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Identifying and understanding how critical landscapes for carbon sequestration respond to development for low carbon energy production: Insight to inform optimal land planning and management strategies

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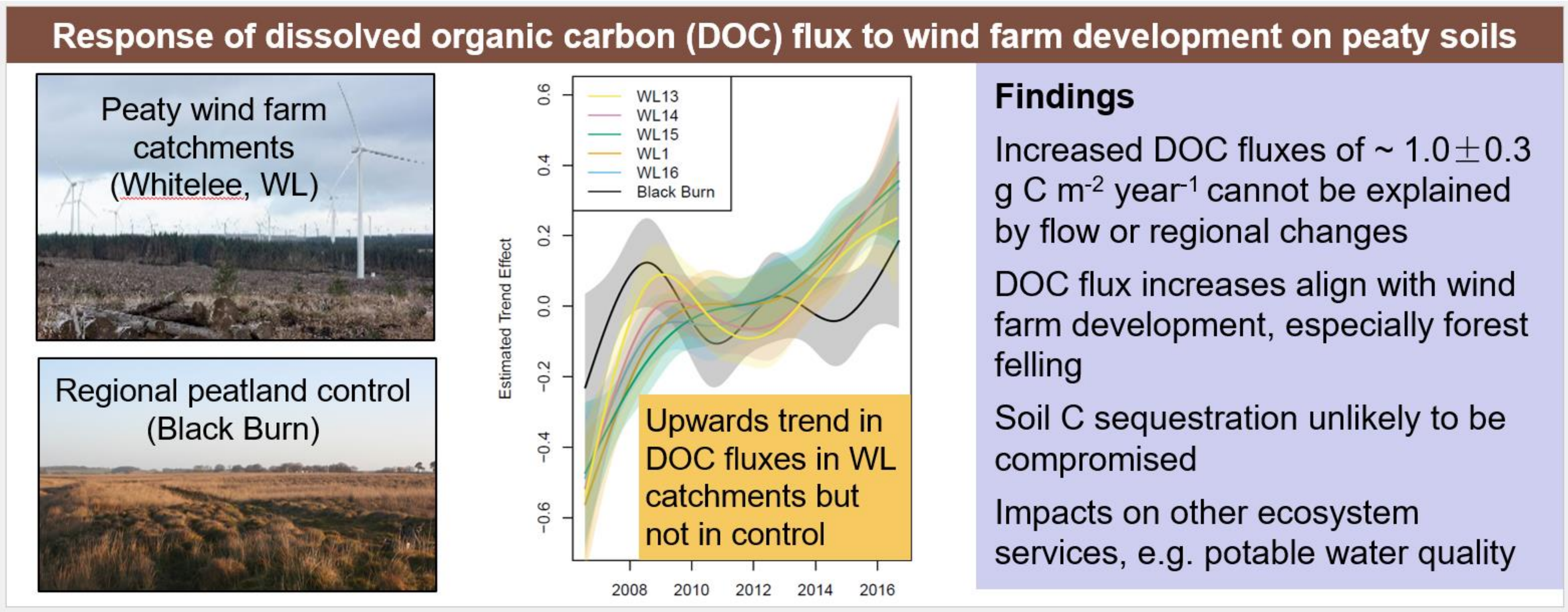
Abstract

Wind farms can mitigate increasing CO_2 emissions by fossil-fuel free energy generation. However, landscape disturbance during development must not have lasting impacts on C sequestration, an ecosystem service. To understand how a critical carbon landscape responds to wind farm development, we monitored for 10 years dissolved organic carbon (DOC) export in five catchments draining Europe's second largest onshore wind farm, Whitelee, UK. The DOC flux trend and seasonality were modelled using Generalised Additive and Mixed Models, novelly using first derivatives of trends to identify responses to wind farm development. Unlike a nearby, minimally-disturbed catchment, Whitelee catchment DOC fluxes increased over the decade, tracking successive phases of wind farm development, particularly forest felling to enable turbine location. Inter-catchment differences in the rate of DOC flux, where slowing suggests recovery (not always evident), reflect differing intensity (timing and spatial reach) of catchment disturbance. However, increased DOC flux approximated 3.5% maximum of C likely sequestered and therefore soil C sequestration is unlikely to be compromised unless the soils are highly degraded and close to not being a C sink. For the greenest energy transition, responsible planning should minimise C losses, even when small, and in a critical landscape may require consideration of impact on other ecosystem services. For example, deterioration in quality of potable water supply occurred within the observation period. Ongoing provisioning of multiple ecosystem services from critical carbon landscapes requires planning. We demonstrate an example of such an approach for wind farm development considering in priority order carbon storage, forest products, potable water supply. Our findings are relevant to integrated landscape planning and management in temperate and high latitude / altitude peatlands globally that are subject to wind energy development and/or forest felling

Keywords: green energy, wind farm, peat soils, ecosystem services, water abstraction, landscape planning

Graphical Abstract



Keywords: green energy, wind farm, peat soils, ecosystem services, water abstraction, landscape planning

1. INTRODUCTION

Reaching Net Zero and beyond to mitigate global warming requires decarbonisation of industrial activity and increased carbon sequestration by nature. For both these needs, natural landscapes are playing a critical role, and at growing scale. For example, global deployment of renewable energy has offered significant advances in decarbonising energy production, with growth in wind and solar supporting a record 30% of global electricity production in 2023 (Ember, 2024). Although only covering 2.84% of Earth's land surface, over 25% of the total terrestrial carbon (C) is stored in peat soils (Chico et al., 2023), and this could increase if degraded peatlands are restored. Managing peat landscapes effectively is therefore of increasing importance, particularly as blanket peatland is now on the European Red List of habitats due to declining quality (Chico et al., 2023) and ~12% of peatland globally is degraded (UNEP, 2022).

Peatland ecosystem services supporting a pathway beyond Net Zero do not always function independently in a landscape. Rather a new critical landscape has emerged that stores carbon, and now supports also energy generation: the deployment of wind farms in landscapes with soils rich in C. This is because peaty soils and blanket peatlands can be found at higher latitudes and at altitude and on hill summits where wind energy potential is greater. However, the deployment of wind farms can cause considerable land disturbance (e.g. Heal et al., 2020). Understanding the long-term impacts of wind farm development on peatlands is needed to ensure that land use for energy production is managed optimally (e.g. Sha et al., 2022) for net atmospheric CO_2 drawdown. Ideally recovery from disturbance would be quick such that C losses are short-term and soon the embedded C footprint of construction is offset by renewable energy generation and the landscape functions to sequester atmospheric C again. If not managed well, the green-energy transition risks being undermined.

One way of assessing the carbon impacts of wind farms on peatlands is to compare the carbon costs of wind farm developments with the carbon savings attributable to the wind farm to calculate the payback time: when the carbon savings of generating electricity by the wind farm development exceed its carbon costs. The Scottish Government commissioned and uses such a tool (Scottish Government, 2022a) in the wind farm development consent process. However, the tool assessment of nature-based C storage response relies on single component management processes in analogous landscapes (such as a response to deforestation, or a response to soils being drained) and on modelled responses (Nayak et al., 2010) – rather than being able to draw on wind farm development case studies of multiple contemporaneous impacts. As such, research is still needed that seeks to understand how carbon storage is affected by the wind farm construction process and uses direct observations at development sites.

A wind farm is a complex site, large in scale often with heterogenous land use. Immediate impact from significant landscape disturbance is to be expected, but measuring how land carbon sequestration is impacted is challenging. Direct measurement of net ecosystem exchange (NEE), although possible (e.g. Armstrong et al., 2015), would be unmanageable to scale up and continue for a decade. Boundary layer mixing by turbine blades violates theoretical eddy covariance systems assumptions and makes this approach challenging. However, DOC fluxes in aquatic drainage systems act as an integrative measure of landscape response. DOC export is a major C loss component in peatlands (Rosset et al 2022), accounting for ~25% of ecosystem carbon NEE (Dinsmore et al., 2010) although this % will be dependent on the NEE balance of the peatland (e.g. Tong et al 2024). As such changes in net C storage should be reflected in DOC export. There is considerable uncertainty over parameterising DOC losses in payback time calculations in the wind farm C assessment tool as research in this area is lacking. If DOC export is inaccurately quantified, then estimates of carbon sink and storage may be erroneous and for wind farms payback times may be substantially underestimated. In addition, increases in DOC concentration ([DOC]) in surface waters can impact on aquatic ecosystems and organisms (e.g. Laudon and Buffam, 2008; Strack et al., 2015) and, where water is abstracted for potable use, the costs of water treatment, and the carbon footprint therein, also increase (e.g. Kritzberg et al., 2020). Thus, the changes in peatland carbon storage that occur through different land management approaches are a serious concern and identifying when catchment management is required to be different to mitigate loss is valuable.

There can be short-term increases in aquatic carbon loss during wind farm development, including DOC and macronutrient flux, associated with forest-felling and borrow pits, identifiable through nested sampling and land-use analysis (Heal et al, 2020) and additionally from compositional differences in DOC that represent source organic matter composition (Zheng et al 2018). Soluble reactive phosphorus loss recovery can occur within 2 years (Heal et al., 2020). However, understanding of DOC export over a longer period and where there are multiple phases of wind farm development is lacking – e.g. to what extent can disturbance be buffered? Does the site fully recover? Is more active management to minimise C loss needed? Without this knowledge the need for voluntary and mandatory mitigation in land management practices cannot be identified.

To tackle such questions, for a decade we monitored DOC flux from five peaty catchments draining the 539 MW Whitelee Wind Farm, a large wind farm of 239 turbines and only within the last decade displaced from its role as Europe's largest onshore wind farm. DOC flux, the product of river [DOC] and flow, provides an integrated signal of internal and external drivers that affect DOC production and flux from peatland - such as landscape disturbance and regional changes in rainfall, respectively. A catchment response can identify a directional change over time, rather than just interannual variation that occurs as internal and external drivers interact. By comparing Whitelee catchment DOC fluxes with flux from the Black Burn, which drains a minimally-disturbed peatland 80 km to the east, we can assess if changing trends in DOC flux from Whitelee are attributable to wind farm development and not external, regional factors, e.g. changes in atmospheric sulfate deposition or climate (Monteith et al, 2007; Monteith et al, 2023). Additionally, this large volume of direct measurements from five adjacent wind farm-impacted catchments, sampled at high frequency over a decade allows us to model how the wind farm site responds to successive phases of development. Our modelling incorporates site, trend, season and interaction between trend and season, with flow as a covariate, to allow inter-catchment differences and similarities in DOC flux to be observed, test theoretical models of recovery, and to project how mitigation may reshape DOC flux responses (Fig. 1).

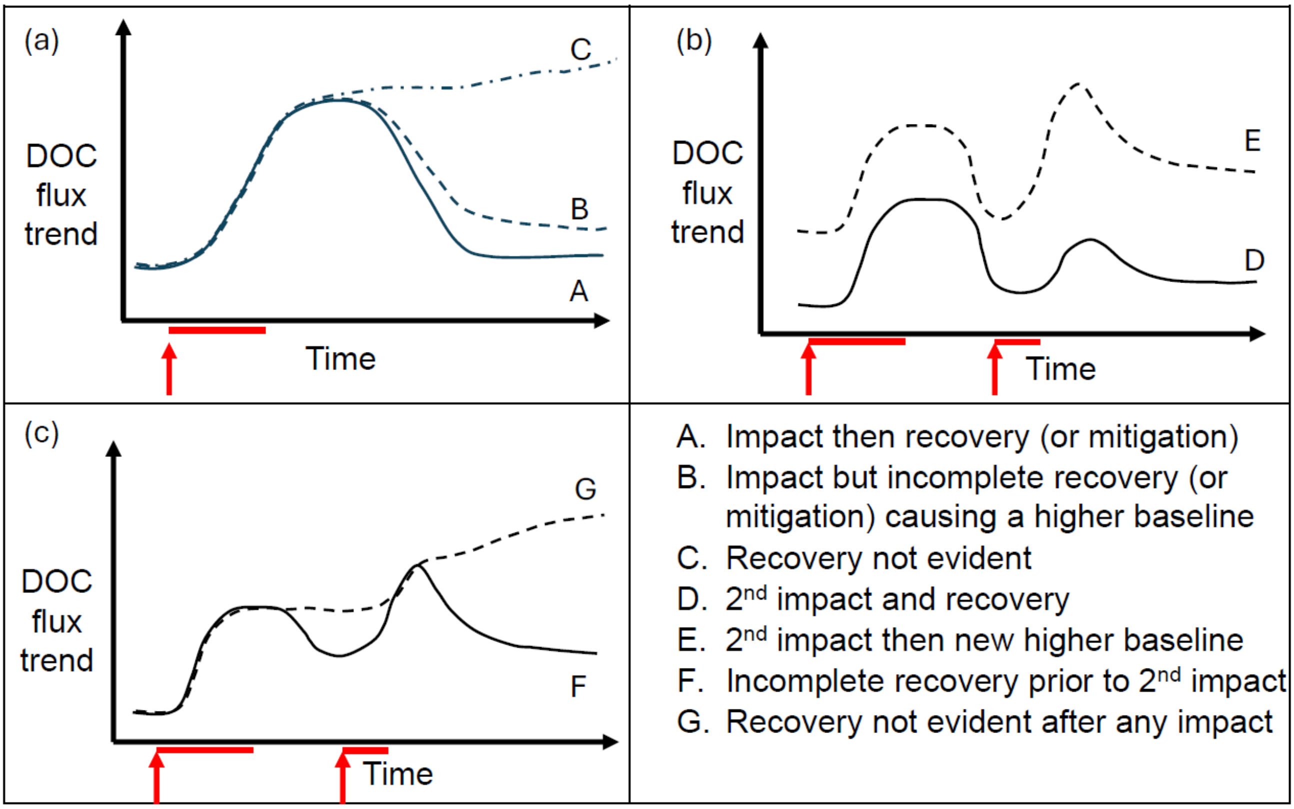


Figure 1. Conceptual responses of catchment DOC flux trend to different land management approaches, both disturbance and mitigation: (a) single impact with three example recovery/mitigation projections, A, B & C; (b) multiple impacts with complete recovery after the first impact and two example recovery/mitigation projections D & E; and (c) multiple impacts with incomplete recovery after the first impact and two example recovery/mitigation projections F & G. The red arrows and horizontal lines depict the hypothetical driver of change – representing the start and duration of change, respectively. Further response permutations are possible, but the conceptualisations above show key features. The sub-annual variation in DOC flux has not been included for simplicity.

We focus on flux because it captures hydrological variability to give an integrated catchment response and we can interpret catchment export trends using our understanding of how land use management affects DOC fluxes. For example:

- In response to development (an impact) there may be a full recovery (A), incomplete recovery (B), or no signal of recovery (C) (Fig. 1a).
- If wind farm construction has taken place over several phases, multiple impacts may be observed followed by full (D), incomplete (E) or no recovery (G), and full recovery may not take place between impacts (F) (Fig. 1b, c).
- If there is a strong regional control on DOC flux then temporally similar responses between Whitelee catchments and Black Burn would be observed (both DOC flux trendlines would be similar).
- If the wind farm development is resulting in increased DOC flux, different responses between Whitelee catchments and Black Burn would be observed (for example G vs. A respectively).
- If there are localised controls on DOC flux in response to different timings and spatial extent of wind farm development activities, differences in the timing of DOC flux trends between Whitelee catchments would be observed.
- If recovery from disturbance is occurring, a decrease in DOC flux would be expected - unless countered by a second phase of development, or external regional controls.

We use the resultant understanding to discuss how significant a change in C flux may be to landscape C sequestration and offer an approach to decision making in ‘critical landscapes’ that accommodates different ecosystem services the landscape must support.

This is the first study to monitor at scale, across multiple catchments, and for such a long period how DOC flux changes from a landscape undergoing development to host a wind farm. This research uses direct observations at development sites to compare how carbon storage is affected by the wind farm construction process, and the findings contribute to the evidence base that can inform how land use for energy production is managed optimally for net atmospheric CO_2 drawdown.

MATERIALS AND METHODS

An overview of the data collection and analysis methods is contained in Figure 2, with more details of the methods given below and in the Supplementary Material.

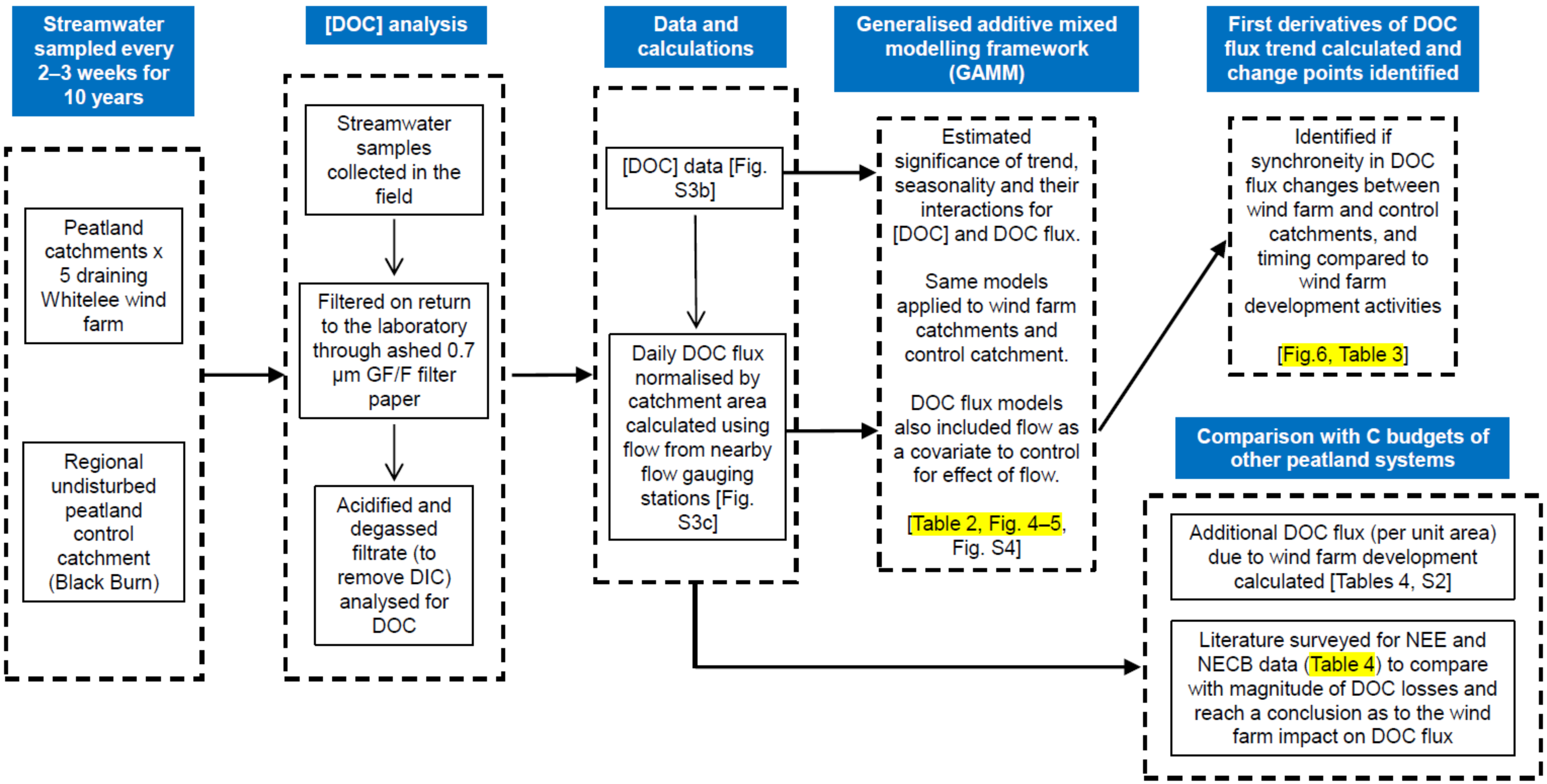


Figure 2. Overview of the research methodology. Blue-shaded boxes indicate the major methods steps. Dashed-line boxes show more detail within each step. Yellow-shaded text indicates where the key quantitative findings are located.

2.1 Study catchment descriptions

2.1.1 Peatland catchments affected by wind farm development

These five study catchments (acronyms WL13, WL14, WL15, WL16, WL1) all drain Whitelee wind farm, located on Eaglesham Moor (55° 40' N, 04° 16' W), 16 km south of Glasgow, central Scotland (Fig. 3). Development of the wind farm started in 2006 and occurred in two main phases: 140 turbines by 2009 (here called “phase 1”) and 75 turbines constructed from November 2010 to February 2013 (here called “phase 2”). At the time of construction, Whitelee wind farm was the largest onshore wind farm in the UK and the second largest in Europe by generating capacity (539 MW) (EWEA, 2013). The Calder Water Community (39 MW, 13 turbines) and West Browncastle (36 MW, 12 turbines) wind farms, immediately adjacent to the south of Whitelee wind farm, were constructed in the WL16 catchment between August 2012 and August 2014.

The wind farms are located on a plateau with a maximum elevation of 376 m. For the period 1991–2020, mean annual rainfall was 1413 mm and seasonal air temperatures ranged from an average of 13.4 ± 1.0 °C in summer (June–August) to 3.2 ± 0.1 °C in winter (December–February) (± values denote standard deviation, Saughall climate station, 3 km from Whitelee wind farm; UK Met Office, n.d.). The underlying geology comprises Carboniferous porphyritic basalts and members of the calciferous sandstone series, mostly overlain with ~3 m of glacial and recent drift deposits and peat, predominantly as blanket bog. The mean peat depth, measured at 161 locations in phase 1, was 1.90 m (SD ± 1.35 m) (CRE Energy, 2002). Assessment of peat maturity at test points across the site, found variations in the degree of decomposition across the site related to peat depth and local hydrology, but in general upper layers recorded a von Post class (1922) of approximately H3, slightly decomposed, in the shallower deposits typically increasing to between H6 and H8, strongly decomposed, at depth (CRE Energy 2005). The Amlaird Water Treatment Works (WTW) operated by Scottish Water provides potable water supplies sourced from Lochgoin and Craigendunton reservoirs located in the WL14 catchment.

The main land use has been Sitka spruce-dominated (*Picea sitchensis*) forestry plantation established in the 1960s–70s. During wind farm construction, large forest tracts were felled to clear land for building turbine foundations (~3000 m^2 surface area disturbance per turbine, Waldron et al., 2009), and to reduce surface roughness that decreases power outputs. Other significant wind farm development activities included: borrow pit excavation followed by restoration; turbine foundation excavation; adjacent hardstanding for turbine maintenance; substations; tracks for access and to carry cabling. Existing forestry tracks were upgraded and new tracks constructed using stone from borrow pits or as floating tracks over peat > 1 m depth and gradients of < 1:10. To mitigate the effects of disturbance, tracks were routed to avoid sensitive areas, and silt fences and settling ponds used to manage suspended solids in runoff. A detailed map of soil types and wind farm turbine locations in the Whitelee catchments is contained in Figure S1.

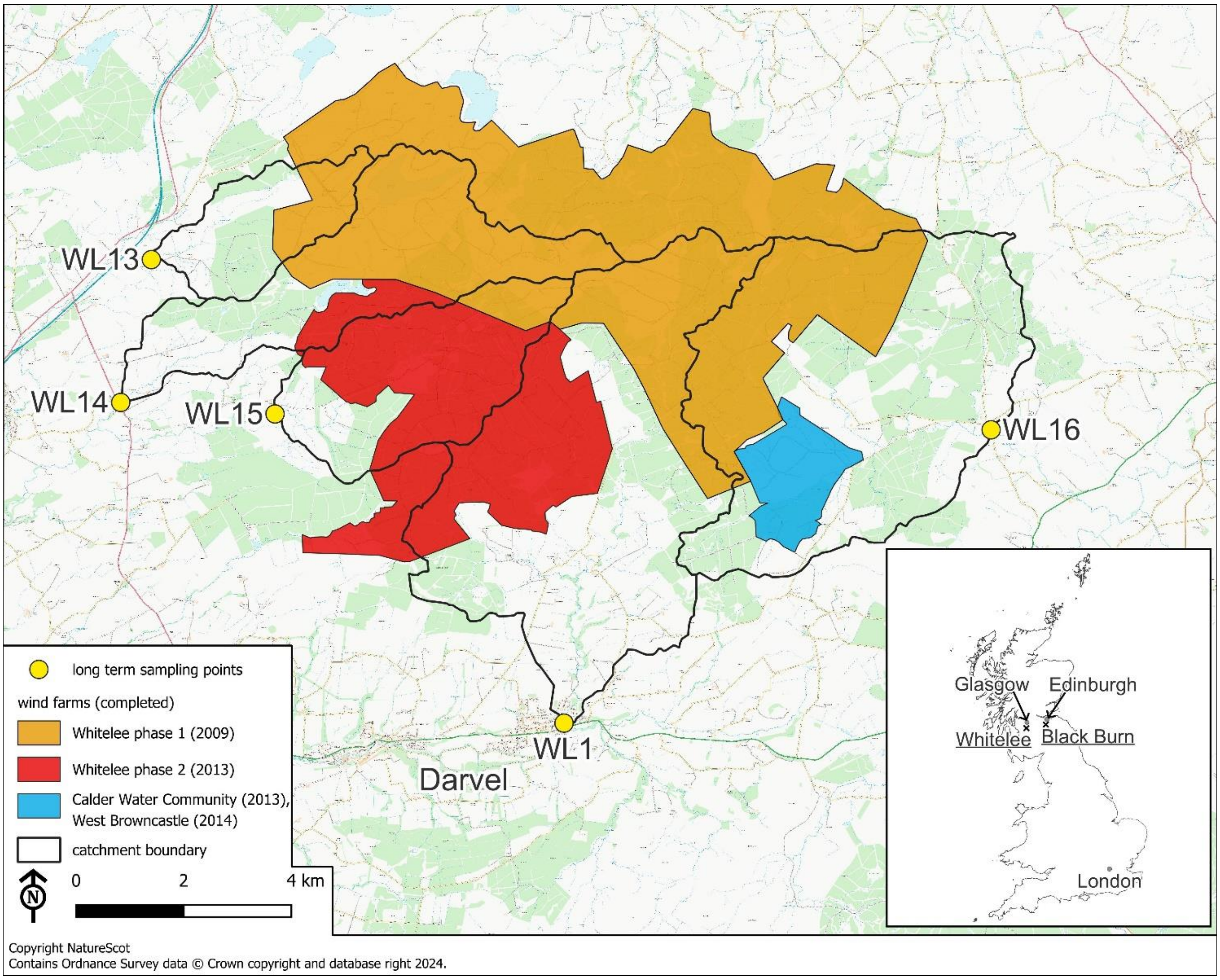


Figure 3. Locations of the Whitelee and Black Burn (control) study catchments in Great Britain (crosses in inset map), Whitelee catchment locations and long-term streamwater sampling points (yellow circles), and locations and timing of wind farm development. Catchment boundaries shown as black lines.

2.1.2 Regional control peatland catchment

Identifying control catchments that were not degraded was challenging since only ~10% of the blanket bog area in the UK of known condition is assessed to be in "good condition" (JNCC, 2019). Additionally, DOC fluxes must be monitored in this catchment contemporaneously with Whitelee. These restrictions limited selection of the regional control catchment to the Black Burn draining Auchencorth Moss (55° 47′ N, 03° 14′ W), 80 km east of Whitelee wind farm (Fig. 3), as the nearest minimally-disturbed peatland catchment with similar physical and climatic characteristics to the Whitelee catchments and critically a long-term flow and [DOC] monitoring programme covering the study period.

A full catchment description is given in Dinsmore et al. (2010). Briefly, Auchencorth Moss is a 3.35 km^2, low-lying (249–300 m), ombotrophic peatland. During 2002–2013, mean annual precipitation was 1018 ± 166 mm and seasonal air temperatures ranged from an average of 13.6 ± 1.1 °C in summer (June–August) to 3.7 ± 1.0 °C in winter (December–February) (± values denote standard deviation; Helfter et al., 2015). Peats, which range in depth from < 0.5 m to > 5 m, cover 85% of the area and are underlain by glacial till and an Upper Carboniferous/Lower Devonian sequence of sandstones and shaly sandstones with minor limestone, mudstone, coal and clay layers. The peat has an organic C content of 30–40% and comprises several organic horizons, some containing poorly decomposed plant fragments and others that are highly humified (Leith et al., 2014). The land use is primarily low-intensity sheep grazing with a small area of peat extraction at the catchment boundary. Vegetation consists of a patchy mix of grass (e.g. *Deschampsia flexuosa*) and sedges (e.g. *Eriophorum vaginatum* and *Juncus effusus*) covering a primarily Sphagnum base layer on a hummock/hollow microtopography; shrubs such as *Calluna vulgaris*, *Erica tetralix* and *Vaccinium myrtillus* are also present. The catchment drains through natural tributaries and overgrown (> 100 years old) drainage ditches into the main stream channel, the Black Burn. In March 2015, 39 drains in the upper catchment were blocked (Dinsmore et al., 2016).

2.2 Streamwater sampling and analysis

Full methods for water sample collection and analysis and quality control procedures are in Supplementary Material S1. Long-term monitoring of the Whitelee catchment macronutrient concentrations commenced in July 2006 (Waldron et al., 2009) and continued until September 2016. Streamwater was sampled every 2–4 weeks at the outlet of the five catchments (Table 1), draining the west and south of the wind farm and with few confounding land-uses. This frequency sampled a full range of river flows (Fig. S2). Streamwater samples were analysed for [DOC], with the first measurement on 3 July 2006 and the final one on 19 September 2016, comprising 167 sampling dates. [DOC] values are missing on 2 dates for WL13 and WL1, 3 dates for WL14 and 9 dates for WL16 as streamwater was not collected from all catchments on all sampling dates. [DOC] was quantified on acidified, degassed streamwater filtrate (Whatman® GF/F 0.7 μm filter size, as in many other studies, e.g. Dyson et al., 2011) using a Thermalox™ total carbon analyser.

At the Black Burn, streamwater was sampled typically every 1–2 weeks, comprising 345 sampling dates. The median sampling frequency was every 8 days, although there were some larger sampling gaps of 40–92 days due to personnel changes, e.g. 92 days between 29 June and 29 September 2016. Streamwater samples were prepared in the same manner as the Whitelee samples (see Dinsmore et al., 2010 for a method description) and [DOC] was quantified from the filtrate using a Rosemount-Dohrmann DC-80 total organic C analyser (2006–2007) or a PPM LABTOC Analyser (2008 onwards).

2.3 Flow and DOC flux estimates

2.3.1 Whitelee catchments

Flow monitoring was conducted directly in all catchments over shorter time periods (as reported in Murray, 2012; Phin, 2016; and Coleman, 2017), apart from WL1. Since flow monitoring was not continuous throughout the study period from June 2006 to September 2016, long-term flows in the Whitelee catchments were estimated from 15-minute flow data for the nearby Scottish Environment Protection Agency (SEPA) gauging station at Newmilns on the River Irvine (61 masl, catchment area 72.8 km^2). The suitability of using relationships between catchment measured flow and the SEPA Newmilns flow for estimating flows vs. the catchment area scaling method was evaluated to select the most appropriate method to generate 15-minute flow estimates for each catchment (Table S1). From 15-minute flow data, mean daily flow values were calculated for each catchment for 1 June 2006–29 September 2016 using the definition of a "water-day" of the UK National River Flow Archive (NRFA, n.d.; the water-day is the 24 hours from 09.00 GMT). This generated 3774 mean daily flow values for each catchment which input to modelling and were used to calculate daily DOC flux.

Mass fluxes of DOC normalised for catchment area were calculated for each Whitelee catchment using mean daily flow and the 2-4-weekly [DOC] data. Relationships between [DOC] and mean daily flow were constructed for each catchment, but none were significant so an interpolation method was used for DOC flux calculations, assuming that [DOC] was constant between consecutive sampling occasions. Daily DOC flux normalised by catchment area (*DOC flux$_d$*, kg C km^{-2} day^{-1}) from each study catchment from 3 July 2006 to 19 September 2016 inclusive was calculated as:

$$DOC\ flux_d = \frac{[DOC] \times \bar{Q}_d \times CF}{CA} \qquad \text{(Eq. 1)}$$

where [DOC] is measured DOC concentration (mg C L^{-1}) at the start or end of the time period, $\bar{Q}_d$ is the mean daily flow (m^3 s^{-1}), *CF* is a correction factor for units and *CA* is catchment area (km^2).

Since [DOC] was measured every 2–4 weeks, the same [DOC] value was used to calculate daily DOC fluxes across the entire time period between measurements. Daily DOC fluxes were calculated both "forwards" and "backwards" using [DOC] at the start or end of the [DOC] measurement period, respectively, to understand uncertainties in DOC fluxes. For example, for [DOC] measurements for 4 May and 24 May 2007, "forwards" daily DOC fluxes between 4 and 23 May inclusive were calculated using the [DOC] for 4 May, whilst "backwards" daily DOC fluxes between 5 and 24 May were calculated using the [DOC] for 24 May. Mean daily DOC fluxes were calculated as the mean of the "forwards" and "backwards" DOC flux for each date 3 July 2006–19 September 2016 inclusive and were used for statistical modelling. Standard deviations of the mean daily DOC fluxes were calculated using the “forwards” and “backwards” DOC flux calculated for each day (n = 2), apart from the dates on which [DOC] was measured. Mean standard deviations (mean of the standard deviations for all days when both forwards and backwards DOC flux were calculated, n = 3236–3565 values per catchment) ranged from 8.5 to 18.6 kg C km^{-2} day^{-1} and median standard deviations from 5.0 to 9.7 kg C km^{-2} day^{-1} for the study catchments.

2.3.2 Black Burn catchment

Since flow measurements for the Black Burn were not continuous for the study time period, mean daily flows were estimated from mean daily flows at the nearest SEPA gauging station, Dalmore Weir on the North Esk (132.1 m asl, catchment area 81.6 km^2). Where estimated mean daily flow values in the Black Burn were 0 m^3 s^{-1} (1199 out of 3776 values), which occurred during low flow periods, they were substituted with 0.0001 m^3 s^{-1}, reflecting the lowest positive mean daily flow estimates. It is likely that the flow estimation approach used is underestimating low flows in the Black Burn since the burn is rarely totally dry. Daily DOC fluxes and their uncertainty were calculated in the same way as for the Whitelee catchments. Mean standard deviation (mean of the standard deviations for all days when both forwards and backwards DOC flux were calculated, n = 3429 values) was 18.1 kg C km^{-2} day^{-1} and median standard deviation was 2.5 kg C km^{-2} day^{-1}. This uncertainty expressed as % relative standard deviation (14%) is similar to the total uncertainty of ~15% for solute fluxes calculated using interpolation in the Hubbard Brook USA experimental upland catchments (Pu et al., 2023).

2.4 Statistical modelling of temporal variability of [DOC] and DOC flux

We chose flexible regression models to compare trends and seasonality of [DOC] and DOC flux between the disturbed and undisturbed catchments. These powerful models suited this research as they relax the assumption of simple linear dependencies between response and covariate. A flexible regression makes no assumptions about the shape of a relationships other than it should be smooth, so is well suited to field responses that change naturally with time (e.g. DOC seasonality, Fig. S3b) or may reflect an imposed environmental change (see Fig. 1). As such the modelling framework involves smoothing but does not require making any specific distributional assumptions about the modelled variable such as normality. The regression uses Analysis of Covariance (ANCOVA) methods within the context of Generalised Additive Models (GAMs) and Generalised Additive Mixed Models (GAMMs) to describe and test relationships across the different sites.

The choice of the smoothing level of the modelled relationships is a trade-off between the bias and the variance, typically assessed using cross-validation (Wood, 2006). In the presence of autocorrelation in the model residuals, a typical characteristic of time series data, the GAM framework was extended to a mixed model (GAMM). Flow was included as an explanatory variable in the DOC flux models so that any differences in modelled trend and seasonality of DOC flux between the study catchments cannot be attributed to flow. Change point analysis using first derivatives was applied to the DOC flux estimated trend to independently identify whether change points existed, and if their timing could be associated with wind farm development activities. The research design and analysis controlled for any changes in climate, atmospheric deposition and flow. Thus, differences in DOC flux responses and their timing in the catchments can only be attributed to landscape disturbance and management.

Full details of the modelling steps and the equations used are in Supplementary Material S2. Briefly, we used additive modelling (Friedman and Stuetzle, 1981) to estimate the trend and seasonality of [DOC] and DOC flux in the Whitelee and Black Burn catchments as smooth functions of time and day of year, respectively. To adjust for autocorrelation in the time series, the errors within each catchment were assumed to follow an Auto-Regressive Moving Average (ARMA) process (Box et al., 1994). Models were fitted using the function gamm for generalised additive mixed models in the mgcv package in R 3.6.3 (Wood, 2012). The degrees of freedom for the trend seasonal components were held the same across the different catchments to ensure valid comparisons. To account for differences in mean [DOC] / DOC flux between catchments, site was added as a parametric term to the model. After validating the model assumptions about the errors, the significance of each term in the model was assessed using the Wald test (Wood, 2006). A term with a *p*-value greater than 0.05 is statistically insignificant and can be removed from the model.

3. RESULTS AND DISCUSSION

3.1 Summary results of [DOC], flow and DOC fluxes

Table 1 shows summary statistics for the measured DOC concentration and estimated mean daily flow, and daily and annual DOC fluxes for each study catchment. The similar magnitudes of DOC concentrations and fluxes in the Whitelee catchments and the minimally-disturbed Black Burn catchment support the selection of the Black Burn as an appropriate control catchment. Mean catchment [DOC] was 21–31 mg C L^{-1}, and highest in WL13, WL15 and Black Burn, the catchments with the highest % of peat-related soils and shortest river flow length to the sampling location (Table 1). Maximum [DOC] was very high in all catchments, exceeding 64 mg C L^{-1} in WL13, WL14 and WL15, and 85 mg C L^{-1} in Black Burn on three occasions in July–August 2007. The elevated [DOC] in the Black Burn in 2007 was not apparent in the Whitelee catchments but was observed in some UK Upland Waters Monitoring Network (UWMN) sites (Monteith et al., 2014). It is attributed to the rewetting of hydrophobic peat after drought, releasing an accumulated soil DOC store produced by enhanced decomposition during drought conditions (e.g. Worrall and Burt, 2004), in this case the prolonged 2004–2006 drought that particularly affected the eastern and southern UK (Marsh et al., 2014). The lowest [DOC] maxima (50–60 mg C L^{-1}) occurred in WL1 and WL16, which have the longest river flow length to the sampling location. Mean and maximum daily flow values (expressed as mm day^{-1} to take account of the different catchment areas) were similar in all catchments, apart from at WL14 where they were lower, attributed to the flow buffering effect of the lochs (lakes) and reservoirs covering ~5% of the catchment area.

Table 1. Key catchment characteristics and summary of catchment flows, DOC concentrations and DOC fluxes, June 2006–September 2016. For mean daily flow n = 3774 and flow values are normalised by catchment area. n for DOC concentrations and daily fluxes varies as streamwater was not sampled from all catchments on all sampling dates. Annual DOC fluxes calculated each year for the 10-year period 4 July 2006–3 July 2016. Whitelee information from Murray (2012) and Zheng (2018) and Black Burn from Dinsmore et al. (2010) and Pickard et al. (2017).

Catchment characteristic	**Catchment name**					
	WL13	**WL14**	**WL15**	**WL1**	**WL16**	**Black Burn**
Sampling point grid reference (elevation masl)	NS250646 (238)	NS249644 (167)	NS252643 (165)	NS257638 (104)	NS265643 (206)	NT213567 (254)
Area (km^2)	5.7	15.1	11.3	30.5	31.1	3.35
Upstream flow length (km)	5.3	8.1	6.0	11.4	10.8	4.7
Peat-related soils (%)	88	78.6	90	62	82	85
Forest cover pre wind farm (%)	42.6	31.7	77.3	50.2	58.4	---
Deforested area (%)	30.3	10.3	33.6	10.7	13.2	---
Turbine number (density, km^{-2})	8 (1.4)	38 (2.5)	38 (3.4)	54 (1.7)	52 (1.7)	0
Mean daily flow, mean ± StDev (min–max) (mm day^{-1})	3.43 ± 3.93 (0.629–46.9)	2.13 ± 2.01 (0.703–24.3)	3.1 ± 4.13 (0.150–48.8)	3.16 ± 4.21 (0.152–49.7)	3.04 ± 4.05 (0.147–47.8)	3.13 ± 5.36 (<0.001–57.5)
[DOC] mean ± StDev (min–max) (mg C L^{-1})	28.1 ± 12.5 (4.7–64.7)	22.8 ± 10.0 (6.7–66.1)	30.7 ± 13.0 (6.6–66.8)	21.1 ± 9.90 (5.2–50.4)	23.6 ± 10.5 (6.5–58.8)	28.2 ± 12.9 (4.3–87.5)
[DOC], n	165	164	167	165	158	345
Daily DOC flux, mean ± StDev (min–max) (kg C km^{-2} day^{-1})	92.4 ± 112 (10.0–1473)	47.9 ± 48.0 (7.10–554)	93.2 ± 133 (4.83–1821)	64.6 ± 93.9 (2.50–1371)	72.6 ± 105 (3.03–1472)	89.5 ± 168 (0.007–1945)
Daily DOC flux, n	3730	3729	3732	3730	3643	3774
Annual DOC flux mean ± StDev (min–max) (g C m^{-2} $year^{-1}$)	33.4 ± 5.3 (22.1–42.2)	17.2 ± 3.2 (12.0–24.7)	33.7 ±7.4 (22.4–49.0)	23.3 ± 5.1 (14.6–33.4)	26.3 ± 5.7 (18.2–47.6)	33.7 ± 7.5 (23.9–48.1)

As expected in headwater catchments dominated by peat soils, flow and [DOC] were highly variable over time in each catchment, varying by two and three orders of magnitude, respectively (Fig. S3a, S3b). Daily DOC flux rates were also highly variable (Fig. S3c) and had a ‘spiky’ pattern, similar to mean daily river flow. However, the highest flux rates occurred when seasonally elevated [DOC] in summer/autumn coincided with high river flow. For example, in WL15, the highest estimated mean daily flow in the entire study period in December 2014 was not associated with the highest daily DOC flux of 1821 kg C km^{-2} day^{-1}, which occurred in August 2011. Similarly, whilst a series of high flows occurred in the Whitelee catchments between November 2015 and February 2016, the disproportionate decrease in DOC flux rates during this period is attributed to flushing of DOC stores from catchment soils.

Daily and annual DOC flux rates were highest in WL13, WL15 and Black Burn, with mean values following the same catchment order as for mean [DOC] (WL15 ≈ Black Burn ≈ WL13 > WL16 > WL1 > WL14) (Table 1). Annual DOC flux rates from the Black Burn during the study period were initially amongst the highest of the study catchments but were increasingly exceeded by those from the Whitelee catchments. In the Whitelee catchments, until 2010, the highest annual DOC flux rates occurred in WL13, but from 2011 onwards WL15 had the highest annual DOC flux rate, most simply interpreted due to these being the sampling points closest to phase 1 and 2 of disturbance respectively.

Scaling up mean annual DOC flux rates for catchment area gives annual DOC fluxes ranging from ~110 t C at Black Burn to ~800 t C at WL16. Mean annual DOC flux rates at ~20–30 g C m^{-2} $year^{-1}$ mostly exceeded the largest annual DOC yields of 20.2 and 14.4 g C m^{-2} $year^{-1}$ from the regionally close Ayr and Cree catchments (also in south-west Scotland) in a 2017 survey of large rivers in Great Britain (Williamson et al., 2023a). The flux rates from our study catchments were similar to the average value of 24.4 ± 21.9 g C m^{-2} $year^{-1}$ of measured DOC flux reported for 62 peaty catchments globally, which ranged from cold to tropical climates and included disturbed sites (Rosset et al., 2022). The mean DOC value of 24.2 g C m^{-2} $year^{-1}$ for the 17 temperate sites (Rosset et al., 2022) lies within the range of average annual DOC flux rates for the study catchments.

3.2 DOC concentration modelling results

The ANCOVA-GAMMs accounted for most of the variability in [DOC] in the disturbed catchments but not in the minimally-disturbed Black Burn catchment (R^2 = 85.3% and 35.6%, respectively). Table 2 shows the significance of the terms in the models for all catchments to assess differences in trends and seasonality.

Table 2. Significance of smoothed terms, including interactions, for all catchments in the ANCOVA models fitted to the [DOC] data (table centre) and the DOC flux data (table right). (Model = Site + trend + season + interaction between trend and season + MA(2). The DOC flux model also includes flow). The effective degrees of freedom (edf) for each term had near identical values for all catchments, as the same level of smoothing has been applied to all sites, allowing valid comparisons between catchments. *WL13 is the reference site for the Whitelee catchments. For the other Whitelee catchments the mean effect is relative to that for WL13. For example, the mean [DOC] at WL14 is 5.40 mg C L^{-1} lower than at WL13.

Catchment	**[DOC] modelling**			**DOC flux modelling**		
	Estimate	**edf**	***p*-value**	**Estimate**	**edf**	***p*-value**
*WL13						
Site mean effect	27.99		< 0.0001	4.104		< 0.0001
Trend effect		5	< 0.0001		5	< 0.0001
Season effect		3.8	< 0.0001		4	< 0.0001
Flow effect		---	---		1	< 0.0001
Trend x season interaction		12	0.788		12	0.455
WL14						
Site relative mean effect	-5.40		< 0.0001	-0.193		< 0.0001
Trend effect		5	< 0.0001		5	< 0.0001
Season effect		3.6	< 0.0001		4	< 0.0001
Flow effect		---	---		1	< 0.0001
Trend x season interaction		12	0.390		12	0.027
WL15						
Site relative mean effect	2.52		0.00014	0.106		< 0.0001
Trend effect		5	< 0.0001		5	< 0.0001
Season effect		3.8	< 0.0001		4	< 0.0001
Flow effect		---	---		1	< 0.0001
Trend x season interaction		12	0.147		12	0.017
WL1						
Site relative mean effect	-6.92		< 0.0001	-0.249		< 0.0001
Trend effect		5	< 0.0001		5	< 0.0001
Season effect		3.7	< 0.0001		4	< 0.0001
Flow effect		---	---		1	< 0.0001
Trend x season interaction		12	0.908		12	0.356
WL16						
Site relative mean effect	-4.58		< 0.0001	-0.182		< 0.0001
Trend effect		5	< 0.0001		5	< 0.0001
Season effect		3.7	< 0.0001		4	< 0.0001
Flow effect		---	---		1	< 0.0001
Trend x season interaction		12	0.374		12	0.041
Black Burn (control)						
Site mean effect	28.17		< 0.0001	3.376		< 0.0001
Trend effect		5	0.0050		5	0.199
Season effect		3.23	< 0.0001		4	< 0.0001
Flow effect		---	---		1.4	< 0.0001
Trend x season interaction		12	0.975		12	0.975

The [DOC] trend was highly significant in the Whitelee catchments ($p < 0.0001$) and in the Black Burn catchment ($p = 0.005$), although, the oscillating [DOC] trend in the Black Burn is very different to the increasing [DOC] trends in the Whitelee catchments (Fig. 4a). The fitted trends in [DOC] from the final ANCOVA model (Fig. 4a) show a marked increase in the wind farm-impacted Whitelee catchments during the study period, whereas in the minimally-disturbed Black Burn catchment there is no evidence of an upwards decadal trend.

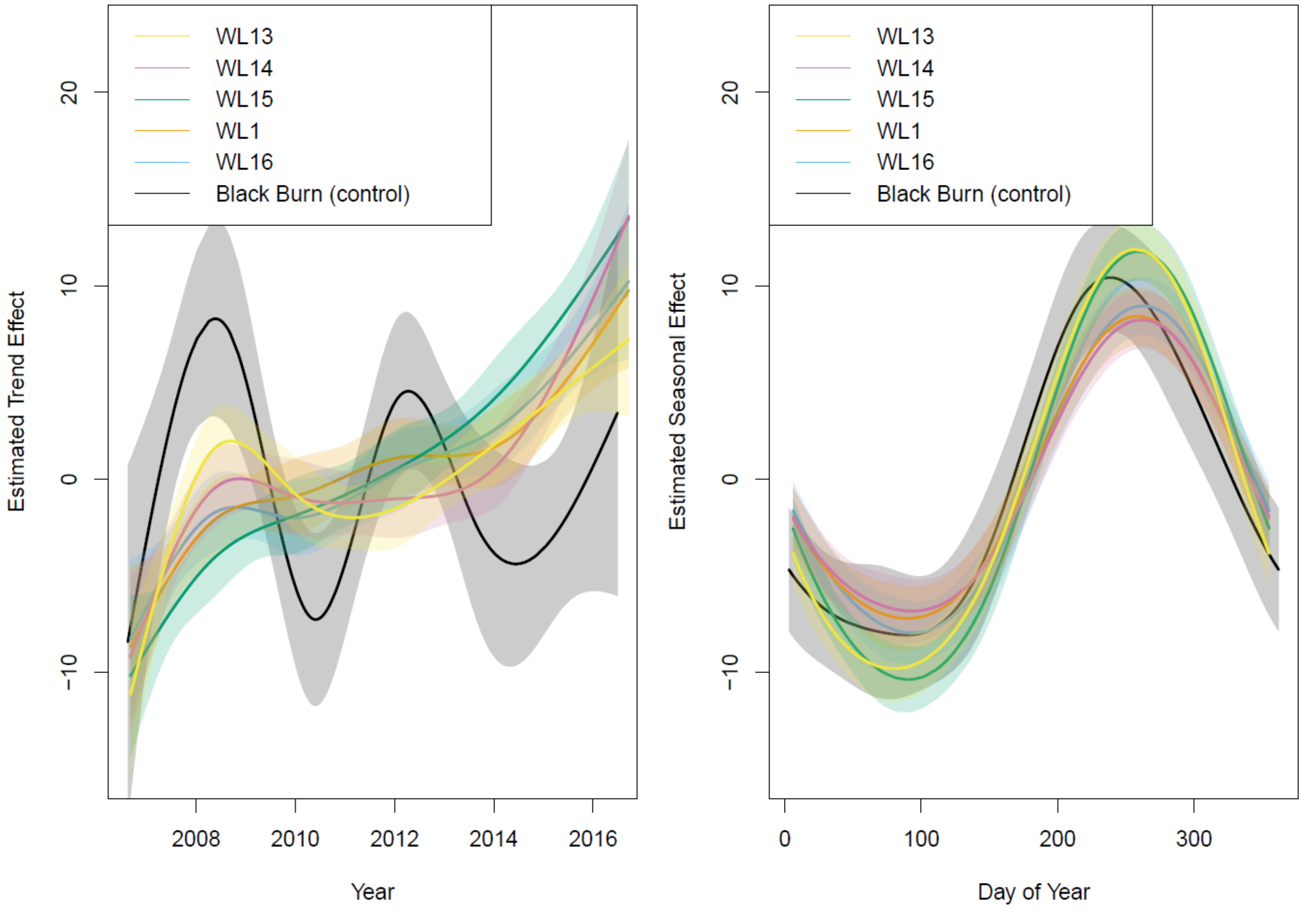


Figure 4. Estimated smooth curves of (a, left) the long-term trend component and (b, right) the seasonal components of DOC concentration in the five catchments draining Whitelee wind farm and in the Black Burn control catchment. The shaded areas represent an approximate pointwise 95% confidence band. The y-axis scale is the same in both plots and represents the fitted effect, after removing mean site effects.

A major driver of the trend of increasing [DOC] reported from the 1990s onwards in many Northern hemisphere temperate peat-dominated upland catchments is the reduction in atmospheric sulfate deposition and hence precipitation ionic strength, resulting in increased dissolution of soluble soil organic matter (e.g. Monteith et al., 2007). However, the rate of increase in [DOC] in response to reduced acid deposition changes was strongest in the late 1980s–1990s and has decelerated in the 2000s in headwater catchments in northern and central Europe (de Wit et al., 2021), in contrast to the increasing [DOC] trend in the Whitelee catchments for 2006–2016. The divergent trend in the Whitelee catchments compared to our control catchment (Black Burn) is notable as the more easterly Black

Burn would have been more impacted by reduced acid deposition, and therefore expected to have an increasing trend in [DOC] compared to the Whitelee catchments (Monteith et al., 2023).

For all catchments the seasonality of [DOC] was highly significant (p < 0.0001), but there was no significant interaction between trend and seasonality. As observed routinely in other UK peat-dominated catchments (e.g. Dawson et al., 2008; Reynolds et al., 1997; Tipping et al., 2007; Worrall et al., 2006), [DOC] had a clear seasonal pattern in all the study catchments, with an annual maximum around day 230–280 (late summer–early autumn) and minimum around day 70–120 (late winter–early spring) (Fig. 4b). Highest [DOC] occurs at the end of the summer due to hydrological flushing of the products of enhanced organic decomposition during warmer summer soil temperatures and/or DOC concentration due to evapotranspiration (Dawson et al., 2002; Tipping et al., 2007). Once the soil DOC store has been exhausted, subsequent hydrological events dilute the DOC pool, causing the annual minimum [DOC]. The magnitude of the trend effect of increasing [DOC] over time at the Whitelee catchments is of similar magnitude to the seasonal change in [DOC] in all catchments. Although the [DOC] seasonal amplitude was larger at WL13 and WL15, overall there was little difference in the amplitude between the disturbed and undisturbed catchments. However, the timing of [DOC] seasonality differed between the disturbed and undisturbed catchments, with the annual minimum [DOC] occurring earlier and the maximum [DOC] later in the Whitelee catchments, compared to the Black Burn.

3.3 DOC flux modelling results

The GAMMs with interactions between the site and the trend and seasonal effects accounted for almost all the variability in DOC flux in the disturbed and minimally-disturbed catchments (R^2 = 94.3% and 98.5%, respectively). The flow term was significant for modelled DOC flux in all catchments, as expected (Table 2). For Whitelee catchments, the DOC flux trend, seasonality, and the interaction between trend and seasonality were highly significant (p < 0.0001), except the interaction terms for WL13 and WL1, which were not significant (p > 0.05) (Table 2). As flow has been controlled for in the model, the significant trends in DOC flux in the Whitelee catchments are not attributed to flow. For Black Burn, seasonality of DOC flux was significant, but no significant trend in DOC flux was identified.

The fitted DOC flux trends after the effect of flow has been accounted for (Fig. 5a) show a clear increase in DOC flux rate in the wind farm-impacted Whitelee catchments but no apparent trend in the minimally-disturbed Black Burn catchment. There is a distinct seasonal pattern in all catchments (Fig. 5b), with lowest DOC flux values near the start of the year (late winter) and highest around day 250 (late summer–early autumn). However, there are some notable differences between the catchments. DOC flux in WL14 has a lower seasonal amplitude compared to the other Whitelee catchments, attributed to buffering by the reservoirs in WL14, and is more similar to that of the Black

Burn. Furthermore, the Black Burn has a lower seasonal amplitude in DOC flux and the minimum DOC flux occurs later in the year (~day 50) compared to most Whitelee catchments (~day 20–30). The magnitude and timing of seasonal DOC flux have implications for water quality management and treatment for colour in potable water supply catchments (Ferretto et al., 2021; Williamson et al., 2023b). For example, where alternative water sources are available, to reduce the treatment costs of colour removal, abstraction from sources with high DOC could be paused if the timing of annual maximum DOC is known. Since the focus of this research was on DOC flux trend, further analysis and discussion of the DOC flux seasonal effect is presented in Supplementary Material S3.

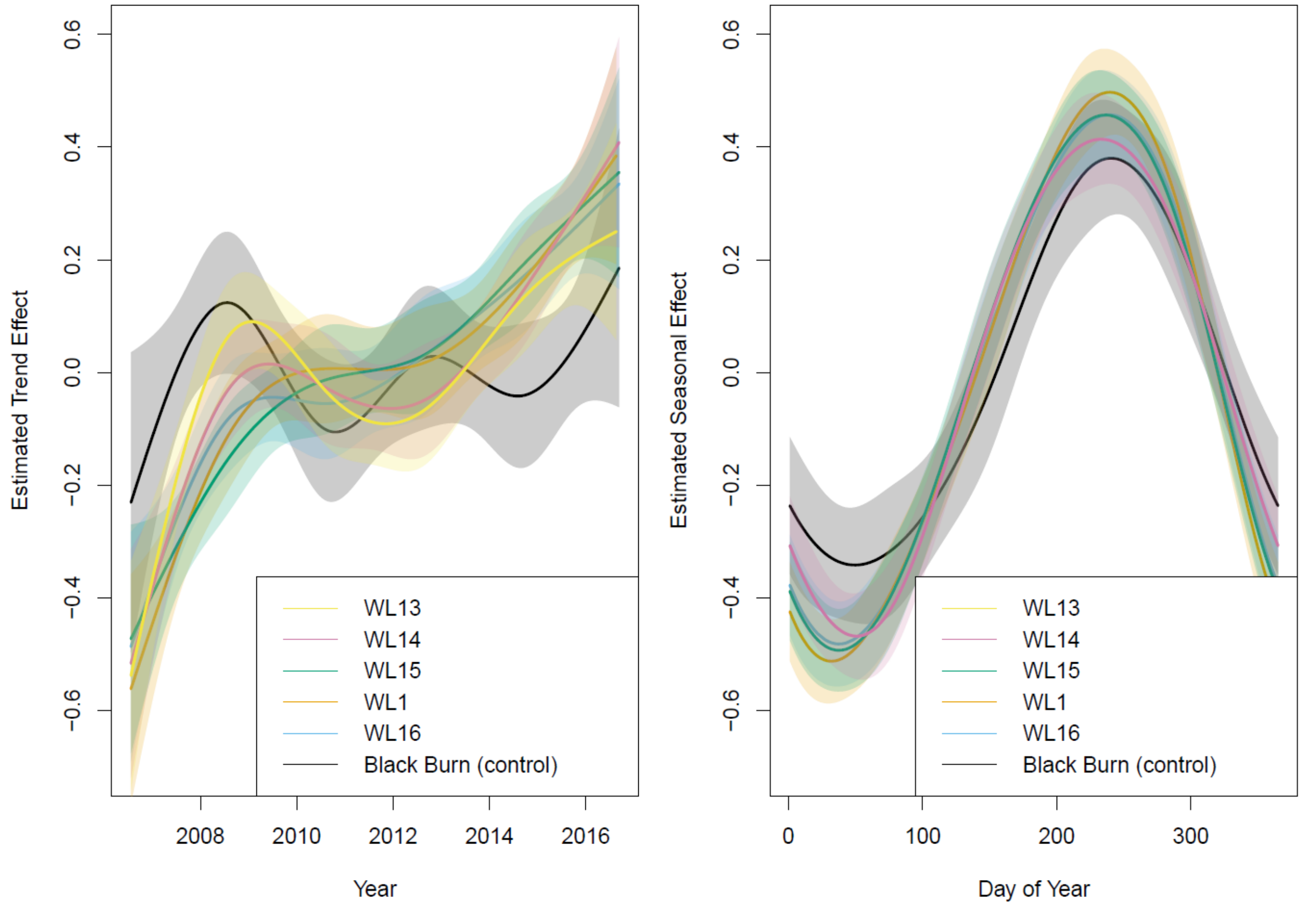


Figure 5. Estimated smooth curves of (a, left) the long-term trend component and (b, right) the seasonal components of DOC flux for the five catchments draining Whitelee wind farm and in the Black Burn control catchment. The shaded areas represent an approximate pointwise 95% confidence band. The y-axis scale is the same in both plots and represents the fitted effect, after removing mean site effects.

3.4 DOC flux trends consistent with catchment interventions

A clear overall upwards trend in DOC flux in all Whitelee catchments is evident from the fitted smooth curves of the DOC flux trend for each catchment (Fig. 6a). In contrast, in the Black Burn there is no evidence of a trend, indicating that regional drivers (such as changes in atmospheric sulfate deposition or climate) do not explain the increasing DOC flux trend in the Whitelee catchments. Since the effect of flow has been controlled for in the model, any differences between catchments must be attributed to drivers other than flow. Thus, the wind farm development appears to underlie the significant upwards DOC flux trend in the Whitelee catchments. The overall increasing Whitelee DOC flux trend shows some inter-catchment differences. Compared to the other Whitelee catchments, WL13 and WL14 have a different trend pattern, with a greater initial rate of DOC flux increase, followed by a more evident decrease before increasing again later in the study period. This is attributed to phase 1 of the wind farm development affecting WL13 and WL14 most acutely and close to the catchment sampling locations so that there was a greater rate of increase in DOC flux in these catchments, followed by a short recovery period. WL13 was not affected by phase 2 of development so the later increase in DOC flux trend is at a slower rate than in the first development phase and has a gradient similar to the other Whitelee catchments.

To assess the effect of wind farm development on the DOC flux trends in the Whitelee catchments, the first derivatives of the DOC flux trend were calculated to examine rates of change over time and identify the time points at which changes in the direction of trend occur (Fig. 6b). If the wind farm development is a key driver of DOC flux trends in the Whitelee catchments, we would expect the time points at which the trend changes direction to be slightly lagged after the start of wind farm development phases. The change points - the dates when the modelled DOC flux trend in a catchment changed from an increasing to decreasing trend or vice versa (shown as vertical dotted lines Fig. 6b) - were analysed to identify if synchroneity in DOC flux trends occurred between the disturbed catchments and the control and, if not, whether the timing of the change points is aligned with known catchment interventions (Table 3).

In phase 1 of the Whitelee wind farm development, DOC flux increases in all the Whitelee catchments. Towards the end of phase 1 development the rate of DOC flux increase decelerates, and in WL13, WL14, WL16 and WL1 the first derivative falls to around zero or below, indicating some recovery (decrease) of the DOC flux following disturbance, particularly for WL13 and WL14 (Fig. 6b). The second phase of the Whitelee wind farm development, primarily affecting WL1 and WL15 (Fig. 3), is associated with an acceleration in the DOC flux trend in all Whitelee catchments. In WL15, the continuous upwards trend in DOC flux is attributed to this catchment being most impacted by phase 2 of the wind farm development (Heal et al. 2020), with insufficient time for recovery between development phases 1 and 2. After cessation of the Whitelee wind farm development in 2013, the increasing trend in DOC flux is maintained in the Whitelee catchments, with no evidence of a recovery in DOC flux to pre-development values. The absence of a trend in DOC flux in the Black Burn catchment indicates that Whitelee catchments show a longer-term DOC flux response to wind farm development, shifting to a new higher baseline. This response is counter to the decelerating trend in [DOC] reported for largely undisturbed but otherwise similar catchments in Europe for the period 2002–2016 and supports further the interpretation that wind farm development increases DOC flux is that (de Wit et al., 2021).

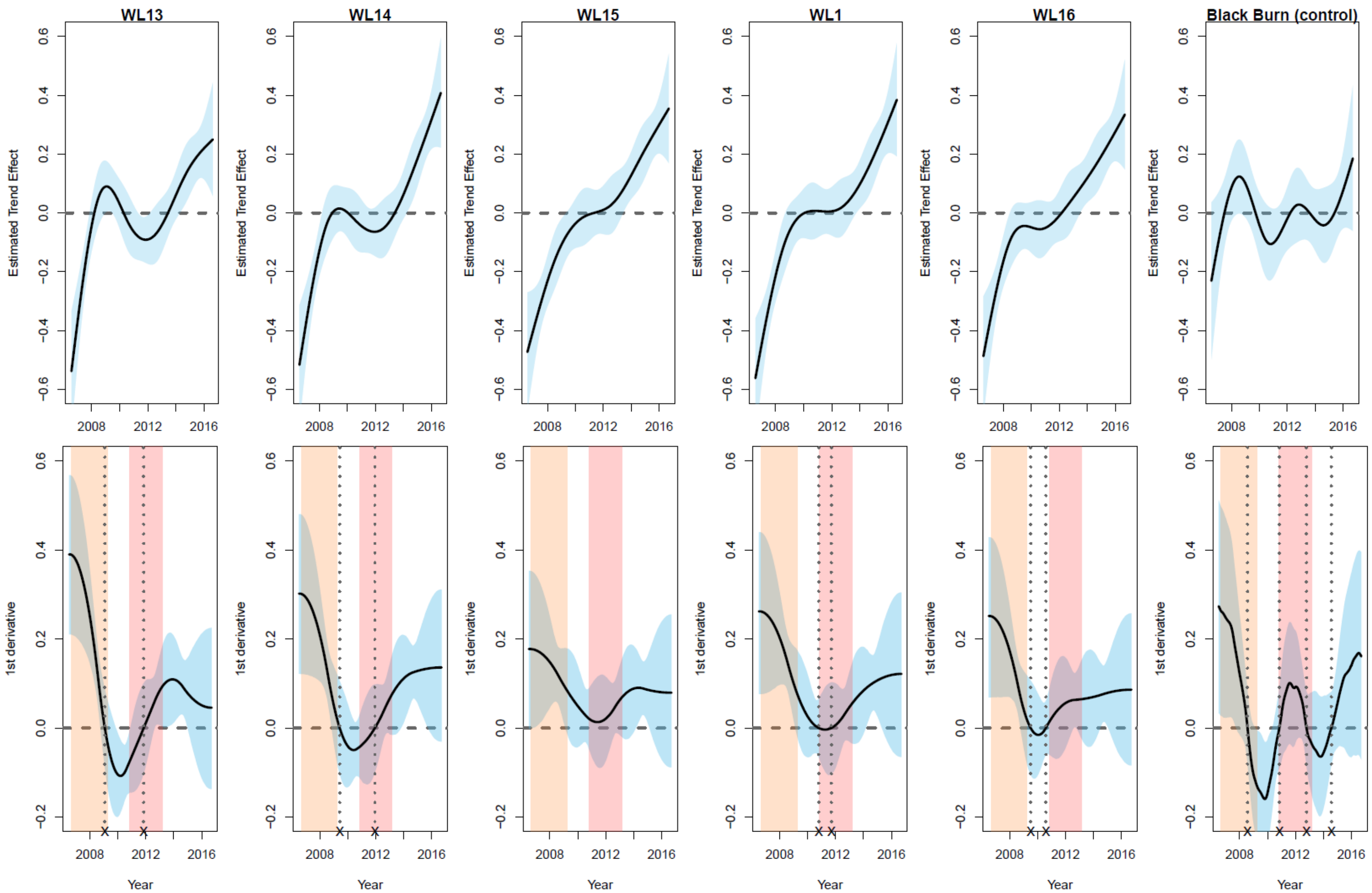


Figure 6. DOC flux for each catchment with flow modelled as a covariate, (a, top) estimated smooth curves of the DOC flux trend, and (b, bottom) plots of the first derivative of the DOC flux trend. The horizontal dashed lines in (a) and (b) show the zero-change line. The vertical dotted lines in (b) are the intersections between the derivative curve and the zero-change lines, i.e. these lines show the dates ("change points") at which the DOC flux trend changed direction from increasing to decreasing or vice versa. There are no vertical lines for WL15 as the DOC flux trend increased throughout the study period. In (a) the shaded areas represent an approximate pointwise 95% confidence band. In (b), the blue shading represents the 95% bootstrapped confidence bands for the derivative functions of the estimated trend effects and the vertical orange- and red-shaded bands show Whitelee wind farm phase 1 and phase 2 development periods, respectively. Note that the Whitelee wind farm phase 2 development did not include WL13 and WL16, though in WL16 development of Calder Water Community and West Browncastle wind farms occurred Aug 2012–Aug 2014.

In the Black Burn catchment, there were four change points in the DOC flux trend that comprised two cycles of an increasing to decreasing flux, followed by an increasing flux at the end of the study period (Fig. 6b). This may be a regional response, which we take it into account when estimating C loss due to the wind farm (see section 3.5). In the Whitelee catchments, apart from WL15, there were two change points (Fig. 6b). The first change point, from increasing to decreasing DOC flux, was in 2009–2010 around the time of the completion of phase 1 of the Whitelee wind farm development, indicating recovery. The second change point, from decreasing to increasing DOC flux, occurred in 2010–2011 within the first year of the Whitelee wind farm development phase 2 commencing. From then until the end of the study period in 2016 there were no further change points in the Whitelee catchments as the DOC flux continued to increase, whereas the Black Burn DOC flux trend fluctuated through two further change points.

The modelled DOC flux trends for the wind farm-impacted catchments WL13, WL14, WL1 are consistent with our scenario F of incomplete recovery after the first intervention followed by shifting to a higher baseline after the second intervention (Fig. 1c). WL16 shows a similar response but the second change point is prior to two separate wind farm developments (Fig. 3) and so a second development phase is not clearly a driver of change. WL16 has the largest catchment area (Table 1) so there may also be a lagged response to activities in phase 1. The modelled DOC flux trend for WL15 is aligned with scenario G (no recovery evident after any intervention, Fig. 1c). The absence of any change points and no recovery in WL15 (Table 3) is attributed to this catchment being most impacted by phase 2 of Whitelee wind farm development, with insufficient time for DOC flux recovery (decrease) in streamwater between development phases 1 and 2.

Other studies have identified elevated [DOC] and DOC fluxes for up to several years after forest felling on peatlands. For example, highly increased DOC fluxes occurred in the first three years following harvesting of forests on drained peatland in Finland (Nieminen et al., 2015) and elevated [DOC] was observed in pore water and surface water in previously afforested peatland up to 17 years after forest felling and restoration commenced (Gaffney et al., 2018; Howson et al., 2021). A global review reported that DOC fluxes from disturbed peatland catchments were significantly higher than from undisturbed sites, with means of 30.7 ± 23.1 and 18.5 ± 19.3 g C m^{-2} $year^{-1}$, respectively (Rosset et al., 2022). However, there have been very few studies of the impacts of successive disturbances on DOC fluxes from peatlands, although a literature review emphasised that multiple stressors from forest management and their interaction can cause cumulative impacts on stream water quality and ecology (Kuglerová et al., 2021).

Table 3. Dates of change points when catchment modelled DOC flux trend changed increasing (↑) to decreasing (↓) or vice versa (see Fig. 6a) during the study period 2006–2016, and interpretative comment on catchment interventions occurring at that time. Whitelee information from Murray (2012), Phin (2016) and Zheng et al. (2018).

Catchment	Change point 1	Change point 2	Change point 3	Change point 4
WL13	24/01/2009 (↑ to ↓) Phase 1 Whitelee wind farm development Oct 2006–Jun 2009 affected 21% of catchment area. Conifer forest felling Nov 2006–Jan 2008 in 30.3% of catchment area. Whole-tree and brash mulching May 2007–Jul 2008.	06/11/2011 (↓ to ↑) High DOC attributed to continuing impact of forest felling (Zheng et al., 2018) and large mobile DOC pool resulting from harvesting techniques (Howson et al., 2021).	---	---
WL14	02/06/2009 (↑ to ↓) Phase 1 Whitelee wind farm development Oct 2006–Jun 2009 affected 60% of catchment area. Conifer forest felling in 2008.	09/12/2011 (↓ to ↑) Phase 2 Whitelee wind farm development Nov 2010–Mar 2013, affected 11% of catchment area. Conifer forest felling in 2013.	---	---
WL15	No change points identified. Continuous upwards trend in DOC flux 2006–2016, with lower gradient in 2010–2012. Phase 1 Whitelee wind farm development affected 12% catchment area, including conifer forest felling in 2006, 2008 and 2009. Phase 2 of the Whitelee wind farm development affected 66% of the catchment area Nov 2010–Mar 2013. Phase 2 activities in the catchment included: track construction Nov 2010–Jun 2012, construction of the substation Nov 2010–May 2011, installation of turbine bases Jan–Sep 2011, digging of cable trenches Aug 2011–Feb 2012, conifer forest felling in 2010, 2011 and 2013.			
WL1	18/10/2010 (↑ to ↓) Phase 1 Whitelee wind farm development Oct 2006–Jun 2009 affected 24% of catchment area. Conifer felling in 2006 and 2008 in 10.7% of catchment area.	10/09/2011 (↓ to ↑) Phase 2 Whitelee wind farm development Nov 2010–Mar 2013, affected 26% of catchment area.	---	---
WL16	03/07/2009 (↑ to ↓) Phase 1 Whitelee wind farm development Oct 2006–Jun 2009 affected 30% of catchment area. Conifer forest felling in 13.2% of catchment area.	04/08/2010 (↓ to ↑) High DOC attributed to continuing impact of forest felling. Change point before Calder Water Community and West Browncastle wind farm developments Aug 2012– 2014.	---	---
Black Burn	26/07/2008 (↑ to ↓) No major catchment interventions	06/11/2010 (↓ to ↑) No major catchment interventions	15/10/2012 (↑ to ↓) No major catchment interventions	28/07/2014 (↓ to ↑) Change point identified before blocking of 39 drains in the upper catchment in Mar 2015

Disentangling the possible effects on DOC flux trends of different activities involved in the wind farm development, such as borrow pit excavation, track construction, excavation for turbine foundations, and forest felling, is complicated. However, this research across different Whitelee catchments (Table 1) with a range of conifer forest cover (~30–80%) and forest felled (10–34%) across individual catchments has indicated that conifer forest felling is a major driver of increased DOC concentration and flux. In the WL13 catchment, higher streamwater [DOC] was measured in 2014–2015 in the sub-catchment dominated by forest felling compared to the other sub-catchment mostly affected by land preparation for wind turbine emplacement and with almost intact forest (Zheng et al., 2018). Furthermore, DOC compositional analysis indicated forest harvesting debris and brash as a major DOC source in the sub-catchment most affected by forest felling (Zheng et al., 2018). In WL15, multiple linear regression analysis of catchment and wind farm development related controls on nutrient concentrations and fluxes in its sub-catchments during the phase 2 development, identified forest felling as the primary wind farm development activity with a significant positive impact on [DOC] and DOC flux (Heal et al., 2020). In WL15, there was near continuous disturbance throughout the Whitelee wind farm development in which conifer forest felling occurred every year from 2006 to 2013 (apart from 2007 and 2012), accounting for the continuous upward trend and the absence of change points in DOC flux. The extensive area and ongoing influence of forest felling on DOC concentrations and fluxes may also explain the occurrence of the second change points in WL13 and WL16, from a decreasing to an increasing DOC flux trend, even though they were not affected by phase 2 of the wind farm development. The observation that conifer forest cover influences DOC concentration and flux in peatland catchments has been reported elsewhere (e.g. Škerlep et al., 2020; Williamson et al., 2023a, 2023b).

Several processes may account for the increased streamwater [DOC] and DOC fluxes associated with forest felling, including: DOC mobilisation from felling debris and exposed and disturbed soil (e.g. Nieminen et al., 2017); warmer soil temperatures after felling stimulating soil microbial activity to generate an enhanced soil DOC pool (Pérez-Batallón et al., 2001; Schelker et al., 2013); a higher water table and increased runoff, due to reduced evapotranspiration arising from lower interception losses after tree removal, resulting in greater mobilisation of DOC from near-surface organic-rich soils (e.g. Laudon et al., 2009; Muller et al., 2015). In this study, direct changes in runoff do not explain DOC flux trends as changes in flow were controlled for in the DOC flux modelling by including flow as a covariate. Analysis of DOC concentrations and composition in streamwater draining the two sub-catchments of WL13 identified higher concentration and ‘fresher’ (less humified) composition in streamwater DOC draining the felled forestry area in 2014–2015 (Zheng et al., 2018) at least 6 years after felling, suggesting that DOC mobilised from felling debris contributed to the increased DOC fluxes in the Whitelee catchments. The practice of on-site mulching of whole trees and felling debris (brash) at WL13 and WL14 (Murray, 2012) is also expected to have augmented the pool of readily mobile DOC in these catchments as reported at other peatland sites with forest felling (Howson et al., 2021).

3.5 Significance of the increasing trends in DOC flux for C losses from the landscape and for other ecosystem services

Whilst the modelled trend shows that DOC flux increases in the Whitelee sites during and after wind farm development, it is important to understand in a carbon-sensitive landscape how much this change in carbon flux represents, and thus the carbon amount that has to be sequestered to counteract net loss. We have approached this by calculating how much extra C the DOC flux from the Whitelee catchments represents, assuming Black Burn represents the regional flux trend of Whitelee independent of wind farm development. This is explained more fully in Supplementary Material S4 and Table S2, but in brief for all catchments we have used the first observation year (2006–07) as the pre-wind farm baseline and calculated the % change in flux (g C m^{-2} $year^{-1}$) compared to three different time periods: i) the final year, 2015–16, ii) the mean of years 2007–15, iii) the mean of years 2007–16. Three time periods have been chosen to provide a measure of uncertainty associated with correctly choosing a comparison end point. The % DOC flux change observed in Black Burn for these time periods is subtracted from the % change at each Whitelee catchment and the residual % increase flux in the Whitelee catchments is attributed to wind farm development activity.

This % change is of mass terrestrial C flux, per unit area, per unit time, which we can compare with estimates of peaty soil C sequestration: net ecosystem exchange (NEE), or net ecosystem carbon balance (NECB) (Table 4, which includes estimates from Black Law, a nearby wind farm on peatland). The different Whitelee catchments and calculation approaches yield similar estimates, with a mean for all catchments and across the three approaches of 1.03 ± 0.28 g C m^{-2} $year^{-1}$, approximately 0.26-3.44 ± 0.07–0.94% of the mean values reported in Table 4 from peatland NEE/ NECB or used in modelling NECB. Although the DOC fluxes have not returned to pre-wind farm values, the ongoing C loss represents a very small component of net ecosystem exchange. For this wind farm, and for solely C lost as DOC, the soils would have to be close to a net source of C (very degraded) for the wind farm development to change this site to a net source of atmospheric CO_2.

We note DOC flux is not the only way in which C is lost and there remain other landscape C fluxes (e.g. particulate organic carbon, and gaseous CO_2 and CH_4) that could be measured for a fuller C budget, allowing a more accurate assessment of overall impact of wind farm development on landscape carbon storage. An overview of these different approaches has been included in the Supplementary Material S5.

Table 4. Estimated loss of Whitelee catchment stored C as DOC flux due to wind farm development calculated through three different time period approaches, in comparison to rates of C sequestration measured from a range of peaty soils, including one hosting a wind farm. The input term for C sequestration of peaty soils in the wind farm carbon calculator is also included. See text and Supplementary Material S4 for calculation details.

Comparison period	**Additional catchment DOC flux due to wind farm (g C m^{-2} $year^{-1}$)**					**Mean**	**StDev**
	WL13	**WL14**	**WL15**	**WL1**	**WL16**		
2015–16	1.34	0.93	2.05	1.50	1.42	1.45	0.40
mean 2007–16	1.10	0.48	1.09	0.87	0.75	0.86	0.26
mean 2007–15	1.06	0.43	0.97	0.79	0.67	0.78	0.25
Mean	1.17	0.61	1.37	1.05	0.95	**1.03**	**0.28**
StDev	0.15	0.28	0.59	0.39	0.41	**0.36**	
Reference	**Other studies**						
Armstrong et al. (2015)	Black Law wind farm, ~ 25 km from Whitelee NEE of all 4 sites throughout the year: -46.0 ± 196.9 mg C m^{-2} h^{-1} (Table 5) which is -402.96 ± 1724.84 g C m^{-2} $year^{-}$						
Nayak et al. (2010)	Wind farm carbon calculator -0.9 t CO_2e ha^{-1} $year^{-1}$ for undrained peat used, which is -90 g C m^{-2} $year^{-1}$						
Artz et al. (2013)	Average net ecosystem C balance from 19 ombrotrophic peatlands from 5 sites in the Northern hemisphere -29.9 ± 9 g C m^{-2} $year^{-1}$ reported in Table 2. NEE is approximately 50% greater on average						
Levy and Gray (2015)	Ombrotrophic blanket bog northern Scotland, NEE/NEBC respectively of -114.00 and -99.37 g C m^{-2} $year^{-1}$(Table 1)						

Whilst terrestrial C sequestration may have the capacity to buffer the impact of a well-built wind farm, the change in catchment drainage biogeochemistry may impact other ecosystem services. Mean DOC concentrations have increased by approximately 25% and with this increased iron concentration (Zheng et al., 2018) and together these 'brown' the waters. In low concentrations, more DOC in freshwaters can be ecologically beneficial, but at higher concentrations (such as in the Whitelee catchments) this can lower phytoplankton diversity and ecosystem productivity, promote pelagic over benthic species and reduce recovery from acidification (e.g. Kritzberg et al., 2020) and this may be undesirable.

Further, if the catchment drainage is used for potable water, as is the case in Whitelee, increased DOC requires more removal to reduce problems with water taste, odour and bacterial contamination, and the risk of potentially carcinogenic chlorinated organic compounds arising from standard disinfection procedures (Fenner et al., 2021 and references therein). This results in increased water treatment costs due to increasing amounts of coagulants, sludge production and disposal, and disinfectants (Matilainen et al., 2010; Ritson et al., 2016). Additionally, water suppliers may need to invest in additional treatment stages (Scottish Water 2020), new treatment works (Xu et al., 2018 and references therein) or find alternative water sources. Surface water is abstracted from WL14 to the nearby Amlaird WTW. Whilst our research design does not allow identification of whether the Whitelee wind farm development directly affected the quality of the WTW input water, in 2012 (mid-way during the monitoring period), the Drinking Water Quality Regulator for Scotland (DWQR) noted that "*a step change has occurred in raw water colour in recent years. During periods of exceptionally high colour usually experienced in Summer months this requires a coagulant dose beyond that of the design of the works and can result in higher than normal levels of iron and organic carbon in the treated water*" such that "*trihalomethanes, iron and manganese have exceeded regulatory standards*" (DWQR, 2012). It is understood that as the works were unable to cope with the changing water composition and to ensure the provision of treatable water, Amlaird WTW was interconnected with other water sources outwith the wind farm catchments as part of a major investment by Scottish Water to improve the resilience of the regional water supply network. Within the overall £120m project this has required complicated engineering (e.g. tunnelling) to gravity feed water by a pipeline across areas of deep peat (Water Projects, 2016).

Although the current study did not include monitoring to assess whether changes in DOC concentration / flux affected aquatic biodiversity and biogeochemical cycling, extensive literature exists already on the detrimental impacts of freshwater browning on aquatic ecosystems, driven by increased organic C concentrations (e.g. Blanchet et al., 2022; Solomon et al., 2015). Key impacts are summarised here. Increased light attenuation results in warming of surface waters and more stable stratification in lakes, altering the thermal regime and dissolved oxygen concentrations (e.g. Pilla et al., 2018). These physical and chemical changes have ecological impacts that include, amongst many, reduced primary production (e.g. Thrane et al., 2014), altered ecosystem metabolism (Wilken et al., 2018), and reduced zooplankton biomass (Tonin et al., 2022) and invertebrate abundance (Arzel et al., 2020). Changes in aquatic carbon cycling may also occur that exacerbate climate change and reduce the resilience of freshwaters, such as increased CO_2 emissions from waterbodies due to increased DOC amounts and susceptibility to photodegradation (Lapierre et al., 2013) and the associated warmer water temperatures (Kosten et al., 2010). These effects occur not only in standing waters, but in headwater streams such as our study catchments that contribute ~75% of global riverine CO_2 and CH_4 emissions (Li et al., 2021).

3.6 Approaches to management of critical landscapes

It is clear from the last section that ecosystem provisioning in critical landscapes needs to draw on sustainable landscape management. Mismanagement contributes to the exceedance of thresholds for critical landscapes and the Earth system may not provide ecosystem services. For example, the latest update to the planetary boundaries framework (Richardson et al., 2023) assesses that the boundaries have been transgressed for the Earth system processes on which our study impacts: land system change, climate change, freshwater change and biosphere integrity. We can identify two ways in which the risk that boundaries exceedance poses ecosystem resilience is reduced.

The first is to develop best practice decision-making to protect, manage and restore the ecosystem services a critical landscape offers. To demonstrate how to do this we constructed a decision-making procedure for managing wind farm development on peatland that accommodates both research from this and other studies, and existing professional guidance industry and regulatory bodies have developed in tandem. In this example (Fig. 7), the priority ecosystem service is wind energy generation, but thereafter C storage, forest products (e.g. wood, pulp) and potable water are priorities in order of decreasing importance. The steps in the flow chart guide decision making from wind farm site identification, through to site planning and measures taken during and after construction to avoid, minimise and mitigate deleterious impacts on the ecosystem services of interest.

To help understand the effectiveness of the steps and measures in the flow chart within the context of land stewardship and nature-based solutions, we incorporated the natural climate solutions (NCS) hierarchy proposed by Cook-Patton et al. (2021). The hierarchy classifies NCS into three categories – protection, improved management and restoration – with protection and improved management offering the most cost-effective methods for nature-based mitigation compared to restoration. Accordingly, where relevant, the steps and measures in Fig. 7 have been colour-coded to indicate their classification in the NCS hierarchy. For example, Step 2, asking whether an existing wind farm can be repowered instead of development occurring at a `new' peatland site, is classified as "protection" since it protects peatland ecosystems at a high risk of losing carbon stores from wind farm development. The decision-making framework thus places greater emphasis in the initial steps on the most effective measures of protection and improved management, with restoration occurring predominantly in the design of the wind farm development once the site has been identified.

The second is to forward plan better land management and pre-emptively accommodate, as yet unknown but likely land use changes. For example, given typical wind farm layout requirements are known, design forest layout with future wind farm development in mind, e.g. plan in access tracks and potential turbine bases so forest felling for wind farm development, and associated soil disturbance is reduced; design fire breaks to be landscapes that also could accommodate turbine bases; plant mixed forestry with deciduous trees closer to possible turbine locations offering undisturbed natural buffer zones to filter and trap organic matter loss.

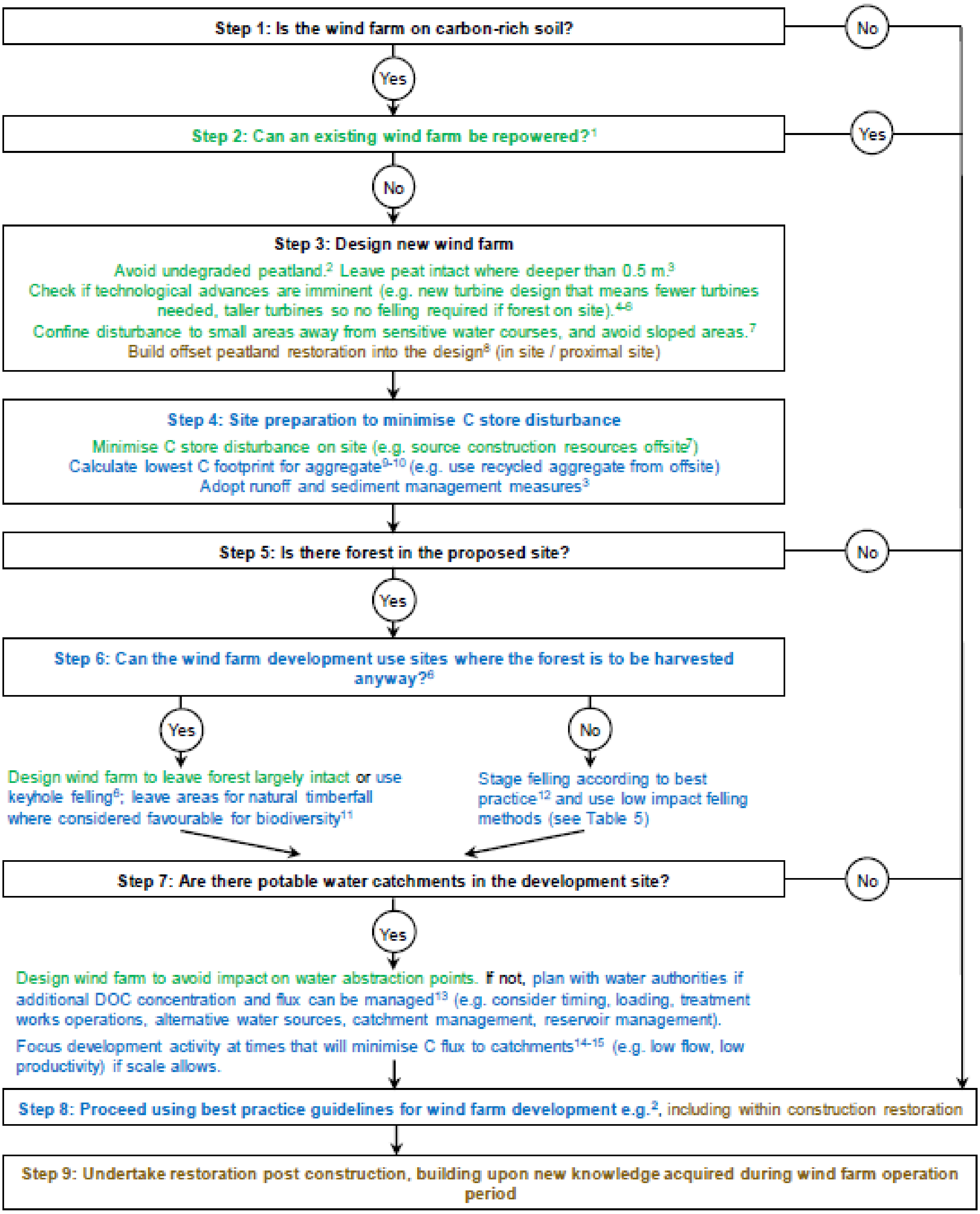


Figure 7. Flow chart of decision-making processes for wind farm development on peatland for three ecosystem services of interest considered in this priority order: carbon storage, forest products, potable water supply. This can also apply to sites without forestry. Where relevant, the text colour reflects the hierarchy of natural climate solutions of Cook-Patton et al. (2021). Steps/actions in green indicate protection, those in blue are improved management, and those in brown indicate restoration.

Superscript numbers indicate references. [1]Waldron et al. (2018). [2]Smith et al. (2014). [3]NatureScot (2024). [4]Scottish Government (2022b). [5,6]Enevoldsen (2016, 2018). [7]Heal et al. (2020). [8]Nayak et al. (2010). [9]Inventory of Carbon and Energy (2019). [10]Zhang et al. (2023). [11]Humphrey et al. (2005).[12]Forest Research (2023). [13]Williamson et al. (2023b). [14]Dawson et al. (2011). [15]Doyle et al. (2019).

Table 5. As directed from the flow chart of decision-making processes (Step 6), a summary of measures reported and recommended for minimising increases in DOC concentration / DOC flux following forest harvesting from studies focused on peatlands. Measures to minimise negative impacts of forest felling on water quality in all landscapes are also explained in Forestry Commission (2019) and Shah et al. (2022).

Measure	Reference
Use of low impact harvesting techniques, e.g. trees felled by hand and winched to forwarder track; felling residues used to strengthen forwarder tracks and subsequently removed for use as biomass for fuel.	Shah and Nisbet (2019)
Timely removal of needles and branches after harvesting, whilst minimising disturbance from brash removal	Gaffney et al. (2022)
Avoid whole-tree mulching (trees chipped from standing), used where trees have little or no commercial value, site access is constrained and/or conventional harvesting may be environmental damaging	Howson et al. (2021)
Reduce artificial hydrological connectivity, such as not maintaining ditch networks and using shallower ditch networks	Härkönen et al. (2023) and references therein
Do not harvest in riparian zone forest and peatland buffer areas and ensure buffer zones are not too narrow	Härkönen et al. (2023)
Create and maintain buffer zones free of brash	Howson et al. (2021)
Phased felling	Shah and Nisbet (2019); Howson et al. (2021)
In boreal forest, time harvesting on frozen ground during winter to decrease soil disturbance	Nieminen et al. (2017)
Spatial extent of harvesting should not exceed 15–30% of catchment area within a 1–10-year period.	Kuglerová et al. 2021
Restore peatland	Härkönen et al. (2023)

Our research offers considerable insight into the trajectory of DOC fluxes after multiple landscape disturbances revealing that, even in complex wind farm development, forestry on peatlands is priming enhanced DOC concentrations and fluxes in streamwater and increasing catchment vulnerability through net export of carbon on its removal (as well as likely in its planting e.g. Williamson et al., 2023b). In wind farms developed on peatland with forest harvesting and few/no mitigation measures, our analysis in section 3.5 indicates ~20–30% increases in DOC fluxes are anticipated for at least the first 10 years due to the development alone, excluding any increase arising from regional factors (e.g. changes in atmospheric sulfate deposition or climate). Such understanding allows mitigation to be planned into wind farm development through site selection and measures to minimise C losses during construction and best support the green energy transition. Given that forest plantation on peatland will be felled anyway and that wind farms sited on peatland will result in some disturbance during development, we suggest that sites with forests scheduled to be felled are selected in preference to pristine or less disturbed peatlands (Step 6 in the decision-making flow chart). In situations where continued generation of forest products is a desired ecosystem service for the peatland landscape, there is some evidence that the use of continuous cover forestry with trees of different ages and also a mixture of conifers and deciduous trees reduces the increase in organic carbon fluxes at harvesting since not all trees are removed at the same time and oxidation of deep peat is minimised (Härkönen et al., 2023).

Although the need to consider multiple ecosystem services in making landscape management decisions is widely recognised and numerous tools have been proposed to assess ecosystem services (e.g. Bagstad et al., 2013), it is not usual to see multiple ecosystems services designed explicitly into landscape management decision tools (Duarte et al., 2020). Such an approach could serve as a way forward in reducing the impact of human activities on natural resources.

3.7 Applicability to other geographical and ecological contexts and methodological advances

Onshore wind development is an important element for meeting the Net Zero Emissions by 2050 scenario (IEA, 2025), which has been adopted as a goal by most countries. It is projected that additional onshore wind capacity will nearly double from 2024 to 2030, particularly in Europe, the USA, India, China and other developing and emerging economies (IEA, 2024). Ireland and Spain in addition to the UK) have been identified as European countries with blanket bog peatlands that may be impacted by wind farm development (Chico et al., 2023). In Sweden, which has the third largest peatland area of European countries (UNEP, 2022) and where wind power is predominantly located in forest-dominated landscapes, there is an ambitious strategy to expand onshore wind power generation (Svensson et al., 2023) that could be anticipated to increase DOC fluxes. Other peatland areas in which onshore wind energy generating capacity is being developed are Patagonia (southern Chile and Argentina) (Cacciuttolo et al., 2024) and the Andes mountains (e.g. in Ecuador (López et al., 2023)). The Tibetan-Qinghai Plateau in China contains ~5000 km^2 peatland (Chen et al., 2014), half of the peatland area in China, and has been identified as having abundant wind resources (Zhuang et al., 2021), mostly yet to be developed.

Since we identified forest felling as the predominant activity within wind farm development giving rise to increased DOC concentration flux, our findings are also applicable to temperate, boreal and high altitude / latitude forest-dominated peatlands in which forest felling occurs. Examples of this ecological context are Ireland and Chile, and the boreal forests of Sweden, Finland, the Baltic countries, Russia, Alaska (USA) and Canada.

Our study methodology and findings are also relevant to the design of future investigations of how land use and management affect DOC concentrations and fluxes, enabling more focused sampling efforts. In this study the same trends in DOC concentration and flux were identified in all the sampled catchments affected by wind farm development, providing confidence for future studies that sampling only one catchment should be representative in identifying trends. This means that sampling and analysis resources could be reduced or redirected towards gaining a greater understanding of landscape disturbance effects on DOC composition as this may profoundly affect aquatic biogeochemistry, ecology and carbon cycling, and reduce the treatability of potable water supplies (Evans et al., 2024; Williamson et al., 2023b). Furthermore, our study demonstrated the utility of using first derivatives applied to the DOC flux estimated trend to independently identify change points, that could then be compared against the timing of wind farm development activities. This modelling approach can be applied to other long-term (≥ 10 years) stream or river water quality datasets to assess whether land use and management changes are associated with changing water quality.

Of course, there are improvements that can be made to future research projects seeking to understand how peatland carbon storage is affected by wind farm development. A more accurate understanding of the impact could come from the construction of a full C flux budget (see the Supplementary Material S5 for an overview of approaches to this). The deployment of [DOC] sensors would support the construction of more accurate DOC budgets (see for example Bass et al, 2023) and due to the large data sets generated offer an opportunity to model the controls on export over shorter time scales – important to project water quality deterioration for example. Analysis of DOC composition would allow more accurate assessment the impacts of increased [DOC] and DOC flux from wind farm development on aquatic ecology and biogeochemistry and on treatability of potable water (Evans et al., 2024). Remotely-sensed geospatial data could be used to explore inter-catchment differences and identify the best locations for sampling, or site sensor technology, and nested sampling design could aid understanding of whether there is a critical scale of disturbance in a given land area beyond which the impact is not buffered.

4. CONCLUSIONS

This research provides insight into the development of a landscape that has previously been used for more traditional ecosystem services such as potable water abstraction and forest products, but now must be considered as a critical energy landscape. Indeed currently there is a planning application for a solar array and green hydrogen production and storage facility next to the wind farm, still on peaty soils (ScottishPower, 2024). Analysis of our detailed observational measurements for a decade at five adjacent catchments sampled synchronously shows that the wind farm construction impact can be detected in catchment drainage waters several years after the wind farm has been constructed. The rate of DOC flux into catchment drainage has continued to increase and, in some catchments, does not show evidence of recovery. However, it appears that the loss of terrestrially-sequestered carbon that this increase represents is small. Therefore, unless the peaty soils are already heavily degraded and almost a net source of atmospheric CO_2, the increased DOC loss caused by wind farm development is unlikely to impact on ongoing carbon sequestration by catchment soils.

However, critical landscapes do not only provide one ecosystem service and we evidence this in considering the impact ‘browning of the water’ appears to have had on the treatment of potable water supplies draining Whitelee wind farm. Thus, at the point of landscape development the impact on other ecosystem services should be considered, as well as the carbon footprint associated with future remediation. It can be complex to address multiple priorities (and to mitigate the fall-out), but how a landscape is valued and priority uses identified is a key consideration for a sustainable future. Progress here would be supported by decision trees that can flex in response to agreed priorities, and by better forward planning where both may need to be underpinned by cost-benefit analysis where priority strategic needs are agreed at the outset.

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DATA AVAILABILITY

DOC concentration data for the Whitelee catchments are available within the PhD theses of Helen Murray, Antony Phin, Martin Coleman and Ying Zheng (in the reference list below). The full data set is hoped to be lodged in the NERC Environmental Information Data Centre as below.

DOC concentration data for the Black Burn are available within:

Dinsmore, K.J., Billett, M.F., Dyson, K.E. (2013). Five year record of aquatic carbon and greenhouse gas concentrations from Auchencorth Moss. NERC Environmental Information Data Centre. https://doi.org/10.5285/3f0820a7-a8c8-4dd7-a058-8db79ba9c7fe

Dinsmore, K.J., Murphey, O., Leith, F., Carfrae, J. (2016). Aquatic carbon and greenhouse gas concentrations in the Auchencorth Moss catchment following drain blocking. NERC Environmental Information Data Centre. https://doi.org/10.5285/88ffbf44-0ec0-41d6-9814-04bc3535cd84

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SUPPLEMENTARY MATERIAL

Identifying and understanding how critical landscapes for carbon sequestration respond to development for low carbon energy production: Insight to inform optimal land planning and management strategies

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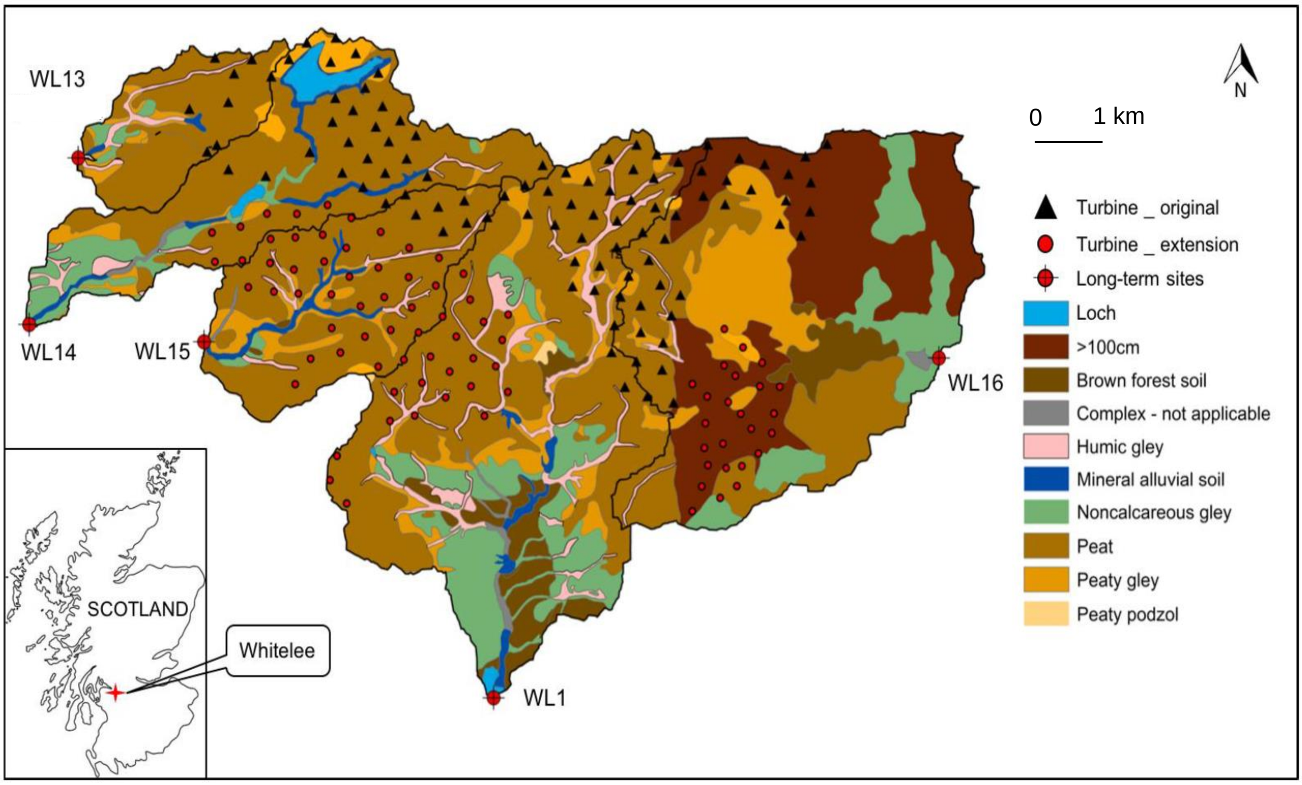


Figure S1. Whitelee catchments long-term streamwater sampling points (large red circles named WL*n*), catchment locations, wind farm development and soil types. Turbines located in the catchments are shown, constructed during the original wind farm development (phase 1: October 2006–May 2009) and the extension (phase 2: November 2010–February 2013), including in the West Browncastle and Calder Water Community wind farm developments in WL16 (August 2012–August 2014). Turbine locations from Digimap® Land Cover Map 2015. Catchment boundaries shown as black lines. Soil data courtesy of Macaulay Land Use Research Institute.

S1. Methods for collection and analysis of streamwater samples

Streamwater samples for carbon species analysis were collected in pre-cleaned 1 L Nalgene® bottles and caps (Whitelee catchments) / glass bottles (Black Burn). Gloves were worn during sampling and all sampling containers and tops were rinsed with streamwater before sample collection. Samples were stored in a cool box with ice packs until return to the laboratory, where they were stored in the dark at 4 °C until processing and analysis. A known sample volume (~0.5 L) was vacuum-filtered within 48 h of collection using pre-ashed Whatman® GF/F 0.7 μm filters for particulate organic carbon (POC) analysis (data not reported here). For quantification of dissolved organic carbon (DOC) in the Whitelee catchment samples from October 2007, a 60 mL aliquot of the filtrate was acidified to pH 3.9 using 0.05 M H_2SO_4 then placed in an ultrasonic bath to degas any dissolved CO_2, before analysis within 3–6 months using a Thermalox™ TOC analyser. The instrument was calibrated using a range of concentrations of the standard potassium hydrogen phthalate that spanned the field site concentration. Standards were also analysed during runs to check for instrument drift. From June 2006 to September 2007, DOC analysis was conducted as outlined in Waldron et al. (2009). Where carbonate was likely to be present in the sample, the filtrate was acidified to pH 4 with 0.1 M H_2SO_4.

It was then filtered through a GF/F filter, and reduced to a concentrate by rotary evaporation (at 50 °C and 50 mbar). The concentrate was subsequently freeze-dried to a powder. wt % C was assayed by analysis of ~2 mg of dried powder on a Costech C/N/S analyser, linked to a ThermoFinnigan continuous flow mass spectrometer (at the Scottish Universities Environmental Research Centre). From the volume of sample filtered, mass of solid residue and the wt % C [DOC] was calculated. Black Burn streamwater samples were analysed for DOC using a Rosemount-Dohrmann DC-80 total organic C analyser (for 2006–2007) and a PPM LABTOC analyser (for 2008 samples onwards), calibrated with potassium hydrogen phthalate standard solutions. All DOC analytical runs were accompanied by 2–3 procedural blanks, consisting of deionised water prepared in the same way as the samples. All concentrations are reported as blank corrected. For the Whitelee catchments, an additional field blank was measured on three occasions, where a 1 L Nalgene® bottle was filled with deionised water and taken to site, to account for any cross contamination that might have occurred through reusing bottles.

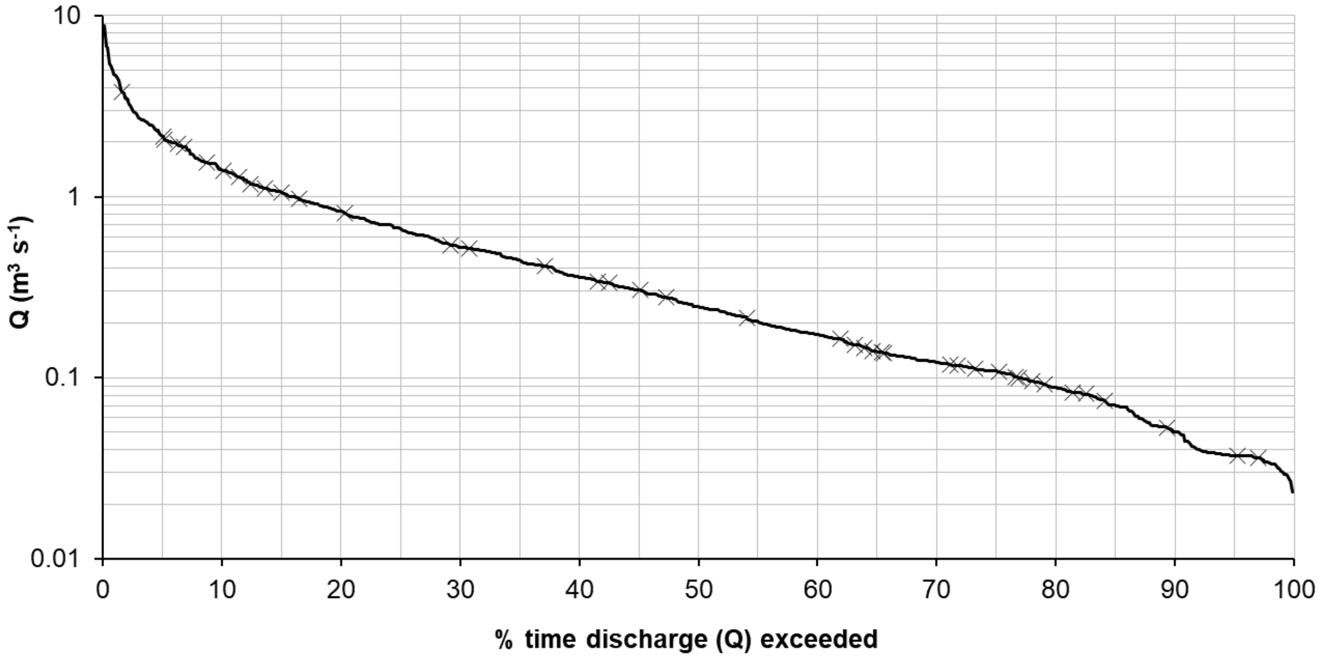


Figure S2. Flow duration curve of the mean daily flow on days of streamwater sampling (x) for WL15 for 1 October 2011–30 September 2013 showing that the streamwater sampling programme was successful in targeting a range of flows. y-axis is discharge (Q) on a $\log_{10}$ scale. (From Heal et al., 2020).

Table S1. Information on the method selected and its rationale to generate the long-term 15-minute flow estimates for each Whitelee catchment, 1 June 2006–29 September 2016. SEPA Newmilns Q and SEPA Q refers to flow measured at the Scottish Environment Protection Agency (SEPA) gauging station at Newmilns on the River Irvine (61 masl, catchment area 72.8 km^2, 6 km from the Whitelee catchments). Flow was quantified in the Whitelee catchments using an area-velocity flow logger (Isco 4150) that recorded water depth (via pressure transducer) and velocity (via Doppler ultrasound method) every 15 min. These were combined with cross-sectional area data to yield flow.

Catchment	Method selected	Equation to estimate flow (m^3 s^{-1})	R^2	Comments
WL13	Linear relationship measured flow vs. SEPA Newmilns Q for the period 5 Nov 2013–7 Jan 2015, from Coleman (2017)	WL13 Q = (SEPA Q x 0.0745) + 0.0321	0.788	Less over-estimation of high flows at WL13 compared to other relationships investigated with SEPA Newmilns flow ($\log_{10}$, semi-log, sqrt) and catchment area scaling method.
WL14	Linear relationship measured flow vs. SEPA Newmilns Q for the period 10 Sep 2010–5 Oct 2010, from Murray (2012)	WL14 Q = (SEPA Q x 0.1007) + 0.1102	0.782	Yields most conservative estimates of mean and high flows at WL14 compared to $\log_{10}$ relationship investigated with SEPA Newmilns flow and catchment area scaling method.
WL15	Catchment area scaling[1]	WL15 Q = SEPA Q x (11.3/72.8)	0.766[2]	Yields most conservative estimates of mean, median and high flows at WL15 compared to relationships investigated with SEPA Newmilns flow (linear, $\log_{10}$, sqrt).
WL1	Catchment area scaling[1]	WL1 Q = SEPA Q x (31.1/72.8)	---	No continuous flow measurements. Catchment scaling method is only option.
WL16	Catchment area scaling[1]	WL16 Q = SEPA Q x (30.5/72.8)	0.888[3]	Yields most conservative estimates of mean and median flows at WL16 compared to the strongest (linear) relationship investigated with SEPA Newmilns flow.

[1] catchment areas from Zheng (2018)
[2] from comparison with measured flow at WL15 (Phin, 2016)
[3] from comparison with measured flow at WL16 (Murray, 2012)

S2. Details of the statistical modelling methods

In this paper, we analysed the temporal variability of [DOC] and DOC flux (y) in the disturbed catchments (the Whitelee catchments) and the regional control catchment (Black Burn) using additive modelling (Friedman and Stuetzle, 1981). An Additive Model is a flexible regression approach that allows estimates of the trend and seasonality in the variable of interest as smooth functions of time. The Generalised Additive Modelling (GAM) framework was chosen to avoid making specific parametric assumptions about the shape of any relationships, which are not known a priori, between the [DOC] and DOC flux variables and the corresponding covariates. This modelling framework involves smoothing and does not require making any specific distributional assumptions about the modelled variable such as normality. The choice of the smoothing level of the modelled relationships is a trade-off between the bias and the variance, typically assessed using cross-validation (Wood, 2006). In the presence of autocorrelation in the model residuals, a typical characteristic of time series data, the GAM framework was extended to a Generalised Additive Mixed Model (GAMM). In this type of model, the errors can be divided into a structured component modelled using a time series model and a non-structured random component. Here, after accommodating for the trend and seasonal components, we used an auto-regressive (AR) model for the structured component of the errors. Verification of the temporal correlation structure was conducted using empirical autocorrelation function (ACF) plots. In the modelling framework, to co-locate the [DOC] and flow measurements in time, a 21-day moving window approach was used. This provides a smooth curve of flow from which we can identify/interpolate flow when DOC is measured. The choice of the window length is analogous to the choice of smoothing in the GAM, and was similarly evaluated in a sensitivity study. Since the goal of the modelling was description, rather than prediction, this approach provides a robust estimate, which balances bias and variance.

The five Whitelee catchments were modelled altogether, and then a separate model, but with the same structure, was applied to the Black Burn catchment.

The smooth trend and seasonality effects in the five Whitelee catchments were modelled using the following additive model:

$$y_{ij} = \beta_0 + \beta_j \, \mathbb{I}_{ij} + f_{1j}(x_{1i}) \, \mathbb{I}_{ij} + f_{2j}(x_{2i}) \, \mathbb{I}_{ij} + \epsilon_{ij} \quad \text{for } i = 1, \dots, n_j \text{ and } j = 1, \dots, 5, \qquad \text{(Eq. S1)}$$

where $\mathbb{I}_{ij} = \begin{cases} 1 & i \in catchment\ j \\ 0 & otherwise \end{cases}$

and x_1 and x_2 are the time on the continuous scale and the day of year, respectively. The functions f_{1j} are non-linear smooth functions of the trend component of the different catchments, which were modelled using cubic B-splines. The functions f_{2j} are non-linear smooth functions of the seasonal component over the day of year at the same catchments. The seasonal smooths were modelled using cyclical cubic splines as there should be no discontinuity between January and December.

Each of the separate smooths in the above model is subject to identifiability constraints, which effectively centres each smooth around zero effect. As such, differences in the mean DOC concentrations or fluxes of the catchments are not accounted for by the smooths. To rectify this, we added to the model the catchment indicator variables $\mathbb{I}_j$ as parametric terms along with the smooths. Each of those indicator variables takes 1 if the i^{th} observation falls in catchment j and 0 otherwise. For model identifiability, the constraint $P_{jj} = 0$ ($\sum_j \beta_j = 0$) was imposed.

To adjust for the autocorrelation in the time series, the errors $\in_{ij}$ within each catchment, in Eq. S1, are assumed to follow an Auto-Regressive Moving Average ARMA(p,q) process (Box et al., 1994):

$$\in_{ij} = \varphi_{1j} \in_{i-1,j} + \cdots + \varphi_{pj} \in_{i-p,j} + \theta_{1j} v_{i-1,j} + \cdots \theta_{qj} v_{i-q,j} + v_{ij}, \qquad \text{(Eq. S2)}$$
$v_{ij} \sim N(0, \sigma^2)$ for $i = 1, \ldots, n_j$ and $j = 1, \ldots, 5$,

where φ_{1j},…,φ_{pj} are the auto-regressive model parameters and ϑ_{1j},…,θ_{qj} are the moving average model parameters, at catchment j. p and q are non-negative integers that determine the order of the ARMA process, determined by the examination of the autocorrelation and partial autocorrelation functions of the residuals resulting from fitting the same model above but assuming white noise errors, i.e. $\epsilon_{ij} \sim N(0, \sigma^2)$.

The model above, given by Eq. S1 and Eq. S2, was fitted using the function gamm for generalised additive mixed models in the mgcv package in R 3.6.3 (Wood, 2012). The flexibility of the smooth terms in the above additive model is determined by the maximum degree of freedom specified for each smooth. The degrees of freedom control the complexity of the smooth term; in the sense that large degrees of freedom imply a more complex smooth term. To ensure valid comparisons, the degrees of freedom for the trend component and that for the seasonal component were held the same across the different catchments. The significance of each term in the model was assessed using the Wald test (Wood, 2006). A term with a p-value greater than 0.05 is statistically insignificant and can be removed from the model.

We decided to model the [DOC] over time for all Whitelee catchments collectively in one Analysis of Covariance (ANCOVA) model to allow for comparisons across sites. We allowed the trend and the seasonal component to vary from one site to another. Each of the separate trend and seasonal component smooths is subject to identifiability constraints, which effectively centres each smooth around zero effect. As such, differences in the mean [DOC] of the sites is not accounted for by the smooths. To rectify this effect, the site was added as a parametric term to the model, along with the smooths. The model can thus be expressed as follows:

$$DOC_i = \beta_0 + \beta_1 \mathbb{1}_{14} + \beta_2 \mathbb{1}_{15} + \beta_3 \mathbb{1}_{1} + \beta_4 \mathbb{1}_{16} + \\ f(dec.date_i)\, \mathbb{1}_{13} + f(dec.date_i)\, \mathbb{1}_{14} + f(dec.date_i)\, \mathbb{1}_{15} + f(dec.date_i)\, \mathbb{1}_{1} + f(dec.date_i)\, \mathbb{1}_{16} + \\ f(doy_i)\, \mathbb{1}_{13} + f(doy_i)\, \mathbb{1}_{14} + f(doy_i)\, \mathbb{1}_{15} + f(doy_i)\, \mathbb{1}_{1} + f(doy_i)\, \mathbb{1}_{16}, \qquad \text{(Eq. S3)}$$

where $\mathbb{1}_S = \begin{cases} 1 & Site\ = S \\ 0 & otherwise \end{cases}$

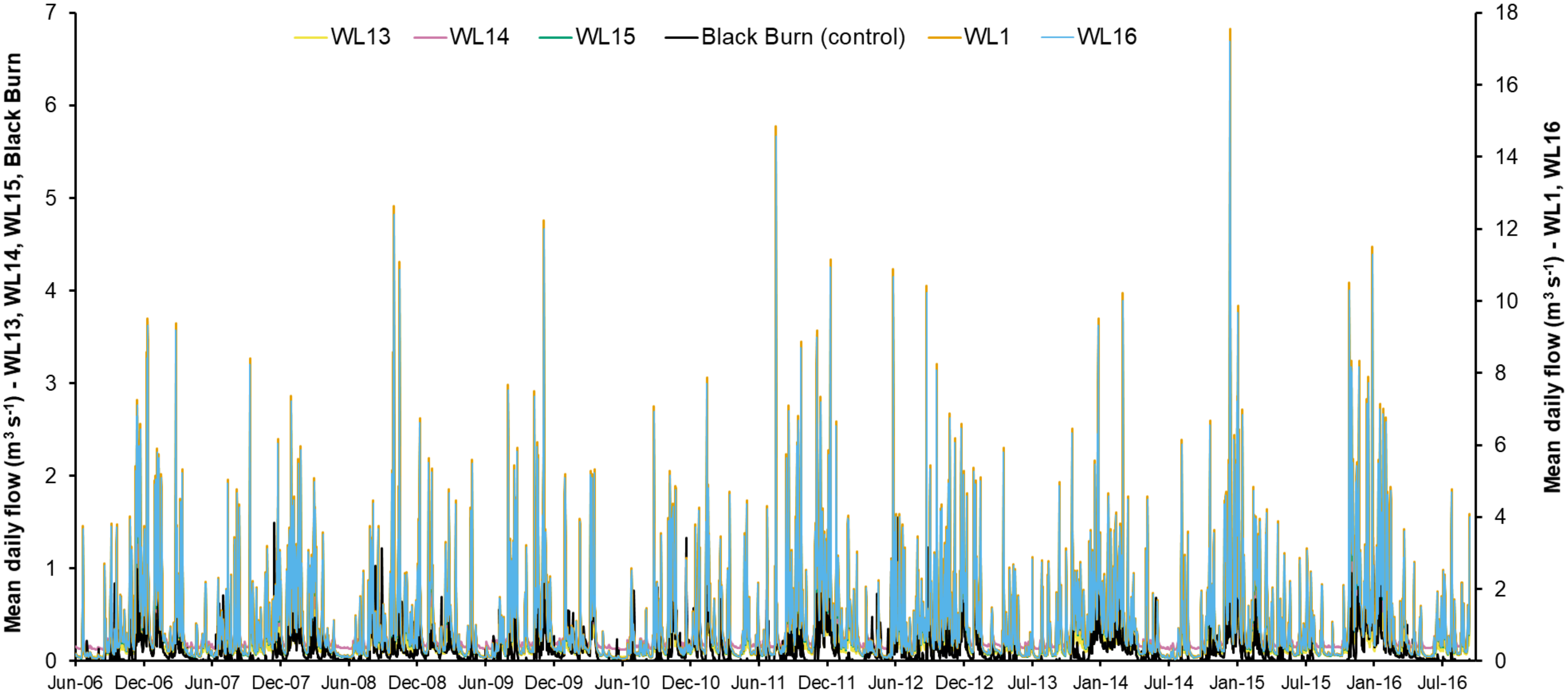
In summary, for [DOC] modelling we:

1. Constructed a preliminary additive model with different intercept and smooth terms for trend, seasonality, and their interaction for each Whitelee catchment (ANCOVA model), fitted to assess differences in the DOC across the Whitelee catchments.
2. Incorporated an ARMA(2,2) autocorrelation structure into the additive model in (1) to adjust for autocorrelation and fitted the model under an additive mixed model framework.
3. Also fitted for the Black Burn catchment an additive model as in (1) with smooth terms for trend, seasonality and interaction between them and the additive mixed model in (2) adjusting for remaining autocorrelation.

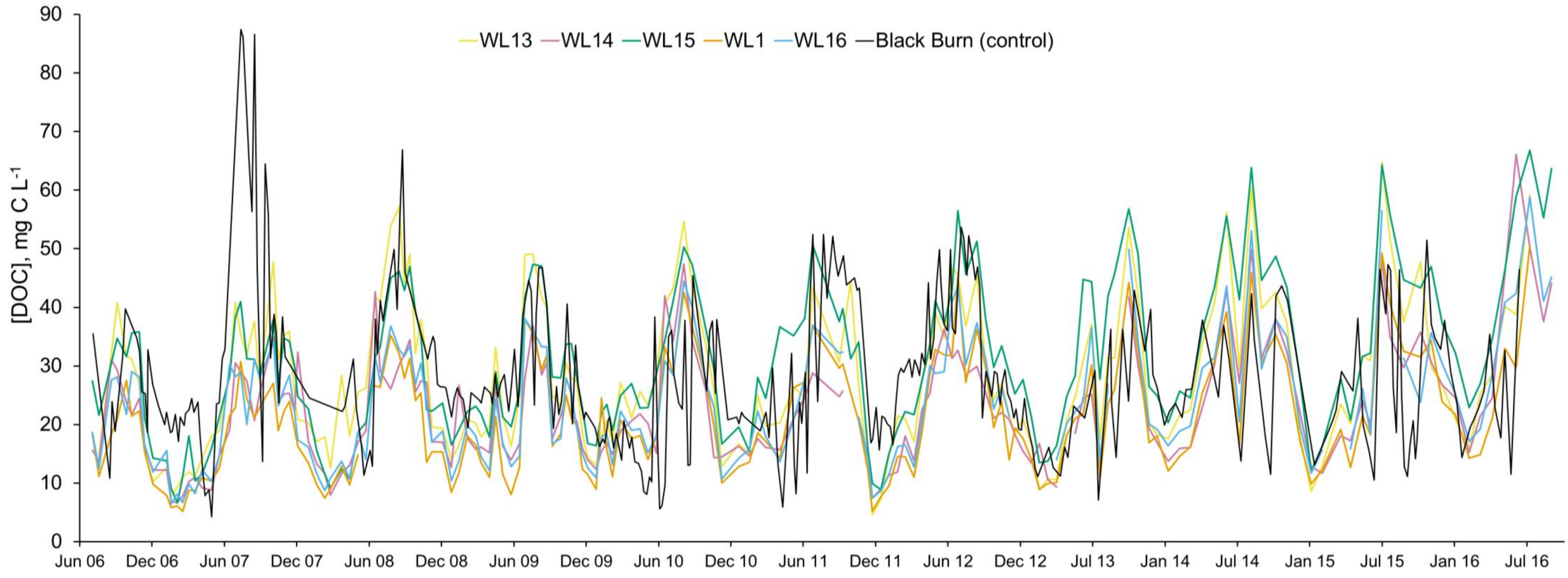
For modelling DOC flux, we included flow as a covariate to tease apart the additional influences of flow and other controls on DOC flux, and:

1. For each catchment, a moving average of DOC flux and flow (both normalised by catchment area) using a 21-day sliding window were obtained then one observation was sampled every 21 days, reflecting the frequency of streamwater sampling for [DOC] analysis.
2. An additive mixed (ANCOVA) model was fitted that allows for each Whitelee catchment different intercept; smooth terms for trend and seasonality; interaction between trend and seasonality; and also a different linear relationship with flow. In addition, a second order Moving Average correlation structure (MA(2)) was used to account for the correlation in the errors within each catchment.
3. The model in (2) with smooth terms for trend and seasonality, interaction between trend and seasonality, a linear relationship with flow, and an MA(2) correlation structure for the errors was fitted to the Black Burn catchment separately. (A first order Moving Average correlation structure (MA(1)) was enough here, but for comparison purposes we used MA(2) as for the Whitelee catchments).
4. The first derivatives of the estimated trend from models (2) and (3) were investigated to assess the causes of the changing rate of DOC flux over time.

(a)


WL13
WL14
WL15
Black Burn (control)
WL1
WL16
Mean daily flow (m³ s⁻¹) - WL13, WL14, WL15, Black Burn
Mean daily flow (m³ s⁻¹) - WL1, WL16
0
1
2
3
4
5
6
7
0
2
4
6
8
10
12
14
16
18
Jun-06
Dec-06
Jun-07
Dec-07
Jun-08
Dec-08
Jun-09
Dec-09
Jun-10
Dec-10
Jun-11
Dec-11
Jun-12
Dec-12
Jul-13
Jan-14
Jul-14
Jan-15
Jul-15
Jan-16
Jul-16

(b)



WL13
WL14
WL15
WL1
WL16
Black Burn (control)
[DOC], mg C L-1
0
10
20
30
40
50
60
70
80
90
Jun 06
Dec 06
Jun 07
Dec 07
Jun 08
Dec 08
Jun 09
Dec 09
Jun 10
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Jun 11
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Jul 16

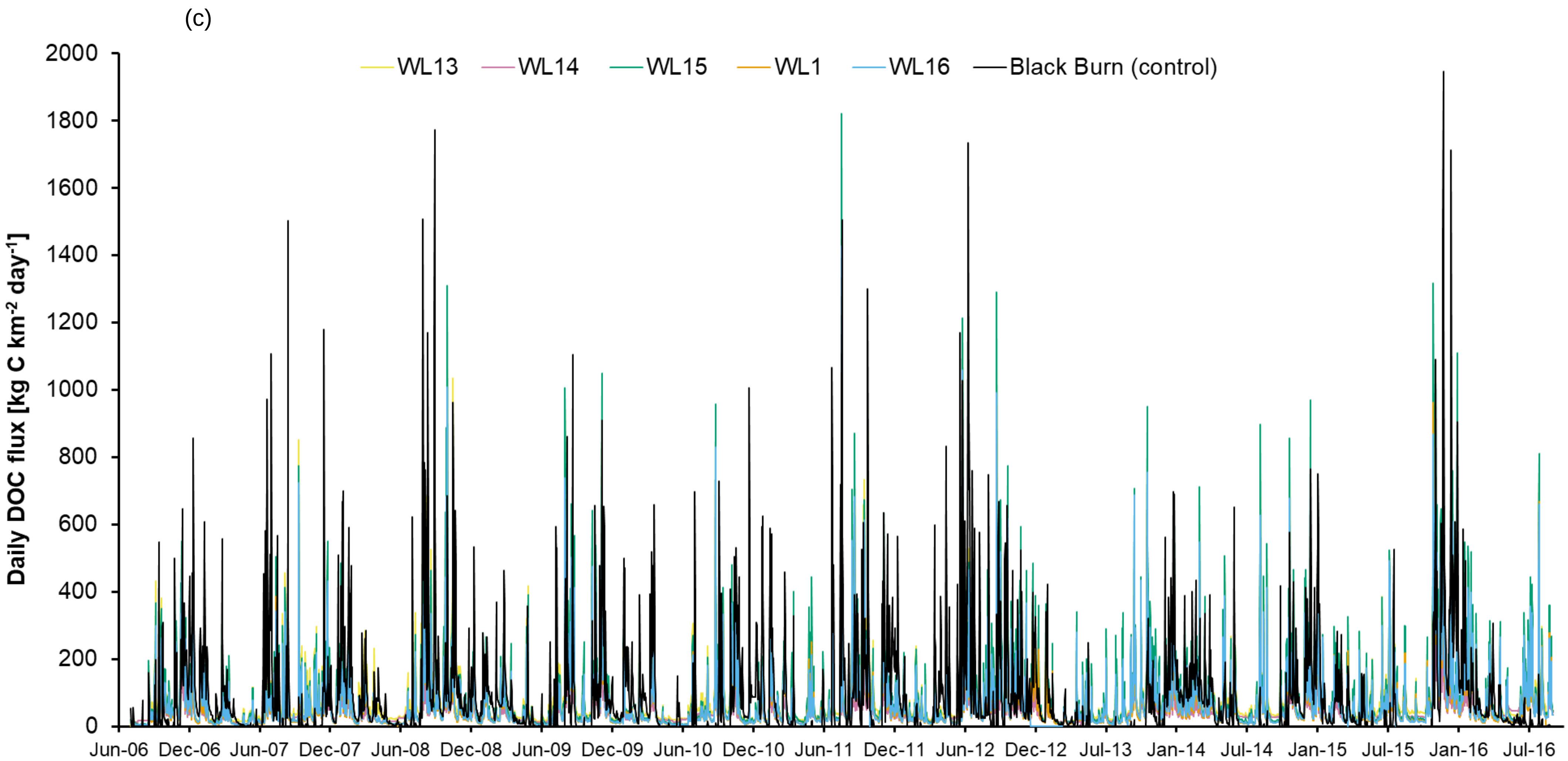

Figure S3. Time series for all catchments June 2006–September 2016. (a) Mean daily estimated flow. Note different y-axis scales: mean daily flow values for WL13, WL14, WL15 and Black Burn are on the left-hand y-axis and those for WL1 and WL16 are on the right-hand y-axis. (b) Measured [DOC]. There are no values for WL16 17 December 2012–12 March 2013 and for Black Burn 29 June–29 September 2016 due to gaps in streamwater sampling. (c) Mean daily DOC flux normalised by catchment area.

S3. DOC flux seasonal effects

Figure S4 shows the fitted DOC flux seasonality for each catchment averaged across the study period, with the timings of the annual DOC flux minimum and maximum highlighted. The seasonality of DOC flux differs between the Whitelee catchments and the Black Burn (control catchment), apart from at WL14, where there is buffering from Lochgoin and Craigendunton reservoirs. The other Whitelee catchments have a greater amplitude of DOC flux and more asymmetry, compared to the Black Burn. There is also a steeper and shorter time duration of the decrease in DOC flux from the annual maximum to minimum values in the Whitelee catchments.

The greater seasonal amplitude of DOC flux in the Whitelee catchments may be attributed to enhanced soil DOC content and mobilisation resulting from conifer afforestation of large proportions of the Whitelee catchments in the 1960s–70s, since other studies have reported positive associations between DOC flux and catchment conifer cover (e.g. Williamson et al., 2023). Furthermore, wind farm development may have increased DOC production in the Whitelee catchments compared to Black Burn due to disturbance creating a larger DOC supply (e.g. from disturbed peat and brash from forest harvesting (Howson et al., 2021)) and enhancing microbial production (e.g. through greater nutrient availability from brash (Kreutzweiser et al., 2008) and warmer, drier conditions in harvested plantation soils (Schelker et al., 2013)). All these factors could enhance the seasonal amplitude of DOC flux in the Whitelee catchments compared to the unafforested Black Burn. Catchment differences in DOC flux seasonality are unlikely to be caused by any differences in DOC release related to spring snowmelt (e.g. as reported for boreal catchments in northern Sweden (Laudon et al., 2004)) since prolonged winter snow cover was rare in the study catchments during the 2006–2016 study period. Any differences in winter freeze-thaw cycles between our upland study catchments are also unlikely to affect DOC flux seasonality since freeze-thaw fluctuations have been shown to have only a minor influence on soil carbon dynamics and DOC losses (Grogan et al., 2004; Matzner and Borken, 2008).

The steeper decline in DOC flux from annual maximum to minimum values in the Whitelee catchments could be attributed to either higher annual rainfall in these more westerly catchments (1413 mm vs. 1018 mm mean annual rainfall, see study catchment descriptions in sections 2.1 and 2.2) – although greater interception by forest cover in the Whitelee catchments could mean that the soil is not wetter – and/or more better hydrological connectivity due to forest ditching and access tracks (Kuglerová et al., 2021) compared to the minimally-disturbed Black Burn catchment.

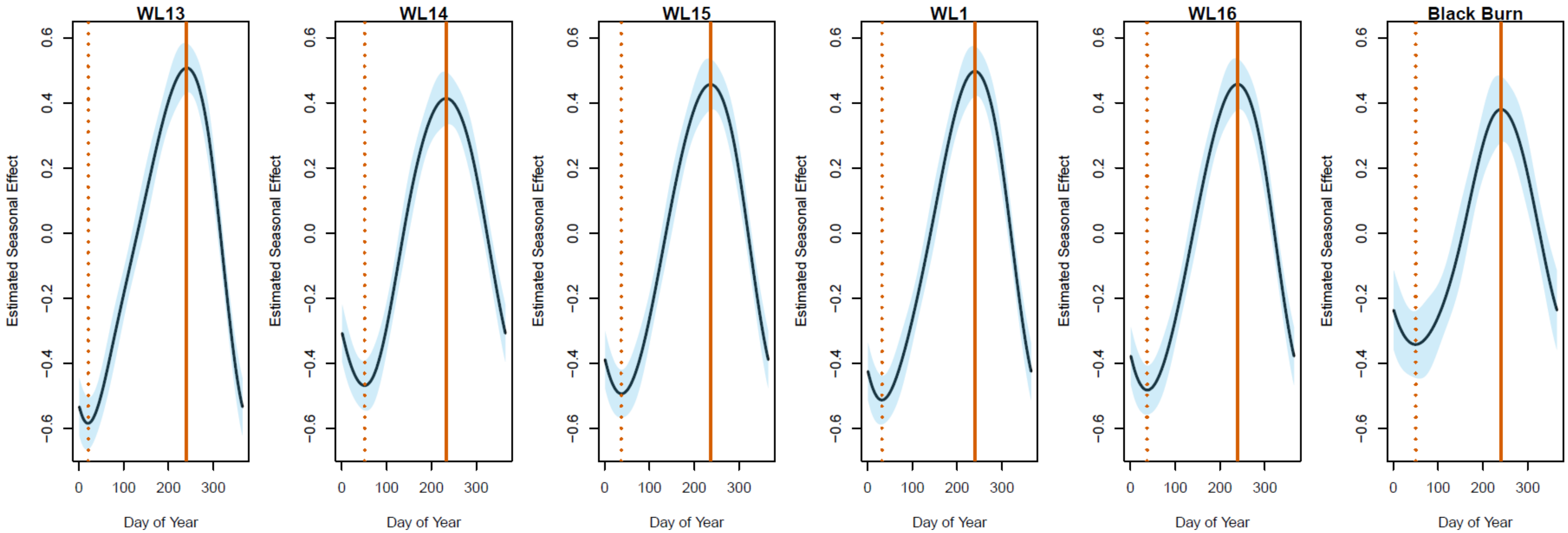

Figure S4. Estimated smooth curves of the seasonality component of DOC flux for each catchment during the study period, with flow modelled as a covariate. The vertical orange-coloured dotted and solid lines show the timing of the annual DOC flux minimum and maximum, respectively. The blue-shaded areas represent an approximate pointwise 95% confidence band.

S4. Estimation of the effect of wind farm development on DOC flux from the Whitelee catchments during the study period

Our calculations of the mass of carbon that must be sequestered every year to compensate for additional DOC loss due to wind farm development uses three approaches for estimation. We compared the first year of sampling to the final year (2015–16) but also to the years leading up to this time (2007–16) and the period of time that excludes the final year (2007-15). We do this as it is possible that the increasing trend in DOC flux in the final year (2015–16) is a regional trend, as this is also apparent at Black Burn. Therefore, it may not be appropriate to solely compare with the final year to estimate change.

The percentage increase in DOC flux for a given catchment compared to 2006–2007 for each of these three end points was calculated. The percentage increase in the Whitelee catchments was corrected for a regional influence by subtracting the % increased flux observed in Black Burn in the three observational periods to give a percent change that is solely due to the wind farm. Thereafter, that percent change was converted to a mass flux for the observational period and a correction made for the 9-year period to give an estimate of annual change in DOC flux due to wind farm development (Table S2).

In Table S2 the following calculation notes apply:

1. Black Burn 2015–16 % change subtracted from WL % change,
2. Black Burn 2007–16 % change subtracted from WL % change,
3. Black Burn 2007–15 % change subtracted from WL % change,
4. Calculated by multiplying 2006–07 flux by % change due to wind farm, 2006–07 compared to 2015–16,
5. Calculated by multiplying 2006–07 flux by % change due to wind farm, 2006–07 compared to mean 2007–16,
6. Calculated by multiplying 2006–07 flux by % change due to wind farm, 2006–07 compared to mean 2007–15.

Table S2. Summary of the workings to calculate the % change in annual DOC flux per unit area for the Whitelee catchments attributable to the wind farm development through comparison with the Black Burn (regional control catchment) for the three different endpoints. The baseline year is 2006–07. See text in S4 for more explanation.

Annual DOC flux (g C m^{-2} $year^{-1}$), 4 July–3 July each year
(corrected for missing days and leap years)

Year	WL13	WL14	WL15	WL1	WL16	Black Burn
2006–07	22.1	12.0	22.4	14.6	18.2	30.4
2007–08	31.6	15.5	25.1	18.0	20.5	28.5
2008–09	35.6	16.7	31.1	21.0	24.3	42.3
2009–10	31.7	15.5	30.4	21.5	22.8	32.2
2010–11	28.2	14.9	28.2	19.8	20.7	27.4
2011–12	37.6	18.0	37.9	28.0	30.5	48.1
2012–13	34.0	16.8	37.9	26.2	30.7	35.7
2013–14	37.4	18.8	39.1	25.8	30.5	27.3
2014–15	33.5	19.2	35.5	24.4	27.0	23.9
2015–16	42.2	24.7	49.0	33.4	37.6	41.5
Mean 2007–16	**34.6**	**17.8**	**34.9**	**24.2**	**27.2**	**34.1**
Mean 2007–15	**33.7**	**16.9**	**33.2**	**23.1**	**25.9**	**33.2**

% Change in annual DOC flux due to wind farm development

	WL13	WL14	WL15	WL1	WL16	Black Burn	***Calculation notes***
% change 2006–07 compared to 2015–16	91	106	119	129	107	36	
% change 2006–07 compared to mean 2007–16	57	48	56	66	49	12	
% change 2006–07 compared to mean 2007–15	53	41	48	58	42	9	
% change due to wind farm, 2006–07 compared to 2015–16	55	70	83	92	70	--	***1.***
% change due to wind farm, 2006–07 compared to mean 2007–16	45	36	44	54	37	--	***2.***
% change due to wind farm, 2006–07 compared to mean 2007–15	43	32	39	49	33	--	***3.***
Mean	**48**	**46**	**55**	**65**	**47**	**--**	
St dev	6	21	24	24	20	--	

Additional DOC flux over 9 years due to wind farm development (g C m^{-2} 9 $years^{-1}$)

	WL13	WL14	WL15	WL1	WL16	Black Burn	***Calculation notes***
Additional DOC flux, 2006–07 compared to 2015–16	12.08	8.38	18.49	13.47	12.77	--	***4.***
Additional DOC flux, 2006–07 compared to mean 2007–16	9.86	4.33	9.81	7.85	6.77	--	***5.***
Additional DOC flux, 2006–07 compared to mean 2007–15	9.58	3.83	8.73	7.14	6.02	--	***6.***
Mean	**10.51**	**5.51**	**12.34**	**9.49**	**8.52**	**--**	
St dev	1.37	2.49	5.35	3.47	3.70	--	

Additional DOC flux per year due to wind farm development (g C m^{-2} $year^{-1}$)

	WL13	WL14	WL15	WL1	WL16	Black Burn	**Mean**	**St Dev**
Additional DOC flux per year, 2006–07 compared to 2015–16	**1.34**	**0.93**	**2.05**	**1.50**	**1.42**	--	**1.45**	**0.40**
Additional DOC flux per year, 2006–07 compared to mean 2007–16	**1.10**	**0.48**	**1.09**	**0.87**	**0.75**	--	**0.86**	**0.26**
Additional DOC flux per year, 2006–07 compared to mean 2007–15	**1.06**	**0.43**	**0.97**	**0.79**	**0.67**	--	**0.78**	**0.25**
Mean	1.17	0.61	1.37	1.05	0.95	--	**1.03**	**0.28**
St dev	0.15	0.28	0.59	0.39	0.41	--		

S5. Different approaches to quantify a fuller C budget

Here we have used DOC as a measure of C loss from catchment soils due to the importance of the flux strength, and the simplicity with which data can be collected over long periods of time. For a more complete C budget, the following methods could be drawn on to quantify net CO_2 and CH_4 greenhouse gas fluxes. Static flux chambers could be installed to measure directly gas evasion in representative areas, for example from representative plant functional types (e.g. Armstrong et al., 2015), or from different ecotopes (e.g. Jordan et al., 2016). Such chambers can also be used on open water systems within peatlands (e.g. Männistö et al., 2019). Advances in technology using automated suspended and moveable chambers could extend the temporal and spatial range of data collected (e.g. Keane et al., 2021) to improve accuracy of modelling. If boundary layer dynamics can be modelled, there may be scope for eddy covariance studies in wind farms. Choosing representative topography and habitat for upscaling could focus on understanding of the important of different habitats, drawing on geospatial environmental data, some collected remotely by satellite or UAV, which can may be used in machine learning modelling to predict greenhouse gas fluxes (e.g. Christiani et al., 2024).

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