The effect of forest-to-bog restoration on the hydrological functioning of raised and blanket bogs

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Abstract

The carbon sequestration potential of peatlands has led to increasing interest in the restoration of bogs previously subjected to plantation forestry. However, little information exists about the effects on hydrological processes of what has become known as forest-to-bog restoration. The hydrological functioning of three afforested, two intact and four forest-to-bog restoration sites was compared at a raised bog and blanket bog location. For the raised bog location, the annual runoff/rainfall coefficient was 59.7% for the intact site, 41.0% for the afforested site, and 53.1% for the oldest restoration site (9 years post-felling). At the blanket bog location, the coefficient was 80.6% for the intact site, 63.0% for the afforested site, and 71.6% for the oldest restoration site (17 years post-felling). Compared to intact bog, median peak storm discharge was significantly greater in the restoration sites for the raised bog location but not for the blanket bog location. Water-table peak lag times were greatest, and water-table depths deepest in the afforested sites and the most recent raised bog restoration site and least in the oldest blanket bog restoration site. The estimated contribution of overland flow in the afforested sites was 2.9% for the raised bog and 11.9% for the blanket bog, increasing to 8.7% and 32.2% at the oldest restoration sites for the raised bog and blanket bog, respectively. Overall, hydrological functioning of the raised bog and blanket bog restoration sites was different from the intact sites but was most similar to intact bog in the oldest restoration sites.

Keywords: peatland, forestry, restoration, water table, streamflow, overland flow, storm event response

1 Introduction

Globally, peatlands are now widely recognised for their potential to mitigate climate change through carbon sequestration (Scharlemann *et al.*, 2014; Yu *et al.*, 2010). In the boreal zone, more carbon is thought to be stored in peat soils than in the above-ground biomass, including natural forests (Apps *et al.*, 1993). Peatlands also provide a range of other important ecosystem services, including nature conservation (Stoneman *et al.*, 2016), freshwater supplies (Xu *et al.*, 2018) and downstream flow maintenance, influencing flood events (Acreman & Holden, 2013). However, peatlands are complex and fragile ecosystems, and many have been impacted by industrial development through changes in land use, air quality and global climate. A tightly-coupled feedback system exists between the peat, the native vegetation and the hydrology (Price *et al.*, 2016), and a suitable balance is required to sustain carbon sequestration via continuous accumulation of new peat in these ecosystems. Changes in land use and climate can disrupt this balance, altering the hydrology and the ecosystem services delivered.

Bogs are ombrogenous peatlands, being predominantly rain-fed. Blanket bogs occur where the underlying topography becomes covered in an extensive layer (blanket) of peat, and they can occur on sloping (up to 20^{o}) or flat terrain. Raised bogs usually occur on more gentle slopes and form a characteristic dome shape with deeper peat in the centre of the peatland (Charman, 2002). Hydrology is very important to the functioning of bogs. Near-surface water-table levels in bogs are widely regarded as the most crucial factor in maintaining the anoxic conditions necessary for peat accumulation (Holden *et al.*, 2015; Joosten *et al.*, 2016), as they slow down decomposition (Clymo, 1983). Also, high water tables maintain the growth of peat-forming plants such as *Sphagnum spp*. and *Eriophorum spp*. (González *et al.*, 2014) which sequester carbon from the atmosphere.

The flow of water in both intact raised bogs and intact blanket bogs is dominated by near-surface and surface pathways (Holden & Burt, 2003a; Ingram, 1982; van der Schaaf, 1999). Water received by precipitation either flows across the surface of the peat as overland flow or as subsurface flow through the shallow peat layers. In contrast, limited flow usually occurs in the denser, deeper layers except where there are macropores and soil pipes (Holden and Burt, 2003b). Saturation-excess overland flow has been found to contribute up to 82 % of streamflow in an intact upland blanket peatland in the UK (Holden & Burt, 2003a; Holden & Burt, 2003b). However, drainage of peatlands to improve the land for grazing and forestry, burning, and other disturbances have been shown to reduce the dominance of overland flow and increase subsurface flow (Acreman & Holden, 2013; Holden *et al.*, 2006; Holden *et al.*, 2015; Prévost *et al.*, 1999) so that the hydrological functioning of a disturbed peatland is quite different from that of an intact one.

Afforestation has been a significant source of peatland degradation throughout the world (Paavilainen & Päivänen, 1995; Strack, 2008), and concern over the changing climate has led to a global increase in peatland restoration (Andersen *et al.*, 2017; Bonn *et al.*, 2016) including restoration of peatlands that have previously been subject to plantation forestry (Anderson *et al.*, 2016). In the UK, 190000 hectares of deep peat was ploughed and planted with non-native coniferous trees between the 1950s and 1980s (Hargreaves *et al.*, 2003). Due to the naturally shallow water tables found in deep peat, artificial drainage was necessary to allow the trees to establish. Drainage and increased evapotranspiration from trees can significantly lower the water table (Anderson & Peace, 2017; Anderson *et al.*, 2016; Gaffney *et al.*, 2018; Muller *et al.*, 2015), reduce water yield and subdue streamflow response to rainfall (Bosch & Hewlett, 1982; Brown *et al.*, 2005; Sahin & Hall, 1996; Zhang & Wei, 2014). However, in some cases, drainage can provide a more efficient pathway for flow, particularly in the early stages of a forest rotation, enhancing water yield and streamflow response (Holden *et al.*, 2004; Robinson, 1986). Many coniferous plantation forests on peatlands in the UK are now reaching maturity, and more information is needed to understand the impacts of land management decisions such as felling and restocking or peatland restoration under 'forest-to-bog' initiatives (Anderson *et al.*, 2016) on the hydrological functioning of different bog types.

Robinson (1986) found forest drainage of the Coalburn catchment in Northumberland, northern England, led to a doubling of baseflow and an increase in annual streamflow by 50-100 mm after the ploughing of peaty soils. In the first five years after the trees were planted, mean storm peak lag times were reduced from 2.2 to 1.7 hours (Robinson, 1998). However, higher evapotranspiration rates as the trees matured and the infilling of drains with sediment, forest litter and vegetation reversed these effects with time. After 45 years, annual streamflow was 350 mm lower than before forestry operations began, although there was only a small difference in water yield for large storms (Birkinshaw *et al.*, 2014). Anderson *et al.* (2000) found baseflows to decrease and total annual streamflow to be reduced by 7% five years after afforestation in deep peat at Bad a' Cheo, Caithness, but the control in the study had also been drained. Other field studies on the hydrological effects of afforestation on peatlands have been undertaken (Archer, 2003; Bathurst *et al.*, 2018; Robinson *et al.*, 2003; Robinson *et al.*, 2012), yet only two paired catchment studies have used a near 100% afforested and near 100% open peatland as a comparison (Bathurst *et al.*, 2018; Marc & Robinson, 2007). Both studies reported smaller total annual streamflow (18 and 24%, respectively) in the afforested catchments relative to open peatland.

Forest harvesting has been associated with changes in streamflows. Sahin and Hall (1996) found a 10% reduction in coniferous forest increased the annual water yield by 20 - 25 mm from a regression analysis of 145 catchments throughout the world. However, such studies in peatland systems are not common. Robinson *et al.* (2003) noted that forest felling in deep peat at Glenturk in Ireland increased moderate peak flows, and there was a tendency for flow peaks and low flows to increase after partial felling at Plynlimon, Wales. However, Robinson *et al.* (2003) found changes in peak flows difficult to detect, which they suggested could be because of increased interception losses from the felled waste, which may also act like dams in furrows and drains attenuating runoff. It is not clear how long is required for the hydrological functioning of sites, which have been felled and left to rehabilitate naturally, to return to that of intact peatlands, or whether other restoration measures such as ditch blocking can reduce the time span required. Furthermore, to our knowledge, the wider impact of forest-to-bog restoration on storm runoff, streamflow regimes and downstream flooding has not been studied.

Clear-felling alone may not raise the peatland water-table level sufficiently toward that of intact peatlands in the short term. Therefore, restoration after forest clearance often includes the damming or infilling of furrows and drains (Anderson & Peace, 2017; Haapalehto *et al.*, 2011; Laine *et al.*, 2011). Water-table recovery following felling and ditch blocking has been reported for blanket bogs (Anderson *et al.*, 2000; Anderson & Peace, 2017; Gaffney *et al.*, 2018; Muller *et al.*, 2015) and raised bogs (Haapalehto *et al.*, 2011; Komulainen *et al.*, 1999;

Menberu *et al.*, 2016). However, we are not aware of a published forest-to-bog restoration study on hydrological function for raised bogs in the UK. Additionally, there is limited understanding of how peatland water-table variability and water-table response to rainfall events differs between forest-to-bog restoration sites, afforested sites and intact sites. Holden *et al.* (2011) found the water-table dynamics of a restored blanket bog where ditch blocking had occurred at a non-forested site was quite different from that in nearby intact bog six years after restoration. We are only aware of one restoration study (Menberu *et al.*, 2016) that has reported the water-table dynamics in afforested peatlands, comprising the infilling of drainage ditches, construction of peat dams and surface barriers, and tree removal if significant growth had occurred since drainage. Menberu et al. (2016) observed water-table depths, fluctuations, hydrograph recession slopes, and measures of groundwater recharge reflected those found in natural peatlands 1-6 years after re-wetting, particularly for nutrient-poor spruce mires. However, they did not specify whether felling had contributed to the recovery.

This study seeks to compare the hydrological functioning of nearby intact, afforested and forest-to-bog restoration sites at a raised bog and blanket bog location. We compare the water balance, streamflow dynamics, water-table dynamics and overland flow occurrence between the different sites. We hypothesise that for each of the two types of peatland (raised bog and blanket bog), the water yield would be greatest for the intact bogs, followed by the restoration sites and least for the afforested bogs. We also hypothesise that water tables would be deepest in afforested sites and deeper and more variable in the restoration sites after clear-felling than for intact bog. As a result, overland flow was expected to be least common on forested sites, followed by the restoration sites and then intact sites. Furthermore, we hypothesise that streamflow storm response would be more subdued (smaller peaks, longer lag times) in afforested sites, followed by restoration sites, than for intact systems. Finally, we hypothesised that sites that had been under restoration the longest would have hydrological functioning that was most similar to intact bogs compared with sites where the restoration was most recent.

2 Methods

2.1 Study sites

The raised bog (RB) location is situated at Flanders Moss, in the floodplain of the River Forth, Central Scotland (Figure 1), one of a series of lowland raised bogs formed by the uplifted former estuary of the river (56° 08'10.5"N, 4°19'28.7"W). The blanket bog (BB) location is at Forsinain, in the 'Flow Country' region of northern Scotland (58°25'35.6"N, 3°52'09.1"W), Europe's largest expanse of blanket peat (c. 4000 km^2). The mean annual precipitation between 1981 and 2010 (Met Office *et al.*, 2018) was 1443.7 mm at Flanders Moss and 1096.9 mm at Forsinain. The mean annual temperature was 8.7 °C at Flanders Moss and 7.4 °C at Forsinain over the same period.

Closed canopy coniferous forestry plantation sites (afforested bog (AB)), open, near-natural bog (intact bog (IB)) and two forest-to-bog restoration sites of different ages (R1 and R2) were included to represent the different land uses. R1 was the oldest restoration site at each location, although the method and timescale of restoration varied between sites and locations (Table 1). There were two afforested sites at Flanders Moss as, after the first site was instrumented (RBAB1), osprey nesting (protected species) restricted access throughout the study period, so a second site (RBAB2) was established. The slopes at each site were broadly comparable, and afforested sites were selected where the whole area was under tree cover.

RBAB2 and the raised bog restoration sites are located on what is known as ‘Flanders Moss West’, whereas the IB site is located on Flanders Moss National Nature Reserve to the east. RBAB1 is located to the northeast of Flanders Moss West on an area known locally as ‘Cardross Moss’. Flanders Moss West was drained in the 1920s to improve conditions for grouse shooting, and in the 1960s and 1970s was planted with Lodgepole pine (*Pinus contorta*) and Sitka spruce (*Picea sitchensis*). RBAB1 had been planted with the same mixture of tree species as Flanders Moss at a similar period, although the catchment was dominated by mature Lodgepole pine. We include data from RBAB1 in this paper because nearby forest operations limited the choice of the catchment area for RBAB2, and equipment problems arose trying to measure the flows at RBAB2. We included RBAB2 in the study so that we could investigate overland flow and water-table dynamics data that we could not regularly sample from RBAB1 because of site access restrictions described above.

Table 1 - Site characteristics at Flanders Moss (RB) and Forsinain (BB).

Site	Description	Felling Dates	Deforestation and Re-wetting Actions	Furrow Spacing (m)	Catchment Area (ha)	Outflow Location	Planting year
RBAB1	Cardross Moss afforested bog (AB)			1.4	0.7	56°09'48.0"N 4°17'03.5"W	~1965
RBIB	Flanders Moss intact bog (IB)				6.0	56° 9'47.00"N 4°10'52.29"W	
RBAB2	Flanders Moss afforested bog (AB)			1.4	0.2	56° 9'10.12"N 4°20'1.54"W	~1965
RBR1	Flanders Moss restoration site 1 (R1)	24/11/2009 - 09/12/2009 01/08/2011 - 18/10/2011	Part conventional harvesting; part low impact harvesting and removal of brash and logs.	1.4	2.5	56° 8'12.88"N 4°19'35.19"W	~1965
RBR2	Flanders Moss restoration site 2 (R2)	01/10/2013 - 31/03/2014	Conventional harvesting (i.e. fell, debranch, extract timber, leave brash).	1.4	26.2	56° 8'27.24"N 4°19'19.27"W	~1965
BBIB	Forsinain intact bog (IB)				1.6	58°25'10.32"N 3°51'41.01"W	
BBAB	Forsinain afforested bog (AB)			1.9	5.1	58°25'30.85"N 3°52'14.67"W	~1980
BBR1	Forsinain restoration site 1 (R1)	2002-2003	Originally felled-to-waste – furrows & collector drain blocked. Brash compressed into furrows.	1.4	1.6	58°25'58.49"N 3°51'18.76"W	~1980
BBR2	Forsinain restoration site 2 (R2)	2014-2015	Mulched – collector drain blocked.	2.3	2.3	58°25'32.21"N 3°51'44.25"W	~1980

RBIB represents the best example of near-natural bog in the area with a mosaic of *Sphagnum* mosses (including some nationally scarce species: *S. austinii*, *S. fuscum* and *S. molle*), sedges, ericaceous shrubs and sundews. RBR2 is the larger of the two restoration catchments at Flanders Moss and was felled over six months between 2013 and 2014 using a conventional harvester and forwarder (Shah & Nisbet, 2019). The main tree stems were extracted from the site, but lesser tree debris (brash) was left to decompose on the peat surface or in furrows and drains. RBR1 was felled in two phases: the first in the winter of 2009 (15%) and the remainder in summer/autumn 2011. The first phase of felling was carried out using standard forest harvester and forwarder techniques, whereas the second phase was carried out by hand and winching the timber out by an overhead Skyline (Shah & Nisbet, 2019). All useable timber and brash were removed from RBR1. No other peatland restoration work took place in the catchments during the monitoring period, although drain blocking and other re-wetting treatments are scheduled for this site.

The vegetation at BBIB was similar in composition to that at RBIB with the addition of liverworts, bog asphodel (*Narthecium ossifragum*) and bogbean (*Menyanthes trifoliata*) in natural pools. At the southern end of BBIB, there was evidence of prior peat cutting, and trees had been planted to the east and west, but mainly it is a good example of near-natural bog, typical to the area. In the 1980s, parts of Forsinain were drained and planted with the same mixture of tree species as the Flanders Moss, but there was a difference in the ploughing/planting phase with the furrows being 50 and 90 cm further apart in BBAB and BBR2, respectively. BBR2 differed from the other restoration sites because it was the only one where the standing trees had been 'whole tree mulched' as harvesting the timber was not economically viable, resulting in a layer of masticated tree debris being left on the peat surface. The main drain at BBR2 had been blocked with a sequence of plastic piling dams at the outflow after mulching in 2014, but further peat dams were added on 23 March 2019, 12 months after monitoring had started in the catchment. BBR1 was originally felled-to-waste in 2002-03 when the trees were comparatively young (~20 years old). The resulting brash was compacted into the furrows, which were blocked with peat dams in 2015-16 at the same time as the main collector drain.

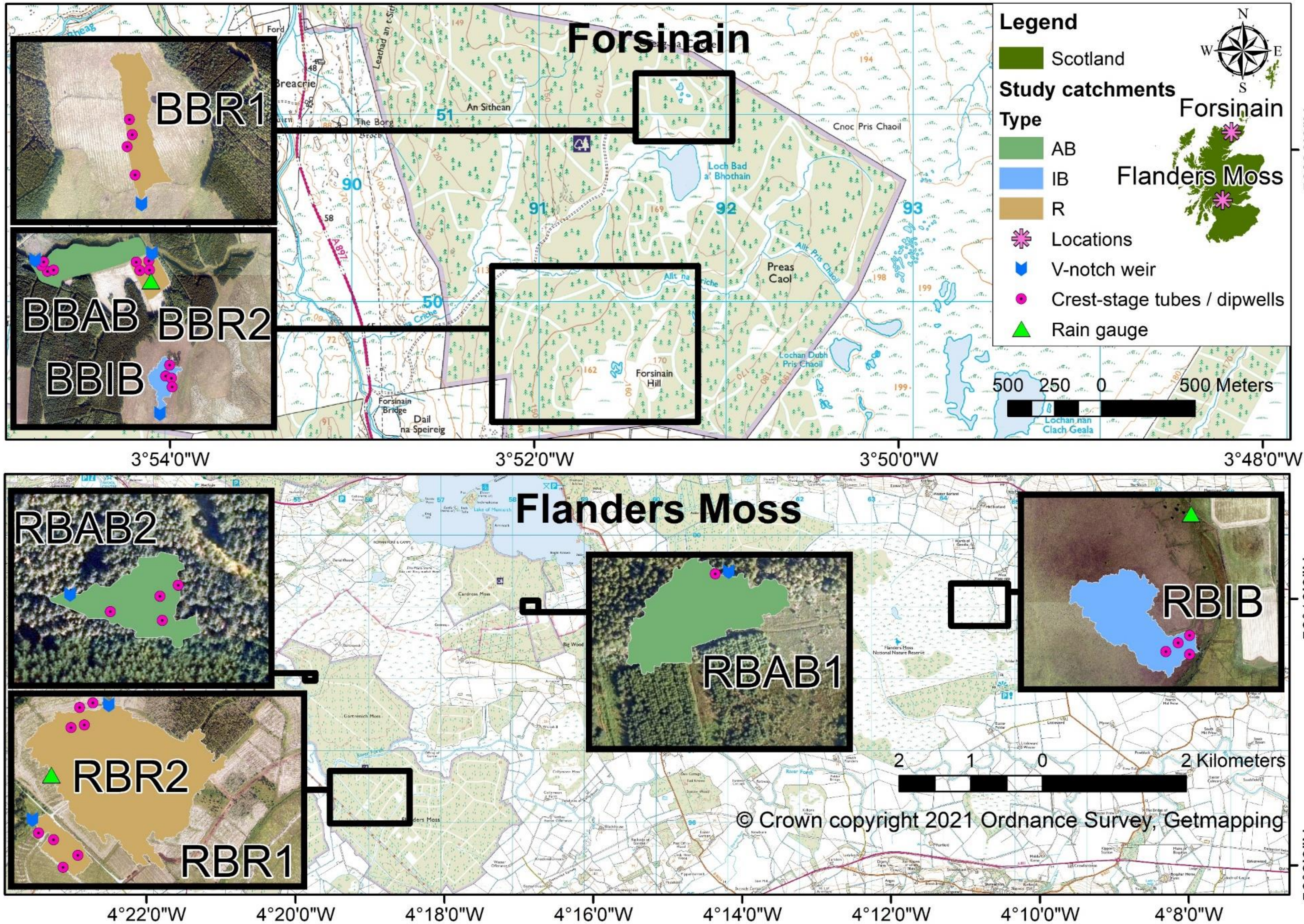

Figure 1 - Study site experimental design at the blanket bog (BB) and raised bog (RB) locations; AB = afforested bog; IB = intact bog; R = under restoration where R1 was restored before R2.

2.2 Field sampling and measurements

Within each site, four sampling nests were created, each comprising one dipwell and one crest-stage tube, which were carefully inserted into the peat after a hole had been augured of a slightly smaller diameter to the tubes. A stratified random sampling procedure was used to allocated locations within each site, using the “Create Random Points” tool in ArcGIS (ESRI, 2017). One of each of the natural bog microforms (hollows, hummocks and lawns) and one of each of the afforested and restored bog surface features (ridges, furrows and original surface) were included in the designated nest locations. The distance between the nests was generally > 30 m. The different surface features associated with afforestation and natural bog microforms including hollows (n = 2), hummocks (n = 3) and lawns (n = 3) in the intact bogs, and ridges (n = 2), furrows (n = 8), and the original surface (n = 14) in the afforested and restored bogs and vegetation were recorded at the time of installation. The dipwells were constructed from PVC tubing, generally > 1 m in length and with 0.5 cm diameter holes drilled at 3.5 cm intervals throughout the length of the tube with four holes at each interval. The base was sealed with a PVC plug. Caps were fitted to the tops of both crest-stage tubes and dipwells to prevent debris and insect ingress. The crest-stage tubes were formed from a short section of PVC tubing with a single ring of 0.5 cm diameter holes inserted so that they were level with the top of the peat layer to collect any overland flow. On each site visit, the crest-stage tubes were examined for evidence of overland flow occurring between visits, recording a presence/absence, and then emptied with a plastic syringe. Additional dipwells were added in furrows (n = 3), and the original surface (n = 2) at RBR2 and in hollows (n = 3) and hummocks (n = 2) at RBIB for further manual measurements to assess spatial variability.

Two tipping bucket rain gauges were installed at Flanders Moss (RBR2: Davis 7852 + Hobo H07-002-04 event logger; RBIB: Davis 6465 + Hobo UA00364 temperature/event logger) to account for any localised rain showers between the sites. A single tipping bucket rain gauge was installed at Forsinain, where the sites were relatively close to each other (BBR2: Davis 6465 + Hobo UA00364 temperature/event logger). The Hobo sensors were installed in the rain gauge housing, so Met Office temperature data was used for more accurate air temperature readings. One dipwell at each site was instrumented with a Level Troll 500 vented pressure transducer (in lawns or the original lawn surface) to record high-temporal water-table measurements, except at RBIB, RBR2 and RBR1, where there were two instrumented dipwells at different periods in the study. Where a single dipwell was used, we assumed that spatial differences in the water-table depth would be insignificant compared to the differences between the different land uses, particularly as lawns were consistently used for the datalogger wells. Data were collected between November 2017 and December 2019, but the synchronised monitoring of all catchments occurred between July 2018 and October 2019.

The catchment areas for each site were delineated in ArcGIS (ESRI, 2017) using high-resolution LiDAR imagery (50 cm x- y resolution except RBIB, which was 1 m resolution). At the outflows of the catchments, 30° V-notch weirs were installed. The weirs were constructed from 5 mm thick aluminium sheet and machined, so there was a sloping bevel at approximately 60° on the front face of the V. The aluminium sheets were held in place by wooden fencing posts driven into the peat. A stilling well was attached to another fencing post in the stream channel behind the weir with a flush fitting cap to prevent unwanted ingress. Level Troll 500 vented pressure transducers were lowered into the stilling wells and allowed to rest on the stream bed. Each pressure transducer was set to log at 15-minute intervals, and they were all synchronised. RBIB was the only site that did not have a well-defined outflow channel, and there was little surface flow except in the wettest periods at the catchment outlet. However, there was a clearly defined catchment determined from the LiDAR imagery. The near-surface discharge at RBIB was therefore calculated using the groundwater flow method based on Darcy's law. The law states that discharge through a porous medium is equal to the hydraulic gradient multiplied by the hydraulic conductivity and the cross-sectional area. The RBR2 catchment outlet was close to the stream's junction with the River Forth, which occasionally backed up into the catchment, affecting the head level at the weir.

On each site visit, water-table depth was recorded manually, using a steel capillary tube with adhesive scale, at each dipwell. Manual measurements were also used to calibrate the dipwells, which had been instrumented with pressure transducers. Similarly, the V-notch weirs were manually calibrated by measuring the time for water falling over the crest to fill a known volume of a receptacle. A camera was secured next to the shallowest weir (RBR1) to monitor site conditions remotely and record any overtopping events that may occur throughout the study period.

For RBIB, the saturated hydraulic conductivity (K_{sat}) was measured using piezometer slug tests in which the time taken for the hydraulic head to recover to equilibrium after a fixed volume is removed or added is measured (Baird *et al.*, 2008; Baird *et al.*, 2004; Surridge *et al.*, 2005). Saturated hydraulic conductivity was measured near the catchment outflow at 20, 40, 60 and 80 cm depths and the cross-sectional area calculated from the width of the lower end of the catchment, delineated in ArcGIS, multiplied by the average peat depth (assuming a uniform depth). Precisely machined piezometers were used for the slug tests with slotted 10 cm intakes and an inner tube diameter of 2.9 cm, another generation of the piezometers featured in Baird *et al.* (2004). The hydraulic gradient was measured continuously between two instrumented dipwells within the catchment by calculating the difference in hydraulic head divided by the distance between the two dipwells. TOPMODEL simulations were run in R (Buytaert, 2018), supplying it with an initial value of subsurface flow (calculated by Darcy's law) and the surface hydraulic conductivity to estimate the overall discharge, including the overland flow component. LiDAR data, at the previously mentioned scales, were used to calculate the topographic index. Simulations were also run for the other catchments to estimate the overland flow contribution by inputting TOPMODEL with the stream water observations from those sites.

2.3 Data analysis

All time-series data were processed in R Studio (RStudio-Team, 2016), and the Kalman filter (Helske, 2017) was applied to water-table data to smooth it for seasonal display purposes. Any calculations were performed on the raw data. Annual water balance summaries were produced from the precipitation and discharge data for each catchment, and annual storage change calculated from the water-table fluctuations and laboratory measurements of the specific yield from Howson et al. (in prep). Runoff/rainfall coefficients were calculated from the total discharge (mm) and total rainfall (mm) for each catchment over the water year 1 October 2018 – 30 September 2019 and for the entire study period 27 November 2017 – 3 December 2019 (supplementary information – Table S1). Potential evapotranspiration (PET) was estimated from Met Office (Met Office *et al.*, 2018) air temperature measurements (Fuka *et al.*, 2018) based on the Priestley and Taylor (1972) equation (Archibald & Walter, 2014). The net radiation was estimated from temperature data using an average surface albedo of 0.18 and a Priestley Taylor constant (α) of 1.26. The closest weather stations with continuous air temperature records over the study period were Bishopton, Glasgow (27.3 km), and Kinbrace (17.5 km), for Flanders Moss and Forsinain, respectively. Actual evapotranspiration (AET) was estimated by subtracting the total discharge from total rainfall viewed within the context of the storage change calculations outlined above.

The baseflow index (BFI), defined as the ratio of baseflow to total stormflow over a given period, was calculated using a Lyne and Hollick (1979) derived baseflow filter ($\alpha = 0.975$) produced by Bond (2019). Flow duration curves were plotted from quantile-quantile plots of the base-10 logarithm of discharge, at 15-minute intervals, divided by mean discharge using ggplot2 for each of the streams. “stat_qq” and “qnorm” were used for the quantile-quantile plots and calculating the percentage exceedance axes breaks, respectively. A constant of 1 was added to allow base-10 logarithms to be calculated when discharge was zero. The discharge was divided by the mean over the whole time series to compensate for the different catchment sizes.

Storm discharge metrics were calculated by processing precipitation and discharge data from a synchronised time-series for each site. The 90% quantile of discharge was taken to extract the significant storm events, and a baseflow separation algorithm ($\alpha = 0.95$) used to determine where the quickflow component was zero to signify the start and end of storm events (Fuka *et al.*, 2018). Any storm events that had missing rainfall or other anomalies which would impact on the calculated metrics were discarded (< 5%), but all streamflow measurements were included. Peak discharge, the time from peak rainfall to peak discharge and the time from peak discharge to where the quickflow had returned to zero were computed to compare storm hydrographs between sites. The hydrograph intensity, used as an indication of flashiness, was also calculated by dividing the peak flow by the product of total storm discharge and a scaling factor of 10^{-6}.

Water-table storm metrics from the instrumented dipwells were calculated similarly to the storm discharge metrics without the aid of a baseflow separation algorithm, and storms selected by taking the 95% quantile of precipitation. In this case, peak lag and recession lag times were taken as the time between the water table rising by 0.1 cm to a peak and the time taken for it to recede to the same level after the rainfall event. Minimum, maximum, and mean monthly values were plotted using ggplot2 and boxplots produced in the same package for the underlying metrics.

Statistical analyses were performed in SPSS (IBM-Corp., 2016), by firstly testing for normality and homogeneity of variance, and where possible parametric ANOVA tests of differences in the mean values of each group were used to test any hypotheses and identify any significant differences between sites and location (Flanders Moss/Forsinain). Where the data deviated from a normal distribution or homogeneity of variance was not satisfied, it was transformed in SPSS, or non-parametric Kruskal-Wallis tests were used. *Post-hoc* tests were used to determine significant differences for parametric tests, and pairwise comparisons were used for the same purposes for non-parametric Kruskal-Wallis tests. Correlations were calculated in SPSS, using Spearman's rank correlation coefficients (r_s), and Mann-Whitney U tests were used for non-parametric analysis of differences between the locations.

3 Results

3.1 Climate conditions during the study

The total monthly rainfall and mean monthly air temperatures from April 2018 until the end of November 2019 are shown in Table 2 for both locations. In 2018, the annual precipitation was 1001 mm and 741 mm at Flanders Moss and Forsinain, respectively, with the driest spring/summer in fifteen years (Met Office *et al.*, 2018). Mean monthly temperatures ranged from 2.7 to 16.6 °C at Flanders Moss and 1.7 to 15.4 °C at Forsinain over the study. The period between April 2018 and August 2018 was unusually warm and dry at both locations, and, in Scotland, the summer of 2018 was the sixth warmest since 1884 (Met Office *et al.*, 2018). At Forsinain, no rain was recorded for 36 consecutive days between 15 June and 21 July 2018.

Table 2 - Temperature and rainfall at Flanders Moss (raised bog) and Forsinain (blanket bog) from January 2018 – December 2019, where Temp = temperature (°C) and P = precipitation (mm).

	Flanders Moss				**Forsinain**			
	Temp (°C)			**P (mm)**	**Temp (°C)**			**P (mm)**
Month/Year	**Mean**	**Min**	**Max**	**Total**	**Mean**	**Min**	**Max**	**Total**
Jan-2018	2.9	-7.0	11.7	168	1.8	-8.6	9.2	75
Feb-2018	2.7	-5.5	11.5	32	1.7	-6.2	9.8	39
Mar-2018	3.5	-5.7	11.3	72	2.6	-7.5	11.1	81
Apr-2018	7.9	-2.9	18.9	64	7.1	-5.3	18.6	41
May-2018	12.0	-1.3	25.2	23	11.3	-2.3	24.4	25
Jun-2018	15.0	5.5	31.7	99	13.0	3.3	27.2	13
Jul-2018	16.6	6.7	26.9	38	15.4	5.5	28.2	47
Aug-2018	14.2	3.9	22.8	72	13.1	2.8	23.3	47
Sep-2018	11.7	1.4	20.1	93	10.5	2.7	24.8	138
Oct-2018	9.1	-3.6	20.1	94	8.0	-4.4	18.8	130
Nov-2018	7.2	-0.7	14.7	154	6.3	-3.3	14.9	53
Dec-2018	4.9	-3.8	12.0	93	4.1	-5.0	12.1	53
2018	**9.0**	**-7.0**	**31.7**	**1001**	**7.9**	**-8.6**	**28.2**	**742**
Jan-2019	3.4	-6.0	11.2	34	2.4	-6.6	10.9	131
Feb-2019	5.9	-7.6	14.9	67	5.6	-6.4	17.3	58
Mar-2019	6.4	-2.9	13.4	147	5.6	-2.7	13.8	107
Apr-2019	8.7	-2.1	23.5	29	8.1	-3.0	21.8	60
May-2019	9.9	-0.6	23.6	65	8.0	-3.6	21.6	98
Jun-2019	13.5	4.1	26.4	52	11.9	3.7	26.0	68
Jul-2019	16.6	6.2	29.6	85	15.0	3.0	27.0	65
Aug-2019	15.6	7.0	27.9	130	13.8	5.3	26.4	154
Sep-2019	12.7	3.2	22.2	72	11.3	1.1	22.7	52
Oct-2019	8.3	-1.7	16.1	104	7.2	-3.6	14.1	65
Nov-2019	4.8	-5.7	12.4	68	3.8	-6.0	9.8	140
2019	**9.7**	**-7.6**	**29.6**	**854**	**8.5**	**-6.6**	**27.0**	**1000**

3.2 Water balance

A comparison of the precipitation, discharge and estimated evapotranspiration for the 2018/19 water year is given in Table 3a. The annual groundwater storage change varied from 0.2 to 13.6 mm and was lowest at BBR2 and highest at BBAB. Therefore, it was a small component of the water balance, and our AET values can be deemed to be a reasonably reliable estimate. The runoff/rainfall coefficient was least for their respective locations in the two afforested catchments, which experienced the greatest evapotranspiration losses. Overall, evapotranspiration losses were significantly higher at the raised bog ($p < 0.001$ Mann-Whitney U Test) than the blanket bog location (412 – 742 versus 212 – 405 mm) and the runoff/rainfall coefficients lower (41.4 – 59.7 versus 64.9 – 80.6%). PET was also higher at the raised bog, and AET at RBIB was almost double that at BBIB. The runoff/rainfall coefficient was comparable between RBIB and RBR2. However, new dams added to the outflow of BBR2 on 23 March 2019 redirected some of the flow away from the weir. Therefore, the runoff/rainfall coefficient is lower than expected and similar to that at BBAB. We include water balance calculations for the blanket bog sites before (Table 3b) and after (Table 3c) this event to highlight the impact on the runoff/rainfall ratio.

*Table 3 (a) - Water balance and mean, maximum and minimum discharge for the eight catchments (1st Oct-18 – 30 Sept-19). (b) - Water balance at the blanket bog before (22/07/18 - 23/03/19) and (c) - after new dams (23/03/2019 - 03/12/2019.). RBAB1 was taken as the afforested catchment for the raised bog. P = precipitation (mm); Q = total annual discharge (mm); Mean Q = mean annual discharge (L s^{-1})/ (mm d^{-1}); Max Q = maximum annual discharge (mm d^{-1}); Min Q = minimum annual discharge (mm d^{-1}); Runoff/rainfall = Q/P (%); PET = potential evapotranspiration using the Priestley and Taylor (1972) equation; AET = actual evapotranspiration P-Q (mm). * - rainfall from the Flanders Moss National Nature Reserve rain gauge.*

	Site	P (mm)	Q (mm)	Mean Q (L s^{-1})	Mean Q (mm d^{-1})	Max Q (mm d^{-1})	Min Q (mm d^{-1})	Runoff/ rainfall (%)	PET (mm)	AET (mm)
a	RBIB	1022*	610	1.2	1.7	10.1	0.2	59.7	692	412
	RBAB1	1267	524	0.1	1.4	16.8	0.0	41.4	692	742
	RBR1	1267	673	0.5	1.8	16.1	0.0	53.1	693	593
	RBR2	1267	751	6.3	2.1	87.9	0.0	59.3	693	516
	BBIB	1093	881	0.5	2.4	35.1	0.0	80.6	596	212
	BBAB	1093	688	1.1	1.9	35.5	0.0	63.0	596	405
	BBR1	1093	782	0.4	2.1	35.5	0.0	71.6	596	311
	BBR2	1093	710	0.5	1.9	31.7	0.0	64.9	596	383
b	BBIB	778	561	0.4	2.3	23.6	0.0	72.1	202	217
	BBAB	778	550	1.3	2.3	21.0	0.0	70.7	202	228
	BBR1	778	549	0.4	2.3	21.4	0.0	70.6	202	229
	BBR2	778	693	0.7	2.8	31.7	0.0	89.1	202	85
c	BBIB	762	607	0.5	2.4	35.1	0.0	79.6	578	156
	BBAB	762	309	0.7	1.2	35.5	0.0	40.6	578	453
	BBR1	762	459	0.3	1.8	35.5	0.0	60.1	578	304
	BBR2	762	197	0.2	0.8	9.2	0.0	25.9	578	565

3.2.1 Seasonal rainfall, runoff and water table

The dry spring/summer of 2018 coincides with a steep drop in the water table at both locations (Figure 2). The magnitude of storm peaks was similar except in a period of intense thunderstorms in the summer of 2019 at the raised bog and spring/summer 2019 at the blanket bog location where RBR2 and BBIB experienced higher discharges than the other sites relative to their catchment areas. Some of these peak discharges at RBR2 probably resulted from backing up of the River Forth into the catchment, so they do not necessarily reflect RBR2 stream dynamics. The most intense rainfall events at both locations occurred on

31 August 2019 for the raised bog, which overtopped the weir at RBR2, and 5 August 2019 at the blanket bog, which led to downstream flooding the next day.

The baseflow components over the study period were variable between the different sites. RBIB had the highest baseflow index (BFI = 0.86) at the raised bog, where there was no obvious channelling of water. The BFI at RBAB1 was 0.70, and the restoration catchments had the lowest BFI (RBR1 = 0.65; RBR2 = 0.52;). The BFI at the blanket bog differed, being highest in the two restoration catchments (BBR1 = 0.68; BBR2 = 0.77) and lowest in the forestry (0.64). The BFI for BBIB was 0.66. The flow duration curves at the blanket bog were comparable across the sites, except for the highest 1 % of flows where there was a step-change in discharge in the AB and R1 sites (Figure 3). The curves for BBIB and BBR2 were a smooth S-shape characteristic of low-variability flows and attenuated runoff. At the raised bog, the gradient of the flow duration curves was very similar for AB and R1, but IB and R2 were somewhat different in their response. RBIB experienced a very gentle gradient curve indicative of low variability in discharge, whereas the curve steepened for RBR2, showing extreme peaks for the top 1 % of flow conditions in comparison to the other streams. However, the River Forth backing up may account for some of these. The flatter curve at RBIB is characteristic of greater influence from groundwater discharge.

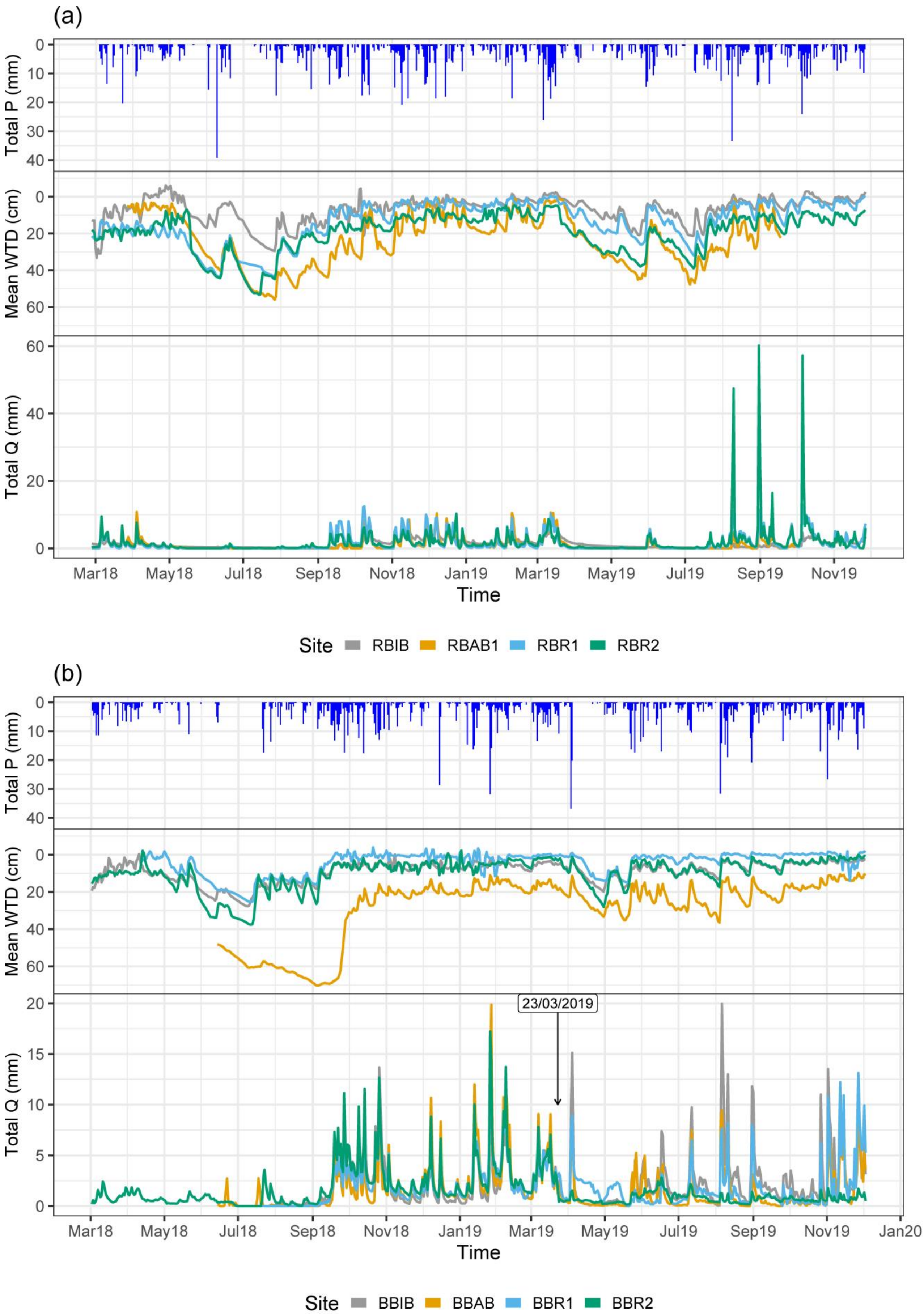

Figure 2 – Total daily precipitation (P), mean daily water-table depth (WTD) and total daily discharge (Q) for the catchments at (a) raised bog and (b) blanket bog locations. New dams were added to the outflow of BBR2 on 23 March 2019, as indicated. Note the difference in the y-axis for total Q between the locations. The discharge was similar at both locations except for extreme events at RBR2.

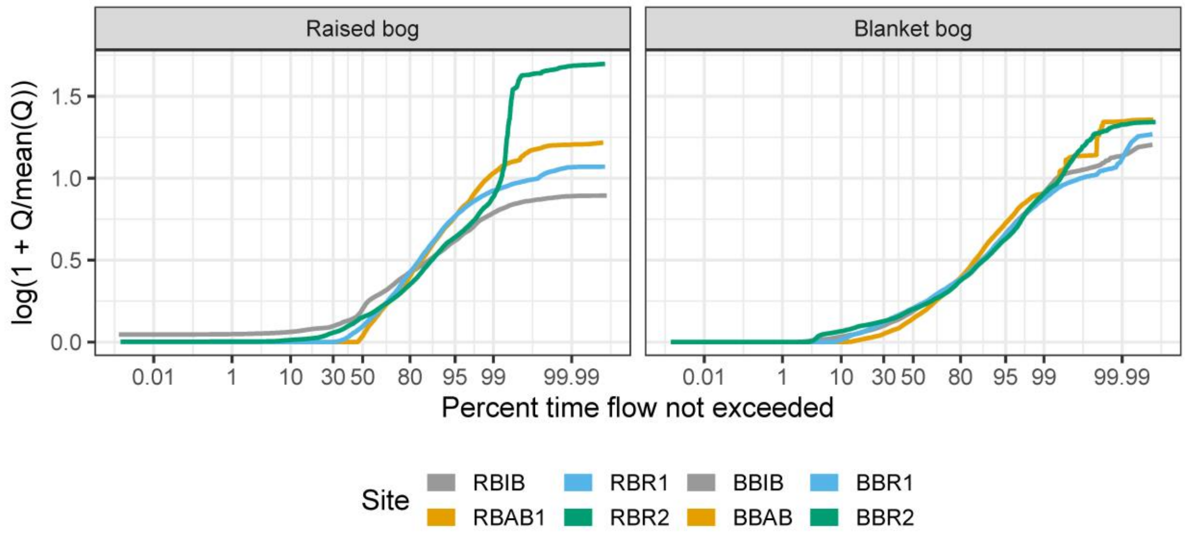

Figure 3 – Flow duration curves at 15-minute time steps for the study streams where RB = raised bog and BB = blanket bog.

3.2.2 Storm response

A summary of the stormflow metrics for the study period is presented as boxplots in Figure 4, and the mean values are included in Table S2 in the supplementary information. Areally-weighted peak storm flows (as mm d^{-1} for the whole catchment) were significantly higher ($p < 0.001$ Mann-Whitney U Test) for the blanket bog (mean = 12.09 ± SD 6.53 mm d^{-1}), where the runoff/rainfall coefficients were greater (Table 3), than the raised bog (mean = 9.77 ± SD 9.78 mm d^{-1}). At the raised bog, peak flows were significantly higher in the two restoration sites than RBIB ($p < 0.001$ Kruskal-Wallis Test), whereas, at the blanket bog, peak flows were significantly higher at BBIB than at BBR1, BBAB, and BBR2 ($p < 0.05$ Kruskal-Wallis Test). Peak lag times were significantly longer ($p < 0.001$ Mann-Whitney U Test) at the raised bog than at the blanket bog and longest in the AB for each location. Peak lag times were shorter than for the IB in the oldest raised bog restoration site (means: RBIB = 14.94 ± SD 6.13 h, RBR1 = 7.07 ± SD 4.66 h), but not in the oldest blanket bog restoration site (BBIB = 6.48 ± SD 9.06 h, BBR1 = 6.61 ± SD 7.73 h). BBIB, BBR1 and RBR1 had significantly shorter peak lag times ($p < 0.05$ Kruskal-Wallis Test) than the other sites at their location.

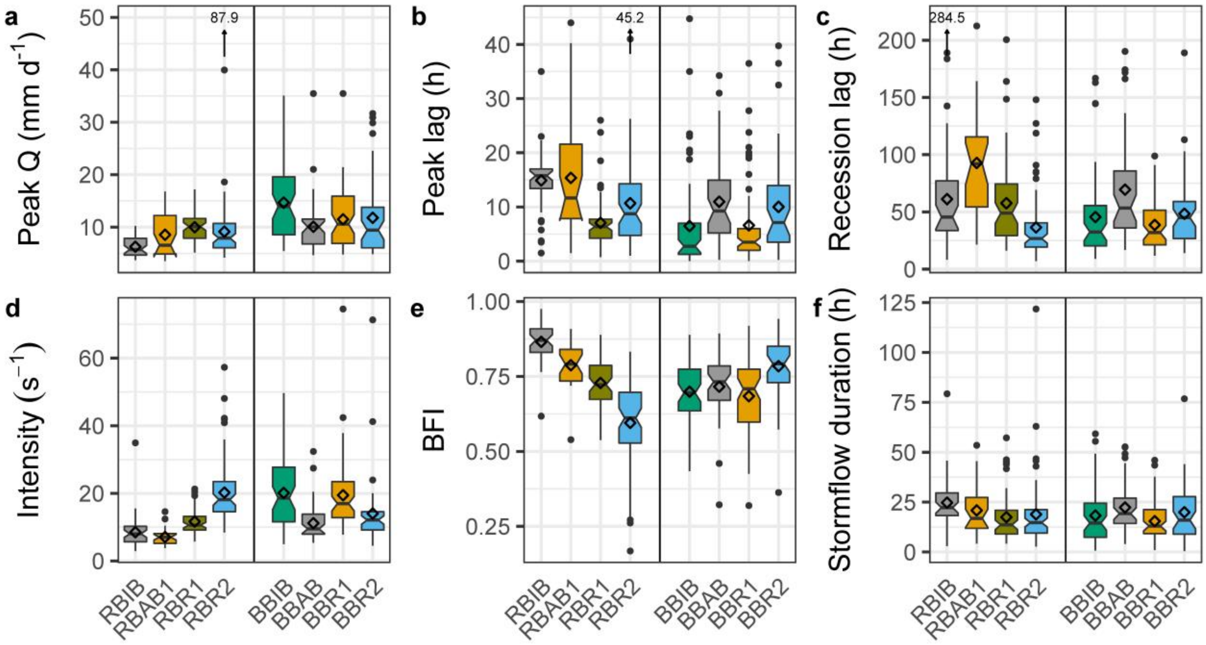

Figure 4 - Boxplots of mean stream storm metrics: a) Peak Q (mm d^{-1}); b) Peak lag (h); c) Recession lag (h); d) Hydrograph intensity (s^{-1}) = peak Q/(total Q for storm x 10^{-6}); e) Baseflow index f) Stormflow duration (h). The diamonds and horizontal lines indicate means and medians, respectively.

Recession lag times were significantly higher in the AB ($p <= 0.05$ Kruskal-Wallis Test) at both locations. RBIB, which had the highest baseflow component, had the second-highest recession lag times at the raised bog after the AB. The hydrograph intensity, which indicates the flashiness of the stream response, was highest for RBR2, BBIB and BBR1, with no significant difference between them. The hydrograph intensity was significantly higher for RBR1 than RBIB ($p < 0.001$ Kruskal-Wallis Test), but it was not significantly different between BBR1 and BBIB. The stormflow duration was a measure of how long stormflow persisted and was highest at RBIB (mean = 24.59 ± SD 12.68 h), but there was no significant difference between the raised and blanket bog locations (means: RB = 20.17 ± SD 14.40 h, BB = 18.70 ± SD 12.87 h).

3.3 Water-table dynamics

At the raised bog, the water table was at the surface longest at RBIB (10.66%), whereas RBAB1 and RBR2 were never at the surface. However, the peat was fully saturated fractionally longer at RBR1 (0.89%) than at RBAB2 (0.12%). At the blanket bog, the water table at BBAB was never at the surface, but it was at the surface for more time at BBR1

(21.66%) and BBR2 (0.95%) than at BBIB (0.25%). The annual mean water-table depth was similar for RBAB2 (21.3 ± SD 14.8 cm), RBAB1 (23.7 ± SD 14.2 cm) and BBAB (28.7 ± SD 17.1 cm). Water-table depths at the restoration sites were intermediate between the intact and afforested sites except for BBR1, where the water table was at the surface for longer than at any of the other sites (Figure 5).

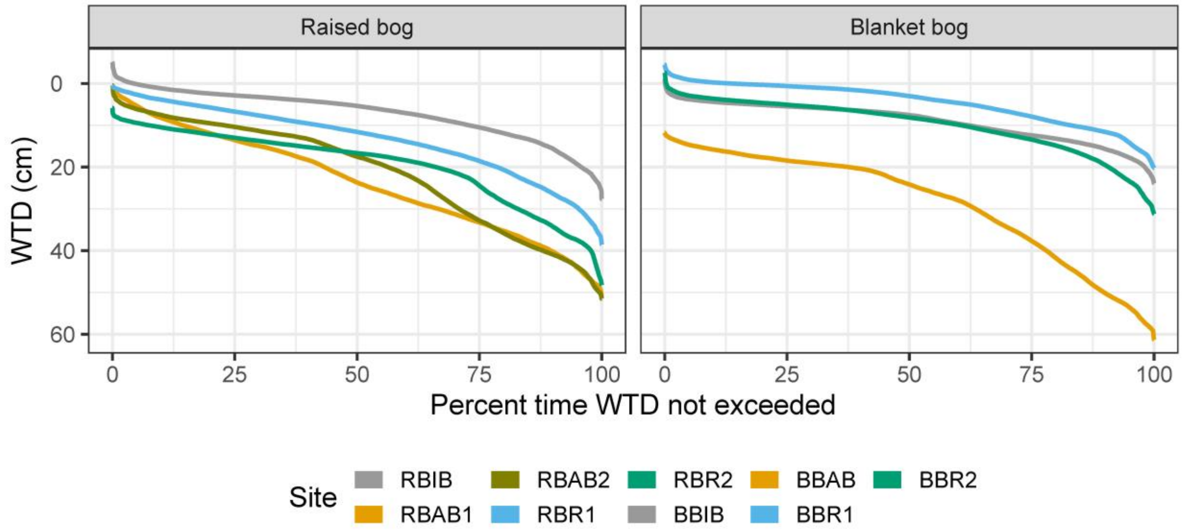

Figure 5 - Water-table depth exceedance curves for the raised bog (RB) and the blanket bog (BB) locations.

A monthly summary of water-table depth at all sites is presented in Figure 6. At the raised bog, the water-table depths at RBR1 were most strongly correlated with those at RBAB2 (r_s = 0.91, p < 0.001, N = 47708), but they were more strongly correlated with RBIB (r_s = 0.67, p < 0.001, N = 59491) than RBAB1 (r_s = 0.65, p < 0.001, N = 49985). The water-table depths at RBR2 were more closely correlated with those at RBAB2 (r_s = 0.89, p < 0.001, N = 47708) and RBAB1 (r_s = 0.79, p < 0.001, N = 51833) than those at RBIB (r_s = 0.65, p < 0.001, N = 49985). However, the water-table depths at BBR1 and BBR2 were more closely correlated with BBIB (BBR1: r_s = 0.76, p < 0.001, N = 51499; BBR2: r_s = 0.83, p < 0.001, N = 61483) than BBAB (BBR1: r_s = 0.65, p < 0.001, N = 51499; BBR2: r_s = 0.79, p < 0.001, N = 51499).

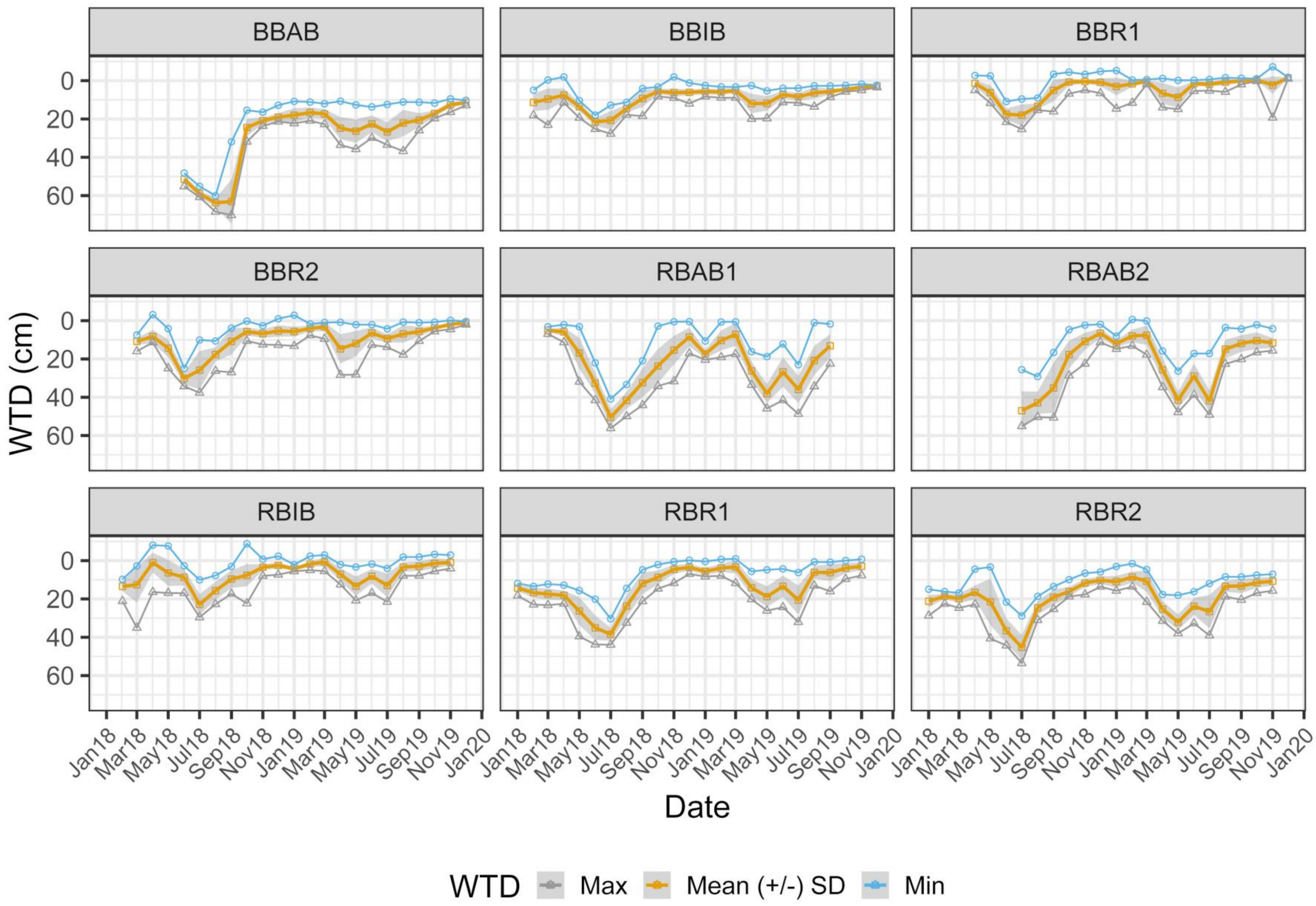

Figure 6 – Mean monthly ± SD, maximum and minimum monthly water-table depth (cm) from the instrumented dipwells. The average was taken from two automated dipwells at RBIB and RBR2, whereas the other sites had a single automated dipwell.

The water-table depth also varied between different forestry surface features and natural IB microforms. In the IB sites, the mean water-table depth across both locations was 5.5 cm in hollows (n = 5), 8.3 cm in lawns (n = 3) and 14.4 cm in hummocks (n = 5). In the forestry, the mean water-table depth was 21.1 cm in furrows (n = 3), 28.4 cm in ridges (n = 1) and 33.2 cm in the original surface (n = 4). For the restoration sites, mean water-table depth was 13.3 cm in furrows (n = 8), 19.5 cm in ridges (n = 1) and 17.7 cm in the original surface (n = 12). However, it is important to note that the surface features were not equally balanced, and ridges were underrepresented in this study.

3.3.1 Water-table fluctuations

The water table was generally deeper at the raised bog than at the blanket bog sites, except at BBAB (Figure 6), and fluctuated more throughout the seasons (Figure 2). At the raised bog, seasonal fluctuations were least at RBIB with the annual standard deviation (7.2 cm) less than at the AB (RBAB2=14.8 cm; RBAB1=14.1 cm) and the two restoration sites (RBR1=10.4 cm; RBR2=10.9 cm). Seasonal fluctuations in water tables were less in the two blanket bog restoration sites than the raised bog restoration sites, but annual standard deviations followed a similar pattern (BBIB=5.7 cm; BBAB=17.1 cm; BBR1=6.4 cm; BBR2=8.3 cm) except the standard deviation at the IB site was closer to R1 than at the raised bog. The water-table depth deviated away from the IB for the AB and R sites in dry periods at both locations, except BBR2 (Figure 2). As the sites began to re-wet, the differences between them decreased, but the AB sites took longer to recover. Water-table dynamics at BBR2 and BBIB were remarkably similar throughout the study period except in the summer drought of 2018, where the water-table depth at BBR2 receded beyond that at BBIB. The water table at BBR1 remained shallower than all the other blanket bog sites for most of the study period.

3.3.2 Storm response

A summary of the water-table storm metrics is presented in Figure 7, and the mean values are provided in Table S3 in the supplementary information. The mean rise in the water table in response to rainfall events was highest at RBAB1 and BBAB, and overall, it was significantly higher at the raised bog than the blanket bog location ($p < 0.001$ Mann-Whitney U Test). The mean peak water-table depth (when the water table was shallowest during each storm) was deepest in the two afforested sites, RBAB1 and BBAB, and both afforested raised bog sites had significantly lower peaks than at RBIB ($p < 0.001$ Kruskal-Wallis Test). RBR2 also had lower peaks in water tables than RBIB ($p < 0.001$ Kruskal-Wallis Test), yet they were not found to be significantly different between RBR1 and RBIB. There was no significant difference in the peak water table between the blanket and the raised bog, but water tables at BBR1 were significantly shallower than at all other sites ($p < 0.001$ Kruskal-Wallis Test).

The average duration between the commencement of rainfall and a detectable water-table rise was greatest at RBAB1 and BBR1 for the raised bog and blanket bog, respectively. However, there was no significant difference in the time to initial water-table rise between the raised bog and blanket bog locations. RBAB1 and BBAB had the longest water-table peak lag times for their location, and at RBAB1, they were significantly greater than all the other sites at the raised bog ($p < 0.05$ Kruskal-Wallis Test). RBR1 had significantly shorter water-table peak lag times than RBIB ($p = 0.009$ Kruskal-Wallis Test), but they were not found to differ significantly between sites at the blanket bog location. At the raised bog, water-table recession rates were significantly higher at RBAB1 ($p < 0.001$ Kruskal-Wallis Test) than the other sites; however, water-table recession at RBR1 was not significantly different from RBIB. At the blanket bog, recession rates for the afforested and both restoration sites were significantly higher than at BBIB ($p < 0.01$ Kruskal-Wallis Test). The 12-hour recession rate was significantly higher at the raised bog ($p < 0.001$ Mann-Whitney U Test) than the blanket bog, but other than that, no significant difference was found for water-table peak lag times and recession rates between the two locations.

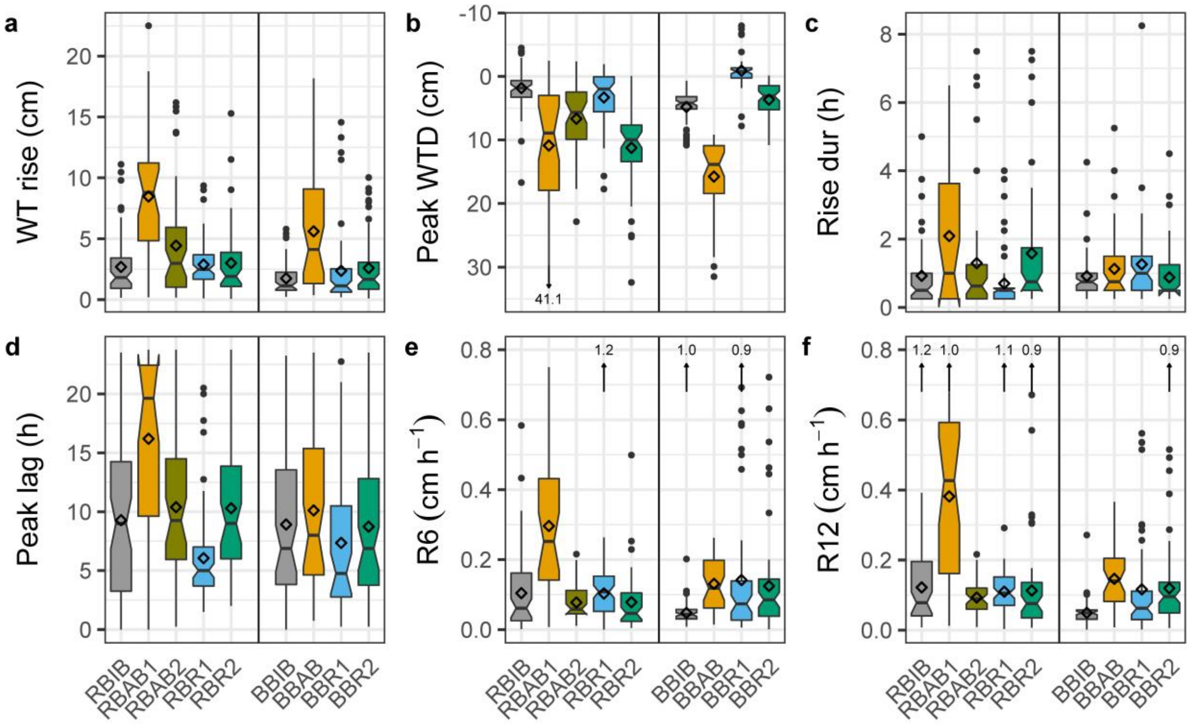

Figure 7 – Boxplots of water-table storm metrics: a) water-table rise (cm); b) peak water-table depth (cm); c) time for a rise of 0.1 cm in the water table (h); d) time from rainfall start to peak water-table depth (h); e) 6 hour recession rate (R6) (cm h^{-1}); f) 12 hour recession rate (R12) (cm h^{-1}). The diamonds and horizontal lines indicate means and medians, respectively.

3.4 Runoff processes

3.4.1 Overland flow

Overland flow was detected most frequently at the two IB sites, as expected. When spatially interpolated for the whole catchments, the overland flow frequencies appeared to be associated with elevation (Figure 8) and the expected topographic direction of flow in the catchments. Percentage overland flow occurrence between site visits was taken from the average of all crest-stage tubes at each site, where water had collected. At RBIB, overland flow was detected as occurring between 63.9% of site visits compared to 45.8%, 24.6% and 51.0% at RBAB2, RBR2 and RBR1, respectively. At BBIB, overland flow was detected between 86.0% of site visits compared to 29.5%, 73.9% and 61.5% at BBAB, BBR2 and BBR1, respectively. For the IB microforms, overland flow was detected on average for 88.2% of site visits on lawns compared to 68% in hollows and 64.4% in hummocks. Overland flow was detected, on average, across all crest-stage tubes in the forestry, on 41.2% of visits in furrows compared to 36.2% for the other surface features. In the restoration sites, the frequency of overland flow detection was 64.6% and 43.7% for furrows and the remaining features, respectively.

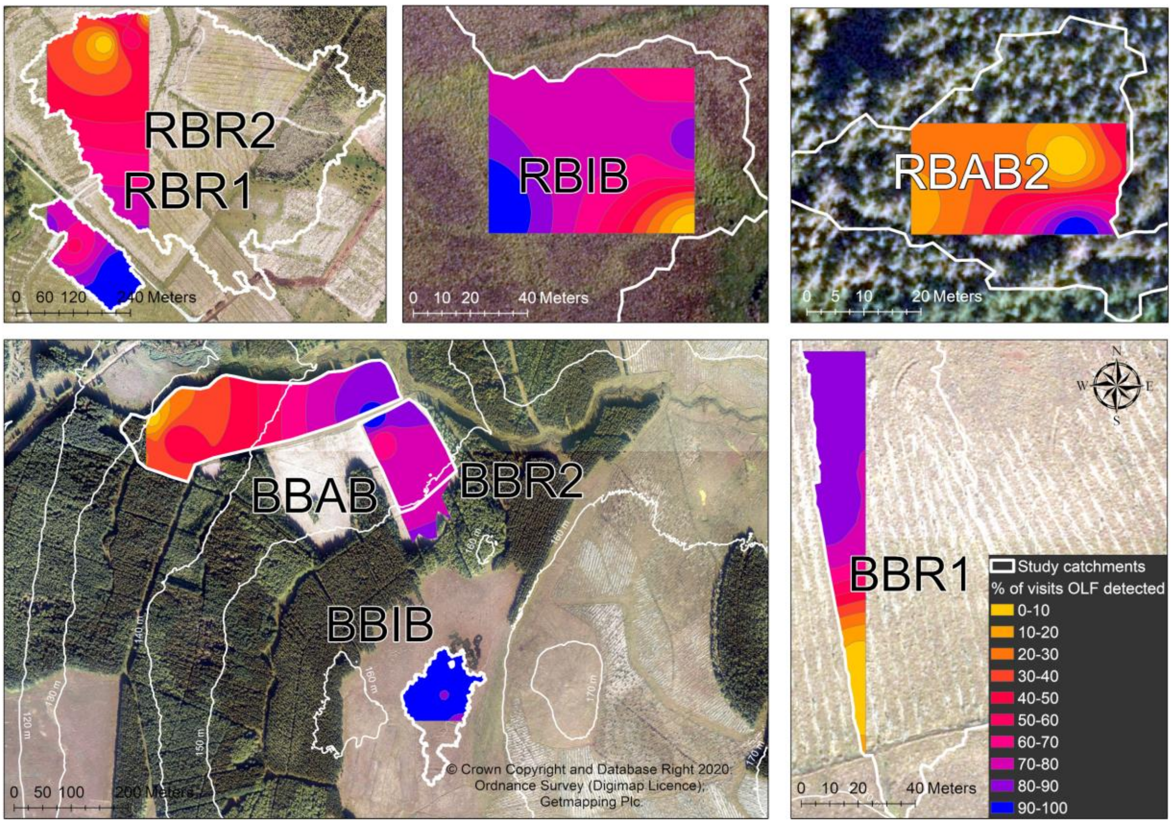

Figure 8 – % of visits overland flow was detected interpolated from the crest-stage tubes for the catchments, excluding RBAB1 where there were access restrictions.

TOPMODEL simulations estimated overland flow contributed to 54.6% of total discharge at BBIB and 19.2% for the RBIB. The percentage contribution of overland flow was lowest in the AB (11.9% BB; 2.9% RB) and was greater in the oldest restoration sites (32.2% BB; 8.7% RB), with the raised bog experiencing a higher percentage change. At BBR2, the contribution of overland flow was 15.3%, whereas RBR2 had the highest contribution of overland flow (34.8%) over the whole catchment.

4 Discussion

4.1 Water balance

For the 2018-19 water year, AET at RBIB was 200 mm greater than BBIB, which was over double the difference in PET estimates (96 mm) between the two sites. The mean annual wind speed is ~0.7 m s^{-1} higher at the blanket bog location (Met Office *et al.*, 2018) but could be expected to be cancelled out by lower temperatures and rainfall, leading to similar evaporative demands at both locations. Evapotranspiration losses for RBAB1 and BBAB were 59% and 36% of total annual rainfall, respectively. The difference in evapotranspiration may in part be related to the age and species of tree with Lodgepole pine planted in 1965 at RBAB1 and Sitka spruce, planted in 1980 at BBAB. Birkinshaw *et al.* (2014) also showed the importance of the age of Sitka spruce stands in controlling the water balance in the later stages of the Coalburn experiment. Overall, results for the AB are within the range of other studies that reported evapotranspiration losses for conifers to be as much as 55 – 80% of the total annual rainfall in some lowland areas of the UK (Calder *et al.*, 2003; Nisbet, 2005) with lower values (18 - 42%) from some upland studies (Anderson & Pyatt, 1986; Johnson, 1995).

At the restoration sites, AET was lowest in the most recent restoration sites and highest at the oldest restoration sites. Between 22 July 2018 and 23 March 2019, AET at BBR2 (85 mm) was considerably lower than BBIB (217 mm), which could be the result of a layer of mulch on the peat surface intercepting sunlight and preserving soil moisture (Prats *et al.*, 2016) or the relative absence of vegetation. At RBR2, there was still a significant quantity of coarse brash covering the peat surface. AET was ~20% higher than at RBIB, which coincides with reports of 15% interception losses of annual rainfall from conventional felling debris (Anderson *et al.*, 1990; Johnson, 1995; Nisbet, 2005). Water losses can also occur where sufficient understory remains after felling (Nisbet, 2005). However, as is common in coniferous plantations in the UK, little understory was present in the afforested sites in this study, and the vegetation was limited to the less hydrophilic bryophyte species and low diversity of vascular plants (Kershaw *et al.*, 2015). The fact that the oldest restoration sites (RBR1 and BBR1) had higher rates of AET than the IB sites could be as a result of differences in the vegetation at the restoration sites and the near-natural bogs (Hancock *et al.*, 2018). At RBR1, non-characteristic bog plants such as rosebay willowherb (*chamerion angustifolium*) had established, and at both raised bog restoration sites, conifer seedlings had regenerated naturally. At BBR1, we recorded purple moor-grass (*Molinia caerulea*) when the dipwells were installed, which Hancock *et al.* (2018) used as a negative indication of restoration success, but not from the dipwell locations in BBIB. As hypothesised, the runoff/rainfall coefficient was greatest in the IB, where evapotranspiration losses were least and water tables shallow, and lowest in the two afforested sites where water tables were deeper because of increased evapotranspiration from the trees. The oldest restoration sites had the next lowest water yield. The lower runoff in the oldest restoration sites compared to the intact sites may be because of more evapotranspiration losses associated with the higher vascular plant density.

4.2 Streamflow response and water-table dynamics

At the blanket bog restoration sites, the blocking of drains and furrows could have attenuated runoff, and reduced peak flows compared to post-felling and pre-blocking. At the raised bog restoration sites, where ditch blocking had not taken place, peak flows were higher than at the intact site, but the vegetation may also be a key factor. The water-table depth at the raised bog restoration sites was more closely correlated with that at the AB, particularly in drought periods. In contrast, the water table at the blanket bog restoration sites was more closely correlated with that at the IB, suggesting that the inclusion of ditch blocking as part of forest-to-bog restoration supports recovery of the hydrological functioning of bogs. RBAB1 had the lowest hydrograph intensity, which matches the hypothesis that the streamflow response to storms would be more subdued in the AB. Hydrograph intensity and the flow duration curves suggest that RBR1, ~9 years after restoration, exhibited a less flashy regime than RBR2, which has been restored for 4 years (despite its much larger catchment size), but not when compared to hydrograph intensity at RBIB.

The higher storm peak lag times at the afforested sites and at the raised bog compared to the blanket bog location coincide with higher water-table rises following storms; the greater water storage capacity would reduce the occurrence of saturation-excess overland flow. Peak lag times for RBR2 were proportionally higher considering the larger catchment area, but the hydrograph intensity was the highest, and the flow duration curves indicated more extreme peaks for the largest storms than at the other raised bog sites. However, following heavy rainfall, the River Forth often backed up on to the RBR2, which may explain the contrasting flow duration curve for that site. Shallow subsurface flow dominated at most of the sites, although 34.8% of flow was found to be overland flow at RBR2, possibly due to steeper slopes on either side of the stream where it flowed through a hollow.

Changes in peat structure due to drying in the tree root zone and ground disturbance may provide new pathways for subsurface flow in afforested and restoration sites. As such, overland flow was detected less frequently from the crest-stage tubes in the afforested and restoration sites when compared to the intact sites, but it was still common. Restoration appeared to reduce baseflows at the raised bog sites compared to the AB, but at the older restoration site (RBR1), BFI appeared to be rising again towards that of the IB. Little difference existed between the blanket bog sites except for BBR2, which had a higher BFI than the other blanket bog sites. The lower BFI in the raised bog restoration sites could be explained by greater compaction from the former tree stands, which were more mature than those at the blanket bog, and interception losses from the brash at RBR2 (Robinson *et al.*, 2003).

There was less difference in the blanket bog catchments sizes than those at the raised bog, and restoration appeared to reduce peak flows at BBR2, which had been restored around 5 years previously. The later addition of new dams by local managers to the outflow of BBR2 reduced average peak flows by a factor of four and resulted in water being redirected away from the weir, thereby changing the catchment area of the weir, causing an apparent reduction in the runoff/rainfall coefficient. BBIB and BBR1 appeared to have the flashiest stormflow response, and BBIB experienced the highest peak flows for its catchment size at both raised and blanket bog locations. BBAB had a more subdued stormflow response than the other blanket bog sites, although the flow duration curve was similar to BBR2. The blanket bog peat was fully saturated for more time in the two restoration sites (where ditch and furrow blocking had occurred) than the IB. Therefore, depending on the restoration techniques/practices used, water tables can be higher in restoration sites than near-natural sites, as also reported by Menberu *et al.* (2016).

Elevated water tables may benefit the restoration process by facilitating the growth of *Sphagnum spp.*, *Eriophorum spp.* (González et al. 2014) and restricting natural conifer regeneration. Conversely, they may also increase stormflow by reducing the water storage capacity within the peat and increasing the likelihood of saturation-excess overland flow. There can also be an adverse effect of increased methane emissions where the water table is at or very near the surface (Hargreaves & Fowler, 1998). Overall, the hypothesis that storm response would be more subdued in bogs under restoration than intact systems is rejected for the raised bog location, but it was less clear at the blanket bog location given the similarity in the flow duration curves and calculated metrics.

There have been few studies on the effects of ditch blocking on stream and river peak flows (Ballard *et al.*, 2012) and lag times in bog systems despite the widespread belief that it might reduce flood risk downstream (Parry *et al.*, 2014). In this study, there were fewer differences in peak flows between the blanket bog sites, but at the raised bog location, where no ditch blocking had occurred, peak flows were significantly higher in the two restoration sites than for the IB ($p < 0.001$ Kruskal-Wallis Test). Peak flows were lower in RBR1 than RBR2, yet the peak lag time was less at RBR1 than the other catchments at the raised bog. However, it is important to bear in mind that the catchment size of RBR1 was ten times smaller than RBR2 and three times smaller than RBIB, leading to shorter lag times. Ditch blocking as part of forest-to-bog restoration could be a factor in reducing average peak flows. However, differences in vegetation cover between the restoration sites and the IB may be a more important factor for lag times and hydrograph intensity (Gao *et al.*, 2016; Grayson *et al.*, 2010), particularly where overland flow begins to become strongly dominant during storms. Holden *et al.* (2008) reported that vegetation and surface roughness were important in controlling overland flow velocities in blanket peat. The effects of ditch blocking can also be very dependent on local conditions (Ballard *et al.*, 2012), and drainage networks have sometimes been found to extend pathways for runoff (Lane & Milledge, 2013).

Water-table fluctuations were least in the IB sites and generally highest in the AB sites. Water tables fluctuated less in the restoration sites than the afforested sites but more than the IB, as shown in other studies (Komulainen *et al.*, 1999; Menberu *et al.*, 2016). Also, the water-table variability was closer to that of the IB in the oldest restoration sites. There were differences between the two locations, with the peak water-table depth being significantly lower than the IB at the raised bog location but not significantly different at the blanket bog location where ditch blocking had occurred. Our hypotheses that we would find deeper and more variable water tables in the AB followed by the restoration sites and that the water tables at the oldest restoration sites would be closer to those at IB are largely accepted. Similarities between the water-table metrics in this study and the ditch blocking study on non-afforested peat by Holden *et al.* (2011) suggest hydrological functioning in forest-to-bog restoration sites is not likely to fully replicate that of near-natural bogs in the short term (<10 years). Our results suggest ditch and furrow blocking may speed up water-table recovery and attenuate runoff, and mulching may be preferable to conventional felling for preserving soil moisture. However, a more focused study on how the different restoration techniques affect hydrological processes is required.

The peat in the drained, afforested sites was fully saturated for the least amount of time, similar to the findings of Menberu *et al.* (2016), but experienced a higher mean water-table rise during storm events. Overall, there was a negative correlation (r_s = -0.466, p < 0.001, N = 360) between storm precipitation/water-table response ratios and water-table depth at the start of larger storms. Therefore, the greater storage capacity with deeper water tables likely explains the higher water-table rise in the afforested sites. The peat may also have experienced a loss in available pore space after drainage and compression by the trees (Anderson *et al.*, 2000; Anderson & Peace, 2017). Differences in the physical peat properties between the raised bog and blanket bog locations (Howson et al., in prep) could explain the higher water tables in the blanket bog restoration sites. Accelerated water-table recovery may occur where there is less available pore space for the water to fill (Meyer *et al.*, 2011; Rezanezhad *et al.*, 2016), but this is unlikely because of the similarity in specific yield (Howson et al., in prep). Overall, the water-table depth after forest-to-bog restoration was similar to that of the IB sites after 5-6 years at the blanket bog location. However, differences still existed in water-table dynamics, and the speed and degree of water-table recovery may depend on the methods of restoration (i.e. if the drains and furrows were blocked in addition to the felling of trees) and physical characteristics of the peat.

5 Conclusions

For the afforested sites, evapotranspiration exerted a dominant control over water yield leading to more subdued streamflow and water-table response to rainfall than for the intact and restored bogs. For sites with no trees, streamflow response to rainfall at the blanket bog restoration sites was more subdued (lower peaks, higher peak lag times) than at the intact blanket bog, whereas at the raised bog restoration sites, streamflow was less subdued than at the intact raised bog. The differences in overland flow occurrence between the intact and the restoration sites were less in the oldest restoration sites than in the most recent restoration sites. Overall, the hypothesis that hydrological functioning would be closest to intact systems in the oldest restoration sites is largely accepted. However, some of the differences between the forest-to-bog restoration sites and the intact bogs we studied suggest a full recovery in hydrological function is not likely to return in the short term (<10 years), although drain and furrow blocking as part of forest-to-bog restoration may provide useful buffering of water tables. An extended time-series study would be required to fully determine whether hydrological functioning changed over long timescales in response to forest-to-bog restoration.

6 Acknowledgements

The project was jointly funded by Scottish Forestry (previously Forestry Commission Scotland) and the School of Geography, University of Leeds. The Forestry Commission provided access to Flanders Moss West and Scottish National Heritage (David Pickett) to Flanders Moss National Nature Reserve. The RSPB kindly granted Forsinard Flows National Nature Reserve access and provided accommodation at the Field Centre at Forsinard. We are grateful for help from Colin Gordon and Dave Darroch (Forest Research), Willie Grant (the superintendent of the River Halladale), Ho Wen Lo and Jacob Ehrlich (University of Leeds) for assistance in the field. Finally, we are grateful to the University of Leeds technicians, including David Wilson (who helped with the site installations) and David Ashley, for technical assistance.

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8 Supplementary information

*Table S1 - Water balance and mean, maximum and minimum discharge for the eight catchments over the whole study period. Catchment monitoring dates are given. RBAB1 was taken as the afforested bog catchment for Flanders Moss. P = precipitation (mm); Q = total annual discharge (mm); Mean Q = mean discharge (L s^{-1})/ (mm d^{-1}); Max Q = maximum discharge (mm d^{-1}); Min Q = minimum discharge (mm d^{-1}) Runoff/rainfall = Q/P (%); PET = potential evapotranspiration using the Priestley and Taylor (1972) equation; AET = actual evapotranspiration P-Q (mm). * - rainfall from the Flanders Moss National Nature Reserve rain gauge.*

Site	Dates	P (mm)	Q (mm)	Mean Q (L s^{-1})	Mean Q (mm d^{-1})	Max Q (mm d^{-1})	Min Q (mm d^{-1})	Runoff/ Rainfall (%)	PET (mm)	AET (mm)
RBIB	27/11/17 - 28/11/19	1942*	1098	1.0	1.5	10.3	0.2	56.5	1386	844
RBAB1	27/03/18 - 30/09/19	1736	598	0.1	1.1	16.8	0.0	34.4	1353	1138
RBR1	26/02/18 - 28/11/19	2030	1020	0.5	1.6	17.2	0.0	50.2	1384	1010
RBR2	26/02/18 - 28/11/19	2030	1149	5.5	1.8	87.9	0.0	56.6	1384	880
BBIB	21/07/18 - 03/11/19	1542	1167	0.4	2.3	35.1	0.0	75.7	785	374
BBAB	14/06/18 - 03/11/19	1555	872	1.0	1.6	35.5	0.0	56.1	951	683
BBR1	02/03/18 - 03/11/19	1542	1007	0.4	2.0	35.5	0.0	65.3	785	534
BBR2	21/07/18 - 03/11/19	1706	1219	0.4	1.5	31.7	0.0	71.5	1227	487

Table S2 - Mean storm metrics for the eight streamflow catchments over the whole study period. N = number of storms; Peak Q = peak storm discharge (mm d^{-1}); Peak lag = duration between peak rainfall and peak Q; Recess lag = duration between peak Q and when the quickflow component had returned to zero; Hydrograph Intensity = peak Q divided by (total storm Q x 10^{-6}); BFI – baseflow index; Storm duration = time quickflow > 0 for the storm event. Standard deviations are in parentheses.

Site	N	Catchment area (ha)	Peak Q (mm d^{-1})	Peak lag (h)	Recess lag (h)	Hydrograph intensity (s^{-1})	BFI	Storm duration (h)
RBIB	44	6.01	6.24 (2.00)	14.94 (6.13)	66.16 (53.18)	8.37 (5.08)	0.87 (0.07)	24.59 (12.68)
RBAB1	28	0.67	8.56 (4.16)	15.35 (10.17)	92.68 (45.84)	7.06 (2.54)	0.79 (0.08)	20.87 (12.19)
RBR1	62	2.46	10.00 (2.74)	7.07 (4.66)	57.56 (37.74)	11.65 (3.64)	0.73 (0.08)	17.47 (12.07)
RBR2	73	26.22	12.17 (15.63)	11.10 (8.42)	37.32 (27.72)	19.76 (9.57)	0.59 (0.14)	19.52 (17.30)
BBIB	55	1.64	14.67 (6.91)	6.48 (9.06)	45.56 (38.01)	20.11 (10.11)	0.70 (0.11)	18.29 (14.25)
BBAB	46	5.08	10.08 (5.06)	10.95 (7.88)	69.34 (44.43)	11.18 (5.36)	0.72 (0.10)	22.32 (12.38)
BBR1	61	1.58	11.50 (5.74)	6.61 (7.73)	38.64 (21.37)	19.45 (10.69)	0.68 (0.12)	15.45 (9.64)
BBR2	46	2.27	11.8 (7.54)	10.01 (9.30)	48.42 (30.94)	14.03 (10.36)	0.78 (0.10)	19.91 (14.55)

Table S3 - Mean water-table storm metrics for all catchments over the whole study period. WT rise = water-table depth before the storm – Peak WTD; Peak WTD = minimum water-table depth for the storm event; Duration = duration from rainfall start to WT rise for a 0.1 cm rise in the water-table; Peak lag = duration between peak rainfall and peak water-table; 6 h recession rate = difference between peak water-table and 6 hours after the peak divided by 6; 12 h recession = rate difference between peak water-table and 12 hours after the peak divided by 12. Standard deviations are in parentheses.

Site	N	WT rise (cm)	Peak WTD (cm)	Duration (h)	Peak lag (h)	6 h recession rate (cm h^{-1})	12 h recession rate (cm h^{-1})
RBIB	64	2.87 (2.86)	1.87 (3.39)	0.93 (1.01)	9.25 (6.34)	0.11 (0.13)	0.14 (0.17)
RBAB1	35	8.00 (5.62)	12.54 (10.54)	2.18 (2.27)	15.40 (7.49)	0.33 (0.22)	0.44 (0.28)
RBAB2	48	4.44 (4.40)	6.67 (5.46)	1.30 (1.77)	10.39 (5.87)	0.08 (0.05)	0.09 (0.05)
RBR1	78	2.98 (1.98)	3.36 (4.29)	0.69 (0.80)	6.39 (4.38)	0.12 (0.14)	0.13 (0.15)
RBR2	52	3.13 (3.07)	11.00 (6.69)	1.60 (1.78)	10.46 (5.71)	0.09 (0.10)	0.13 (0.17)
BBIB	63	1.98 (2.30)	4.63 (2.72)	0.91 (0.64)	8.96 (6.46)	0.06 (0.12)	0.05 (0.05)
BBAB	39	5.61 (4.78)	15.77 (6.10)	1.13 (1.13)	10.11 (7.00)	0.13 (0.08)	0.15 (0.08)
BBR1	53	2.56 (3.45)	-1.05 (2.86)	1.24 (1.24)	7.74 (6.50)	0.17 (0.22)	0.13 (0.17)
BBR2	61	2.68 (2.63)	3.54 (2.89)	0.87 (0.83)	8.90 (6.42)	0.13 (0.15)	0.13 (0.15)