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Hedgerow effects on CO₂ emissions are regulated by soil type and season: implications for carbon flux dynamics in livestock-grazed pasture

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Abstract

In this study we assess the potential for farmland hedgerows to provide climate mitigation via carbon (C) storage, using soil carbon dioxide (CO₂) efflux to improve upscaling validity. Two contrasting sites, freely-draining (FD) *versus* seasonally-wet (SW), situated in mixed-livestock farms (Conwy, Wales, UK), were selected. We measured soil CO₂ efflux associated with three field boundaries: hedgerow on SW soil; hedgerow on FD soil; stone wall (abiotic control) on FD soil, quantifying the influence of distance from field boundary and grazing occurrence (grazed pasture *versus* un-grazed zone adjacent to hedgerows) on annual C budgets based on soil CO₂ flux and net primary productivity. For the FD site, the annual C budget showed that pasture was a net source of C emissions ($11 \pm 1.5 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$) and the un-grazed zone adjacent to the hedgerow a net sink ($-0.9 \pm 2.2 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$). For the SW site, pasture acted as a small net sink of C ($-0.1 \pm 1.3 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$) and the hedgerow zone a net source ($5.8 \pm 0.8 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$), due entirely to a spike in soil CO₂ efflux associated with a relatively unusual summer drought. To investigate the effect of this observed summer drought on more typical (for the UK maritime climate) annual C source-sink dynamics, we modelled soil CO₂ efflux for a summer-drought-excluded year for both FD and SW soils. With greater hedgerow cover (modelled prediction compared with a baseline of no hedgerows),

annual CO₂ flux became more negative (greater net sink) in fields on FD soil (by 1 t CO₂ ha⁻¹ yr⁻¹ at 8% hedgerow cover), with drought limiting the effect size. In SW soils, greater hedgerow cover also led to a more negative annual CO₂ flux (by 0.4 t CO₂ ha⁻¹ yr⁻¹ at 8% hedgerow cover) when drought was excluded, but a more positive flux (net C source) with drought included (by 0.5 t CO₂ ha⁻¹ yr⁻¹ at 8% hedgerow cover). This study illustrates the importance of the interaction between soil type and seasonal events such as drought on the ability of hedgerows to act as a net C sink.

Keywords: Agriculture; Agroforestry; Carbon budget; Carbon dioxide; Grassland; Landscape.

1. Introduction

Agricultural activities account for ~15% of the total global greenhouse gas (GHG) emissions that contribute to climate change (IPCC, 2014). Currently, agriculture is responsible for ~50 Mt carbon dioxide (CO₂) equivalent or 10% of total UK emissions (Defra, 2017), predominantly attributed to the livestock sector. Land management strategies that increase climate change regulation are therefore of major interest to policy makers (Thiel et al., 2015). One such strategy is agroforestry, defined as the practice of growing trees together with livestock and/or crops for a variety of benefits, including silvopasture, riparian planting, shelterbelts and hedgerows (Kim et al., 2016). The potential of existing and new farmland hedgerows, defined as lines of trees and shrubs typically managed by regular cutting (Baudry et al., 2000), to provide climate change mitigation via carbon (C) storage has been increasingly recognised over the past decade (Wolton et al., 2014; Scholefield et al., 2016). Despite this, accurate estimates of hedgerow C storage in temperate agroecosystems are rare, with both above- and below-ground biomass (Thiel et al., 2015; Axe et al., 2017), soil CO₂ efflux (Thiel et al., 2017) and soil organic carbon (SOC) storage (Amadi et al., 2016; Ford et al., 2019) largely

unknown, or assessed in combination with other agroforestry systems (Ma et al., 2020). Assessment of hedgerow C budgets at the landscape scale are infrequent and either rely on modelled data (Falloon et al., 2004), focus solely on soil organic carbon (SOC) stock (Walter et al., 2003), or have limited scope for extrapolation to a European setting (Smuckler et al., 2010).

In agroforestry systems, soil CO₂ efflux was consistently found to be greater under or adjacent to trees, hedgerows or shelterbelts than within arable cropped or pasture fields further away from the woody plants (Peichl et al., 2006; Amadi et al., 2016; Amadi et al., 2017; Baah-Acheamfour et al., 2016; Baah-Achemfour et al., 2017; Thiel et al., 2017), with relatively few studies showing the opposite trend (Franzleubbers et al., 2017). Greater soil CO₂ efflux adjacent to trees was attributed to: i) enhanced fine root turnover and rhizodeposition increasing availability of C-rich root exudates for the microbial community (Stevenson et al., 2004; Peichl et al., 2006; Maier et al., 2011), with soil CO₂ efflux positively associated with total SOC and particulate organic matter (Bailey et al., 2009); or ii) a modification in the soil physical structure [i.e. a reduction in soil bulk density (Amadi et al., 2016), and decrease in soil moisture content that combine to create aerobic conditions that promote decomposition processes (Amadi et al., 2017)]. In contrast, the soil CO₂ efflux in broadleaved or coniferous forests was reduced by up to 20%, compared with neighbouring grassland or pasture (Raich and Tufekcioglu, 2000; Smith and Johnson, 2004; Kellman et al., 2007; Hiltbrunner et al., 2013), attributed largely to a vegetation-mediated reduction in soil temperature (Smith et al., 2003).

In addition to the location of trees in the agricultural environment, silvopastoral carbon storage and soil CO₂ efflux are influenced by several factors including soil type, livestock-

grazing intensity and climatic conditions. SOC is positively associated with soil clay content (Jobbagy and Jackson, 2000), as mineral-associated soil organic matter (SOM) is physically protected from microbial decomposition by adsorption onto silt and clay minerals within the soil (Lavallee et al., 2019). Livestock grazing, particularly in temperate dry-cool climates (aerobic soil), is often associated with increased allocation of plant resources below-ground, with enhanced below-ground biomass and root turnover (Kemp and Michalk, 2007), coupled with 'hotspots' of CO₂ emissions from livestock dung (Lin et al., 2009), leading to greater soil CO₂ efflux from intensively-grazed than extensively- or un-grazed grasslands (Abdalla et al., 2018). In contrast, livestock-grazing in temperate moist-cool climates, where seasonally water-logged soils (prone to compaction via livestock trampling) are common, promotes anaerobic soil conditions with suppressed CO₂ efflux and enhanced SOC storage (Wiesmeier et al., 2013; Abdalla et al., 2018).

In this study we assess the contribution of hedgerows to annual C budgets of livestock-grazed pasture land (in the UK maritime climate) on two contrasting soil types, with a particular focus on soil CO₂ efflux. We hypothesise that soil CO₂ efflux is: i) closely associated with soil temperature, soil moisture (aerobic *versus* anaerobic conditions) and distance from hedgerow, which is linked to grazing occurrence; ii) influenced more by the presence of a biotic than an abiotic field boundary (via a decrease in soil temperature and moisture close to the hedgerow relative to more distant pasture). In addition, we aim to combine data on soil CO₂ efflux with proxies for above- and below-ground net primary productivity to quantify the contribution of hedgerows to annual C budgets, for a range of hedgerow land cover scenarios (1-8% cover).

2. Material and methods

2.1 Study area and sampling design

The study area consisted of two sites, located on two tenanted mixed livestock farms (primarily Welsh mountain sheep, with some beef cattle) within the county of Conwy, in Wales, UK, both within the River Conwy catchment. These two sites were chosen to represent two contrasting soil drainage types present within the study catchment (Fig. S1): i) seasonally-wet soil (SW) with impeded drainage (53.033457°, -3.747871°); and ii) free-draining soil (FD) (53.037096°, -3.712010°), with soils of intermediate drainage excluded. Soils were classified for each site using a combination of the UK Soilsclapes soil map, World reference base for soils, and previous field measurements (Table 1): i) SW site – slowly permeable silty-clay stagnosol; and ii) FD site – silty-clay loam cambisol. The two sites were characterised by semi-improved pasture fields with a mixture of productive grass species (e.g. *Lolium perenne*), in most cases mixed with clover (*Trifolium* spp., which is N-fixing), forbs and mosses, bordered by either hedgerows or stone walls as typical field boundaries (Fig. S1). Both sites were categorised as poor (low fertility grade 4 or 5) agricultural land (Agricultural Land Classification of England and Wales, 2018). The maritime climate of Conwy (north-west Wales) is characterised by greater rainfall than most UK regions, with mean annual precipitation close to 2,500 mm (<https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-actual-and-anomaly-maps>). Conwy air temperatures are generally mid-range for the UK, with mean monthly maximum and minimum temperatures of 12 and 6 °C respectively.

Three study field boundaries were selected across the two sites (Fig. S1): i) hedgerow on SW soil, ii) hedgerow on FD soil, iii) stone wall (as an abiotic control) on FD soil. Only field boundaries running perpendicular to a slope of consistent gradient (5 to 10°) were considered

for selection, with boundaries adjacent to fields with known field drains excluded. Field boundaries were also excluded if there was evidence of bare earth or soil poaching adjacent to the boundary (≤ 3 m perpendicular to the boundary) indicating congregation of livestock (associated with enhanced nutrient inputs and localised compaction). Study plots were located ≥ 5 m away from gateways or gaps in the boundary through which livestock and vehicles could travel (which are associated with high levels of localised soil compaction). Characteristics of each boundary are summarised in Table 1. For each boundary, one 30 x 20 m study plot was selected, incorporating an area 10 m upslope and 10 m downslope of the study boundary (Fig. 1). Three transect lines (10 m apart) were set up within each study plot, running perpendicular to the boundary, with each sampling point referenced relative to the centre of the hedgerow or the edge of the stone wall. At six sampling points along each transect line (three upslope and three downslope, see Fig. 1), cylindrical collars (100 mm diameter, 100 mm length) for measurement of soil respiration were inserted into the soil to a depth of 50 mm, one month prior to the start of the study. The study design was structured, comparative observational, not experimental or manipulative.

2.2 Monthly measurements

Daytime soil CO₂ efflux was recorded once per month for one year, from July 2017 to June 2018. Soil respiration from all three boundaries was recorded within 48 hours during similar weather conditions. Soil CO₂ efflux was measured at each collar sampling point, after above-ground biomass was clipped to ground level, by a portable CO₂ gas analyser, either a LI-COR survey system [via a 10 cm survey chamber (8100-102, LI-COR) attached to the analyser control unit (LI-8100A, LI-COR)] or an EGM-5 [(PP SYSTEMS) attached to a 10 cm soil respiration chamber (SRC-2, PP SYSTEMS)]. For each month's measurements only one type of

gas analyser was used for all soil CO₂ efflux measurements. These two portable CO₂ gas analyser systems were compared under field conditions in agricultural grasslands in the River Conwy catchment equivalent to the present study and found to produce extremely similar results with no significant difference ($p = 0.98$) by Mills et al. (2011). Linear fluxes were calculated using SoilFluxPro (v4.0.1, LI-COR Biosciences). The accuracy of CO₂ (ppm) detection in both LI-COR and EGM-5 gas analysers was measured using British Oxygen Company (BOC, UK) standard gases (250, 500, 1250 and 2500 CO₂ ppm). To bring fluxes from each gas analyser in line with the standard gases the following conversions were used for the LI-COR [field measured soil CO₂ efflux rate ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) $\times 1.04$ = corrected soil CO₂ efflux rate ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)] and EGM-5 [field measured soil CO₂ efflux rate ($\text{g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) $\times 1.02$ = corrected soil CO₂ efflux rate ($\text{g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$)]. The EGM-5 corrected soil CO₂ efflux rate was converted from $\text{g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ to $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ by multiplying the soil CO₂ efflux rate ($\text{g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) by 6.312.

Adjacent to each sampling point, soil temperature (10 cm depth, Checktemp thermometer) and soil moisture content (0-8 cm depth, Theta Probe ML2x and Moisture Meter HH2, Delta-T Devices Ltd) were recorded once during each monthly gas-flux sampling occasion.

2.3 Estimation of C availability

Soil cores (0.15 m deep, 0.05 m diameter) for the measurement of SOC stock were sampled once (during autumn) alongside each un-grazed sampling points for both the SW ($n = 3$) and the FD ($n = 3$) hedgerows. Soil cores were also collected in the grazed pasture at 1.2 m from the boundary fence of the SW and FD hedgerows as part of measurements published in Ford et al. (2019). SOC concentration (g kg^{-1} of dry soil mass) was calculated using the conversion factor of 0.55 of SOM mass, with SOC stock (kg C m^{-2}) of the 0-0.15 m depth re-calculated on

an equivalent soil mass (ESM) basis, a layer of 1,000 t ha⁻¹ as in Lee et al. (2009); for a full description of the methods see Ford et al. (2019). Here, SOC stock is expressed on an ESM basis in kg C m⁻² to allow SOC stock to be compared uncoupled from the influence of soil compaction.

2.4 Data analysis

Linear mixed-effects models were used to determine associations between: i) soil CO₂ efflux rate and six potential explanatory variables [soil temperature, soil moisture, month, slope position (upslope *versus* downslope), distance (perpendicular distance from boundary at 0.7 m, 2 m and 10 m, see Fig. 1) and grazing occurrence (two-level categorical variable incorporating proximity to hedgerow: close-to-hedgerow un-grazed zone at 0.7 m from the hedgerow from which livestock were excluded by the fence *versus* further-from-hedgerow grazed pasture at 2 m or 10 m from the hedgerow, see Fig. 1)]; ii) soil temperature and four explanatory variables (distance, grazing occurrence, month and slope position); and iii) soil moisture and four explanatory variables (distance, grazing occurrence, month, slope position). These analyses were carried out for soil adjacent to the following three field boundaries: i) SW hedgerow; ii) FD hedgerow; iii) FD stone wall and iv) data from all three field boundaries combined. For all sets of linear mixed-effects models, normal distribution of modelled variables was assessed visually using quantile-quantile plots with variables log transformed to improve fit where necessary. Best model fit was selected on the basis of lowest Akaike Information Criteria (AIC) value. Likelihood-ratio-based pseudo-R-squared values were also calculated for each model, using R package 'MuMIn' (Bartoń, 2018). Results were presented using the ANOVA output of the mixed effects models for ease of interpretation. All statistical analysis was carried out in R (R Core Team, 2018).

Further analysis was carried out for the growing season of May-September (October-April data excluded), when hedgerows were in full leaf, using a step-wise regression approach. Step-wise regressions ‘forwards and backwards’ were carried out in the ‘MASS’ package (Venables and Ripley, 2002) using linear models of i) soil CO₂ efflux rate (response variable) and five potential explanatory variables (soil temperature, soil moisture, slope position, distance and grazing occurrence); ii) soil temperature (response variable) and three explanatory variables (slope position, distance and grazing occurrence); iii) soil moisture (response variable) and three explanatory variables (slope position, distance and grazing occurrence). As month influences both soil temperature and moisture it was excluded as an explanatory variable due to potentially confounding effects. This analysis was carried out (using May to September data only) for: i) the SW hedgerow; ii) the FD hedgerow; iii) the FD stone wall; iv) data from all three field boundaries combined. Explanatory variables were only entered into the step-wise regression if hierarchical partitioning (<http://cran.r-project.org/package=hier.part>) analysis assessed them to have $\geq 5\%$ independent effects. Results of the stepwise regression displayed a ‘final model’ selected by lowest AIC, usually with fewer variables than the ‘initial model’. From this model the individual contribution of each remaining environmental variable to the overall variation explained was calculated using the ‘lmg’ function of the ‘relaimpo’ package (Grömping, 2006) using simple unweighted averages as recommended.

The apparent temperature sensitivity of soil respiration, assumed here to be equivalent to soil CO₂ efflux (as in Domínguez et al., 2017), was assessed for: i) SW pasture (further-from-hedgerow, grazed); ii) SW close-to-hedgerow (un-grazed); iii) FD pasture (further-from-hedgerow, grazed); iv) FD close-to-hedgerow (un-grazed); v) data from both hedgerows (i-iv) and stone walls combined, for two scenarios: drought period-included (12 month dataset)

and drought period-excluded (11 month dataset). Soil respiration (soil CO₂ efflux) data were fitted against soil temperature (at 10 cm depth) using an exponential function: $SR = ae^{bT}$, where SR is soil respiration, T is soil temperature, and a and b are fitted constants. Q₁₀ values (increase in soil respiration per 10 °C increase in temperature) were calculated as $Q_{10} = e^{10b}$ (Suseela et al., 2012).

2.5 Annual carbon budgets

An annual C budget was calculated for two land cover types: i) un-grazed zone close to hedgerows; and ii) livestock-grazed pasture further from hedgerows, on two contrasting soil types, FD (brown earth) and SW (stagnogley). Annual soil CO₂ efflux rates were calculated from monthly means (12 months inclusive) for the grazed pasture [10 m sampling point (mean value of upslope and downslope) perpendicular to field boundary] and the un-grazed zone adjacent to the hedgerow [0.7 m sampling point (mean value of upslope and downslope) perpendicular to field boundary and protected by the livestock-exclusion fences] for both FD and SW sites and converted into t CO₂ ha⁻¹ yr⁻¹. As CO₂ efflux was not recorded in December for the SW site, modelled values (using the drought-included Q₁₀ relationship, Table 3) were used to provide realistic data for this site-month combination. Results were expressed as drought period-included (12 month dataset, detailed above) and drought period-excluded [calculated as above but with field-measured soil CO₂ efflux rates for the drought period of May and June removed and replaced with modelled values (using the drought-excluded Q₁₀ relationship, Table 3) to give a 12-month dataset] scenarios, to illustrate the potential impact of seasonal drought. The two month period May-June 2018 was defined as a drought period in the River Conwy catchment, with a mean Standardized Precipitation Index (SPI) of -1 (Centre for Ecology and Hydrology UK Drought Tool <https://eip.ceh.ac.uk/apps/droughts/>,

baseline comparison data 1961-2010), and mean precipitation rate of $< 1.5 \text{ mm day}^{-1}$ (based on *in-situ* weather stations). Over the 20-year 2000-2020 period a May-June drought of similar magnitude was relatively unusual (15%). In contrast, July-August 2017 was not considered a drought period (as the River Conwy catchment had a mean SPI of 1) despite relatively low soil moisture being recorded on the measurement days for July and August in SW soil in the present study.

Above-ground net primary productivity (ANPP) values for the livestock-grazed pasture and the un-grazed hedgerow zone were taken from the measurements made in semi-improved pasture and broadleaved woodland respectively at other sites in the Conwy River catchment by Smart et al. (2016). Values for fine-root biomass to 15 cm depth were taken from the measurements made in semi-improved pasture and broadleaved woodland in the Conwy River catchment by Smart et al. (2017) with this depth assumed to account for 100% of grass and 70% of hedgerow woody plant fine-root biomass (broadleaved woodland data; Macinnis-Ng et al., 2010) respectively, with fine-root hedgerow woody plant biomass adjusted accordingly (total root biomass = root biomass 0-15 cm depth $\times 1.3$). Below-ground net primary productivity (BNPP) was calculated from the adjusted fine-root biomass, using a root turnover rate of 0.5 yr^{-1} suitable for both grassland and woodland habitats (Gill and Jackson, 2000), and converted to $\text{t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$. Soil CO_2 efflux rates (C source), and ANPP and BNPP (C sinks) were combined to give a comparative flux estimate (either net C source or sink with a value in $\text{t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$) for each combination of soil type (SW / FD), cover type (pasture / hedgerow) and drought condition (included / excluded).

2.6 Scaling up

To predict the estimated effect of enhanced hedgerow cover on annual C budgets at a landscape scale, changes in the CO₂ flux estimate under a range of greater hedgerow cover scenarios (based on a model 1-ha field) compared with a baseline of 100% pasture (0% hedgerow cover) were calculated. Values for seasonally-wet (SW) *versus* free-draining (FD) soils under two drought scenarios (included *versus* excluded, for full description see section 2.5) for either pasture or hedgerow features were extrapolated from net C source / sink value calculations (as in section 2.5) and adjusted according to the relative percentage of pasture and hedgerow cover. For example, a hedgerow density of 50 m ha⁻¹ with 2 m width is equivalent to 1% hedgerow cover (99% pasture cover), reflecting current UK hedgerow density. Hedgerow densities of 200 m ha⁻¹ (4% cover) and 400 m ha⁻¹ (8% cover) were presented as two possible options on the projected hedgerow cover continuum. For this scaling-up exercise all hedgerows were assumed to be double-fenced to exclude livestock.

3. Results

3.1 Monthly measurements

Soil CO₂ efflux rate was significantly associated with four variables: i) grazing occurrence (positive, $P < 0.001$); ii) soil temperature (positive, $P < 0.05$); iii) soil moisture (negative, $P < 0.05$); iv) month ($P < 0.001$), with two significant interaction terms [month x soil temperature ($P < 0.001$) and month x soil moisture ($P < 0.001$)], for the year-long dataset for soil adjacent to all three contrasting field boundaries combined (SW hedgerow, FD hedgerow and FD stone wall). This model explained close to three quarters of the variation in soil CO₂ efflux rate ($r^2 = 0.74$). When the three field boundaries were considered separately, soil CO₂ efflux was significantly associated with soil temperature, moisture and month for soil adjacent to both the SW and FD hedgerows (Figs. 2a, 3a), with grazing occurrence an additional explanatory

factor for the FD hedgerow. These combined models explained over 80% of the variation in soil CO₂ efflux ($r^2 = 0.82-0.88$). For soil adjacent to the stone wall (FD), 74% of the variation in soil CO₂ efflux was explained by soil temperature and moisture. Apparent temperature sensitivity of soil respiration (soil CO₂ efflux) was greater in the livestock-grazed pasture than in the un-grazed zone associated with the hedgerow for both SW and FD sites (Table 2).

Soil temperature was significantly related to both perpendicular distance from boundary and month, for each of the three field boundaries (Figs. 2b, 3b, 4b), with temperature consistently greatest further away from the boundary edge. In addition, for each hedgerow category (SW and FD) soil temperature was positively associated with grazing occurrence. Soil moisture was significantly positively associated with grazing occurrence and with month for each (SW and FD) hedgerow category with 80 and 87% of variation in soil moisture explained respectively (Figs. 2c, 3c). The main difference was apparent between the un-grazed soil close to the hedgerow inside the boundary fence, which exhibited significantly lower soil moisture content than soil either 2 m or 10 m from the hedgerow, in the livestock-grazed part of the field (outside the boundary fence). There was no significant association between soil moisture and distance from boundary for the stone wall control site, which had no fencing (Fig. 4c). Slope position (upslope *versus* downslope) from the boundary was not significantly associated with soil CO₂ efflux rate, soil temperature or soil moisture for either the three field boundary types combined or for each separately.

3.2 Estimate of SOC stock

For the FD site SOC stock was very similar between the grazed pasture ($6.0 \pm 0.2 \text{ kg C m}^{-2}$) and the un-grazed zone adjacent to the hedgerow ($6.2 \pm 0.1 \text{ kg C m}^{-2}$). For the SW site SOC stock

was greater adjacent to the hedgerow (un-grazed) than in the grazed pasture (21 ± 2 and $10 \pm 0.3 \text{ kg C m}^{-2}$ respectively).

3.3 Growing season only

For the growing season (May-September), results were largely in line with those for the annual datasets (*section 3.1*), with distance from hedgerow/grazing occurrence (grazed *versus* un-grazed) a significant independent (as assessed by hierarchical partitioning) explanatory variable of soil CO_2 efflux, soil temperature and soil moisture for both SW and FD hedgerows (Table 3); all increased with distance/grazing (Figs 2 and 3).

3.4 Annual carbon budgets

For the FD (brown earth) soil site, the annual C budget (based on soil CO_2 efflux and net primary productivity) showed a marked difference between livestock-grazed pasture and the un-grazed zone adjacent to the hedgerow (Fig. 5), with the pasture acting as a net source of C ($10.8 \pm 1.5 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$) and the hedgerow zone as a net sink ($-0.9 \pm 2.2 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$). This result is entirely due to the large reduction in annual soil CO_2 efflux rate adjacent to the hedgerow compared with the grazed pasture (of 20.5 and $33.9 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$ respectively). For the drought period-excluded scenario the pasture remained a net source of C of comparative magnitude but the strength of the C sink in the soil adjacent to the hedgerow was increased six-fold to $-6.5 \pm 0.7 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$. For the SW (stagnogley) site, the annual C budget also showed a marked difference between livestock-grazed pasture and the un-grazed zone adjacent to the hedgerow (Fig. 5), but in the opposite direction to the FD site, with the pasture acting as a small net sink of C ($-0.12 \pm 1.3 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$) and the hedgerow as a net source ($5.8 \pm 0.8 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$). For the drought period-excluded scenario the pasture

remained a small net C sink, with the hedgerow zone reverting from a net source to a large net C sink ($-9.9 \pm 0.3 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$).

3.5 *Scaling up*

With greater hedgerow land-cover, in fields on FD soil the estimated annual CO_2 flux became more negative (greater net sink), based on modelled prediction compared with a baseline of no hedgerows, with drought reducing the size of this effect (Fig. 6). In fields on SW soils, greater hedgerow cover also caused a more negative annual CO_2 flux estimate under the 'drought-excluded' scenario (by $\sim 0.4 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$ at 8% hedgerow cover), but caused a more positive flux (net C source) under the summer 'drought period-included' scenario (by $\sim 0.5 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$ at 8% hedgerow cover).

4. Discussion

4.1 *Soil CO₂ efflux: Temperature, moisture and grazing occurrence*

In this study, soil CO_2 efflux was closely associated with soil temperature and soil moisture, but affected more by grazing occurrence (close-to-hedgerow un-grazed zone *versus* further-from-hedgerow grazed pasture), than distance from hedgerows *per se*, leading to a partial acceptance of our first hypothesis. Seasonal soil temperature was one of the key abiotic factors regulating soil CO_2 efflux in this study, with an increase in soil temperature associated with greater soil CO_2 efflux, as expected in temperate ecosystems (Smith et al., 2003). Daytime soil temperatures during the May to September growing season were reduced by 3–5 °C within the un-grazed zone adjacent to both study hedgerows (Figs. 2, 3), relative to the short-sward pasture, which is likely to be due to the combined sheltering effect of the hedgerow itself and the understorey plant layer. This vegetation-mediated buffering of

extreme temperatures is well recognised (Stevenson et al., 2004) and largely explains the lower soil CO₂ efflux associated with hedgerows for the majority of the year. Annual apparent temperature sensitivity of soil respiration (soil CO₂ efflux) was high (Q₁₀ values of 5-10) in comparison with the global biome mean of 1.43-2.03 (Zhou et al., 2009), but indicative of ecosystems where seasonality is marked (Domínguez et al., 2017) and Q₁₀ values are regulated by vegetation activity (Wang et al., 2010). Here, apparent Q₁₀ values were greater in livestock-grazed pasture than in the un-grazed zone adjacent to hedgerows in both contrasting soil types (Table 3) indicating that other variables (e.g. soil moisture) may partially regulate the temperature dependency of soil CO₂ efflux adjacent to un-grazed hedgerows.

Soil CO₂ efflux was negatively associated with soil moisture content, as is usual in agricultural grasslands (Abdalla et al., 2018), with soil CO₂ efflux far greater in FD than SW pasture. Constantly aerobic FD pasture is often associated with greater resource allocation below ground and enhanced fine root turnover, leading to greater soil CO₂ efflux than for SW soils that are periodically anaerobic, where below-ground allocation and turnover are minimised and soil organic C storage is enhanced (Jobbagy and Jackson, 2000; Wiesmeier et al., 2013; Abdalla et al., 2018). Soil moisture was reduced by the presence of hedgerows, with effects most marked within the 2-m un-grazed zone associated with the hedgerow itself, due to a probable combination of woody plant roots extracting soil moisture from the soil (Kowalchuk and de Jong, 1995) and an enhanced water infiltration rate due to the absence of grazing compaction (Marshall et al., 2009). Recent evidence shows that soil moisture levels moderate the temperature dependency of soil CO₂ efflux (Lellei-Kovacs et al., 2016), particularly at soil temperatures > 10° C, common during spring and summer in UK uplands. During the initial summer drought-period, the CO₂ efflux of SW soils was much greater in the recently dry un-grazed hedgerow zone than in the relatively wetter pasture, despite a soil temperature

differential of $< 2^{\circ}\text{C}$. This mirrors results from UK shrubland (Lellei-Kovacs et al., 2016) and illustrates the importance of incorporating soil moisture into predictive models of soil CO_2 efflux.

Soil CO_2 efflux was positively correlated with the occurrence of livestock grazing in this study, with effects particularly noticeable for the growing season. This grazing effect can be explained partly by the influence of livestock on soil temperature-moisture dynamics (as detailed above), but the independent grazing effects identified (Table 4) may be due to preferential allocation of plant resources below-ground (Kemp and Michalk, 2007) and/or CO_2 emissions from livestock dung (Lin et al., 2009). In addition, grazing occurrence was a stronger indicator of soil CO_2 efflux than distance from hedgerow, indicating a swift transition between the un-grazed zone associated with the study hedgerows and grazed pasture at the hedgerow livestock-exclusion fence, particularly in FD soil (Fig. 3). It is possible that soil CO_2 efflux was also influenced by broad differences in root exudate C sources available to the soil microbial community (Stevenson et al., 2004) based on the proximity of sample points to woody or pasture plants, but this was not measured directly.

4.2 Biotic versus abiotic field boundaries

Soil CO_2 efflux was influenced more by the presence of a biotic than an abiotic field boundary, with the un-grazed zone associated with the hedgerow characterised by lower soil temperature and soil moisture relative to more distant pasture, supporting our third hypothesis. Summer soil temperature was reduced, but only by $\leq 1^{\circ}\text{C}$, in the immediate vicinity of the stone wall (Fig. 4), illustrating the difference between biotic and abiotic field boundaries in their buffering of soil temperature and thus regulation of soil CO_2 efflux. Despite limited evidence of stone walls reducing run off and enhancing water infiltration rates

(Kovář et al., 2011; Rodrigo-Comino et al., 2019) this type of abiotic field boundary did not influence soil moisture dynamics in the present study.

4.3 Carbon budgets

Annual C budgets of hedgerows and livestock-grazed pasture, on two contrasting soil types typical of UK uplands, were calculated by combining data on soil CO₂ efflux with proxies for above- and below-ground net primary productivity. In this study, livestock-grazed pastures acted as a small net C sink in SW soil and a net source in FD soil, in line with the results of Abdalla et al. (2018), with the drought period having only a minimal effect on carbon sink-source dynamics. Hedgerows, including the soil of their adjacent un-grazed zones, were net C sinks under the drought period-excluded scenario, storing 6-10 t CO₂ ha⁻¹ yr⁻¹ (Fig. 5), which is substantially lower than the 15-40 t CO₂ ha⁻¹ yr⁻¹ stored in agroforestry systems according to a review of C budgets (Kim et al., 2016), although this review included a broad range of agroforestry types and did not include hedgerows specifically. Under the drought period-included scenario, hedgerows remained a net, though smaller in magnitude, C sink in FD soil (of ~1 t CO₂ ha⁻¹ yr⁻¹) but became a net C source in SW soil (~6 t CO₂ ha⁻¹ yr⁻¹) due to a doubling of CO₂ efflux (relative to the drought-excluded scenario). This huge spike in soil CO₂ efflux in SW soil occurred entirely within the first month of the drought period and coincided with a sudden switch from moist-cool to dry-warm soil conditions. This mirrors the transition from flooded to non-flooded conditions in forested wetland and seasonally flooded forests (Miao et al., 2013; Barbosa et al., 2017), where newly aerobic soil stimulates root growth and decomposition of SOM (including necromass accumulated via root death during anaerobic period), enhancing autotrophic/microbial respiration and subsequent CO₂ efflux (Peichl et al., 2006; Amadi et al., 2017).

4.4 Scaling up

Here, we assessed the potential for farmland hedgerows to provide climate change mitigation via carbon storage, attempted previously by Falloon et al. (2004), using measured soil CO₂ efflux to improve the validity of upscaling. At present total UK hedgerow land cover is ~400,000 km (Elliot et al., 2014; Scholefield et al., 2016), equivalent to 50 m ha⁻¹ across the whole agricultural land area. At this level (1% land cover) the impact of hedgerows on the C budget at a landscape scale was minimal for both FD and SW soils. However, the capacity of the farmland landscape to act as a C sink was enhanced by increased hedgerow cover (in scenarios up to 8% cover). This effect is in accord with previous studies of pastures in Europe and USA, where a positive contribution of hedgerows to SOC storage was reported (Walter et al., 2003; Smuckler et al., 2010; Lacoste et al., 2015). Although this pattern holds for both soil types under the drought period-excluded scenario, the capacity for hedgerows to contribute to a net C sink at the landscape scale was reversed in SW soil during periods of drought (Fig. 6); in these conditions increased hedgerow cover resulted in greater C emissions. Although informative, with potentially important implications for C storage capacity, data for the early-summer drought scenario are from only 2 months duration, associated with a relatively unusual climatic event for the UK maritime conditions, and should therefore be extrapolated with caution. Relative abundance of soil types is also relevant to this upscaling exercise. If we exclude peat soils [where neither livestock-grazing or tree-planting is advised (Ostle et al., 2009)], FD and SW (including impeded drainage) soil types each equate to approximately one half of Welsh upland land-cover respectively (Hallett et al., 2017). However, as this approach amalgamates related soils into two broad categories on the basis of drainage, future work could take a more nuanced approach by studying pasture-hedgerow dynamics across a wider range of UK or upland soil types.

4.5 Implications for policy makers

Carbon budgets were modelled for a range of hedgerow land cover scenarios (1-8%), either including or excluding the effect of a naturally-occurring early-summer drought period. Under the drought-excluded scenario, the un-grazed zone adjacent to hedgerows acted as a net C sink in both the contrasting soil types, allowing an increase in C stored with greater hedgerow cover. Taken in isolation this result could be used as evidence to promote hedgerow planting on agricultural land, regardless of soil type, in an attempt to meet climate change mitigation targets via C storage. However, during drought conditions, hedgerow-associated soil CO₂ efflux increased markedly, effectively ‘pausing’ the effect of hedgerow cover as a C storage mechanism. Moreover, on the SW soils characteristic of some upland farms in the UK (Hallett et al., 2017), the sudden change in soil temperature and moisture dynamics associated with a drought period triggered a spike in CO₂ efflux that turned hedgerows into a net annual C source. As a result, greater hedgerow cover (up to 8%) could potentially increase net C emissions, although our evidence base for this conclusion is limited. Our study illustrates the importance of considering the impact of soil type and seasonal extreme events such as drought on the capacity of hedgerows to act as a net C sink, with clear implications for policy makers and land managers tasked with meeting the objective of minimising the net CO₂ emissions from farmland.

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References

- Abdalla, M., Hastings, A., Chadwick, D.R., Jones, D.L., Evans, C.D., Jones, M.B., Rees, R.M., & Smith, P. (2018). Critical review of the impacts of grazing intensity on soil organic carbon storage and other soil quality indicators in extensively managed grasslands. *Agriculture, Ecosystems and Environment*, 253, 62-81.
- Elliot, J., Evans, C., Moxley, J., Chadwick, D., Steve, A., Durrant, C., Moorby, J., Misslebrook, T., Smith, S., Styles, D., & Jones, D. (2014). *Review of land use climate change: An assessment of the evidence base for climate change action in the agriculture, land use and wider foodchain sectors in Wales*. ADAS UK Ltd., Leeds, U.K., 151pp. (CEH Project no: NEC05228).
- Agricultural Land Classification of England and Wales (2018). <https://beta.gov.wales/sites/default/files/publications/2018-02/agricultural-land-classification-frequently-asked-questions.pdf>.
- Amadi, C.C., Van Rees, C.J., & Farrell, E. (2016). Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agriculture, Ecosystems and Environment*, 223, 123-134.
- Amadi, C.C., Farrell, R.E., & Van Rees, K.C.J., (2017). Greenhouse gas emissions along a shelterbelt-cropped field transect. *Agriculture, Ecosystems and Environment*, 241, 110-120.
- Axe, M.S., Grange, I.D. & Conway, J.S., (2017). Carbon storage in hedge biomass – A case study of actively managed hedges in England. *Agriculture, Ecosystems and Environment*, 250, 81-88.

479 Baah-Acheamfour, A., Carlyle, C.N., Lim, S-S., Bork, E.W., & Chang, S.X. (2016). Forest and
 480 grassland cover types reduce net greenhouse gas emissions from agricultural soils. *Science of*
 481 *the Total Environment*, 571, 1115-1127.

482 Baah-Achemfour, M., Chang, S.X., Bork, E.W. & Carlyle, C.N. (2017). The potential of
 483 agroforestry to reduce atmospheric greenhouse gases in Canada: Insight from pairwise
 484 comparisons with traditional agriculture, data gaps and future research. *The Forestry*
 485 *Chronicle*, 93, 180-189.

486 Bailey, N.J., Motavalli, P.P., Udawatta, R.P., & Nelson, K.A. (2009). Soil CO₂ emissions in
 487 agricultural watersheds with agroforestry and grass contour buffer strips. *Agroforestry*
 488 *Systems*, 77, 143-158.

489 Barbosa, R.I., Volkmer de Castilho, C., de Oliveira Perdiz, R., Damasco, G., Rodrigues, R., &
 490 Fearnshide, P.M. (2017). Decomposition rates of coarse woody debris in undisturbed
 491 Amazonian seasonally flooded and unflooded forests in the Rio Negro-Rio Branco Basin in
 492 Roraima, Brazil. *Forest Ecology and Management*, 397, 1-9.

493 Bartoń, K. (2018). *MuMIn: Multi-Model Inference*. [https://cran.r-](https://cran.r-project.org/web/packages/MuMIn/MuMIn.pdf)
 494 [project.org/web/packages/MuMIn/MuMIn.pdf](https://cran.r-project.org/web/packages/MuMIn/MuMIn.pdf).

495 Baudry, J., Bunce, R.G.H. & Burel, F. (2000). Hedgerows: An international perspective on their
 496 origin, function and management. *Journal of Environmental Management*, 60, 7-22.

497 Defra (2017). *Agricultural statistics and climate change*. Department for Environment, Food
 498 and Rural Affairs.
 499 [https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment](https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/666073/agriclimate-8edition-8dec17.pdf)
 500 [data/file/666073/agriclimate-8edition-8dec17.pdf](https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/666073/agriclimate-8edition-8dec17.pdf)

501 Domínguez, M.T., Smith, A.R., Reinsch, S. & Emmett, B.A. (2017). Inter-annual variability of
 502 soil respiration in wet shrublands: do plants modulate its sensitivity to climate? *Ecosystems*,
 503 20, 796-812.

504 Falloon, P., Powlson, D. & Smith, P. (2004). Managing field margins for biodiversity and carbon
 505 sequestration: a Great Britain case study. *Soil Use and Management*, 20, 240-247.

506 Ford, H., Healey, J.R., Webb, B., Pagella, T.F. & Smith, A.R. (2019). How do hedgerows
 507 influence soil organic carbon stock in livestock-grazed pasture? *Soil Use and Management*,
 508 doi: 10.1111/sum.12517.

509 Franzleubbers, A.J., Chappell, J.C., Shi, W. & Cubbage, F.W. (2017). Greenhouse gas emissions
 510 in an agroforestry system of the southeastern USA. *Nutrient Cycling in Agroecosystems*, 108,
 511 85-100.

512 Gill, R.A., & Jackson, R.B. (2000). Global patterns of root turnover for terrestrial ecosystems.
 513 *New Phytologist*, 147, 13-31.

514 Grömping, U. (2006). Relative importance for linear regression in R: the package relaimpo.
 515 *Journal of Statistical Software*, 17, 1-27.

516 Hallett, S.H., Sakrabani, R., Keay, C.A. & Hannam, J.A. (2017). Developments in land
 517 information systems: examples demonstrating land resource management capabilities and
 518 options. *Soil Use and Management*, 33, 514-529.

519 Hiltbrunner, D., Zimmermann, S. & Hagedorn, F. (2013). Afforestation with Norway spruce on
 520 a subalpine pasture alters carbon dynamics but only moderately affects soil carbon storage.
 521 *Biogeochemistry*, 115, 251-266.

522 IPCC (2014). Climate Change 2014: Synthesis Report. *Contribution of Working Groups I, II and*
 523 *III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* [Core
 524 Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

525 Jobbagy, E.G. & Jackson, R.B. (2000). The vertical distribution of soil organic carbon and its
 526 relation to climate and vegetation. *Ecological Applications*, 10, 423-436.

527 Kellman, L., Beltrami, H. & Risk, D. (2007). Changes in seasonal soil respiration with pasture
 528 conversion to forest in Atlantic Canada. *Biogeochemistry*, 82, 101-109.

529 Kemp, D.R. & Michalk, D.L. (2007). Towards sustainable grassland and livestock management.
 530 *Journal of Agricultural Science*, 145, 543-564.

531 Kim, D-G., Kirschbaum, M.U.F., & Beedy, T.L. (2016). Carbon sequestration and net emissions
 532 of CH₄ and N₂O under agroforestry: Synthesizing available data and suggestions for future
 533 studies. *Agriculture, Ecosystems and Environment*, 226, 65-78.

534 Kovář, P., Vaššová, D. & Hrabalíková, M. (2011). Mitigation of surface runoff and erosion
 535 impacts on catchment by stone hedgerows. *Soil and Water Research*, 6, 5-16.

536 Kowalchuk, T.E., & de Jong, E. (1995). Shelterbelts and their effect on crop yield. *Canadian*
 537 *Journal of Soil Science*, 75, 543-550.

538 Lacoste, M., Viaud, V., Michot, D. & Walter, C. (2015). Landscape-scale modelling of erosion
 539 processes and soil carbon dynamics under land-use and climate change in agroecosystems.
 540 *European Journal of Soil Science*, 66, 780-791.

541 Lavallee, J.M., Soong, J.L., & Cotrufo, M.F. (2019) Conceptualizing soil organic matter into
 542 particulate and mineral-associated forms to address global change in the 21st century. *Global*
 543 *Change Biology*, 26, 261-273.

544 Lee, J., Hopmans, J. W., Rolston, D. E., Baer, S. G., & Six, J. (2009). Determining soil carbon
545 stock changes: simple bulk density corrections fail. *Agriculture, Ecosystems and Environment*,
546 134, 251-256.

547 Lellei-Kovács, E., Botta-Dukát, Z., de Dato, G., Estiarte, M., Guidolotti, G., Kopittke, G.R.,
548 Kovács-Láng, E., Kröel-Dulay, G., Larsen, K.S., Peñuelas, J., Smith, A.R., Sowerby, A., Tietema,
549 A., & Schmidt, I.K. (2016). Temperature Dependence of Soil Respiration Modulated by
550 Thresholds in Soil Water Availability Across European Shrubland Ecosystems. *Ecosystems*, 19,
551 1460-1477.

552 Lin, X., Wang, S., Ma, X., Xu, G., Luo, C., Li, Y., Jiang, G., & Xie, Z. (2009). Fluxes of CO₂, CH₄ and
553 N₂O in an alpine meadow affected by yak excreta on the Qinghai-Tibetan plateau during
554 summer grazing periods. *Soil Biology and Biochemistry*, 41, 718-725.

555 Ma, Z., Chen, H.Y.H., Bork, E.W., Carlyle, C.N. & Chang, S.X. (2020). Carbon accumulation in
556 agroforestry systems is affected by tree species diversity, age and regional climate: A global
557 meta-analysis. *Global Ecology and Biogeography*, DOI: 10.1111/geb.13145

558 Maier, M., Schack-Kirchner, H., Hildebrand, E.E. & Schindler, D. (2011). Implications for flux
559 models. *Agricultural and Forest Meteorology*, 151, 1723-1730.

560 Marshall, M.R., Francis, O.J., Frogbrook, Z.L., Jackson, B.M., McIntyre, N., Reynolds, B.,
561 Solloway, I., Wheeler, H.S. & Chell, J. (2009). The impact of upland land management on
562 flooding: results from an improved pasture hillslope. *Hydrological Processes*, 23, 464-475.

563 Macinnis-Ng, C.M.O., Fuentes, S., O'Grady, A.P., Palmer, A.R., Taylor, D., Whitley, R.J., Yunusa,
564 I., Zeppel, J.B., & Eamus, D. (2010). Root biomass distribution and soil properties of an open
565 woodland on a duplex soil. *Plant and Soil*, 327, 377-388.

566 Miao, G., Noormets, A., Domec, J-C, Trettin, C.C., McNulty, S.G., Sun, G., & King, J.S. (2013).
 567 The effect of water table fluctuation on soil respiration in a lower coastal plain forested
 568 wetland in the southeastern U.S. *Journal of Geophysical Research Biogeosciences*, 118, 1748-
 569 1762.

570 Mills, R., Glanville, H., McGovern, S., Emmett, B., & Jones, D.L. (2011). Soil respiration across
 571 three contrasting ecosystem types: comparison of two portable IRGA systems. *Journal of*
 572 *Plant Nutrition and Soil Science*, 174, 532-535.

573 Ostle, N.J., Levy, P.E., Evans, C.D., & Smith, P. (2009) UK land use and soil carbon
 574 sequestration. *Land Use Policy*, 26, 274-283.

575 Peichl, M., Thevathasan, N.V., Gordon, A.M., Huss, J., & Abohassan, R.A. (2006). Carbon
 576 sequestration potentials in temperate tree-based intercropping systems, southern Ontario,
 577 Canada. *Agroforestry Systems*, 66, 243-257.

578 R Core Team (2018). *R: A language and environment for statistical computing*. R Foundation
 579 for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.

580 Raich, J.W., & Tufekcioglu, A. (2000). Vegetation and soil respiration: Correlations and
 581 controls. *Biogeochemistry*, 48, 71-90.

582 Rodrigo-Comino, J., Seeger, M., Iserloh, T., González, J.M.S., Ruiz-Sinoga, J.D., & Ries, J.B.
 583 (2019). Rainfall-simulated quantification of initial soil erosion processes in sloping and poorly
 584 maintained terraced vineyards – Key issues for sustainable management systems. *Science of*
 585 *the Total Environment*, 660, 1047-1057.

586 Scholefield, P., Morton, D., Rowland, C., Henrys, P., Howard, D., & Norton, L. (2016). A model
 587 of the extent and distribution of woody linear features in rural Great Britain. *Ecology and*
 588 *Evolution*, 6, 8893-8902.

589 Smart, S.M., Reinsch, S., Mercado, L., Blanes, M.C., Cosby, B.J., Glanville, H.C., Jones, D.L.,
 590 Marshall, M.R., & Emmett, B.A. (2017). Plant aboveground and belowground standing
 591 biomass measurements in the Conwy catchment in North Wales (2013 and 2014). *NERC*
 592 *Environmental Information Data Centre*. [https://doi.org/10.5285/46bb0117-ed5d-4167-](https://doi.org/10.5285/46bb0117-ed5d-4167-a375-d84d1237cf21)
 593 [a375-d84d1237cf21](https://doi.org/10.5285/46bb0117-ed5d-4167-a375-d84d1237cf21)

594 Smart, S.M., Reinsch, S., Mercado, L., Blanes, M.C., Cosby, B.J., Glanville, H.C., Jones, D.L.,
 595 Marshall, M.R., & Emmett, B.A. (2016). Plant structural measurements in North Wales and
 596 Northwest England 2013 and 2014. *NERC Environmental Information Data Centre*.
 597 <https://doi.org/10.5285/8899768c-cc5a-4885-a88b-c08374ee568e>

598 Smith, D.L., & Johnson, L. (2004). Vegetation-mediated changes in microclimate reduce soil
 599 respiration as woodlands expand into grasslands. *Ecology*, 85, 3348-3361.

600 Smith, K.A., Ball, T., Conen, F., Dobbie, K.E., Massheder, J., & Rey, A. (2003). Exchange of
 601 greenhouse gases between soil and atmosphere: interactions of soil physical factors and
 602 biological processes. *European Journal of Soil Science*, 54, 779-791.

603 Smuckler, S.M., Sánchez-Moreno, S., Fonte, S.J., Ferris, H., Klonsky, K., O'Geen, A.T., Scow,
 604 K.M., Steenwerth, K.L., & Jackson, L.E. (2010). Biodiversity and multiple ecosystem functions
 605 in an organic farmscape. *Agriculture, Ecosystems and Environment*, 139, 80-97.

606 Stevenson, B.A., Sparling, G.P., Schipper, L.A., Degens, B.P., & Duncan, L.C. (2004). Pasture
607 and forest soil microbial communities show distinct patterns in their catabolic respiration
608 responses at a landscape scale. *Soil Biology and Biochemistry*, 36, 49-55.

609 Suseela, V., Conant, R.T., Wallenstein, M.D., & Dukes, J.S. (2012). Effects of soil moisture on
610 the temperature sensitivity of heterotrophic respiration vary seasonally in an old-field climate
611 change experiment. *Global Change Biology*, 18, 336-48.

612 Thiel, B., Smukler, S.M., Krzic, M., Gergel, S., & Terpsma, C. (2015). Using hedgerow
613 biodiversity to enhance the carbon storage of farmland in the Fraser River delta of British
614 Columbia. *Journal of Soil and Water Conservation*, 70, 247-256.

615 Thiel, B., Krzic, M., Gergel, S., Terpsma, C., Black, A., Jassal, R., & Smukler, S.M. (2017). Soil
616 CO₂, CH₄ and N₂O emissions from production fields with planted and remnant hedgerows in
617 the Fraser river delta of British Columbia. *Agroforestry Systems*, 91, 1139-1156.

618 Venables, W.N., & Ripley, B.D. (2002). *Modern applied statistics with S*. Fourth Edition.
619 Springer, New York. ISBN 0-387-95457-0.

620 Walter, C., Merot, P., Layer, B., & Dutin, G. (2003). The effect of hedgerows on soil organic
621 carbon storage in hillslopes. *Soil Use and Management*, 19, 201-207.

622 Wang, X., Piao, S., Ciais, P., Janssens, I.A., Reichstein, M., Peng, S., & Wang, T. (2010). Are
623 ecological gradients in seasonal Q₁₀ of soil respiration explained by climate or by vegetation
624 seasonality. *Soil Biology and Biochemistry*, 42, 1728-1734.

625 Wiesmeier, M., Hübner, R., Barthold, F., Spölein, P., Geuß, U., Hangen, E., Reischl, A., Schilling,
626 B., von Lützw, M., & Kögel-Knabner, I. (2013). Amount, distribution and driving factors of soil

627 organic carbon and nitrogen in cropland and grassland soils of southeast Germany (Bavaria).
628 *Agriculture, Ecosystems and Environment*, 176, 39-52.

629 Wolton, R., Pollard, K., Goodwin, A., & Norton, L. (2014). *Regulatory services delivered by*
630 *hedges: The evidence base*. Report of Defra project LM0106. 99pp.

631 Zhou, T., Shi, P., Hui, D., & Luo, Y. (2009). Global pattern of temperature sensitivity of soil
632 heterotrophic respiration (Q_{10}) and its implications for carbon-climate feedback. *Journal of*
633 *Geophysical Research*, 114, G02016, doi:10.1029/2008JG000850.

634

Tables

Table 1. Characteristics of the two contrasting soil types in this study.

Soil type ¹	Seasonally-wet (SW)	Free-draining (FD)
Soil classification (UK) ²	Stagnogley	Brown earth
Soil classification (Worldwide) ³	Stagnosol	Cambisol
Soil texture ⁴	Silty-clay	Silty-clay loam
Sand / silt / clay (%) ³	0-20 / 40-60 / 40-60	0-20 / 40-73 / 27-40
pH ⁴	5.7 ± 0.1	5.5 ± 0.1
Bulk density (g cm ⁻³) ⁴	0.64 ± 0.04	0.89 ± 0.04

¹As referred to in this paper

²UK Soilsclapes soil map (<http://www.landis.org.uk/soilsclapes/>)

³World reference base for soils (WRB; <http://www.fao.org/soils-portal/soil-survey/soil-classification/world-reference-base/en/>; <http://www.fao.org/3/i3794en/i3794en.pdf>)

⁴Field measurements <10 m from study sites (Ford et al., 2019)

Table 2. Characteristics of the three field boundary categories used in this study, for exact location see Fig. S1.

Boundary	SW Hedgerow	FD Hedgerow	FD Stone wall
<i>Site characteristics</i>			
Location	SW site	FD site	FD site
Soil type	Stagnogley	Brown earth	Brown earth
Drainage	Seasonally-wet, impeded	Free-draining	Free-draining
Slope	Shallow (~5°)	Steep (~10°)	Steep (~10°)
Grazing	Sheep all year, cattle (May-June)	Sheep all year, cattle (March-November)	Sheep all year, cattle (March-November)
Pasture	Semi-improved grass with patches of <i>Juncus</i> spp.	Semi-improved	Semi-improved
Silage cut ¹	Yes (but not during study period due to drought)	No	No
<i>Boundary characteristics</i>			
Hedgerow composition	<i>Prunus spinosa</i> (60%), <i>Corylus avellana</i> (40%)	<i>Crataegus monogyna</i> (70%), <i>P. spinosa</i> (15%) <i>C. avellana</i> (15%)	<i>na</i>
Hedgerow understory	<i>Urtica dioica</i> , <i>Galium aparine</i>	<i>U. dioica</i> , <i>Cirsium vulgare</i> , <i>Ranunculus repens</i>	<i>na</i>
Management	Biennially cut, H ~2 m	Biennially cut, H ~2 m	<i>na</i>
Age	40 years	10 years	In situ ~100 years
Size	W = 2 m, H = 2 m	W = 1 m, H = 2 m	W = 0.6 m, H = 1.2 m
Fence	Double ² , 2 m wide	Double ² , 2m wide	<i>na</i>

SW = seasonally-wet soil, FD = free-draining soil

W = width, H = height, *na* = non-applicable

¹Silage cut refers to annual management where semi-improved pasture is routinely cut and removed for use as silage (winter animal feed).

²Double fenced at 2 m wide refers to the total width of the livestock exclusion zone across both sides of the hedgerow (see Fig. 1).

Table 3. Apparent temperature sensitivity of soil respiration (soil CO₂ efflux) expressed as Q₁₀ values for both grazed (G) pasture and the un-grazed (U) zone adjacent to the hedgerow on both seasonally-wet (SW, stagnogley) and free-draining (FD, brown earth) soils. Two scenarios, drought period-included (12-month dataset) and drought period-excluded (10 months with May and June removed) are presented.

	Drought included		Drought excluded	
	Q ₁₀	R ²	Q ₁₀	R ²
SW pasture (G)	10.3	0.58	8.4	0.58
SW hedgerow (U)	7.4	0.42	5.7	0.58
FD pasture (G)	7.2	0.83	6.4	0.81
FD hedgerow (U)	5.3	0.51	5.0	0.65

Table 4. Best fit models of soil CO₂ efflux, temperature and moisture for soils adjacent to three contrasting field boundary categories, two fenced hedgerows (inside fence, un-grazed at 0.7 m from hedgerow; outside fence, livestock-grazed pasture at 2 m and 10 m) on seasonally-wet and free-draining soil respectively and one stone wall (livestock-grazed pasture at 0.7 m, 2 m and 10 m), using data from May to September when hedgerows are in full leaf. ANOVA outputs of step-wise regression models are presented with explanatory variable information. Models for the stone wall with soil temperature or soil moisture as response variables are not shown as grazing did not vary and there was no significant association with distance from boundary.

Boundary type	Response	Explanatory v1	Explanatory v2	Explanatory v3	F	Sig	R ²
All (n = 265)	SR	SM (30%)	G/U (40%)	ST (30%)	25.7	***	0.23
SW Hedgerow	SR	SM (64%)	G/U (36%)	-	9.72	***	0.18
FD Hedgerow	SR	SM (8%)	G/U (92%)	-	41.5	***	0.49
FD Stone wall	SR	ST (100%)	-	-	7.73	**	0.08
All (n = 265)	ST	G/U (100%)	-	n/a	59.8	***	0.18
SW Hedgerow	ST	D (100%)	-	n/a	14.73	***	0.15
FD Hedgerow	ST	G/U (100%)	-	n/a	20.7	***	0.19
All (n = 265)	SM	G/U (69%)	D (31%)	n/a	14.57	***	0.10
SW Hedgerow	SM	G/U (100%)	-	n/a	88.6	***	0.50
FD Hedgerow	SM	G/U (100%)	-	n/a	14.0	***	0.14

v = variable, F = F statistic, Sig = significance (**P < 0.01, ***P < 0.001), SW = seasonally-wet soil, FD = free-draining soil, SR = soil CO₂ efflux, ST = soil temperature, SM = soil moisture, G/U = grazed/un-grazed categorical variable, D = distance from boundary (0.7 m, 2 m or 10 m). (%) associated with v1, v2 and v3 values refers to % of model R² explained by each variable.

Figure legends

Fig. 1. Sampling schematic for biotic (hedgerow) and abiotic (stone wall) boundaries. SW = seasonally-wet soil, FD = free-draining soil. The upslope and downslope parts of each transect start from the centre of the hedgerow or directly adjacent to the edge of the stone wall. The area adjacent to hedgerows within the livestock-exclusion boundary fence is un-grazed.

Fig. 2. Monthly measurements of soil adjacent to a hedgerow on seasonally-wet soil for a) soil CO₂ efflux, b) soil temperature and c) soil moisture at three perpendicular distances from the hedgerow [0.7 m (un-grazed); 2 m (grazed); 10 m (grazed)]. The r^2 value of the proportion of variation explained is given for the best-fit mixed effects model, with explanatory variable(s) and interaction terms listed underneath. For panel b, letters (x, y, z) adjacent to lines denote significant differences between the three distance categories included in the legend (there were no significant differences with distance for panels a and c). *** = $P < 0.001$, Temperature = soil temperature, Moisture = soil moisture, Grazing = grazing occurrence (yes/no), Distance = perpendicular distance from hedgerow, M = month, x = interaction between variables. Monthly means for each distance are presented with error bars showing the standard error of the mean ($n = 3$). The grey shaded box indicates period of drought.

Fig. 3. Monthly measurements of soil adjacent to a hedgerow on free-draining soil for a) soil CO₂ efflux, b) soil temperature and c) soil moisture at three perpendicular distances from the hedgerow [0.7 m (un-grazed); 2 m (grazed); 10 m (grazed)]. The r^2 value of the proportion of variation explained is given for the best-fit mixed effects model, with explanatory variable(s) listed underneath. For panel b, letters (x, y, z) adjacent to lines denote significant differences between the three distance categories included in the legend (there were no significant differences with distance for panels a and c). *** = $P < 0.001$, Grazing = grazing occurrence

(yes/no), Distance = perpendicular distance from hedgerow, M = month. Monthly means for each distance are presented with error bars showing the standard error of the mean ($n = 3$). The grey shaded box indicates period of drought.

Fig. 4. Monthly measurements of soil adjacent to a stone wall on free-draining soil for a) soil CO₂ efflux, b) soil temperature and c) soil moisture at three perpendicular distances from the wall [0.7 m (grazed); 2 m (grazed); 10 m (grazed)]. The r^2 value of the proportion of variation explained is given for the best-fit mixed effects model, with explanatory variable(s) listed underneath. For panel b, letters (x, y, z) adjacent to lines denote significant differences between the three distance categories included in the legend (there were no significant differences with distance for panels a and c). ** = $P < 0.01$, *** = $P < 0.001$, Distance = perpendicular distance from wall. Grazing is not included as an explanatory variable as all distances are grazed. Monthly means for each distance are presented with error bars showing the standard error of the mean ($n = 3$). The grey shaded box indicates period of drought.

Fig. 5. Annual carbon (C) budget schematic illustrating the estimated effect of hedgerows and livestock-grazed pasture on the C balance of both seasonally-wet (stagnogley) and free-draining (brown earth) soils. Annual soil CO₂ efflux (SR) rates were calculated from monthly means (12 months inclusive) for the grazed (G) pasture and the un-grazed (U) zone adjacent to the hedgerow (protected by the livestock-exclusion fences) and shown in the drought period-included (✓) sections of the schematic. Soil CO₂ efflux for the drought period-excluded scenario (✗) was calculated from monthly means (July-April) but with field-measured soil CO₂ efflux rates for the May-June drought period removed and replaced with modelled values (using the drought-excluded Q_{10} relationship) to give a 12-month dataset. Proxies for above- and below-ground net primary productivity (ANPP & BNPP) were calculated from published

data from the Conwy catchment. All figures are expressed in $\text{t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$, with $\pm$ standard error of the mean ($n = 3$) in parentheses. The symbols + and - denote a source of CO_2 to the atmosphere and a sink (storage) of CO_2 in plant biomass or soil respectively, with the values and arrows in black boxes indicative of net (plant + soil) ecosystem exchange. Methane and nitrous oxide fluxes are not included in these values.

Fig. 6. Change in CO_2 flux estimate under projected increased hedgerow cover scenarios (based on a model 1-ha field) compared with a baseline of 100% pasture (0% hedgerow cover). Values for seasonally-wet (SW) *versus* free-draining (FD) soils under two drought period scenarios (included *versus* excluded) were extrapolated from a C balance calculated from published above- and below-ground net primary productivity (ANPP & BNPP) for the Conwy catchment and measured annual soil CO_2 efflux rates to determine net C source / sink values (see Fig. 5). Hedgerow cover of 1% is equivalent to 50 m ha^{-1} (double-fenced to exclude livestock) at 2 m width, reflecting typical current UK hedgerow density. Hedgerow cover of 4% = 200 m ha^{-1} (at 2 m width), 8% = 400 m ha^{-1} (at 2 m width). Means for each hedgerow cover scenario are presented with error bars showing the standard error of the mean.

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737 **Fig. 1.**

Fig. 2.

Fig. 3.

Fig. 4.

Comparative flux estimate (t CO ₂ ha ⁻¹ yr ⁻¹)				
↑ = C source ↓ = C sink	Seasonally-wet soil		Freely-draining soil	
	Drought ✓	Drought ✗	Drought ✓	Drought ✗
Pasture (G) ANPP - 17.3 ± 1.7 BNPP - 5.8 ± 0.9	SR + 23.0 ± 1.3 - 0.12 ± 1.3	SR + 18.2 ± 1.1 - 4.9 ± 1.2	SR + 33.9 ± 1.8 + 10.8 ± 1.5	SR + 29.9 ± 0.9 + 6.8 ± 1.2
Hedge (U) ANPP - 17.0 ± 0.4 BNPP - 4.4 ± 0.6	SR + 27.1 ± 1.9 + 5.8 ± 0.8	SR + 11.5 ± 0.4 - 9.9 ± 0.3	SR + 20.5 ± 6.0 - 0.9 ± 2.2	SR + 14.9 ± 1.5 - 6.5 ± 0.7

Fig. 5.

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749 **Fig. 6.**

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Appendix.

Supplementary appendix for 'Hedgerow effects on CO₂ emissions are regulated by soil type and season: implications for carbon flux dynamics in livestock-grazed pasture'.

Fig. S1. Location of the two study sites and three field boundary categories used in this comparative observational study. FD site = site characterised by free-draining soil (brown earth), SW site = site characterised by seasonally-wet soil with impeded drainage (stagnogleys). Hedgerow boundaries (lines of trees) can be identified fairly easily from aerial images with stone wall and fence boundaries more difficult to distinguish.

Google Earth Pro V 7.3.2. (20th June 2018). Ysbyty Ifan, UK. 50.026679°, -3.743259, Eye alt 4.11 km.
DigitalGlobe 2018. <http://www.earth.google.com> [18th December 2018].