergy*, 2(4), 180–191. https://doi.org/10.1111/j.1757-1707.2010.01053.x

Verlinden, M. S., Broeckx, L. S., Van den Bulcke, J., Van Acker, J., & Ceulemans, R. (2013). Comparative study of biomass determinants of 12 poplar (Populus) genotypes in a high-density short-rotation culture. *Forest Ecology and Management*, 307, 101–111. https://doi.org/10.1016/j.foreco.2013.06.062

Wang, D., Li, J. S., & Rao, M. J. (2006). Winter wheat canopy interception under sprinkler irrigation. *Science of the Total Environment*, 39(9), 1859–1864.

Wang, G., Jager, H. L., Baskaran, L. M., & Brandt, C. C. (2018). Hydrologic and water quality responses to biomass production in
the Tennessee River Basin. *Global Change Biology Bioenergy*, 10, 877–893. https://doi.org/10.1111/gcbb.12537

Weih, M., & Nordh, N. E. (2002). Characterising willows for biomass and phytoremediation: Growth, nitrogen and water use of 14 willow clones under different irrigation and fertilisation regimes. *Biomass and Bioenergy*, 23(6), 397–413. https://doi.org/10.1016/S0961-9534(02)00067-3

Welsh Government. (2018). Welsh agricultural statistics 2016. Retrieved from https://doi.org/10.1039/c0mt00106f

Welsh Government. (n.d.). Agricultural land classification map. Retrieved from https://beta.gov.wales/agricultural-land-classification-predictive-map

Yang, Q., & Zhang, X. (2016). Improving SWAT for simulating water and carbon fluxes of forest ecosystems. *Science of the Total Environment*, 569–570, 1478–1488. https://doi.org/10.1016/j.scitotenv.2016.06.238

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