



# Net carbon dioxide emissions from an eroding Atlantic blanket bog

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**Abstract** The net impact of greenhouse gas emissions from degraded peatland environments on national Inventories and subsequent mitigation of such emissions has only been seriously considered within the last decade. Data on greenhouse gas emissions from special cases of peatland degradation, such as eroding peatlands, are particularly scarce. Here, we report the first eddy covariance-based monitoring of carbon dioxide (CO<sub>2</sub>) emissions from an eroding Atlantic blanket bog. The CO<sub>2</sub> budget across the period July 2018–November 2019 was 147 (±9) g C m<sup>-2</sup>. For an annual budget that contained proportionally more of the extreme 2018 drought and heat wave, cumulative CO<sub>2</sub> emissions were nearly double (191 g

C m<sup>-2</sup>) of that of an annual period without drought (106 g C m<sup>-2</sup>), suggesting that direct CO<sub>2</sub> emissions from eroded peatlands are at risk of increasing with projected changes in temperatures and precipitation due to global climate change. The results of this study are consistent with chamber-based and modelling studies that suggest degraded blanket bogs to be a net source of CO<sub>2</sub> to the atmosphere, and provide baseline data against which to assess future peatland restoration efforts in this region.

**Keywords** Peatland · Blanket bog · Peat erosion · Carbon dioxide · Net ecosystem exchange · Restoration · Eddy covariance

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## Introduction

Peatlands are the world's most effective terrestrial carbon store, with around 600 Gt (15–30% of the world's soil carbon) stored, despite such ecosystems only occupying around 3% of the terrestrial land area (Limpens et al. 2008; Yu 2012; Leifeld and Menichetti 2018). Peatlands in their intact states that are still located within suitable bioclimatic envelopes accumulate and store carbon because photosynthetic uptake of C exceeds the losses via respiration and anaerobic decomposition (e.g., Waddington et al. 2015). Degradation by direct human disturbance, such as drainage, grazing and land use conversion, as well as the influence of climate change, can switch

peatlands from net sink to net sources of C. Globally, peatlands turned from a net carbon sink to a net source around the 1960s, due to extensive drainage and land use conversion, particularly in Europe and South-East Asia (Leifeld et al. 2019). Expert assessments compiled by Loisel et al. (2021) assert that the combined effects of land use and climate change-driven changes in global temperature, precipitation patterns and fire risk will result in even stronger impacts on peatland carbon stocks in the near (to 2100) and far (2100–2300) future. Cumulative emissions from drained peatlands are estimated to have reached  $80 \pm 20$  Pg CO<sub>2</sub>e in 2015 (Leifeld et al. 2019), with further increases due to continued drainage of peatlands in many areas of the world. This may lead to total peatland emissions of up to  $249 \pm 38$  Pg by 2100 (Leifeld et al. 2019). Moreover, global warming is likely to reach 1.5 °C between 2030 and 2052 if it continues to increase at the current rate (IPCC 2018) which, together with projected changes to precipitation patterns (e.g., Giorgi et al. 2019), make the fate of this vast global peatland C stock highly uncertain.

Peatland restoration has been recognised as a cost-effective way to mitigate CO<sub>2</sub> emissions alongside improving the intrinsic value of these threatened habitats (Paustian et al. 2016). Leifeld and Menichetti (2018) used the 2013 IPCC Wetland Supplement default (Tier 1) emission factors combined with a bespoke global mapping effort of the various IPCC land use categories on peat soils to arrive at an estimate of 0.31–3.38 Gt CO<sub>2</sub>-equivalent of emissions per annum from the world's peatlands, with much of it stemming from CO<sub>2</sub>. Further refinement of this estimate is clearly required, particularly for the less intensively managed peatland categories where emission factors carry very high uncertainties. Peat erosion, in particular, is difficult to allocate to a specific IPCC land use category and is often excluded from broad land cover mapping efforts. Peat erosion occurs around the world (e.g., Yunker et al. 1991; Wilson et al. 1993; Pastukhov and Kaverin 2016; Selkirk and Saffigna 2018), although the literature on erosion rates and resulting C losses is heavily dominated by studies from the UK and Ireland (e.g., Bradshaw and McGee 1988; Birnie 1993; Carling et al. 1997; Bragg and Tallis 2001; Evans and Warburton 2007; Tomlinson 2009; Evans and Lindsay, 2010). Consequently, the scale of peat erosion across the world has not been quantified, and even estimates for the

UK and Ireland are frequently reliant on outdated and spatially poorly resolved data sources. An estimate of the proportion of eroded peat for the UK suggested a figure of around 11% (3283 km<sup>2</sup>) of the total area to be affected by significant erosion, with the proportion for Scotland likely to be around 14% (2732 km<sup>2</sup>; Evans et al. 2017).

The cause of peat erosion is not always clear. While erosive processes in peatlands are driven by the action of wind and rain (sometimes combined with direct initiation through peat instability), human impacts via fire management, overgrazing, drainage can initiate erosion (Parry et al. 2014). In mountain areas, tree removal on the lower slopes of a catchment can also be a contributing factor in initiating erosion in the peatlands in the upper part. There is considerable discussion amongst the scientific community as to the degree to which historic human land use has been the direct cause of erosion with evidence of impacts ranging from a view that initiation largely predates major anthropogenic impacts, although some effects of early deforestation may have contributed (e.g., Ellis and Tallis 2001) to alignment with increased human activities from the mid-to late 1700s (Stevenson et al. 1990, Tallis, 1998).

Peat erosion contributes to direct losses of carbon via losses of particulate matter. In addition, erosion gullies divert water and thus constitute a form of drainage that may affect plant photosynthetic responses and soil respiration. While direct C losses from eroding peatlands through particulate and dissolved organic carbon in fluvial and windborne erosion paths have understandably received much scientific attention, less is known about the direct gaseous emissions from these carbon dense ecosystems. Existing data on GHG emissions from eroded peatlands stem entirely from chamber-based studies (McNamara et al. 2008; Clay et al. 2011; Worrall et al. 2011; Dixon et al. 2014; Gatis et al. 2019), which require careful inclusion of the many different components of erosion features in the landscape and upscaling. Without robust understanding of the current emissions from eroded peatland ecosystems, the cost effectiveness of restoration work through gully blocking, reprofiling and re-vegetation of bare areas cannot be determined, and neither can overall GHG abatement potential be estimated from such activities.

Here, for the first time, CO<sub>2</sub> fluxes measured using the eddy covariance (EC) technique are presented for

an eroding oceanic blanket bog located in the Cairngorms, Scotland. Our objectives for this study were to: (1) quantify the magnitude of the net ecosystem exchange (NEE) of carbon dioxide (CO<sub>2</sub>); (2) explore effects of short-term variation in local climate on CO<sub>2</sub> fluxes; and (3) set the quantitative baseline against which the abatement potential of future restoration management can be evaluated.

## Methods

### Site location, description, and climate

The study took place on a large high-altitude plateau blanket bog in the eastern part of the Cairngorms National Park, Scotland, UK (56.93° N, −3.16° E, 642 m asl). The climate at the site is maritime temperate montane and is located on the edge of Köppen-Geiger class Dfc (cold, no dry season, cold summer, as per Beck et al. 2018). The nearest long-term UK Met Office station is Braemar (57.006 N, −3.396° E, 327 m amsl) which is 10 km NE of the site. Braemar station has a mean annual (1981–2010) maximum and minimum air temperature and precipitation of 10.7 °C, 3 °C and 932 mm year<sup>−1</sup>, respectively. The long-term mean annual temperature (1981–2010) was 6.73 °C, with observations showing a significant warming trend over the 1959–2020 period. The wider Cairngorms area experiences significant snowfall, and between 30 to more than 60 snow days each year, although a significant decline in snow days and snow depth has been observed over recent decades (Rivington and Spencer 2020), especially at higher altitudes. Snowfall in the region in recent years has been generally sporadic, rather than providing a single snow-covered period. It is not possible to provide estimates at the site prior to the monitoring period, but they are likely to fall on the lower end of the range. Peat erosion is widespread (Fig. 1, SI Fig. 1) and bare peat gullies vary between 0.5 and 3 m in their depth and 2 to > 10 m in their width. The surface area occupied by bare peat in erosion features was mapped using high resolution aerial photography (25 cm resolution, Get-Mapping) dated 2019 and further by high resolution LiDAR (12 cm horizontal, 3 cm vertical) in November 2020. Ground control locations of vegetated and bare peat areas exceeding a minimum area of 5 m<sup>2</sup> were collected with a handheld GPS (Garmin Etrex,

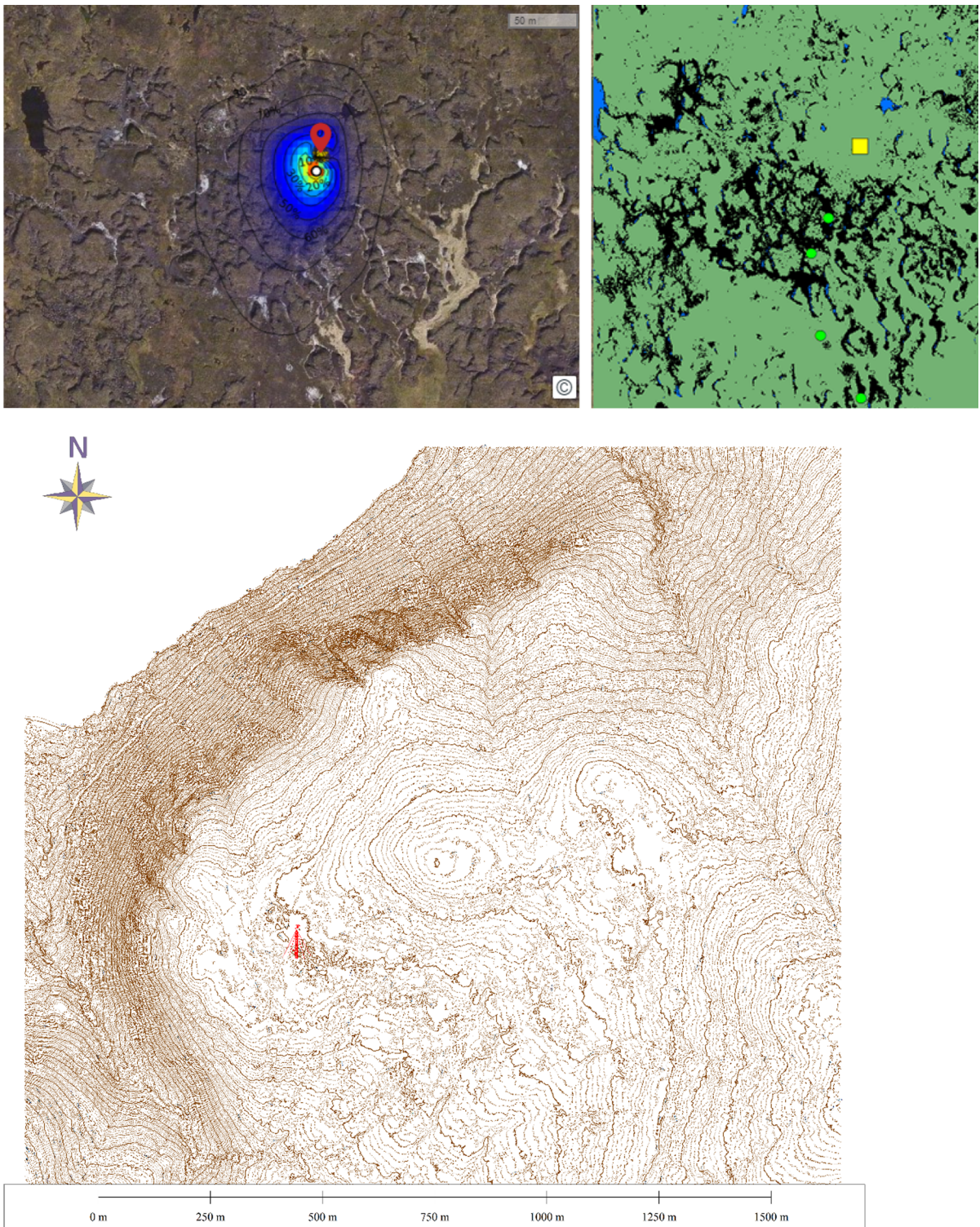
locational accuracy 3 m). Supervised classification analysis, using the ground control point locations, was carried out using the semi-automated classification plug-in (Congedo, 2016) in QGIS version 2.18.4.

Peat depth ranges from <0.2 to >2 m across a neighbouring area in the catchment similar in slope and aspect to the flux tower footprint, based on a 100 m grid-based survey (Peatland Action data, S. Corcoran, Cairngorms National Park Authority, pers. Comm.) which sampled gullies as well as vegetated peat. It is not possible to estimate average peat depth in the gullies versus vegetated areas from these data as the data did not include attributes to differentiate these terrain types and the positional accuracy of the GPS used is insufficient to assess this through overlays with aerial photography. The peat depths in the wider catchment area are largely similar, with a few notably deeper points of up to 6 m. The peat pH was between 3.1 and 3.4. Water levels were unknown prior to this study. Vegetation composition is typical of eastern Scotland elevated blanket bog [largely *Calluna vulgaris*—*Eriophorum vaginatum* blanket mire, described as M19 as per the British National Vegetation Classification (NVC; Rodwell 1991)], with a mean canopy height of 0.25 m. The wider landscape is predominantly managed for red deer (*Cervus elaphus*) and red grouse (*Lagopus lagopus*), with some managed burning (muirburn) applied occasionally, though none within the 2 km<sup>2</sup> area surrounding the site within the monitoring period reported on in this work. Other grazing animals are mountain hare (*Lepus timidus*) and occasional roe deer (*Capreolus capreolus*).

### Eddy covariance measurements

Sensible and latent heat fluxes (LE and H, respectively) and NEE were monitored using an enclosed path EC system. The EC system combined a Windmaster (Gill Instruments Ltd, Lymington, UK) sonic anemometer at 3.2 m above the peatland surface for the components of atmospheric turbulence ( $u$ ,  $v$ ,  $w$ ; m s<sup>−1</sup>) and sonic temperature (T<sub>sonic</sub>; °C), with an LI-7200 enclosed path H<sub>2</sub>O/CO<sub>2</sub> gas analyser (Licor Biosciences, Lincoln, Nebraska, USA L) to measure atmospheric mass density of vapor (g H<sub>2</sub>O m<sup>−3</sup>) and CO<sub>2</sub> (mg CO<sub>2</sub> m<sup>−3</sup>) as well as barometric pressure (Pair; kPa). The horizontal separation between the sonic anemometer and the Li-7200 was





0.1 m. The length of LI-7200 inlet tube was 1.0 m, optimised for measurements of  $\text{CO}_2$  mass density

over  $\text{H}_2\text{O}$  (e.g., LI-7200 RS instruction manual, Li-COR 2015). Air temperature ( $T_a$ ;  $^{\circ}\text{C}$ ) and relative

◀**Fig. 1** Top left: footprint across the eroded area as estimated using Kljun et al. (2015) within Tovi for the 2018 data series. Tower location is at the marker. Footprint contours are shown in 10% intervals from 10 to 90% with a heatmap indicating up to the 50% interval. Top right: classification product in the footprint area, showing vegetated peat in green, water bodies in blue, and bare peat in gullies in black. The flux tower location is shown with a yellow square and ground validation in the vicinity of the tower are shown as green circles (this is an excerpt from a wider classified scene, hence only a small proportion of the ground validation points have been shown). Lower: wider 2 km<sup>2</sup> area, with 1 m contours (LiDAR derived, thicker contours are 5 m intervals), tower location as marked

humidity (RH; %) were measured at 2 m above the peatland surface using a Rotronic Hygroclip (HC2-S3, Rotronics, Darlaston, Wednesbury, UK). EC sensors were sampled and logged at 20 Hz using a CR3000 Micrologger (Campbell Scientific Inc., Logan Utah). The system was installed on a fully vegetated piece of land as the surfaces of the bare peat gullies tend to move during the winter months due to frost heaving. The available fetch in the prevailing wind direction (SW) was > 1000 m, in all wind directions except N where a cliff edge limits the fetch to ca. 400 m. Canopy height (0.25 m) was assumed to be static throughout the year; the vegetation is semi-evergreen with light to locally moderate deer grazing pressure.

A range of micrometeorological measurements were carried out on the same mast as the EC data collection. Net radiation ( $R_{net}$ ;  $W m^{-2}$ ) and its incoming and outgoing short- and long-wave components ( $SW_{in}$ ,  $SW_{out}$ ,  $LW_{in}$ , and  $LW_{out}$ , respectively;  $W m^{-2}$ ) were measured above the canopy (2.06 m above the peatland surface) using an SN500 Net Radiometer (Apogee Instruments Inc. Logan, Utah, USA). PAR was measured at 2.25 m using a SKP215 Quantum Sensor (Skye Instruments, UK). Soil physical parameters were measured close to the EC mast (within a radius of 5 m). Soil heat fluxes ( $G$ ;  $W m^{-2}$ ) were monitored using two HFP01SC soil heat flux (Hukseflux BV, BV, Delft, The Netherlands) plates installed at 0.03 m below the surface. Soil temperature ( $T_{soil}$ ; °C) was measured at a depth of 2, 5, 10, 20 and 50 cm below the peatland surface using an STP01 soil temperature profile (Campbell Scientific Inc., Logan, Utah, USA). Soil volumetric water content (VWC;  $m^3 m^{-3}$ ) and additional soil temperature measurements were made using four Digital Time Domain Transmissometry (TDT) probes (TDT Soil Water Content

sensor, Acclima, Idaho, USA) installed vertically at 5, 10, 15 and 20 cm depth. Precipitation was measured using a SBS500 unheated tipping bucket rain gauge (P; 0.2 mm sensitivity; Environmental Measurements Ltd, Newcastle, UK) installed in an open area free of any obstruction. Water table depth was measured using a CS451 pressure sensor (Campbell Scientific Inc., Logan, Utah, USA) installed in a 32 mm diameter PVC well with alternating 3 mm holes at 50 mm distance on four rows along the profile). The aforementioned sensors were scanned at 0.1 Hz and logged as 30-min means (sums for precipitation). An additional three wells were installed on vegetated ground within 20 m of the mast (1 m capacitance water level loggers, Odyssey, New Zealand) starting autumn 2019, and a further four standalone water level loggers were installed in dipwells approximately 1 m away from gully edges next to two pairs of gullies in the wider eroded catchment in spring 2020.

#### EC data handling

Thirty-minute flux densities (hereafter fluxes) of sensible and latent heat (LE and H) and net ecosystem CO<sub>2</sub> exchange (NEE) were computed from the raw EC data using EddyPRO® Flux Calculation Software (v 7.0.6; LI-COR Biosciences, Lincoln, Nebraska; Fratini and Mauder 2014). Raw EC data were screened for statistical outliers (Mauder et al. 2013) and other physically implausible values (Vickers and Mahrt 1997). Sonic anemometer data were rotated using a two-dimensional coordinate rotation procedure (Wilczak et al. 2001) and corrected for imperfect cosine response (Nakai and Shimoyama 2012). A planar rotation was also tested but did not provide significant improvements. No boost bug correction was required for the Windmaster deployed at the site (firmware version 2329.700.01). Time lags between the vertical wind speed and concentration measurements were removed using a cross-correlation procedure. Uncorrected fluxes were calculated as the mean covariance between the vertical wind speed ( $w$ ) and the respective atmospheric scalar using 30-min block averages (Baldocchi 2003). Fluxes were corrected for high (Moncrieff et al. 1997) and low frequency cospectral attenuation (Moncrieff et al. 2004). H fluxes were corrected for the influence of atmospheric humidity (Schotanus et al. 1983). LE, then CO<sub>2</sub>, fluxes were adjusted for fluctuations in atmospheric



density (Webb et al. 1980). Random uncertainties for 30-min flux observations related to sampling error were estimated as standard deviations derived from a variance of covariance approach (Finkelstein and Sims 2001). CO<sub>2</sub> storage was assumed negligible at the low observation height and NEE was assumed equal to the turbulent CO<sub>2</sub> flux. The micrometeorological sign convention is adopted where positive values represent fluxes from ecosystem to atmosphere and negatives describe the opposite state.

Quality control (QC) procedures were applied to ensure only high-quality turbulent flux data were retained for analysis. Thirty-minute flux data were screened for statistical outliers using the median absolute deviation approach (Sachs 2013) following recommendations in Papale et al. (2006). Fluxes were also excluded when: the results of the stationarity (steady-state) test result deviated by more than 100% (Foken et al. 2004); and when fluxes were outside the range  $-200 < H > 450 \text{ W m}^{-2}$ ,  $-50 < \text{NEE} > 30 \text{ } \mu\text{mol CO}_2 \text{ m}^2$ , and  $-50 < \text{LE} > 300 \text{ W m}^{-2}$ . Periods of low turbulent mixing were identified using a friction velocity ( $u^*$ ) threshold approach (Papale et al. 2006; Reichstein et al. 2016), and CO<sub>2</sub> fluxes were excluded when  $u^* < 0.17 \text{ m s}^{-1}$ . A 2D footprint model was also calculated as per Kljun et al. (2015). Overall energy balance closure (EBC; SI Fig. 2) for the system was 0.8, which is within the range attained for EC sites globally (Leuning et al. 2012; Stoy et al. 2005; Wilson et al. 2002), and acknowledging that the length of the inlet tube of the Li-7200 was optimised for CO<sub>2</sub> flux measurements.

Gap-filling of flux data and the partitioning of NEE into estimates of gross ecosystem production (GEP) and total ecosystem respiration were performed using the REddyProc Package version 0.8-2/r14 (Reichstein et al. 2016) for the R statistical Language (R Core Team 2017). Data gap-filling of H, LE, and NEE was performed using the marginal distribution sampling (MDS) approach (Reichstein et al. 2005). To enable annual sums to be computed, gaps in air temperature were gap-filled using interpolation of observations from the Braemar station (57.011,  $-3.396, 327 \text{ m asl}$ ). Details of the MDS and flux partitioning algorithms have been described in detail and evaluated elsewhere (Desai et al. 2008; Moffat et al. 2007; Papale et al. 2006; Reichstein et al. 2005) and are not repeated here. Standard night-time-based flux partitioning was used, although we note

that this approach has been questioned recently for peatland ecosystems by Järveoja et al. (2020). Gaps in prognostic micrometeorological variables (SWin, Tair) required for MDS gap-filling were filled using observations obtained at Braemar. Tair was used as the driving temperature for flux partitioning as a number of data gaps were present in the Tsoil record. Uncertainty for individual gap-filled fluxes was estimated as the standard deviation of the observations averaged to fill data gaps (Reichstein et al. 2016). No uncertainties were estimated for GEP and TER as the partitioned CO<sub>2</sub> fluxes represent modelled quantities. During the time period of monitoring, no wildfires occurred, and as grazing pressure could not be determined, we assume that NEE equals NEP.

#### Light and night-time respiration response analyses

NEE during the growing season (April–October) was used to parameterise a modified Michaelis–Menten equation as a function of incoming short-wave radiation (SWin), using:

$$NEE(SWin) = \frac{-\alpha SWin}{1 - (SWin/900) + (\alpha SWin/900)} + Rm$$

where  $\alpha$  ( $\mu\text{mol CO}_2 \text{ J}^{-1}$ ) is the ecosystem quantum yield,  $GPP_{900}$  ( $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ) is the rate of photosynthesis when SWin is  $900 \text{ W m}^{-2}$  and  $Rm$  ( $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ) is the mean ecosystem respiration rate (Carrara et al. 2003; Falge et al. 2001). We also fitted an exponential temperature response model (Lloyd and Taylor 1994) using the night-time NEE (assumed to be equivalent to ecosystem respiration), and evaluated the residuals against water table depth as per the expectation that the site would conform to findings of others that peatland nocturnal respiration is driven primarily by temperature and water table (e.g., Laine et al. 2007).

## Results

### Spatial representativeness

The flux footprint (predominantly into the South-westerly direction) included the area affected by erosion at all times (Fig. 1), but spring observations in particular included less of the eroded area and more

of the fully vegetated surface closest to the tower location. The classification analysis of the wider unrestored area surrounding the flux tower footprint returned proportions of 83.9% vegetated ground, 15.3% bare peat in erosion gullies (in 2 dimensions, i.e., not taking into account gully sides) and 0.8% standing water bodies. The footprint captured a visually representative area of the wider eroded catchment (7 km<sup>2</sup>, based on a 5 m DTM water shed model, SI Fig. 1). Mean  $x$  peak and 90% of the footprint were 46.7 and 128.1 m at night, and 46.2 and 126.5 m during daytime, respectively. The maximum 90% of the  $x$  footprint was 1414 m at night and 770 m during daytime.

### Environmental conditions

Environmental conditions at the Balmoral station are shown in Fig. 2. The mean annual air temperature at the Balmoral station during the monitoring period was approximately 1.1 °C cooler than the nearest long term weather station at Braemar ( $y = 0.9223x - 1.1195$ ,  $r^2 = 0.87$ ), however the relationship breaks down at the cold end due to the mountain plateau versus valley bottom locations of the two respective stations (influence of e.g., temperature inversions). There is also significantly more precipitation at Balmoral than at Braemar, however we did not quantify the difference because the rain gauge at our station is not heated and therefore likely to underreport, or at least have a temporal mismatch, during snowfall periods.

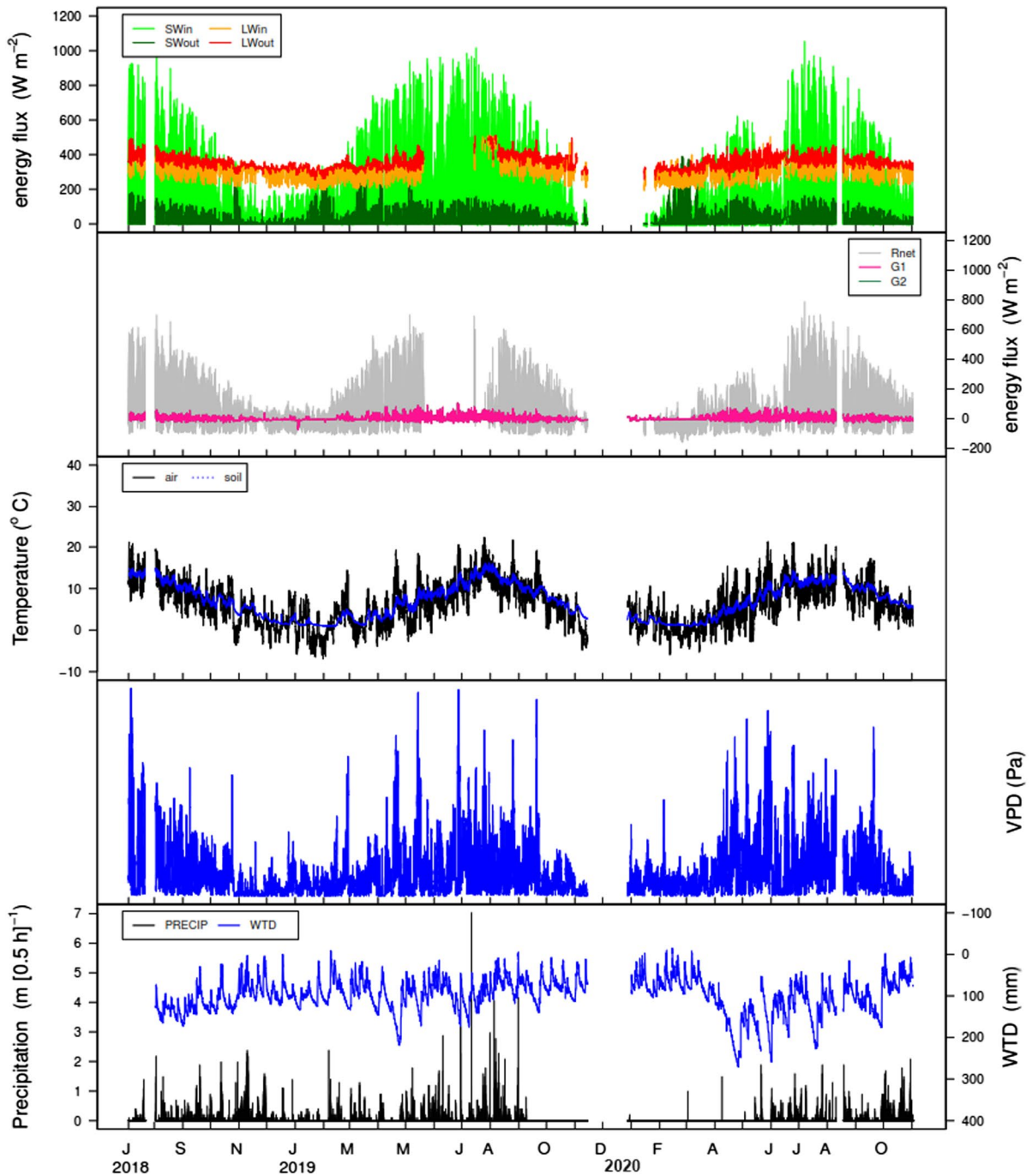
Based on long-term observations from the nearby Braemar weather station (SI Fig. 3), 2018 was ranked 22nd for total annual precipitation out of 60 years of observation (1961–2020), and also followed the 5th driest year (2017) within that record. Relative to the most recent 30-year average period (1981–2010), 2018 was characterised by much lower-than-average precipitation for the first half of the year but heavy rainfall during November (SI Fig. 3). 2018 had a relatively cool start to the year, followed by an abrupt change to an extreme heat wave for the months May to July, peaking at 3.7 °C above the 1981–2010 maximum temperature during June. This was followed by relatively average temperatures during the autumn period and a slightly warmer than average winter 2018–2019. In the biomet data series from our higher altitude site, these unusual conditions were reflected

in the water table dynamics, which displayed a much lower water table depth between July and November 2018 than in the other 2 years (Fig. 2). The year 2019, in contrast, was at the wetter end of the range, occupying rank 49 out of 60 years of observation (1961–2020) for Braemar, with 6 months of higher-than-average precipitation, mostly during the first half of the year. The only deviations in minimum or maximum air temperature occurred during the winter periods. In the data from our monitoring site, this is evident in a period of high air and soil temperatures in February 2019 compared with the other 2 years of observations.

We observed only a brief period of water table drawdown during late April 2019 (Fig. 2). 2020 was another year with higher-than-average spring and summer air temperatures at Braemar, as well as a milder than average early winter. Precipitation records at Braemar also show a sustained drought between March and June 2020, followed by higher-than-average summer rainfall and another short drought in September (SI Fig. 3). At our higher altitude site, water table depth records suggest significant drought between April and September 2020 and some short periods of higher-than-average air and soil temperatures in the same timeframe, compared with the other years of observation (Fig. 2).

### Data capture

Data capture over the monitoring period up to 14th of November 2019 resulted in 91% coverage in flux observations. Biomet data capture was between 92 and 97% except for long wave radiation (79%) due to a faulty sensor, and snow depth (88%, Fig. 2). All data between the 14th of November to the 27th of December 2019 were unfortunately lost due to a major power supply issue. During the same time, peatland restoration activities were carried out in nearby areas, which produced wind-blown peat particles that entered the gas analyser, and which severely affected the spectral response seen in data from the Li-7200RS. Due to COVID travel restrictions at this time, it was not possible to fix this issue until August 2020 and hence NEE and LE data until mid-August 2020 had to be discarded. Biomet data capture continued during this time and was 88–91% except for shortwave (88%) and longwave (76%) radiation components. Remaining observations for the period July



**Fig. 2** Environmental variables monitored across the full monitoring period from July 2018 to November 2020. In order, from top to bottom: short- and long-wave incoming and outgoing energy flux components: soil heat flux relative to net radiation; air and soil temperature vapour pressure deficit and

precipitation/water table depth. The longwave incoming and outgoing radiation data are incomplete due to a faulty sensor; data for such periods have been omitted. Similarly, data for the autumn of 2019 to early summer 2020 for precipitation are suspect due to a wire that was partially damaged by rodents

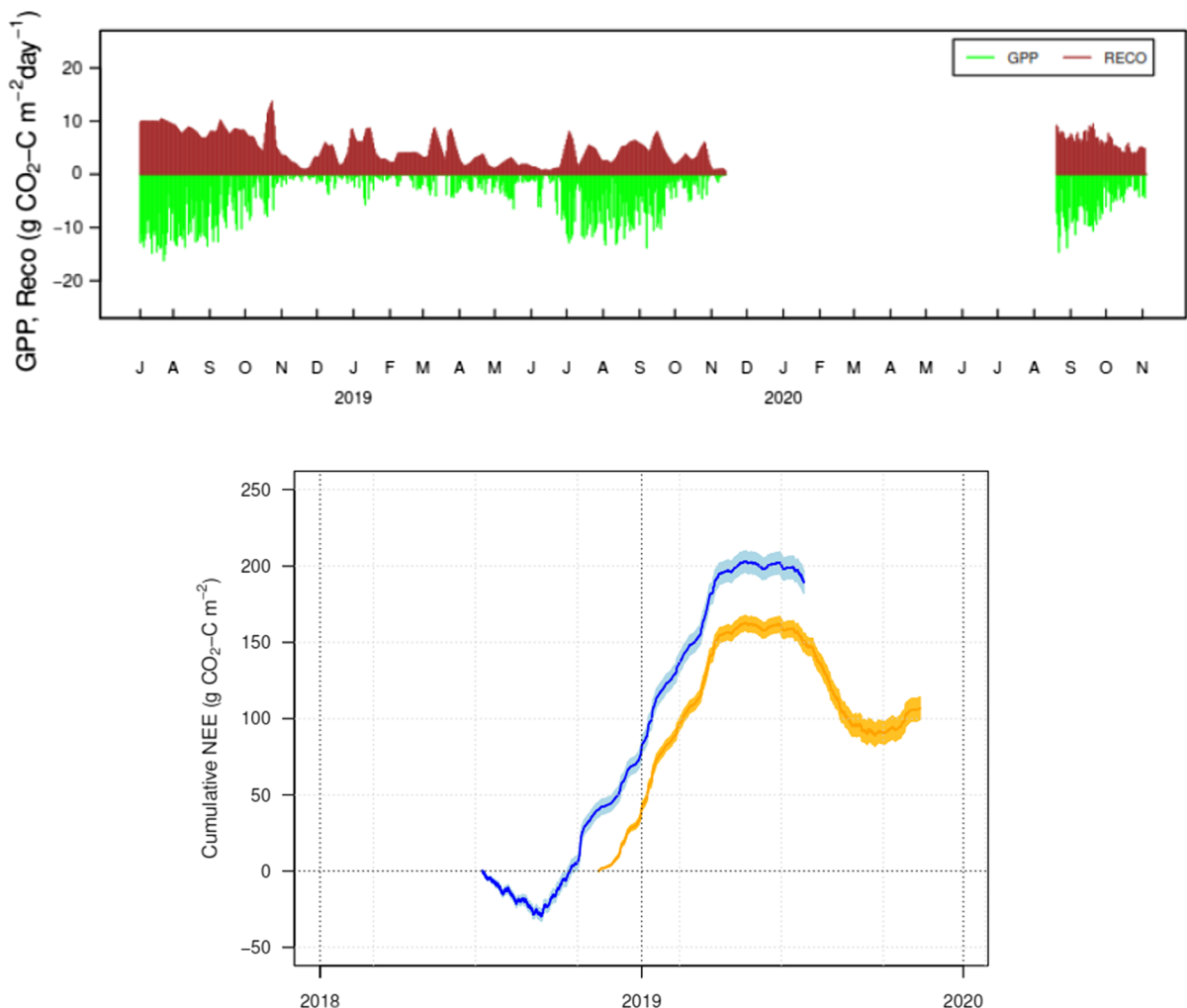


2018 until November 2020 following QC were 35% (CO<sub>2</sub>), 19% (LE), 61% (H), 80% (Rg), 87% (Tair), 87% (Tsoil) and 82% (VPD) (Fig. 2).

#### Seasonal/diurnal patterns and light/night-time temperature response of CO<sub>2</sub> fluxes

In general, net ecosystem exchange of CO<sub>2</sub> (NEE) was positive, except for daytime periods between May and October (SI Fig. 4). The range of NEE was  $-14.06$  to  $13.90 \mu\text{mol m}^{-2} \text{s}^{-1}$ , with maximum net

uptake from the atmosphere during daylight hours in the months of July and August and maximum emissions to the atmosphere during nocturnal periods during autumn, early winter and early spring. The 2018 drought resulted in a slight decrease in net uptake during daylight hours and slightly higher nocturnal net emissions compared with the other 2 years of observations. Due to the altitude of the site and hence frequent cloud cover, maximum modelled partitioned GEP varied significantly between years, with maxima occurring between July and September (Fig. 3).



**Fig. 3** Partitioning of GPP and  $R_{\text{eco}}$ , based on the standard night-time respiration approach (upper panel) for the entire monitoring period excluding the period where the CO<sub>2</sub>/H<sub>2</sub>O analyser was partially blocked. Cumulative NEE is shown in the lower panel, for two 365-day periods, starting either at the beginning of the monitoring period (4 July 2018, blue line) or

ending on the last currently available monitoring date without significant data gaps (14 November 2019, orange line). Uncertainty is shown with the shaded area. Some implausible positive GPP values have been removed from the upper panel for clarity but can be identified in the deposited data. These were an artifact of the partitioning method (see text)

Occasional spikes in emissions were observed in winter periods, corresponding to periods following partial or complete snow cover within the footprint. Modelled  $R_{\text{eco}}$  was noticeable higher during 2018 than 2019, as was GPP during the main growing season (Fig. 3). 2018 was a drought year, with nearly twice as much precipitation during the autumn of 2019 compared with 2018, after an early summer period of warmer than normal air temperatures in 2018 (Fig. 3). These circumstances resulted in a significantly lower summer water table in 2018 than in 2019 and 2020, lasting into the next year (Fig. 2, plotted as annual values in SI Fig. 4).

Modelled parameters for the NEE light response over the summer periods within the 18 months of data capture (Table 1) were roughly in line with other published parameters for peatland ecosystems with semi-natural vegetation cover (e.g., Pelletier et al. 2015). Due to the relatively uniform distribution of the gullies amongst the landscape within the footprint (Fig. 1) we were unable to build separate models for partial footprints that may have contained differential proportions of eroded surface area. Mean  $GPP_{\text{max}}$  over summer periods within the 28-month monitoring period was  $7.65 \mu\text{mol m}^{-2} \text{s}^{-1}$ ,  $\alpha$  was  $0.031 \mu\text{mol CO}_2 \text{J}^{-1}$ , and the mean respiration rate  $R_m$  was  $1.68 \mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$ . Monthly values are given in Table 1.

The temperature response of nocturnal NEE showed only a very weak correlation with air or soil temperature, and similarly, very weak correlation of the residuals of the temperature-NEE response when analysed against water table depth dynamics (not shown). We attempted further data analysis by omitting periods with and immediately after snow fall or extreme cold snap periods (which are highly episodic in nature and often interspersed with periods of milder weather, and in some years occurring as late as mid-May). Snow of any significant depth

tends to only accumulate in the gullies, which are also prone to significant frost heave. Omitting periods with or subsequent to snowfall did not improve temperature response fits. We also tried omitting data from nights with fewer than 7 available half-hours of night-time NEE (e.g., Helbig et al. 2019), as well as choosing soil temperature instead of air temperature, however neither of these improved the model fit.

#### Net ecosystem carbon dioxide balance

Overall, periods of net  $\text{CO}_2$  uptake from the atmosphere were limited to summer periods between late June and late August (Fig. 3), and even during this time, daily fluxes were frequently small net emissions to the atmosphere. Overall, the site was a net source of carbon dioxide of  $147 \text{ g } (\pm 9) \text{ CO}_2\text{-C m}^{-2}$  during the 18-month period of July 2018 to November 2019 (Fig. 3). Annual C balances calculated using a moving window for all 365 days intervals within this period ranged between 106 and  $191 \text{ g CO}_2\text{-C m}^{-2} \text{ year}^{-1}$ . Higher annual C balances were obtained in cumulative 365-day budgets that included the end of the summer 2018 drought, while during 365-day periods with starting dates closer to November 2018, the site was a lower net source of  $\text{CO}_2$ . Finally, we assumed that any additional biomass offtake term through grazing would be negligible, due to the low density of grazing animals and lack of significant browsing damage, however we do not have data to allow us to verify this.

**Table 1** Monthly light use response parameters for periods with data availability during the monitoring period

Month	Alpha			$A_{\text{max}}$			Rm			$r^2$		
	2018	2019	2020	2018	2019	2020	2018	2019	2020	2018	2019	2020
May	NA	0.01	NA	NA	1.61	NA	NA	0.44	NA	NA	0.28	NA
Jun	NA	0.01	NA	NA	2.26	NA	NA	0.34	NA	NA	0.22	NA
Jul	0.04	0.02	NA	7.48	6.18	NA	2.42	1.19	NA	0.94	0.52	NA
Aug	0.04	0.02	0.04	8.76	7.74	9.54	1.99	1.09	1.87	0.83	0.58	0.81
Sep	0.03	0.02	0.04	7.50	7.47	6.59	2.15	1.21	1.58	0.59	0.62	0.74
Oct	0.04	0.02	0.03	3.11	3.57	5.16	1.91	0.92	1.24	0.2	0.34	0.6

NA insufficient data for month

## Discussion

Observed net ecosystem exchange (NEE) of carbon dioxide in context of other reported values

This is the first report of carbon dioxide (CO<sub>2</sub>) fluxes, measured using the eddy covariance technique, in an eroded mountain peatland ecosystem. The magnitude of instantaneous CO<sub>2</sub> fluxes and the light use response at this eroded blanket bog is within the range of previously reported figures for degraded peatlands in the temperate zone (e.g., Lafleur et al. 2003). Specific observations of CO<sub>2</sub> fluxes from eroded peatlands are very sparse, based on chamber measurements, and either report only summer fluxes or suffer from significant data gaps over winter months due to the challenges of accessing mountain environments and carrying out manual measurements of carbon dioxide flux during snow cover. Our study represents the first publication of carbon dioxide emissions from eroded peatland that includes all of the landscape elements and at least one period of near complete winter fluxes, and suggests that net losses are generally higher than in previous, chamber-based or mapping-based, reports (Table 2).

Overall, the reported CO<sub>2</sub> fluxes from this eroded blanket bog fall into the lower end of the same range as reported for the IPCC Wetland Supplement for temperate peat extraction sites, i.e., sites where all vegetation has been removed: the Tier 1 emission factor is 280 (110–420) g C m<sup>-2</sup> year<sup>-1</sup> (95% CI in brackets, IPCC 2014). Wilson et al. (2015) subsequently

reported lower Tier 2 emission factors of 170 (± 47) g C m<sup>-2</sup> year<sup>-1</sup> for industrially extracted, entirely bare, sites and 164 (± 44) g C m<sup>-2</sup> year<sup>-1</sup> for domestic peat extraction sites. The high emissions from this site, where only ~ 15% of the surface was occupied by bare peat, are therefore concerning. The approach taken for inclusion of peatland emissions in UK Inventory reporting (Evans et al. 2017), due to the paucity of data from eroded peatland at the time, was to produce a weighted average emission factor for eroded peatlands, comprising calculated Tier 2 emission factors for slightly modified, semi-natural peatland and extracted, bare, peat surfaces. This approach resulted in much lower estimated CO<sub>2</sub> emissions for eroded peatland (20 g C m<sup>-2</sup> with a 95% confidence interval of – 10 to 60) than the results of this study suggest. It is therefore likely that direct carbon dioxide emissions from eroded peatlands in the UK are underestimated in the current UK Emissions Inventory submission. Finally, although no prescribed burning or wildfires occurred within the footprint during the years of monitoring, for a complete C budget we would also require information on the net methane fluxes to or from the atmosphere and any net losses via aquatic routes. The latter are likely to be substantial as there is evidence of ongoing erosion after rainfall events, but data on this are not available at present for the site. To the best of our knowledge, no direct observations on methane losses from eroding peatlands exist. Although Worrall et al. (2009) report a full C budget for their site, methane fluxes were estimated using an empirical relationship with water table depth. The

**Table 2** Reported direct carbon dioxide emissions from eroding peatlands

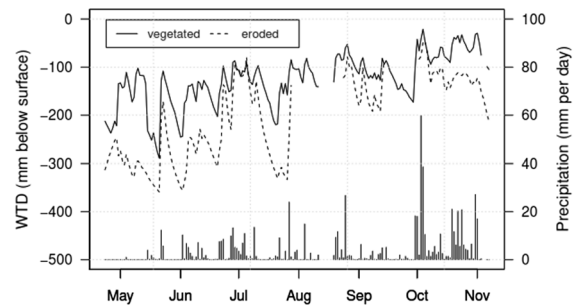
References	Monitoring location and measured emissions	Technique
Gatis et al. (2019)	Dartmoor, UK – Vegetated hags (29 and 20 g C m <sup>-2</sup> ; Year 1 and 2, respectively. 95% CI of – 570 to 762 and – 873 to 1105 g C m <sup>-2</sup> – Eroded peat pans (7 and 8 g C m <sup>-2</sup> , Year 1 and 2, respectively. 95% CI of – 147 to 465 and – 136 to 436 g C m <sup>-2</sup>	Manual chambers, based on growing seasons only (2 years)
Dixon et al. (2014), Clay et al. (2011)	Bleaklow plateau, UK – Bare peat (flat area, average of 24 mg CO <sub>2</sub> m <sup>-2</sup> h <sup>-1</sup> ) – Bare peat gullies (average of 32 mg CO <sub>2</sub> m <sup>-2</sup> h <sup>-1</sup> )	Manual chambers (4 years, no annual budget modelled), year-round but limited winter coverage
Clay et al. (2010)	Moor House, UK – Vegetated blanket bog (62–95 g C m <sup>-2</sup> year <sup>-1</sup> )	Manual chambers (2 years)
This study	Balmoral, UK – Eroding blanket bog (106–191 g CO <sub>2</sub> -C m <sup>-2</sup> year <sup>-1</sup> )	Eddy covariance (18 months)



UK GHG Inventory methodology (Evans et al. 2017) calculates net losses from eroding peatlands on the basis of a proportional flux from bare peat areas (for which the data mostly stem from cutover sites) and heather-dominated blanket bog; and uses values of 18.8 and 19.3 g C m<sup>-2</sup> year<sup>-1</sup>, for DOC and POC lost off site via the aquatic route, respectively, based on the data synthesis of Evans et al. (2015).

#### Effects of short-term variation in local climate on CO<sub>2</sub> fluxes

In a recent synthesis of GHG fluxes from peatlands, and to which this work contributed with a shorter time series of the data presented here, Evans et al. (2021) concluded that water table depth was the highest explanatory factor of net ecosystem exchange of carbon dioxide over the medium term and thus peatland rewetting could contribute significantly to efforts to reduce emissions from degraded peatlands. Water table depth monitoring at the Balmoral site only began in early August 2018 and hence we can only calculate the corresponding NEE for the moving window period from this point onwards. For the first 365-day window (4th August 2018–2019), the annual loss of CO<sub>2</sub> was 175 g C m<sup>-2</sup> at a mean annual water table depth of -10.15 cm. For the latest possible 365-day window (14th November 2018–2019), the site lost 106 g C m<sup>-2</sup> of CO<sub>2</sub> at a corresponding mean annual water table depth of -8.97 cm. However, as water table depth monitoring in this phase was only carried out in the direct vicinity of the instrumentation, it is likely that the mean annual water table depths are not applicable to the wider footprint. Indeed, additional monitoring results within the wider footprint, once the full complement of loggers was installed in spring 2020, show that there was a significant water level drawdown on vegetated areas adjacent to bare peat gullies during summer months (Fig. 4). The true annual mean water table depth is therefore likely to be significantly lower. Additional measurements will be required to shed further light on the likely true water table dynamics at this site, potentially in conjunction with footprint-wide hindcasting into the monitoring period reported on in this work, using modelled water table dynamics from the reported relationship between peatland water table dynamics with synthetic aperture radar C-band backscatter (e.g., Bechtold et al. 2018; Lees et al. 2021).



**Fig. 4** Differences in water table depth dynamics between the water table dynamics measured directly adjacent to the flux tower (black line) against the water table dynamics adjacent to erosion gullies lower, stippled line), shown relative to precipitation events (black bars) during 2020. Observations shown are means of  $n=3$  (flux tower and wider vegetated area) and  $n=4-6$  (adjacent to erosion gullies) water level loggers

#### Atypical observations

Some atypical observations in our study system do warrant discussion and further evaluation. In contrast to published observations of relationships of night-time NEE with temperature in natural, degraded and rewetted peatlands (e.g., Herbst et al. 2013, Laine et al. 2007; Helbig et al. 2019), we were unable to find a satisfactory fit to an exponential Lloyd–Taylor model, using either air or soil temperature, nor by omitting periods during or immediately following snowfall or potential frost-heave, or by omitting nights with fewer than seven observations. We hypothesize that the most likely cause of these observations is sensor mismatch of the air/soil temperature probes and the pressure transducer (for water table depth) relative to the wider footprint, as these are placed in the area immediately surrounding the installation, which is fully vegetated and, on average, displays a more stable and higher water table than gullied areas within the footprint (Fig. 4). It is therefore possible that the reason for our inability to model night-time respiration is that respiration in the gully systems and vegetated areas are subject to temporally divergent dynamics. Gatis et al. (2019) reported, using chamber-based measurements over two growing seasons in an eroded blanket bog peatland, that both photosynthetic CO<sub>2</sub> uptake and ecosystem respiration were lower in the eroded areas (peat pans) than in the vegetated areas (in their case, the hagged remnant areas of blanked bog surface).

As the distribution of gullies within the footprint was relatively uniform amongst the prevalent wind directions, we were unable to partition the contribution of gullies versus vegetated areas akin to e.g., Pelletier et al. (2015). Gullies also act as short-term water bodies during periods of high precipitation, as well as a reservoir of snow for up to several weeks after snowmelt on the vegetated ground, so it is likely that there is a higher degree of insulation during such periods, whereas the ground heat flux during periods when the bare (black) surface is exposed is likely to be higher than on vegetated peat (e.g., Price et al. 1998). Future planned improvements to the site through additional instrumentation in a representative gully close to the existing system will hopefully allow us to better explain the observed temperature responses in this ecosystem.

In around 5% of the partitioned fluxes, we also observed some spurious and relatively large, positive, modelled GPP values after partitioning the NEE using the standard night-time method (Reichstein et al. 2005). These were largely observed in the January–April 2019 period (removed for visual clarity from Fig. 3, but available in the deposited dataset). During this period, several rapid freeze–thaw cycles occurred, combined with short-lived periods of snowfall (Fig. 2). The standard night-time partitioning method does not cope well with fast changing ecosystem responses such as rapid freeze–thaw cycles, due to the 15-day averaging period. More complex ways of modelling the partitioning of NEE, such as the dual source model of Wohlfahrt and Galvagno (2017), or the artificial neural network approach of Tramontana et al. (2021), may produce better results and also take into account recent findings of differential diel patterns of peatland ecosystem respiration (Järveoja et al. 2020). The data series for this publication was however relatively limited in duration and therefore we did not attempt more complex partitioning methods at this stage.

Finally, peatland restoration work (gully reprofiling, ground smoothing, mowing for donor mulching material) was carried out during the monitoring period, although this did not extend into the footprint. It is possible that some minor interference from this work is occasionally included in our measurements, although we excluded any data where spectral responses clearly indicated loss of high-frequency data. There will be no further restoration activities

in the 500 m radius surrounding the fetch and hence future data will shed further light on whether the difficulties in modelling the night-time temperatures were in relation to disturbances caused by nearby movement of soil.

#### Abatement potential with restoration management

In contrast to the substantial emissions in the eroded state observed here and reported to a lesser degree previously, carbon dioxide fluxes from hydrologically intact or near-intact peatlands are generally reported to be strongly net negative to the atmosphere, i.e., carbon sinks (for blanket bog e.g., Levy and Gray 2015; Helfter et al. 2015; Sottocornola and Kiely 2005, 2010). Our findings therefore support an argument for restoration of this eroded site. The most convincing example of the benefits of rewetting to date stems from Canadian peatland studies of long-term experiments by Nugent et al. (2019), Lee et al. (2017) and Strack and Zuback (2013). In a blanket bog context, Hambley et al. (2019) show net CO<sub>2</sub> flux, monitored by eddy covariance, from former forestry-converted sites returning to similar values to that of a nearby undisturbed low altitude blanket bog. Wilson et al. (2016) and Renou-Wilson et al. (2019) similarly show significant mitigation of carbon dioxide emissions following rewetting of former peat extraction sites.

The likely impacts of a ‘do nothing’ scenario as opposed to restoration of this and other eroded blanket peatlands in the global context of climatic change demands further consideration. It is presently unclear whether erosion can be stabilised or reversed (Harris and Baird 2019; Milner et al. 2020). In addition, restoration of upland and mountain peatland locations carries a not inconsiderable cost due to the limited accessibility in space and time. On the other hand, current scenarios for the UK include both an increased frequency and severity of extreme rainfall events (Lowe et al. 2018) as well as summer droughts and potentially higher incidence of wildfires (Perry et al. 2022). Widespread drying of European peatlands over the last 200 years, as a consequence of compound factors such as drainage or other management in combination with climatic changes, has been demonstrated by Swindles et al. (2019) using testate amoeba records. Heavy rain events have been documented to lead to new peat erosion (e.g., Hulme and Blyth 2017) and hence direct carbon losses, leaving the newly eroded

peat surface exposed to other forces that could cause additional wind-blown and wind-driven rain erosion (Foulds and Warburton 2007) and increased on-site gaseous emissions (Evans et al. 2006; Gallego-Sala and Prentice 2013). Indeed, observations on this site suggested significant impact of the Storm Frank intense rainfall (29–30 December 2015) around the area of what is currently the flux tower footprint. There were also negative impacts on restoration work on a nearby location, where the rainfall washed out nearly all of a thin mulch of vegetation that had been laid onto bare peat areas in an initial restoration effort (Peatland Action data, S. Corcoran, Cairngorms National Park Authority, pers. Comm.). Using a peat erosion model (PESERA-PEAT) with seven different global climate models, Li et al. (2017) found that erosion rates for many Northern peatlands may increase by the 2080s.

The pre-restoration state of our monitored eroded mountain blanket bog is that of a significant net CO<sub>2</sub> source to the atmosphere and annual carbon budgets that contained more of the 2018 drought period were nearly twice as high as budgets that included a period of more average climatic conditions. If our findings are representative of the wider state of eroded peatlands across the UK, such areas cover 3283 km<sup>2</sup> in the UK (Evans et al. 2017) and thus may presently be considerable contributors to the UK's net CO<sub>2</sub> emissions from peatlands and the land use (LULUCF) sector generally. This work therefore contributes to a growing body of evidence that current total peatland emissions across Europe are largely characterised by the significant emissions from degraded areas, which completely overrides any remaining sequestration potential from undamaged peatland areas (Leifeld and Menichetti 2018), and thereby strengthens calls for mitigation of their carbon emissions. Future discussions and prioritisation of restoration activities, however, also need to interrogate the carbon benefits of restoration of eroded upland peatlands against the significant fossil fuel emission cost of restoration. At present, restoration of such eroded peatlands is still at early stages, with few sites that have been rewetted more than a handful of years ago, and so the carbon benefits of restoration can also not yet be assessed.

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**Author contributions** RA and RM were responsible for experiment design, GDS and MC carried out site maintenance, RA, MC and RM shared data processing and analysis elements, RA led the writing of this manuscript, with contributions from all co-authors.

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**Data availability** The annual budget of a shorter, daily, timeseries of this site was used in a recent synthesis publication (Evans et al. 2021). The 30-min processed data for this manuscript are available from the UK Environmental Information Data Centre (EIDC), with the identifier: <https://doi.org/10.5285/a65f6241-bfc3-430a-ae93-ccb7c63c1a53>.

#### Declarations

**Conflict of interest** There are no competing interests to declare. None of the authors received direct remuneration for any advisory roles undertaken or had any financial or non-financial interests in organisations that may be affected by the results of this study.

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