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Muddying the waters: Impacts of a bogflow on carbon transport and water quality

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ABSTRACT

Landslides of peat have been recorded throughout Britain and Ireland for centuries. Whilst these events are not uncommon, land degradation can amplify their magnitude and frequency and, crucially, their immediate impacts are rarely documented. A 20,000 m^3 bogflow event that occurred on land undergoing development in the Irish border area in November 2020 was monitored at high frequency in the major receiving river system (384 km^2). Samples collected every seven hours over a 28 day period at a site 37 km downstream were analysed for suspended sediment (SS), particulate organic carbon (POC) and dissolved organic carbon (DOC and UV-derived fractions), synchronous with hydrometeorological data and turbidity. There was no impact of the bogflow on DOC concentrations or loads. However, concentrations of SS and POC in the first samples after the bogflow were 825 mg/L and 346 mg C/L, respectively, and fish kill was estimated at 100 %. Analysis of detrended SS and POC loads suggested the main impacts of the bogflow on water quality lasted just eight days. Over this period, an additional 1318 t of SS and 608 t of POC were transported as far as the monitoring point, equating to 325 % more SS and 925 % more POC than would have been expected otherwise under the same river flow conditions. The carbon loss and water quality impacts were short lived, but nevertheless severe, and highlight the vulnerability of peatlands and the risks when these environments are inappropriately managed.

1. Introduction

Globally, approximately 2.7 petagrams (Pg) of carbon (C) is lost from the land to aquatic systems each year (Battin et al., 2009). Of this, 0.6 Pg C is buried in lake sediments, with 0.9 Pg C ultimately reaching the oceans (Aufdenkampe et al., 2011; Battin et al., 2009). The remaining 1.2 Pg C is remineralised and released into the atmosphere as CO_2 (Aufdenkampe et al., 2011; Battin et al., 2008). Whilst these values remain poorly constrained, they highlight the important role that inland waters play in the transport and cycling of organic matter derived from terrestrial sources. As a consequence of both climate change and land use change, the quantity of organic matter entering streams and rivers is predicted to increase (Roebuck et al., 2019; Wilson and Xenopoulos, 2009). In particular, infrastructure projects, temperature extremes, and the increased frequency of major storm events are predicted to exacerbate soil erosion (e.g. Evans et al., 2006; Li and Fang, 2016; Nearing et al., 2004; Wang et al., 2012), resulting in episodic increases in riverine sediment loads and return of C to the atmosphere.

One particularly vulnerable ecosystem containing some of the densest deposits of organic matter and C on the planet is peatlands. Accounting for just 3 % of the world's land area but constituting over 30 % of global soil C (Gorham, 1991; Rydin and Jeglum, 2006), many peatlands rely on cool damp conditions in order to continue functioning as C sinks (Lindsay, 2010). During hot dry periods, peat can dry and shrink, sometimes irreversibly, as a result of lowered water table depth (Morton and Heinemeyer, 2019). This drying can lead to reduced vegetation cover and the creation of cracks and gullies (Evans et al., 2006). During subsequent heavy or prolonged rainfall events, which are predicted to increase under climate change (Kendon et al., 2014), the lack of vegetation can increase surface runoff and erosion (Evans et al., 2006), whilst cracks and gullies can cause subsidence and canalise basal peat (Holden et al., 2004; Ronkanen and Kløve, 2008), potentially destabilising the entire peat body (Warburton et al., 2004).

Similarly, anthropogenic activities which dry or disturb the peat can

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have major impacts on its stability. Tree planting disrupts the soil structure and dries the peat, both in the initial excavations to plant forestry and in the root growth of the trees (Braekke, 1983; Sloan et al., 2018). Evapotranspiration from trees is also much greater than from peatland vegetation (Sarkkola et al., 2010), which greatly reduces the water table depth, exposing deeper layers of peat to drier and more oxic conditions (Pyatt et al., 1992; Sloan et al., 2018). Construction projects, such as wind farm developments or roads to service wind farms or forestry, also disturb peat structure and affect its stability (Lindsay and Bragg, 2005). Peat excavations and the addition of heavy materials, such as wood, stone aggregates, concrete or steel, create physical barriers that disrupt water flow through the peat matrix, which may result in supersaturation and flooding of upslope areas, increasing pressure on downslope peat (Lindsay, 2010; Lindsay and Bragg, 2005). Whilst roads across peatlands often utilise a floating road construction to reduce barrier effects (Reilly and Buggy, 2021), the weight of the materials still compresses the peat, altering flow pathways through the peat and leading to water build up upslope (Bell, 2000; Lindsay and Bragg, 2005). Ultimately, the pressure of the water, the supersaturation of the peat and subsurface erosion, whether individually or in combination and whether caused by forestry, construction or climatic events, can lead to catastrophic failure of the peat, with substantial sections, or even whole hillsides, turning into landslides (Lindsay and Bragg, 2005; Lindsay and Freeman, 2008; Long et al., 2011; Warburton, 2015; Warburton et al., 2004).

Where these events connect with waterbodies, they have potentially serious implications for both water quality and the functioning of aquatic ecosystems. Suspended sediments can have a shading effect, reducing photosynthetic productivity in the water column or streambed (McSweeney et al., 2017; Otto and Enger, 1960). They also provide nucleation sites for microbial colonisation, supporting the formation of microbial flocs (Niederdorfer et al., 2016). Dissolved organic matter adsorbs to soil and fine sediment particles in streams (Aufdenkampe et al., 2001; Hunter and Battin, 2016), altering carbon and nitrogen cycling pathways both in the water column and on the streambed (Hunter et al., 2016). As such, increased suspended sediment loads within freshwater systems have the potential to alter downstream transport of organic matter (Aufdenkampe et al., 2001; Simon et al., 2002) and cause proliferation of both pathogenic and non-pathogenic microorganisms (e.g. Simon et al., 2002; Wotton, 2007), negatively affecting water quality.

This is of particular concern in catchments where water is abstracted for drinking. Whilst surface water intended for human consumption is usually treated before distribution, sudden changes in contaminants in the intake water can be difficult to adequately treat (Ritson et al., 2016). Additionally, certain fractions of dissolved organic matter are difficult to remove during conventional treatment and can react with disinfection chemicals to form potentially carcinogenic by-products, such as trihalomethanes (Singer, 1999). This can be exacerbated by both high and fluctuating concentrations. Therefore, maintaining peatlands as intact and well-hydrated units through sensitive land management, restoration of surface vegetation and resilience to climate change can ultimately benefit drinking water quality (Ritson et al., 2016).

Mass movement of peat have been recorded throughout the United Kingdom and Ireland for centuries (e.g. Dykes and Warburton, 2008; Dykes et al., 2008; Lindsay and Bragg, 2005; Long et al., 2011; McCahon et al., 1987; Warburton, 2015 and references therein). However, there are few studies on the immediate water quality consequences and aquatic carbon loss of peat slides entering river systems. In November 2020, a landslide occurred at Meenbog (Co. Donegal) on the Irish border depositing large amounts of peat into the Mourne Beg tributary of the River Derg catchment. Whilst bogflows (likely the most appropriate term for the peat movement according to definitions by Dykes and Warburton (2007)) such as this are not overly unusual in Ireland (Dykes, 2022), it should be noted that this one arose within a section of commercial forestry and at a site under wind farm development, both of which have previously been implicated in increasing the chance of slippage (Lindsay and Bragg, 2005). Coincident with the period in which the bogflow occurred, a high-frequency water quality monitoring programme was sampling downstream for a separate pesticide study (Atcheson et al., 2022; Cassidy et al., 2022; Farrow et al., 2022; Morton et al., 2021). Therefore, the aim of this study was to quantify and assess the carbon loss and water quality pressures caused by the bogflow on a major river system. Whilst a limitation of this study is that the experimental set-up was designed to target a different parameter, and thus there was no *a priori* hypothesis relating to carbon transfer, the monitoring instrumentation was nonetheless ideally placed to audit water quality in a large river system, regardless of contaminant or cause. This coincidence, therefore, uniquely enabled benchmarking of the effects and associated impacts of a bogflow for comparison with similar extreme events in both natural and engineered settings.

2. Methods

2.1. The Meenbog site and bogflow

Meenbog is a blanket bog in Co. Donegal (Ireland), largely planted with commercial coniferous forestry that was undergoing wind farm development in unplanted and clear-felled blocks. The point of failure was close to the top of the hill, was unplanted at the time of the bogflow and was uphill of a partially constructed access road, for which tree trunks were laid out in preparation for a gravel layer to follow (visible in Derg Media (2020)). The wider area contains open areas of blanket bog, some of which have been extensively hand-cut, and includes the site of the 1963 bogflow on Barnesmore bog, Co. Donegal (Colhoun et al., 1965). The Meenbog site borders the Barnesmore Bog Natural Heritage Area (NHA) to the south-west (NPWS, n.d.-a) and Croaghonagh Bog Special Area of Conservation (SAC) to the north-west (DAERA, n.d.). The Sruhangarve River drains the sub-catchment, feeding into the Mourne Beg River then the River Derg (Fig. 1), which are part of both the River Finn SAC (NPWS, n.d.-b) and the River Foyle and Tributaries SAC (DAERA, n.d.).

The movement of a mass of peat at Meenbog began at approximately 08:00 GMT on 13th November 2020 and continued for several hours. The point of failure was at approximately 54°43′02″ N 7°52′35″ W (Irish National Grid: H 08,015 85572) and affected an area of approximately 100 m by 100 m (Fig. 2). Assuming an average peat depth across the area of 2 m (which is likely based on extrapolation of depths given in AGEC Ltd (2017)), this would give a total failure volume of 20,000 m^3 (although *cf.* Dykes (2022) who estimates a displaced volume of 65,000 m^3). The bogflow was largely a movement of wet basal peat but carried rafts of surface peat, complete with standing trees, up to several kilometres downstream (Derg Media, 2020; Rooney, 2020). The failure was less than 500 m from the Sruhangarve River, which enabled the bogflow to move through the Mourne Beg River and into the River Derg to the sample location within about 5 h (arriving between 12:45 and 13:00 GMT, according to NI Water, pers. comm.). For further description of the bogflow, see Dykes (2022).

2.2. Water sample collection

Water samples were collected using an ISCO 6712FR refrigerated autosampler (Teledyne ISCO, Lincoln, NE, USA) with a 24-bottle configuration from the River Derg at the Derg water treatment works (WTW) abstraction point, which was approximately 37.1 km downstream from the site of the bogflow (the length of flow pathways between locations was determined by measurement of watercourse sections as defined in the OSNI Basemap (Ordnance Survey of Northern Ireland - © Crown Copyright and Database Right), which was undertaken in ArcGIS 10.6 (Esri, 2017)). Every 7 h, approximately 500 ml of water was pumped from the river into a 1 L polypropylene bottle. Samples were taken between 14:00 GMT on 10th November 2020 and

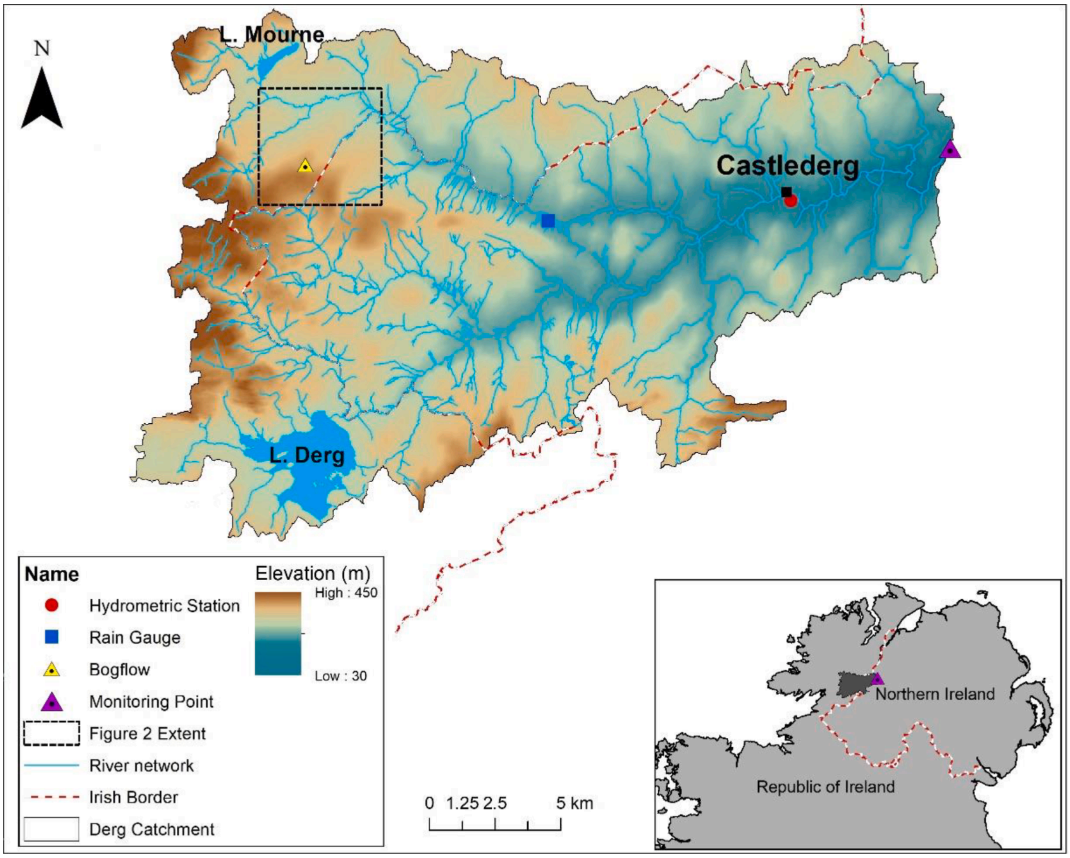

Fig. 1. Map of the River Derg catchment with locations of the Meenbog bogflow (yellow triangle), the rain gauge (blue square), the river discharge monitoring point (red circle) and the water monitoring point (purple triangle). Inset shows the location of the Derg catchment within the island of Ireland. The Mourne Beg River flows from Lough Mourne past the rain gauge location and converges with the River Derg, which flows from Lough Derg, just west of Castlederg. Fig. 2 shows the area inside the dashed square in more detail. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

07:00 GMT on 8th December 2020. Samples were retrieved weekly, stored at 4 °C and filtered within 10 days of collection from the river. Filtrate was then frozen at −18 °C until analysis, which was within 24 days of initial collection for all samples.

Due to the samples being collected for another purpose, approximately 350 ml of the 500 ml samples were analysed for this study. On all occasions, the samples were left to settle before the water was withdrawn, with removed volumes recorded and added to the analysed volume when calculating concentrations and loads. However, the sediment in the first three samples following the bogflow did not settle and remained fully suspended throughout the sample (Fig. S1). Therefore, the quantity removed on site was assumed to be fully mixed and the volume was not added to the analysed quantity. The day following sample retrieval, 130 ml from each sample was centrifuged, with the solids added back to the main sample and this volume added to the analysed volume.

2.3. Sample analysis

Water samples were filtered under vacuum through pre-ashed (combusted in a furnace overnight at 550 °C), pre-weighed 0.7 μm filters (Whatman glass microfiber filters, Grade GF/F, 47 mm diameter, Sigma-Aldrich, Dorset, UK). Due to the amount of suspended matter in them, some samples were split into aliquots, each of which was filtered separately. Filters were dried overnight at 110 °C and re-weighed to determine the amount of SS. They were then combusted in a furnace at 550 °C overnight and re-weighed to determine the proportion of organic matter by loss-on-ignition, which was then used to calculate POC following Ball (1964). Due to the opportunistic nature of this study, it was not possible to perform a comparison of this method to a direct one for the quantification of POC. However, use of the loss-on-ignition technique to estimate the organic carbon content of soils and sediments is very well established (e.g. Billett et al., 2012; Dawson et al., 2002; Friggens et al., 2020; Li et al., 2019; Robroek et al., 2010). Indeed, Chambers et al., (2011, p.5) stated that estimating peat carbon indirectly "can reduce some uncertainty since the variation in peat [organic matter] (OM) content… …is greater than the variation in the [organic carbon] content of OM, and measurement of OM content by loss-on-ignition analysis… …is straightforward, fast, and inexpensive". Given that it is highly likely that the vast majority of the sediment, especially over the initial period following the bogflow, was peat and that the standard equation from Ball (1964) assumes 45.4 % carbon in the organic matter, if anything, the carbon within the suspended sediment collected was underestimated in this study.

Dissolved carbon concentrations were determined from the filtrate by high-temperature oxidation using a Multi N/C 3100 TOC analyser (Analytik Jena AG, Jena, Germany). Dissolved organic carbon (DOC) concentrations were determined by difference between total dissolved carbon (TDC) and dissolved inorganic carbon (DIC). TDC was quantified by high-temperature oxidation of a 500 μl aliquot of the filtrate at 750 °C over a platinum catalyst. DIC concentrations were then determined by acidification of a 500 μl sample of the filtrate with 10 % phosphoric acid solution. The CO_2 produced through either combustion (TDC) or acidification (DIC) was then quantified using the instrument's nondispersive infrared CO_2 detector. From each sample, three replicate injections were performed to determine both TDC and DIC, and the system was rinsed with 1500 μl of ultra-pure water between injections. A six-point calibration of the instrument was performed using solutions of potassium

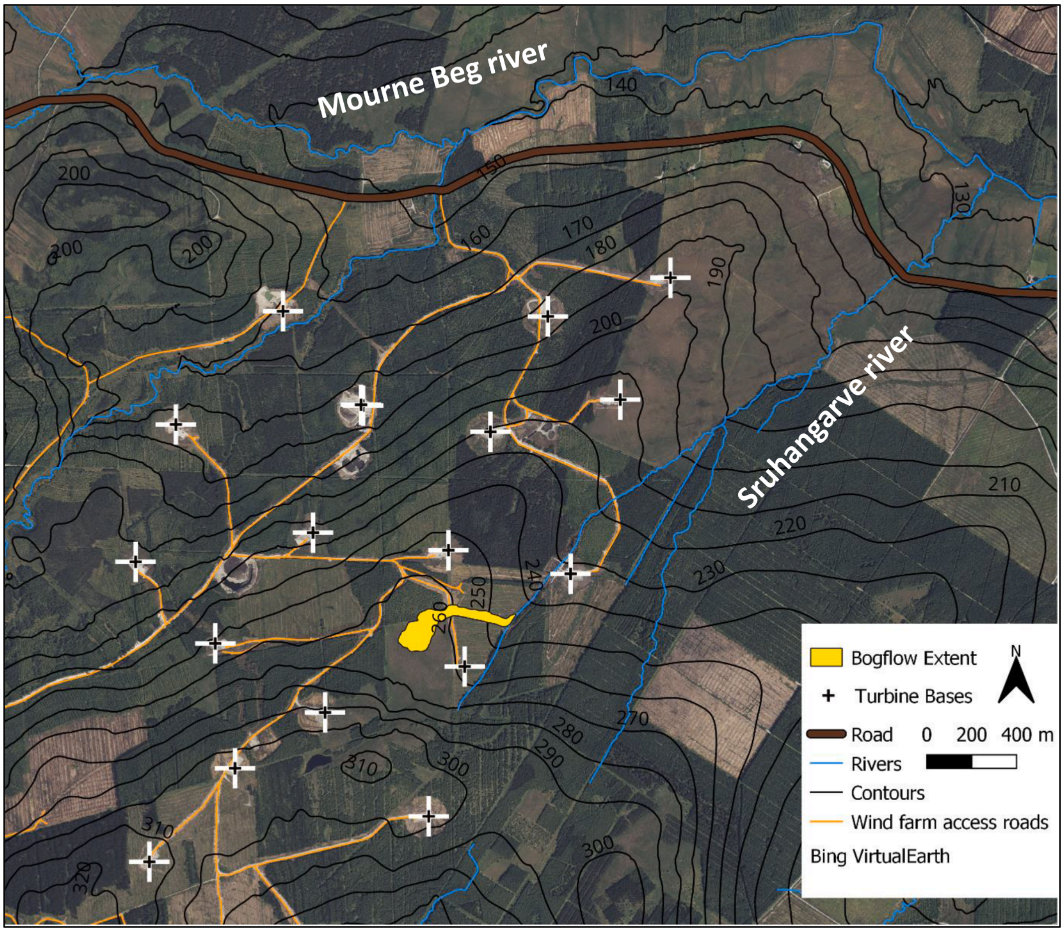

Fig. 2. A detailed view of the area inside the dashed square from Fig. 1. The extent of peat that moved as part of the bogflow was approximately 100 m x 100 m, with a long scar stretching downhill to the Sruhangarve River. The bogflow swept through and breached one of the newly constructed "floating" wind farm access roads. The Mourne Beg River, into which the Sruhangarve drains and which later joins the River Derg, is shown at the top of the map, flowing west to east.

hydrogen phthalate (total C concentrations: 0 mg/L, 1.0 mg/L, 2.5 mg/L, 5 mg/L, 10 mg/L and 20 mg/L), and a combination of sodium carbonate and sodium hydrogen carbonate (total inorganic C concentrations: 0 mg/L, 1.0 mg/L, 2.5 mg/L, 5 mg/L, 10 mg/L and 20 mg/L) as standards. Ultra-pure water (Milli-Q) samples were used as procedural blanks and Consensus Reference Material obtained through the QUASIMEME inter-laboratory calibration scheme (https://www.wepal.nl/en/wepal/Home/Proficiency-tests.htm) was used as quality control checks on the analytical procedures.

Subsamples of the filtrate were analysed for absorbency at 254 nm, 400 nm, 465 nm and 665 nm on a spectrophotometer (U-5100, Hitachi High-Tech Corporation, Fukuoka, Japan) coupled with an autosampler. Blank samples of deionised water were also analysed and their values subtracted from each reading.

2.4. Environmental data

Rainfall was measured by an ARG100 rain gauge (Environmental Measurements Limited, North Shields, UK) and recorded every 15 min on a Tinytag data logger (TGPR-1201; Gemini Data Loggers Ltd, Chichester, UK). The rain gauge was located at 54°41′56″ N 7°43′59″ W, 16.6 km downstream of the bogflow. These rainfall data were combined with minimum and maximum air temperature, wind speed and sunshine data from the nearest accessible synoptic weather station at Malin Head, Co. Donegal (55°22′20″N 7°20′20″ W; 75 km from Castlederg) to estimate daily soil moisture deficit (SMD) using the model by Schulte et al. (2005). SMD predictions for the poorly-drained soil class were used here.

Stage height and rated discharge data for the River Derg were obtained from the only permanent gauging station within the catchment, situated near Castlederg (54°42′19.6″ N, 7°35′21.8″ W), 8.8 km upstream of the WTW intake and the sample collection point (total catchment area to this point was 384 km^2). The downstream discharge at the sampling point was corrected using a time-adjusted proxy of 84 min following Atcheson et al. (2022).

For context, a water quality sonde (C3 Submersible Fluorometer; Turner Designs, San Jose, CA, USA) was deployed at the sampling point and recorded turbidity (minimum detection range 0.05 nephelometric turbidity units (NTU), range 0–1500 NTU) at hourly intervals between 13th October 2020 and 31st January 2021.

2.5. Data analysis

Concentrations of SS were converted to mg/L and POC and DOC were converted to mg C/L. Analyses were conducted in a combination of Excel (Microsoft Office 15) and R software (R Core Team, 2016). Seven-hourly loads of each component were calculated by linear interpolation of the time series using:

$$L = \sum_{1}^{n} k(c_i t_i q_i) \quad [1]$$

where

L is the load for a 7-hourly sampling period.

c_i is the instantaneous concentration of the component (SS, POC or DOC) in the ith sample (mg/L).
t_i is the time period represented by the sample (s).
q_i is the flow in the ith sample period (interpolated from 15 min data using ± 3.5 h from c_i) (m^3).
k is a conversion factor to correct for units and time interval.

Absorbency at 254 nm was divided by the DOC concentration to obtain specific ultra-violet absorbency ($SUVA_{254}$) values and expressed in L mg/C m^{-1}. $SUVA_{254}$ is often used by water companies as a proxy for the aromaticity of DOC and to determine the need for or amounts of enhanced coagulation and softening prior to treatment (Weishaar et al., 2003). Water colour was expressed in Hazen units by multiplying absorbency at 400 nm by 12, following Watts et al. (2001). The relative proportions of fulvic to humic acids were expressed as E4:E6 ratios by dividing absorbency at 465 nm by that at 665 nm (Thurman, 1985).

To determine the impact of the bogflow on DOC, SS and POC, the loads were regressed against total 7-hourly discharge and the linear regression equations were used to calculate the predicted loads, which were then subtracted from the measured loads. These detrended loads were used to visually identify days of exceptionally high loads. To calculate the additional SS and POC loads caused by the bogflow, the regression was redone without the days attributed to the bogflow and the linear equation was reapplied to 7-hourly discharge to calculate the "normal" loads for all periods. These were subtracted from the measured loads on the bogflow-affected days, from which the additional percentage of matter was calculated for the affected days and for the 28-day monitoring period.

3. Results

3.1. Time-series data

Rainfall on the day of the bogflow totalled just 1.2 mm but regional SMD was −10 cm (i.e., supersaturated). In fact, the soil was saturated (SMD < 0 cm) from 30th September 2020 until well after the bogflow occurred, although SMD was similar over the same period in both 2018 and 2019. However, total rainfall in the 30 days preceding the bogflow was much higher in 2020 (285.8 mm) than in 2018 (90.6 mm) or 2019 (138.2 mm) and included two days where rainfall exceeded 30 mm: 33.8 mm fell on 2nd November 2020 and 35.6 mm on 11th November, just two days before the bogflow. This latter rainfall provided a storm peak of 162 m^3 s^{-1} at the Castlederg gauging station (Fig. 3).

Measured concentrations of DOC were unaffected by the bogflow,

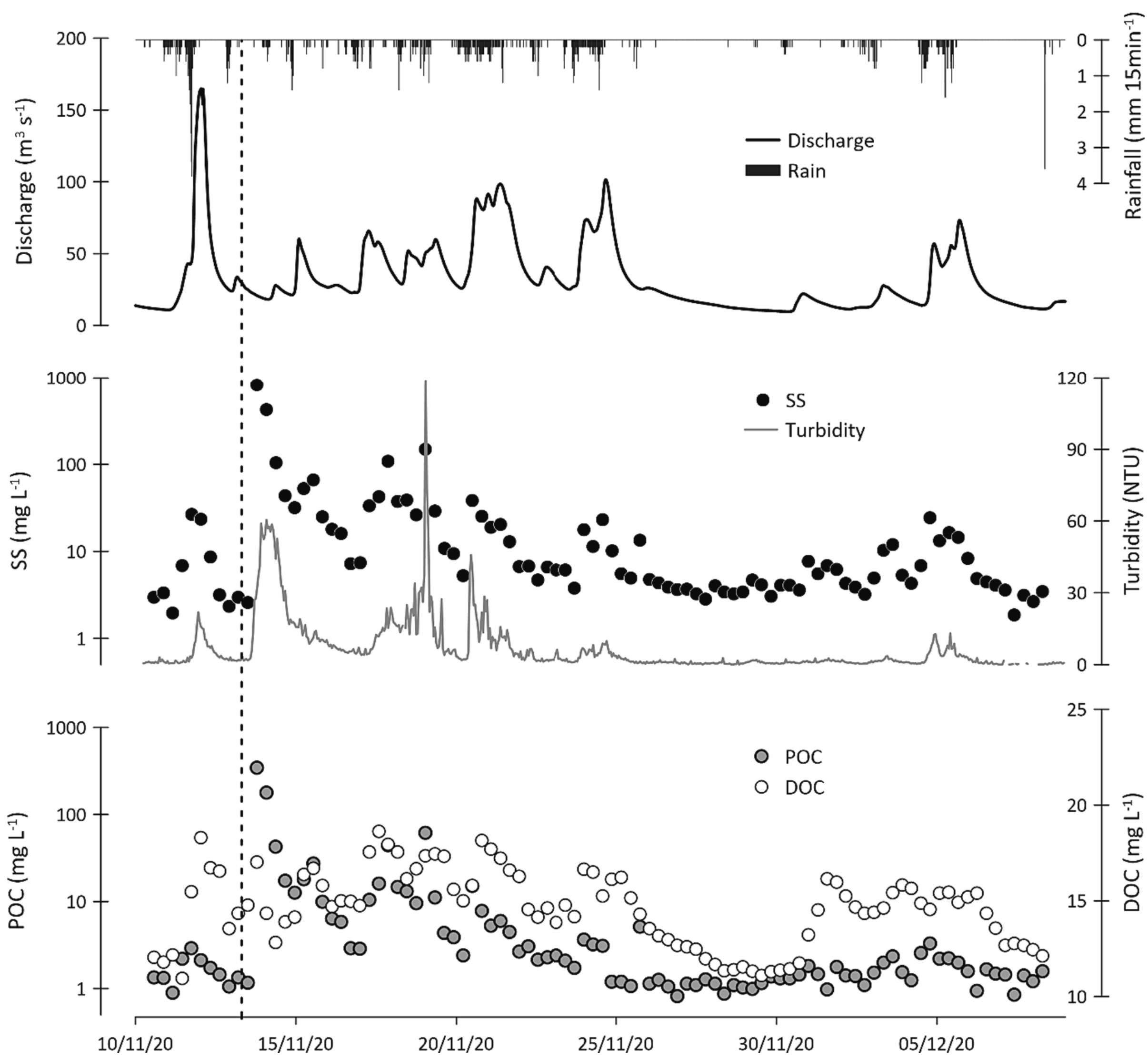

Fig. 3. Rainfall (15 min totals), instantaneous discharge, SS, POC, DOC concentrations (7 h intervals), and turbidity (1 h intervals) immediately before, during, and following the bogflow. The bogflow occurred on 13th November 2020, as indicated by the dashed line.

correlating with discharge, and ranged from 11.0 mg C/L on 11th November to 18.6 mg C/L on 17th November (Fig. 3). Although the inorganic C concentrations (DIC) were much lower than DOC (range: 1.3–8.8 mg C/L), they followed an inverse relationship with the DOC concentrations and were highest under the lowest flows but were also unaffected by the bogflow (data not shown). The DOC fractions within the filtrate also appeared largely unaffected by the bogflow. $SUVA_{254}$ values ranged from 0.034 L mg C^{-1} m^{-1} at 00:00 GMT on 24th November to 0.049 L mg C^{-1} m^{-1} at 22:00 GMT on 12th November and tended to increase when discharge increased, albeit with a delay of 6–24 h, although the magnitude of the $SUVA_{254}$ value change was not mirrored by the change in discharge (data not shown). Similarly, Hazen was largely governed by discharge rates, again with a delay in change and decoupled magnitude, and ranged from 0.71 Hazen at 03:00 GMT on 30th November and 1.6 Hazen at 14:00 GMT on 17th November (data not shown). Conversely, the E4:E6 ratio had no obvious relationship with discharge and was very variable, with the highest ratio of 10.4 occurring at 09:00 GMT on 28th November. However, the lowest E4:E6 ratio of 5.3 was recorded at 19:00 GMT on 13th November. Interestingly, the preceding three samples and the following three samples all had E4:E6 ratios below 6.5. Whilst there were a few other samples over the 28 days of monitoring that had E4:E6 ratios below 6.5, this was the only period, covering almost two days, when there were more than two samples in a row with values less than 6.5 (Fig. S2). Taken with the fact that the lowest ratio occurred in the first sample taken following the peat failure, this suggests that the E4:E6 ratio was affected by the bogflow, albeit briefly, but possibly before as well as after.

In contrast, SS and POC concentrations, and turbidity were greatly impacted by the bogflow. Across a longer measurement period (October 2020 to February 2021), turbidity ranged from effectively zero NTU to 118.79 NTU and tended to change with discharge, albeit with a lag and some variability (Fig. S3). During the 28 days covered by the water

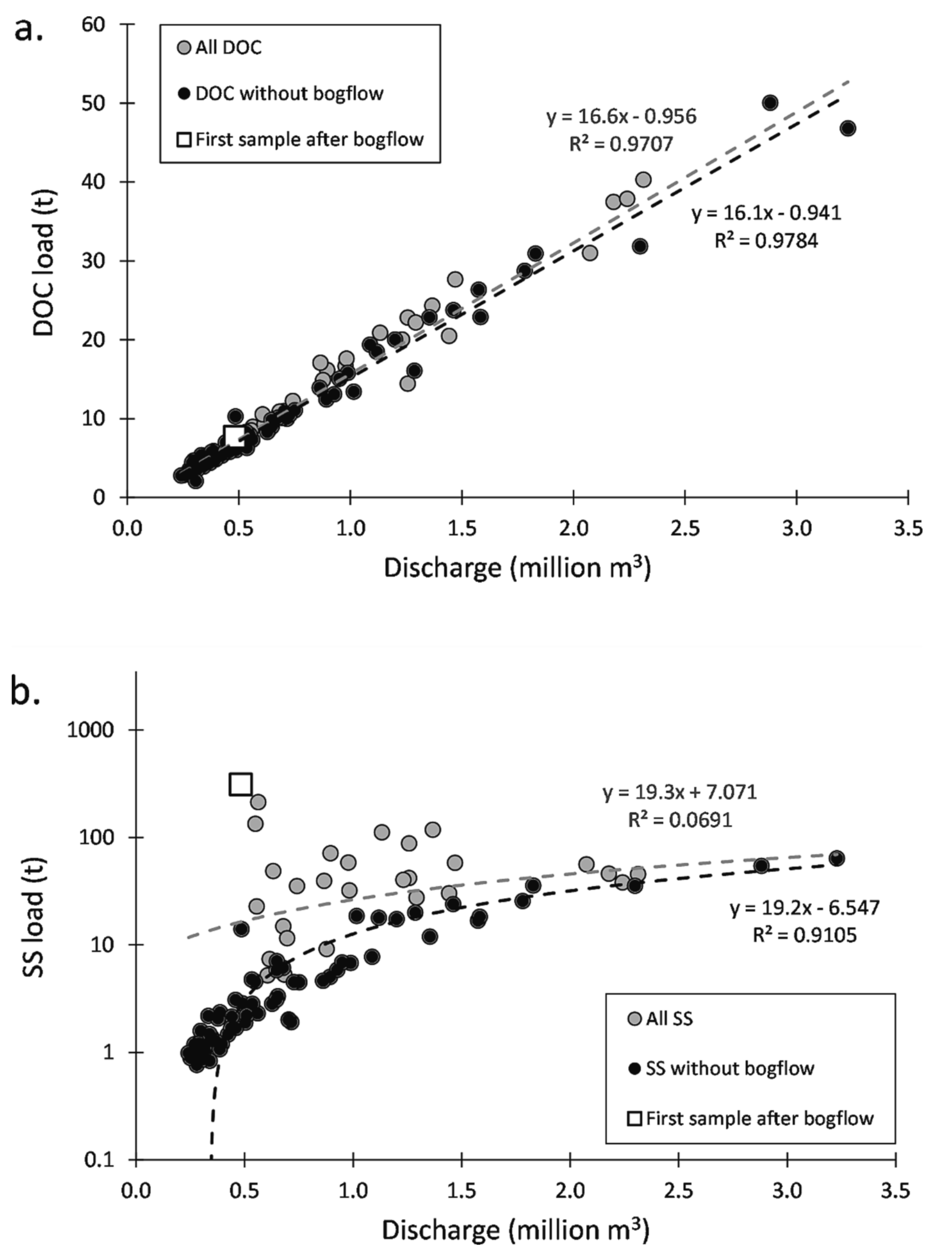

Fig. 4. a-c. comparison of 7-hourly discharge to 7-hourly doc load (a.), ss load (b.), and poc load (c.) for all 28 sampling days (i.e., All DOC, All SS and All POC on legends—grey circles) and the same data excluding the eight days (identified through detrending—black circles) the bogflow was deemed to have an impact with linear regression lines fitted to the data (grey dashed lines for all, black dashed lines excluding the eight days). For each parameter, the first sample after the bogflow had passed the monitoring point is shown as a white square. Note that the y-axes for SS and POC are on a log scale.

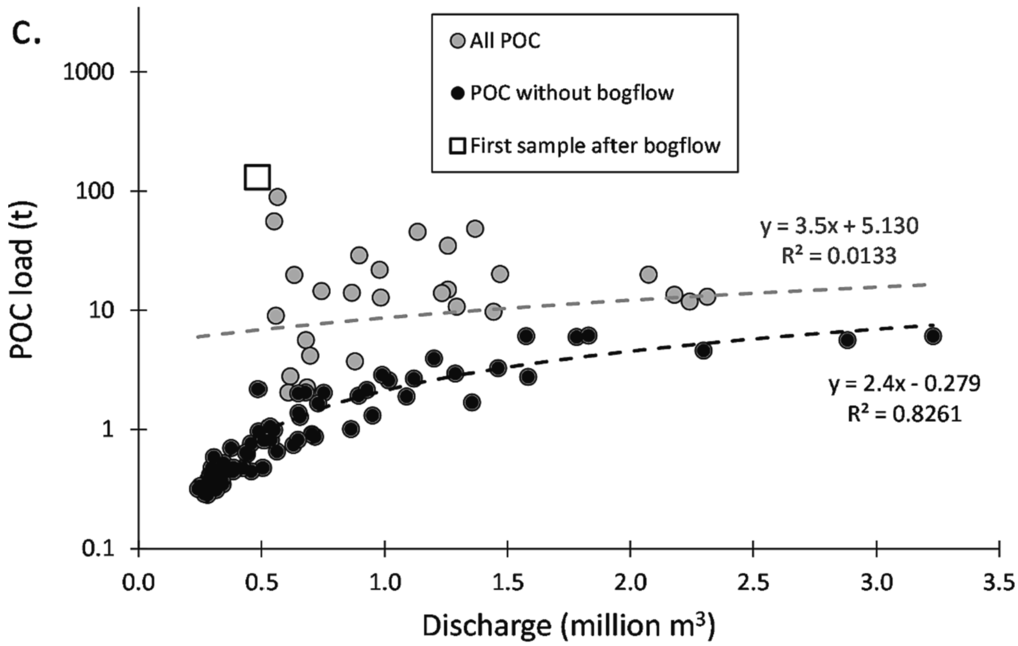

Fig. 4. (*continued*).

samples, the minimum and maximum turbidity values were 0.49 NTU and 118.79 NTU, occurring on 6th December at 23:00 GMT and 19th November at 01:00 GMT, respectively (Fig. 3). The highest SS (824.5 mg/L) and POC (346.1 mg C/L) concentrations measured were at 19:00 GMT on 13th November, which was the first sample taken after the initial wave of peat passed the sampling point (Fig. 3). In comparison, the lowest concentrations of SS and POC were 1.9 mg/L on 7th December at 10:00 GMT and 0.82 mg C/L on 26th November at 22:00 GMT, respectively. The first three samples taken after the bogflow were opaque and did not settle, although the third (taken at 09:00 GMT on 14th November) was brown rather than black (Fig. S1). Correspondingly, the SS and POC concentrations were much lower in this sample compared to the first two. Despite discharge falling at the time the bogflow passed the monitoring point, turbidity increased from 2.06 NTU to 60.47 NTU in 13 h. A second, much higher but briefer peak of turbidity occurred on 19th November at 01:00 GMT, coinciding with a moderate flow increase (Fig. 3), most likely representing a remobilisation event of the peat that had initially been deposited on the riverbanks, as it followed a peak in river flow. This was corroborated by the dark brown colour of this water sample (Fig. S1) and the correspondingly high POC and SS concentrations (Fig. 3).

3.2. Sediment and carbon loads

The highest 7-hourly DOC load was 50 tonnes (t) C, occurring at 01:00 GMT (i.e., ± 3.5 h) on 12th November, over 24 h before the bogflow and coinciding with the highest 7-hourly discharge of the monitoring period and second highest concentration (Fig. S4). Indeed, the bogflow did not seem to affect the DOC loads or measured concentrations; conversely, the loads and concentrations were mainly driven by the discharge rate ($R^2 = 0.97$) (Fig. 4a).

SS and POC 7-hour loads were highest on 13th November (310 t C and 130 t C, respectively). Despite the loads having almost halved by the second sample after the bogflow (02:00 GMT on 14th November), they were still very high (135 t SS and 56 t POC) compared to the majority of the measurement period, although these loads were matched on 19th November (118 t SS and 48 t POC) by the possible remobilisation event (Fig. S4). The SS concentrations and loads and, to a lesser extent the POC concentrations and loads, also indicate a slightly smaller remobilisation event on 17th November at 21:00 (Figs. 3 and S4), although this did not result in an increase in turbidity. Indeed, comparison of the SS concentrations against the percentage of this carbon represented by POC suggests a similar composition as in the first samples after the bogflow (Fig. S5).

Comparison of the SS and POC concentrations and discharge before the bogflow with the rest of the monitoring period suggests that the major impacts of the bogflow only lasted eight days (corresponding to 28 of the 96 samples). This comparison is corroborated by the detrended loads and change in the strength of regression between the 7-hourly loads and discharge including and excluding the eight days of the bogflow effects: for SS, $R^2 = 0.07$ for all 28 days and $R^2 = 0.91$ excluding the eight days (Fig. 4b); for POC, $R^2 = 0.01$ for all 28 days and $R^2 = 0.83$ without 13th-21st November (Fig. 4c). Nevertheless, assuming the majority of the material from the bogflow was released in the first eight days following the event, an additional 1318 t of SS and 608 t of POC reached at least as far the monitoring point. This was 76 % and 90 % of the SS and POC, respectively, measured on those eight days and 59 % and 78 % of the loads, respectively, recorded during the full 28 days of monitoring (in comparison, when calculated in the same way, additional DOC during the eight affected days was 8 %, largely due to slightly higher concentrations compared to flow magnitude on 17th-19th November and therefore likely due to natural variation, and it was just 3 % across the whole 28 days). In comparison, 325 % and 925 % more SS and POC, respectively, passed the monitoring point in the first eight days than would have been expected if the bogflow had not occurred.

4. Discussion

Landslides occur across the world and, where peatlands are present, slides and slips of peat on slopes are not uncommon, particularly in Ireland (Dykes, 2022; Dykes et al., 2008; Lindsay and Bragg, 2005; Long et al., 2011). However, there is little warning about when such an event is likely to occur and, as such, any monitoring of the impacts does not begin until days, sometimes weeks, later. Consequently, this is possibly the only study that has explicitly monitored the land-to-water transfer of a bogflow (or, indeed, any mass movement of peat) on water quality in a major river.

It is worth noting that the high rainfall in the 30 days preceding the bogflow and the two very wet periods 11 and 2 days before the failure occurred, combined with supersaturation of the peat, may have been the ultimate trigger for failure. This is in contrast to Dykes (2022) who concluded that there was no evidence of a rainfall trigger for the Meenbog bogflow. However, Dykes (2022) used rainfall data from a meteorological station outside of the River Derg catchment and more than twice the distance from the failure site than the rain gauge used in this paper, which may explain the disparity. Nevertheless, the failure was most likely caused by a number of factors where antecedent rainfall and SMD conditions caused an already disturbed and weakened system to move.

4.1. DOC and its composition

A surprising finding was the lack of impact the bogflow had on DOC and the composition of the dissolved compounds within the filtrate (i.e., $SUVA_{254}$ and Hazen). DOC concentrations within peat soils tend to be relatively high compared to those in streams and rivers (Heinemeyer et al., 2019), particularly in forested areas (Flynn et al., 2022). As such, when large volumes of peat entered the river during the bogflow, it might have been expected to result in a concomitant increase in DOC concentrations within the river—but this was not observed. This contrasts with Grieve and Gilvear (2008), who found the DOC concentrations in streams flowing from areas where wind farms had been constructed were higher than from undisturbed areas. Whilst no direct comparison was made in this study to similar areas not affected by the bogflow, DOC concentrations in the River Derg remained below 20 mg C/L, which are certainly not unusual for peaty catchments (e.g. see Worrall and Burt (2007) for a review of DOC at 315 sites covering a range of conditions across Great Britain, including peaty catchments).

As the water was sampled 37 km downstream of the point of failure, it is possible that the turbulence of the movement sufficiently oxidised the water (Guasch et al., 1998) so that any additional DOC had largely decomposed by the time it reached the sampling point (Berggren et al., 2022). However, this would have been very quick (the bogflow was first detected by the WTW just 5 h after the event occurred) and is unlikely to have only affected any additional DOC released. The DOC could have adsorbed to the great excess of POC and suspended sediment particles within the water column (Aufdenkampe et al., 2001; Hunter and Battin, 2016), or within the sample bottles. If this were the case, the POC present would have likely bound all the DOC and actually reduced the concentrations, but this appeared not to be the case, particularly given that the relationship between DOC loads and 7-hourly discharge was very similar regardless of whether all 28 days were included or the eight days when SS and POC were affected were excluded ($R^2 = 0.97$ *cf.* $R^2 = 0.98$; Fig. 4a). In the month preceding the bogflow, rainfall was very high compared to the previous two years, with daily rainfall exceeding 30 mm for two days in the 11 days prior to the bogflow. It is, therefore, more likely that the DOC that had built up within the peat during warmer months was largely flushed from the soil before the peat mass entered the river. For example, Scott et al. (1998) demonstrated that most DOC export occurred during high rainfall events in autumn. Regardless of the reasons, the discovery that DOC concentrations (and associated filtrates $SUVA_{254}$ and Hazen) appeared unaffected by the bogflow may be useful information for water companies, due to the propensity of some DOC fractions to form potentially carcinogenic trichloromethanes (Clay et al., 2012; Ritson et al., 2016; Singer, 1999).

The variability of the E4:E6 ratio may initially appear to present more of a challenge to drinking water treatment. However, previous studies have also documented large variability in E4:E6 ratios compared to other measures of DOC character within natural waters (O'Driscoll et al., 2006; Park et al., 1999; Peacock et al., 2014). Additionally, the E4:E6 ratio has been shown to vary with storm events (Grayson and Holden, 2012), which may explain the variability over the 28 days of measurement. However, as the E4:E6 ratio has long been considered as a measure of humification (Peuravuori and Pihlaja, 1997), where lower ratios (2–5) indicate more decomposed humic acids and higher ratios (over 8) indicate more of the less decomposed fulvic compounds (Thurman, 1985), the fact that the lowest ratio occurred in the first sample after the bogflow may indicate that the E4:E6 ratio was briefly affected by this event: the bogflow would have released a large load of basal peat which would have been more decomposed than the usual peat contribution to DOC and should have had a lower ratio, which it did. Whether this would impact on water treatment or river ecology, however, is unknown and, given the transient nature of the ratio change, is unlikely to have a great effect.

4.2. POC and SS

Unlike the DOC and its compositional fractions, SS, POC and turbidity were heavily impacted by the bogflow. POC concentrations from peatland areas in the UK and Ireland tend to range between about 0.5 to 4 mg C/L, although can occasionally exceed 20 mg C/L (Billett et al., 2012; Dawson et al., 2004; Turner et al., 2016). As all of the measured POC concentrations in the River Derg were below 4 mg C/L in the three days before and from nine days after the bogflow occurred, apart from one concentration of 5.2 mg C/L on 25th November, this suggests that the area usually produces typical POC concentrations. The very high concentrations in the eight days following the bogflow (Fig. 3) demonstrate the impact the bogflow had on POC and SS release. Similarly, two of the three highest turbidity values in the 18 months of measured data in the River Derg occurred within six days of the bogflow (Figs. 3 and S3). However, there were lower POC and SS concentrations and turbidity values within this eight day period, mainly coinciding with lower river flows (Fig. 3), suggesting that, after the first few days, the majority of sediment was being remobilised from riverbanks or the riverbed having been deposited there during the initial wave. This is corroborated by the varying proportions of carbon in the SS being attributed to POC in the first eight days after the bogflow. The initial samples after the bogflow and those indicating the remobilisation events had over 40 % carbon (Fig. S5), whereas other samples within the first eight days following the bogflow had as little as 27 % carbon in the POC. This strongly suggests input from different sources even during this period of elevated carbon and sediment input.

Despite being relatively brief (i.e., days rather than weeks or months), these large increases in turbidity, SS and POC represent a challenge for both drinking water treatment and aquatic life in the rivers. Nonetheless, the recorded values represent the only quantities of SS and POC that passed the monitoring point, 37 km downstream of the site of the bogflow. It is unlikely that all released peat reached the monitoring point within the first eight days. For example, there was visual evidence of peat deposits along much of the banks of the Sruhangarve River three months after the bogflow (Fig. S6), with smaller quantities likely along the banks of the upper Mourne Beg River and an unknown amount incorporated into river bottom sediments. Additionally, as plant-derived unbound POC, which likely comprised most of the POC released by the bogflow, tends to travel as an undercurrent near the riverbed (Repasch et al., 2022) and the autosampler tube intake was deliberately suspended midwater for the purposes of the other ongoing sampling, the water samples analysed probably underestimated the true POC concentrations in the water column.

It is, however, possible to calculate an estimate of the proportion of peat released that did reach the monitoring location. Assuming that the bulk density, fraction of organic matter and C content of organic matter of the peat at the site of failure is most closely represented by those used in the "haplotelmic cubic metre of peat" defined by Lindsay (2010) (under the assumption that the forestry at Meenbog is likely to have drawn down the water table to a point where the top layer (acrotelm) was no longer viable), and taking the total area of failure as 100 m by 100 m with an average depth of 2 m, the quantity of C released by the bogflow would be 1,163 t. As previously mentioned, any additional DOC released by the bogflow was negligible, meaning the majority of this C would be represented by POC. Thus, 52 % (608 t) of the estimated C released by the bogflow had been moved more than 37 km downstream within eight days of the event. Similarly, an estimated 2,300 t of SS (assuming this is equivalent to dry weight in Lindsay (2010)) was released, of which 57 % (1,318 t) was measured at the monitoring point as above baseline SS within eight days of the bogflow.

4.3. Implications

This quantity of SS and the concentrations associated with it (up to 825 mg/L of SS 37 km downstream) had severe consequences for river ecology. The whole of the River Derg network and the rivers beyond (Foyle tributaries and estuary) are designated as protected areas for Atlantic salmon (DAERA, n.d.) and, following the bogflow, fish kill in the River Derg was estimated at 100 % (Loughs Agency, pers. comm.). As basal peat is rich in iron and aluminium, it is possible that these concentrations were elevated to toxic levels, as occurred in Teesdale following the 1983 peat slide which resulted in a fish kill (McCahon et al., 1987). It is also likely that these concentrations affected aquatic invertebrates (McCahon et al., 1987). In the present study, metal concentrations were not measured and so it is not possible to comment further on the impacts of heavy metal toxic effects. However, it is likely that the gills of the fish and filter-feeding apparatus of invertebrates were clogged by SS such that a significant portion of the fish kill was caused by suffocation. Either way, the fish population may take years to recover to previous levels and the rest of the aquatic community could also be disrupted for a substantial length of time. Kaushal et al. (2018) review similar impacts and recovery responses from pollution pulses caused by extreme climate and weather events – following this, it is likely that the carbon effects to the benthic system following bogflows will need more thorough investigations in targeted surveys and/or experiments. These authors also propose a more strategic response to surveillance in vulnerable catchments, using networks of *in situ* sensors, which should make the capture of such events more likely when they do occur.

Whilst the effects on river life and water quality were the most immediate and obvious consequences of the bogflow, impacts on C transport and release are just as important and will likely play out over greater timescales. As mentioned above, only 52 % of the estimated C released by the bogflow was measured passing the monitoring point in the first eight days following the event. Of the remaining 48 %, some is likely to have been washed off the banks and passed though the monitoring point in small quantities over an extended period of time, some will have been retained on the site of the bogflow having merely been shifted out of position and exposed (Fig. S6), and some will have settled out of the water column and been incorporated within river sediment, as well as being washed through further and been buried within lake and ocean sediments (Aufdenkampe et al., 2011).

Only 5–25 % of eroded sediment ever reaches the oceans and, while a substantial quantity of this is incorporated within freshwater sediments (Aufdenkampe et al., 2011), much of the organic C is either metabolised or photolysed and outgassed as CO_2 (Cole et al., 2007; Richey et al., 2002). Since the majority of the additional solid material detected after the bogflow was POC, it is likely that a large portion was oxidised to CO_2 and released as a "hot moment". Battin et al. (2009) estimated that, of the 2.7 Pg C yr^{-1} passing from the land into inland waters, 1.2 Pg C yr^{-1} (44 %) was outgassed, largely as CO_2, although a small amount would also be released as methane, which is a much more potent greenhouse gas. Some of the CO_2 produced is likely to have remained within the water column (Hope et al., 2001), leading to increased acidification. Even if the increase was relatively brief and slight, it could have put additional pressures on any remaining aquatic organisms within the system, hindering recovery. If the proportion of 44 % of the C input into the River Derg from the bogflow being released as CO_2 was applied to just the POC measured at the monitoring point, this would mean the loss of 267.6 t C to the atmosphere in just eight days (although the actual outgassing may have occurred over a longer period). This is equivalent to the estimated annual loss of CO_2-C from 157 ha of drained Irish/British peatlands under industrial extraction (Wilson et al., 2015).

5. Conclusions

Vulnerable upland peat environments are subject to large scale geomorphic change through land slips and bogflows. These can be exacerbated by land cover change and development, especially during and following extreme weather conditions. The river water quality impacts of these changes are often unquantified. In this study, a land-to-water transfer of peat was captured by a companion investigation where monitoring equipment was in operation. The bogflow was ultimately triggered by a period of intense rainfall events and regional soil supersaturation on a hillside undergoing development. The investigation found that:

- A bogflow released approximately 20,000 m^3 of peat, equivalent to 2,300 t of suspended sediment, and 1,163 t of carbon into a major salmonid river system.
- Within eight days, approximately 1,318 t of suspended sediment and 608 t of particulate organic carbon were transported 37 km downstream. This was approximately 325 % and 925 % more, respectively, than would have otherwise been expected over the eight day period.
- Concentrations of up to 825 mg/L of suspended sediment and 346 mg C/L of particulate organic carbon, measured directly after the bogflow.
- There was no concurrent increase in DOC concentrations or loads (nor DOC filtrate fractions) as a consequence of the bogflow.

There is no doubt that the ecological consequences of the event were severe and likely to be long lasting. Furthermore, the peat mass lost would likely have been mostly oxidised thus adding to a "hot moment" of CO_2 emission. This study, by chance, has quantified the water quality pressures and carbon loss when there is land use development on vulnerable peatland environments that are subject to extreme climatic events. Ironically, in this case, both afforestation and wind farm development, designed to offset carbon emissions, are implicated in large scale carbon release and ecosystem destruction.

CRediT authorship contribution statement

Phoebe A. Morton: Conceptualization, Formal analysis, Investigation, Methodology, Writing – original draft, Writing – review & editing. **William Ross Hunter:** Formal analysis, Methodology, Writing – review & editing. **Rachel Cassidy:** Conceptualization, Formal analysis, Funding acquisition, Writing – review & editing, Methodology, Supervision. **Donnacha Doody:** Conceptualization, Funding acquisition, Supervision, Writing – review & editing. **Kevin Atcheson:** Formal analysis, Investigation, Methodology, Writing – review & editing. **Phil Jordan:** Conceptualization, Formal analysis, Funding acquisition, Writing – review & editing, Methodology, Supervision.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Phoebe Morton reports financial support was provided by Special EU Programmes Board.

Data availability

The data used in this study are confidential.

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Appendix A. Supplementary material

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