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Rewetting impact on the hydrological function of a drained peatland in the boreal landscape

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ABSTRACT

There is a growing interest in peatland restoration as a nature-based solution to mitigate hydrological extremes. To counter the impacts of past peatland degradation and ongoing climate change trajectories, governmental authorities propose rewetting of drained peatlands as a key tool to enhance landscape resilience against floods and droughts by improving water storage. Despite a growing body of literature on this topic, the effectiveness of rewetting to enhance peatland hydrological functions remains insufficiently documented, especially in the boreal region. Therefore, this study utilized high temporal resolution groundwater table level and streamflow data to investigate the impact of peatland rewetting using a before-after-control-impact (BACI) approach. This investigation was conducted on a historically drained peatland located at the Trollberget Experimental Area (TEA) in northern Sweden. The primary aim of the experimental study was to examine the impact of rewetting on (1) the groundwater table level response, (2) runoff dynamics, and (3) water storage and hydrological buffer capacity. Our results showed that peatland rewetting led to a significant increase in the groundwater table level by 60 mm compared to the control. Flow duration curve (FDC) analysis demonstrated that the low-flow threshold increased by up to 150% at the rewetted sites. Furthermore, our findings suggested that rewetting resulted in an increase in the groundwater table level threshold at which stream runoff is generated. Additionally, our findings showed a noteworthy shift in the monthly runoff coefficient, with an increase during dry months and a decrease during wet periods. Combined, these observations point towards an enhancement in the peatland's water storage and hydrological buffer capacity as a positive outcome of the rewetting efforts, but also highlight that within the first three years, full hydrological restoration did not occur.

1. Introduction

Natural wetlands play a significant role in water purification, flood control, and climate change mitigation (Bullock and Acreman, 2003; Fluet-Chouinard et al. 2023). Along with these benefits, their importance for maintaining baseflow and improving groundwater recharge is often highlighted as some of the most fundamental processes of pristine wetlands (Kadykalo and Findlay, 2016). Peatlands are the dominant type of wetlands in northern latitudes (Locky and Bayley, 2006; Bring et al., 2022), and cover approximately 15% of the boreal forest region (Helbig et al., 2020; Harris et al., 2020). With their high porosity and large storage capacity, boreal peatlands provide a unique and valuable ecosystem service by retaining water during storm events, making them crucial for mitigating the hydrological processes that contribute to floods and droughts (Acreman and Holden, 2013; IPCC, 2021). In the context of climate change, air temperatures will increase, and extreme precipitation events will likely become more frequent making peatlands even more important for buffering hydrological extremes (IPCC, 2021). The hydrology of peatlands also influences plant species diversity and composition (Goud and Moore, 2018; McPartland et al., 2019), nutrient availability (Macrae et al., 2013), soil oxidation–reduction (Mitchell and Branfireun, 2005), gas exchange processes (Waddington et al., 2009), and water quality (Holden, 2005). Understanding the fundamental hydrological processes in peatlands is therefore important for predicting the consequences of climate change and/or different potential management and restoration actions (Kimmel et al., 2010).

Wetland degradation due to human drainage, in combination with ongoing climate change, has led to wetlands being among the most threatened ecosystems in the world (Fluet-Chouinard et al. 2023). On a global scale, around 50 million hectares (11%–15%) of peatlands have been degraded by human activities (Frolking et al., 2011; Leifeld et al., 2019). In the northern regions, particularly Scandinavia and the UK,

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peatlands have been degraded primarily by drainage to improve forestry and/or agricultural practices (Ramchunder et al., 2012; Swindles et al., 2019; Härkönen et al., 2023). Drainage leads to changes in peat hydrological properties due to decreasing pore volume and increasing bulk density associated with peat mineralization and subsidence (Silins and Rothwell, 1998; Regan et al., 2019; Liu et al., 2022). In the short term, lowering the groundwater table level (GWL) through drainage likely increases the water storage capacity and therefore mitigates runoff response during extreme events. Long-term studies, however, have indicated that drained catchments eventually lose more water, leading to lower baseflow during drought (Holden et al., 2006; Loisel and Gallego-Sala, 2022). Drainage also affects vegetation composition (Schrautzer et al., 2013), greenhouse gas emissions (Dinsmore et al., 2009; Kwon et al., 2022), hydraulic conductivity (Price et al., 2003; Shantz and Price, 2006; Rezanezhad et al., 2016; Ahmad et al., 2020; Balliston and Price, 2023) and results in altered discharge dynamics in downstream locations and hence increased risk of flooding (Holden et al., 2004).

Due to the enhanced understanding of the importance of peatlands as a nature-based solution for buffering extreme hydrological events, restoration to more natural conditions has been suggested as one of the best management strategies to avoid downstream hydrological catastrophes (Holden et al., 2004; Holden et al., 2007). The restoration of wetlands has the potential to bring back other lost ecosystem services and increase the heterogeneity of wetland functions and biodiversity compared to their degraded state (Zedler, 2003; Temmerman et al., 2013). The urgency for a global effort to restore wetlands has been emphasized in various studies (Erwin, 2009; Loisel and Gallego-Sala, 2022; Bring et al., 2022) and noted by the United Nations when they decided to declare 2021–2030 the "Decade on Ecosystem Restoration" (Waltham et al., 2020; Kettenring and Tarsa, 2020; Aronson et al., 2020). Recent hot and dry summers have also reinforced the interest in peatland restoration projects (Bring et al., 2020). As a result, the Swedish government allocated about 25 million euros for wetland restoration after the severe drought of 2018 as support for enhancing the water supply in the forested landscape. The primary perception underlying most wetland restoration projects is that blocking ditches will slow down runoff and drainage from the peatland. This will cause GWL to rise and water residence times to lengthen (Allott et al., 2009). However, to date, the efficiency of restoration for increasing the hydrological functioning of peatlands is not well understood. Like many ecological restoration projects, the evaluation of peatland restoration outcomes has often been limited due to the considerable costs and time constraints associated with conducting detailed, long-term monitoring of GWL post-restoration (Wilson et al., 2010). Additionally, conflicting conclusions about peatland restoration success have been reported. Some studies have shown that restoration has increased the GWL and peatland water storage properties (Wilson et al., 2011; Schimelpfenig et al., 2014; Menberu et al., 2018), while others reported that restoration does not always result in increased GWL (Holden et al., 2017) or water storage capacity (Shuttleworth et al., 2019). Hence, the effectiveness of peatland restoration in raising the GWL appears to be influenced by various factors, including the degree of peat degradation (Holden and Burt, 2003) and local conditions, such as topography, soil, and climate (Price et al., 2003; Wilson et al., 2010; Bring et al., 2022) that have not been properly evaluated.

Considering the extensive network of ditches, which spans one million kilometers in Sweden alone (Laudon et al., 2022), blocking them all would be logistically impossible and prohibitively expensive. Hence, there is an urgent need to find ways to prioritize restoration efforts. Furthermore, the lack of understanding regarding the effectiveness of hydrological restoration in peatlands for mitigating drought, combined with the substantial funding allocated to restoration in Sweden, makes the current policies uncertain. In light of this, the primary aim of this study was to examine how peatland rewetting impacts its hydrological functioning. Here, we used continuous GWL and discharge data to investigate changes in peatland GWL and the corresponding runoff response associated with ditch blocking of a previously drained boreal peatland. Our analysis followed a before-after-control-impact (BACI) approach to comprehensively examine these changes. We hypothesized that the rewetting of the drained peatland would enhance the hydrological buffer capacity depicted by 1) a rise in GWL, 2) increased baseflow, 3) less variable GWL and discharge, and 4) increased water storage that altogether result in augmented water retention during wet seasons and sustained streamflow during dry seasons.

2. Materials and methods

2.1. Study sites

The research was conducted in boreal northern Sweden and included a rewetted peatland that had been historically drained approximately 100 years ago. In addition, two undrained nearby peatlands were used as control sites to assess the impact of the rewetting on the hydrological conditions. The rewetted and control sites have similar topography, weather conditions, and vegetation.

The rewetted peatland is a nutrient-poor, minerogenic fen, located at the Trollberget Experimental Area (TEA) (64.15 N, 19.92E) approximately 45 km northwest of the city of Umeå, in northern Sweden (Laudon et al., 2023) (Fig. 1A, B). The study area has a cold temperate humid climate, characterized by long winters with permanent snow cover from mid-November to late April. The mean annual temperature (30 years mean from 1991 to 2020) is + 2.4° C and the mean annual precipitation is 638 mm, with about 30% falling as snow (Peichl et al., 2023). Much of the surrounding forest is covered by conifers such as Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*). The peatlands are underlain by glacial till and gneissic bedrock. The bulk density of the drained peatland site exhibited a range of values spanning from 0.05 to 0.13 g/cm3 in the top 55 cm of the peat profile and generally increased with distance from the central ditch and with peat depth (Casselgård, 2020). The average peat depth at the site is 2.41 m. The peatland is divided into two catchments, R1 and R2, with drainage areas of 47 and 60 ha, respectively (Fig. 1C). The peatland is dominated by *Sphagnum* spp. together with some sedges (*Carex* and *Eriophorum* spp.) and dwarf shrubs (*Calluna vulgaris* and *Vaccinium uliginosum*). Based on the land use history of the site, the main ditches within the peatland were dug around 1905, with the majority of the catchment surrounding the peatland drained for forestry in the mid- 1930 s. The peatland was rewetted by filling and blocking the ditches in November 2020. The ditches were filled using peat from the site with additional dams built at regular intervals using the tree logs harvested from the site. The logs were placed horizontally but perpendicular to the ditch, except at the two outlet locations where the logs were inserted vertically into the peat and layered additionally with geotextile. In addition, the sparse tree cover that grew on the peatland was cut to reduce evapotranspiration and complement the ditch blocking (Laudon et al., 2023).

The control catchment, C4 in Krycklan Catchment Study (KCS, 64.25 N, 19.46E) is a nutrient-poor, minerogenic fen located approximately 10 km from the rewetted catchment. This catchment is an integrated part of the Svartberget field infrastructure and has a long history of research beginning in the early 1980 s. C4 has a high-areal proportion of peat soils (51%) and drains an 18-ha catchment (Laudon et al., 2021). The bedrock, which is similar to TEA is dominated by sedimentary veined gneiss and overlain by glacial till of varying thickness. The peat vegetation cover is dominated by *Sphagnum* spp.

The second control catchment, Degerö Stormyr, which is a nutrient-poor, minerogenic fen (64.11 N, 19.33E), is located approximately 24 km from the TEA, at the Kulbäcksliden Experimental Forest (Fig. 1C, 1D; Noumonvi et al., 2023). Degerö Stormyr encompassing an area of 650 ha, is an undisturbed peatland and the average peat depth is between 3 and 4 m deep (Nilsson et al., 2008). The peatland is underlain by a relatively impermeable layer of mineral glacial till and gneissic bedrock

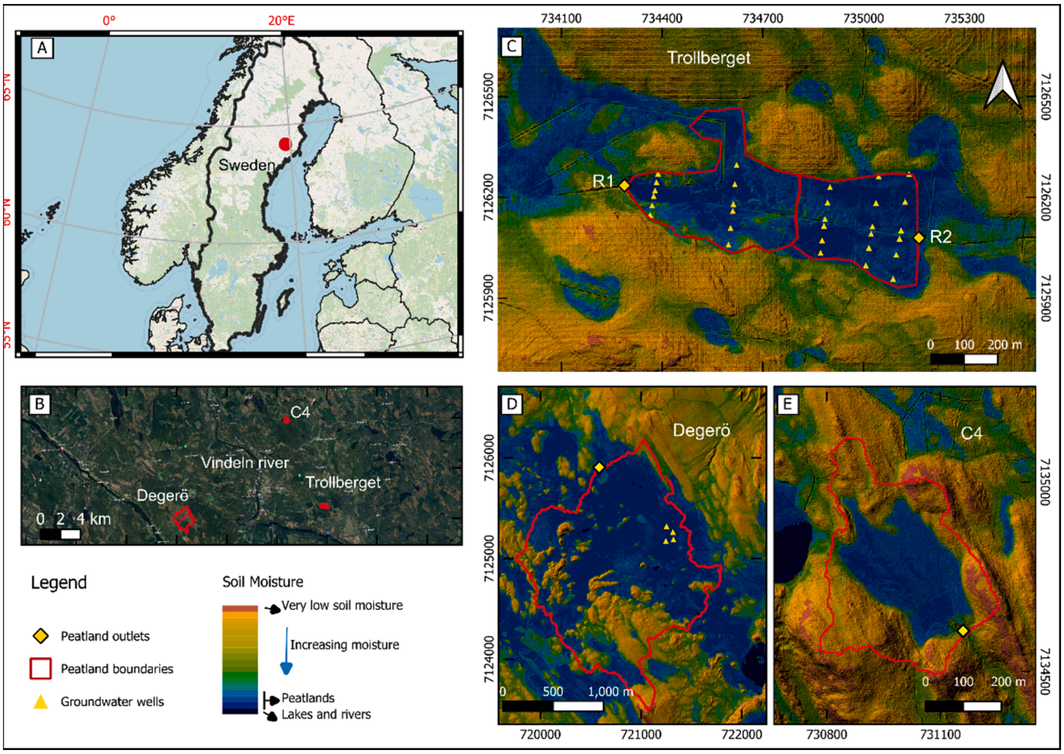

Fig. 1. Geographic location of the study sites in Sweden (A), the location of the three sites in relation to each other (B), the outline of the Trollberget rewetted peatland (catchments R1 and R2) with the location of groundwater wells (C), the outline of the Degerö control site with the location of groundwater wells (D), and the outline of the C4 site (E). The base map for C, D, and E is the SLU soil moisture map (Ågren et al., 2021) blended with a hillshade from a 2 m resolution digital elevation model.

(Malmström, 1923). The vegetation cover in the mire is dominated by lawn and carpet plant communities, including *Sphagnum* spp., sedges, and dwarf shrubs.

2.2. GWL monitoring in the rewetted site

At Trollberget, five groundwater dipwell transects were established perpendicular to the main ditch with six monitoring wells along each transect (30 in total). The wells were placed approximately at distances of 10, 50, and 100 m on both sides of the main ditch. GWL were monitored manually on a biweekly basis from October 2019 to October 2023 during the snow-free season (May–October). All manual measurements of GWL were calculated as the distance from the ground surface to the top of the water table using a measuring tape with a water-sensitive tip (Weiss Bandmab Measuring Tape). The procedure involved measuring the distance between the top of the dipwell and the water level (a), as well as the distance from the top of the dipwell to the ground surface (b). The GWL was then calculated by subtracting the top on the ground surface from the measured water table as GWL=a − b.

Continuous (i.e. hourly) GWL logging was performed in 15 wells using a Solinst Levelogger pressure transducer. Hourly GWL data were corrected for barometric pressure using Solinst™ baraloggers in the air. The barometric compensation process was carried out automatically using the Levelogger Software 4.5.1 Data Wizard. For this conversion, the manual GWL measurements were used to calibrate and check automated measurements. At the end of this processing, our data set included a GWL time series one year pre-rewetting (October 2019 to November 2020) and three years post-rewetting (November 2020 to October 2023). In the Degerö control site, the same time series of GWL from four wells that are part of the ICOS-Svartberget system (https://www.icos-sweden.se/data) were used. Due to technical issues with the groundwater loggers, no groundwater data for recent years was available for the C4 control catchment in the Krycklan Catchment Study.

2.3. Hydrological and meteorological data

Rainfall was logged every 10 minutes using a tipping bucket (ARG 100, Campbell Scientific, USA) as part of the reference climate monitoring program at the Svartberget meteorological station (64°14′ N, 19°46′ E, 225 m a.s.l) (Laudon et al. 2013). For this study, the data were resampled to daily values. Snow depth data measured at Degerö control site were obtained from ICOS database (www.icos-sweden.se/data) for the corresponding period. Additionally, stream discharge measurements have been recorded since 2019 at the outlet of each catchment (R1 and R2, see Fig. 1C). Stage height was recorded hourly at each catchment using pressure transducers (Expert 3400, MJK A/S, Denmark). The transducers were placed in stilling ponds of each 90-degree V-notch weir. The automatic water height time series for the two outlets were corrected to account for logger offset using manual measurements of reference water height. These manual readings were performed at biweekly intervals during snow-free conditions. The stage height time series of each logger was quality-controlled manually and corrected for the influence of ice and unrealistic values due to occasional downstream damming. Moreover, independent stage-discharge rating curves were derived using volumetric methods. The reliability and agreement between calibrated versus predicted discharges are shown in Supplementary Fig. 1. Specific discharge (mm/day) was calculated based on the measured discharge and the catchment area. The catchment areas were obtained using the D8 algorithm (O'Callaghan and Mark, 1984) on a 0.5 m resolution digital elevation model (DEM), derived from airborne Light Detection and Ranging (LiDAR) measurements (Laudon et al. 2021).

For comparative analysis of discharge responses, we utilized the discharge data from the control sites C4 (Laudon et al. 2021) and C18 at Degerö catchment (Noumonvi et al., 2023). The outlet of C4 is equipped with a V-notch weir situated in a heated dam house, and discharge measurements and calibrations were carried out following the same protocol and interval as those applied at Trollberget. The weir at the C18 catchment is located inside a small house set up on a flume, allowing for continuous stage height measurements throughout the year (Leach et al., 2016). Discharge was calculated by applying a stage height-discharge rating curve to hourly water level measurements. The C18 rating curve was calibrated using manual discharge measurements using salt dilution made during different flow conditions.

2.4. Data analysis

Due to the presence of large snow cover and deep soil frost, winter and spring months were excluded from the data analysis because of data gaps. Consequently, our emphasis was primarily on assessing the effects of rewetting on hydrological responses during the summer and fall seasons. These periods are anticipated to undergo significant changes in GWL and are also times when drought conditions are most likely to occur (Wilson et al., 2010). The GWL time series for pre- and post-rewetting was used to visually assess GWL differences between control and rewetted sites. To facilitate this comparison, continuous GWL data from all the dipwells were averaged to obtain one single time series. The statistical design used in this study focuses on the BACI experimental design as used previously in hydrological studies (Laudon et al., 2023). For this analysis, we calculated the relative GWLs (rewetted minus control) across the time series, thus the effect of annual rainfall has been accounted for as weather conditions are assumed to be similar within this small geographical area. Any statistical differences in the median GWL pre- and post-rewetting were tested using the non-parametric Wilcoxon test with Bonferroni correction ($p < 0.05$). Furthermore, significant differences in snow depth from January to May and rainfall data from June to October among pre- and post-rewetting years were determined using Wilcoxon test with a Bonferroni adjustment ($p < 0.05$).

To assess the impact of peatland rewetting on streamflow regime, a Flow Duration Curve (FDC) approach was employed to examine the frequencies of daily streamflows, particularly focusing on low flow for pre- and post-rewetting. The 'hydroTSM' R package was utilized for this analysis. The shape of the curve provides information about the variability of flow in streams: a curve with an overall steep slope indicates high variability in flow, whereas a flatter slope indicates less variability in flow over time. To characterize the information in the FDC, it was partitioned into three segments (Smakhtin, 2001). The first segment represents high flows (0–20% exceedance probabilities of flow) and the middle part is mid-range flows (20–70%) illustrated by flows from moderate rainfall events. The third segment (70–99%) is related to the sustainability of baseflow in the dry period.

To evaluate the impact of peatland rewetting on baseflow dynamics, the Baseflow Index (BFI) was computed for both control and rewetted sites over pre- and post-rewetting periods spanning from June to October. The BFI is derived as the ratio of total baseflow to total streamflow, calculated according to the equation:

$$BFI = 1 - \frac{\sum Q_b}{\sum Q}$$

where Q is the total streamflow, and Q_b is the baseflow. The BFI value ranges between 0 and 1, with higher values indicating a greater proportion of baseflow contributing to streamflow. To isolate the baseflow component from the total streamflow data, the one-parameter Lyne-Hollick digital filter (Lyne and Hollick, 1979) was employed, with a filter parameter set to 0.98 and 9 passes across the hourly data, as described by Ladson et al. (2013).

To assess whether peatland rewetting has increased water storage and hydrological buffer capacity, the GWL and discharge time series were regressed using an exponential model and, R squared, p values, and the inflection points of the fitted curves were extracted for the control and rewetted sites for pre- and post-rewetting. Additionally, the monthly runoff coefficient (calculated as total runoff divided by total rainfall) was determined for both control and rewetted sites (R1 and R2). The relative difference from the control sites was then calculated to assess the impact of rewetting. Statistical analysis, data processing, summary statistics, and plotting were performed using R (version 4.1.2, R Core Team 2023).

3. Results

3.1. Precipitation and snowpack

In the study area, snow accumulation generally started in December (Fig. 2a). Snowmelt commenced in mid-March during the pre-rewetting and the second post-rewetting year, while occurring in early March in the first post-rewetting year and early April in the third post-rewetting year. Remarkably, the first post-rewetting year exhibited the highest snow accumulation, reaching 85 cm in February. Maximum snow accumulation measured 62 cm, 59 cm, and 66 cm in the pre-rewetting, second post-rewetting, and third post-rewetting years, respectively. Statistical analysis using pairwise Wilcoxon tests indicated no significant differences in snow accumulation between the pre-rewetting and second and third post-rewetting years ($p > 0.05$) while snow accumulation was significantly higher in the second year post-rewetting ($p < 0.05$) (Fig. 2b). Moreover, snow accumulation in the second post-rewetting year was significantly lower than in the first year and the third post-rewetting year ($p < 0.05$). Additionally, pairwise Wilcoxon tests indicated no significant differences in rainfall between pre- and post–rewetting years ($p > 0.05$) (Fig. 2d). Furthermore, cumulative precipitation during the pre- and post-rewetting periods were similar, particularly in the summer months (Fig. 2c).

3.2. Effect of rewetting on GWL

To test our hypothesis regarding the impact of rewetting on GWL, we conducted pairwise comparisons of relative GWL across various pre- and post-rewetting years. Post-rewetting, the GWL was significantly higher ($p < 0.05$, Fig. 3, Table 1), as well as less variable (Fig. 3). In 2020, before rewetting, the relative difference in GWL was 130 mm. One year post-rewetting (2021), this difference significantly decreased to 40 mm. The GWL in 2022 (two years post-rewetting) at the rewetted and control sites was approximately the same (−85 mm and −83 mm, respectively). Overall, the mean GWL at the rewetted site rose 64 mm by 2023 (three years post-rewetting) compared to 2020 (pre-rewetting year), while the mean GWL at the control site decreased by 30 mm over the same time frame.

Additionally, there was a notable decrease in GWL variation between the rewetted and control sites, as evidenced by the post-rewetting GWL time series (Fig. 4). In the second post-rewetting year, the rewetted site displayed minimal divergence from the control site in the mean GWL. During the pre-rewetting period, the GWL at the rewetted site consistently remained below −70 mm (with a minimum depth of −405 mm) from the surface of the peatland, while the control site GWL varied between −187 and 8.5 mm (Table 1). Moreover, in the pre-rewetting period of 2020, GWL at the rewetted site experienced two rapid declines, once at the end of June and once at the end of August, due to prolonged periods of low rainfall input (see Fig. 4). A decrease in GWL was visible at the control site; however, the decline was less pronounced and did not reach comparable depths. Notably, there were no instances of flooding events, where the GWL rose above the soil surface, at the rewetted site before rewetting (indicated by the blue bands in Fig. 4). After rewetting, the rewetted site experienced prolonged flooding.

The duration curves revealed that the fluctuation in GWL in the three post–rewetting years was less during both summer and fall periods (Fig. 5). Specifically, in the pre-rewetting year, the GWL at the rewetted site was −337 mm 75% of the time during the summer, while at the control site it was −143 mm. The control site also experienced drier conditions during the pre-rewetting year; for example, GWL was above −100 mm 50% of the time in 2020, while it was above −100 mm about

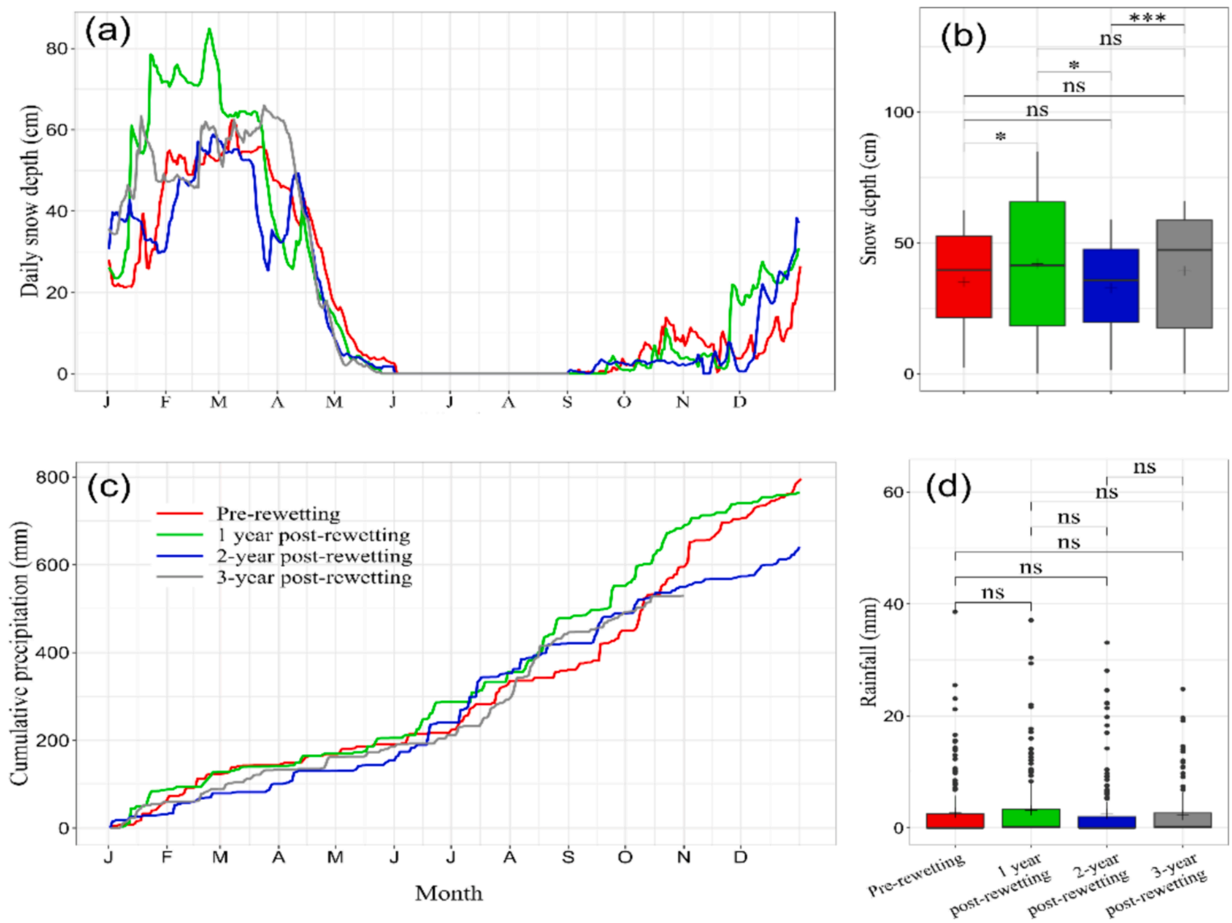

Fig. 2. (a) Mean daily snow depth (cm), (b) boxplot of snow depth (cm), (c) daily cumulative precipitation (mm), and (d) boxplot of rainfall (mm). The box is the interquartile range (IQR) with the median (line in box) and average (black +sign); the whiskers show the 1.5 IQR value. Black dots represent outliers beyond the whiskers.

70% of the time in 2021. One year post-rewetting, GWL was −199 mm at the rewetted site and −123 mm at the control site for 75% of the time. This difference was even less two years following rewetting, with −112 mm at the rewetted site and −111 mm at the control site for 75% of the time. The shallowest GWL at the rewetted and control sites occurred in the second post-rewetting year. In the summer of the third post-rewetting year (2023), both the rewetted and control sites experienced a decline in GWL, with GWL being at −251 mm and −198 mm at the rewetted and control sites, respectively, for 75% of the time.

During the fall season of the pre-rewetting year, the GWL at the rewetted site remained above −247 mm for 75% of the time, while the control site maintained a GWL within −115 mm for 75% of the time. One year post-rewetting, the GWL was above −71 mm for 75% of the time at the rewetted site, similar to the control site, which was −70 mm. The GWL duration curve of the rewetted site aligned with the control site over the three post-rewetting years, and GWL never dropped below −163 mm at the rewetted site during the fall season.

3.3. Effect of rewetting on streamflow response

Overall, the hydrographs for the two catchments (R1 and R2) of the rewetted site exhibited a flashy response with high peak flow spikes before rewetting, whereas the control catchment had relatively flattened hydrographs with lower peaks (Fig. 6). Differences were apparent between the two catchments, with R1 exhibiting a more pronounced flashy response with higher peak flows compared to R2. Hydrograph responses were generally more pronounced during the fall season. An exception to this trend was observed in the summer of second post-rewetting year (2022), where exceptionally high peaks of 24.5 mm/day, 13 mm/day, and 4.15 mm/day were recorded at R1, R2, and the control site, respectively. These peaks coincided with a period of intense precipitation, reaching 33 mm/day in July 2022. Interestingly, 3 years post-rewetting, the hydrographs of the rewetted catchments displayed muted responses, with lower peaks compared to those observed in the control site.

To examine our hypothesis that peatland rewetting enhances baseflow and reduces discharge variability, FDCs were applied. In this analysis, we chose not to separate individual post-rewetting years due to consistent trends observed across the three years. Distinct variations in the FDCs between the sites pre- and post-rewetting were found (Fig. 7). The rewetted and control sites exhibited different patterns of variability in FDCs for different flow levels. The FDCs at high flows exhibited similar behaviour, whereas intermediate and low flows showed greater differences between control and rewetted sites during the pre-rewetting year. In general, the FDC in catchment R1 appeared smoother than that of R2. At the rewetted catchment R1, there was an increase of 100% at high flow (0.58 to 1.16 mm/day for 20% exceedance probability) and 157% at low flow (0.14 to 0.36 mm/day for 70% exceedance probability) thresholds, respectively. At the rewetted catchment R2, a 69% increase in high flow (from 0.56 to 0.95 mm/day for 20% exceedance probability) and a 120% increase in low flow (0.05 to 0.11 mm/day for 70% exceedance probability) thresholds were observed. For comparison, the control site displayed an 87% increase in high flow (from 0.83 to 1.56 mm/day for 20% exceedance probability) and an 85% increase in low flow (from 0.14 to 0.26 mm/day for 70% exceedance probability) thresholds, respectively.

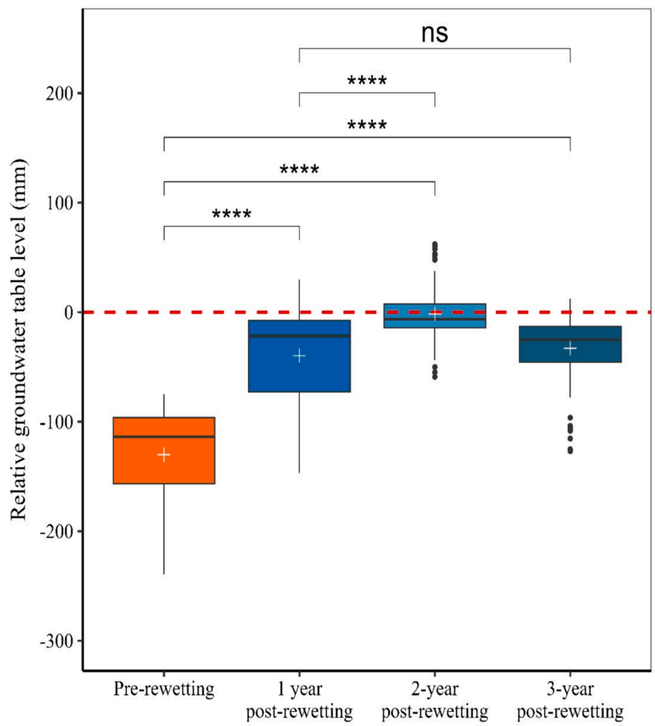

Fig. 3. Differences in groundwater table level between the rewetted and control sites for pre- and post-treatment (i.e., rewetting) periods. The difference is computed as treatment minus control; thus positive values indicate that GWL is greater at the treatment site than at the control site, while negative values indicate the opposite. The stars indicate the levels of significance in Wilcoxon test (*$p \leq 0.05$, **$p \leq 0.01$, ***$p \leq 0.001$, ****$p \leq 0.0001$) between individual years, while "ns" stands for not significant. For description of box plot elements, refer to Fig. 2 caption.

Table 1
Summary statistics of groundwater table level for control and rewetted sites during pre- and post-rewetting periods.

Site	State	Min mm	Max mm	Mean mm	Std. deviation mm	p-values
	2020					
Degerö	Control	−187	8.5	−86	48.7	
Trollberget	Pre-rewetting	−405	−70	−216	88.2	**<0.001**
	2021					
Degerö	Control	−170	2.7	−63	43	
Trollberget	1 year post-rewetting	−271	−3.7	−103	68.3	**<0.001**
	2022					
Degerö	Control	−147	−28.8	−83.5	28.6	
Trollberget	2-years post-rewetting	−187	−12.4	−85.1	37.6	0.75
	2023					
Degerö	Control	−234	−29	−119	59	
Trollberget	3-years post-rewetting	−312	−48	−152	78	**<0.001**

Bold p-values indicate a statistical significance at $p < 0.001$ (Wilcoxon test).

A notable rise in BFI was observed at both catchments of the rewetting site (i.e., R1 and R2) post-rewetting (Fig. 8). Before rewetting, the control site exhibited the highest BFI at 0.47, while the BFIs at R1 and R2 were 0.35 and 0.30, respectively. One year post-rewetting, there was an increase in BFI across all sites, albeit the rewetted catchments still demonstrated lower BFI compared to the control site. Interestingly, two years post-rewetting, R1 and the control site displayed identical BFIs, while R2′s BFI remained unchanged. The most striking change occurred three years post-rewetting, wherein both rewetted sites exhibited higher BFI compared to the control site, with R1 reaching a BFI of 0.72 and R2 reaching a BFI of 0.61.

3.4. *Groundwater-discharge relationship*

To test the hypothesis that rewetting increased water storage and hydrological buffer capacity, mean daily GWL and discharge data are plotted for summer and fall, both pre- and post-rewetting (Fig. 9). The results indicated a strong relationship between mean streamflow responses and mean GWL at both the control and rewetted sites. A threshold relationship was identified between mean specific discharges and GWL at all sites, demonstrating that elevated flows only occur once GWL exceeds a certain threshold level from the peat surface (Fig. 9). Furthermore, distinct differences were noted pre- and post-rewetting for rewetted catchments R1 and R2, while no change was observed in groundwater-discharge relationship at the control site. The inflection point of the fitted line and the minimum GWL required to generate flow at rewetted sites varied between the pre- and post-rewetting periods (Supplementary Table 1). Notably, all examined relationships were found to be statistically significant at $p < 0.01$. The coefficients derived highlighted a robust correlation between GWL and discharge at the control site during both the pre- and post-rewetting periods. Interestingly, the correlation was particularly robust for R1 ($r^2 = 0.8$) pre-rewetting. However, following rewetting, this correlation decreased to 0.3, indicating a shift in the dynamics of the relationship. Before rewetting, the GWL threshold for activation of elevated flow was −150 mm and −50 mm at the rewetted and control sites, respectively. After rewetting, the thresholds for both rewetted catchments became similar to the control site.

3.5. *Effect of rewetting on the monthly runoff coefficient*

The monthly runoff coefficients for the two catchments (R1 and R2) of the rewetted site relative to the control site exhibited a clear change following rewetting (Fig. 10, Supplementary Table 2). In general, the second and third post-rewetting years were distinctly different from the pre-rewetting period. Furthermore, the two rewetted catchments, R1 and R2, displayed distinct differences during the pre-rewetting and first post-rewetting periods, particularly regarding the runoff generated during the wet months. After rewetting, there was a substantial increase in runoff coefficient during dry months such as June and July, while wet months like August, September, and October showed a decrease in runoff coefficient at both rewetted sites. However, there was not a clear trend between each post-rewetting year at R1, as there was an increase in runoff coefficient in August and September one year post-rewetting, indicating that it was more productive in generating runoff than the control site.

4. Discussion

Despite significant scientific and political interest in peatland rewetting in Sweden, there is a limited body of literature addressing rewetting effects on hydrological functioning. Thus, there is a substantial knowledge gap regarding the impact of peatland rewetting on alterations in GWL, streamflow responses, and the scale of these modifications. We found that the rewetting process has led to notable changes in peatland hydrology; increasing GWL, reducing streamflow variations, and increasing water storage and hydrological buffer capacity towards values found at our natural peatland control sites.

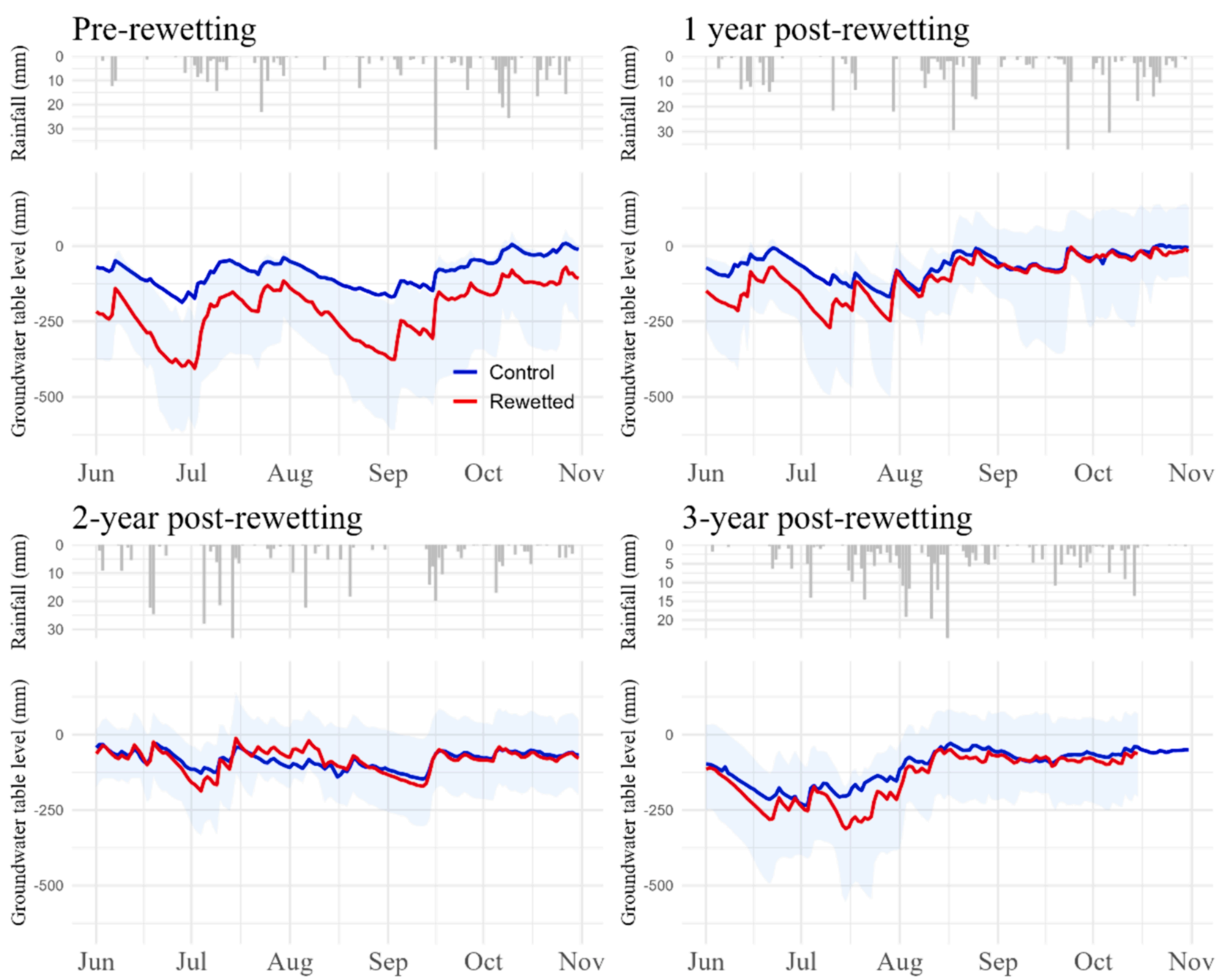

Fig. 4. Groundwater table level dynamics for the rewetted (red line) and control site (blue line) during the June-October period for the pre-rewetting year (2020) and the three post-rewetting years (2021, 2022, and 2023). The data for the rewetted site were available until October 15, 2023. The blue bands show max and min of GWL for all the dipwells at the rewetted site. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

4.1. Climatic context

During both the pre- and post-rewetting periods, no significant difference was observed in mean rainfall inputs, and there was likewise little variability in cumulative rainfall input. Additionally, analysis of snowpack for the pre- and post-rewetting years indicated that the pre-rewetting year was not characterized by drought conditions, as the snow depth was comparable to that of the post-rewetting years. Hence, any alterations in runoff and groundwater level data can be attributed to the impact of rewetting.

4.2. Effect of rewetting on GWL dynamics

Results support our first hypothesis, indicating a significant increase in GWL following the rewetting of the drained peatland. We found that peatland rewetting has resulted in a 64 mm increase in mean GWL at the rewetted site three years post-rewetting, compared to the pre-rewetting conditions. Moreover, we showed that the rise of the GWL remained stable at the rewetted site and that the difference in the GWL between the rewetted and control sites decreased remarkably with time (Fig. 3). In addition, we found that extreme fluctuations in summer GWL declined after rewetting. Comparing the rewetted site data to the control site suggests that GWL is returning to a more natural state. The observed changes in GWL stability suggest a potential increase in the water storage of peat caused by ditch blocking, leading to sustained high GWL levels between rainfall events and therefore remaining more stable during dry spells. Furthermore, the use of impermeable dam materials has facilitated the retention of water within the peatland, creating a permanent barrier against rapid drainage towards the stream, which, before rewetting, would have flushed through the site more quickly.

Nevertheless, the analysis of the GWL duration curve (Fig. 5) indicates a slight increase in the difference between GWLs at the rewetted and control sites in the third post-rewetting year. It is important to note that the third post-rewetting year was unusually warm and dry, especially during the early part of summer of 2023 and this may have contributed to the observed drop in GWL at the rewetted site. Hence, it is possible that the rewetted site, having been drained for a century, still may not function as a natural peatland during exceptionally dry periods, and a full hydrological recovery will take substantially longer. Several factors linked to prolonged drainage, such as peat oxidation and compaction, contribute to this extended recovery period (Liu and Lennartz, 2019). Peat oxidation, for instance, leads to increased bulk density, which in turn affects the site's ability to effectively retain and release water (Price et al., 2003). Consequently, even after rewetting, the site's hydrological recovery process may be protracted, requiring considerably more time than initially anticipated.

While our results, in general, are in line with several previous studies that have shown that restoration creates shallower GWL in the

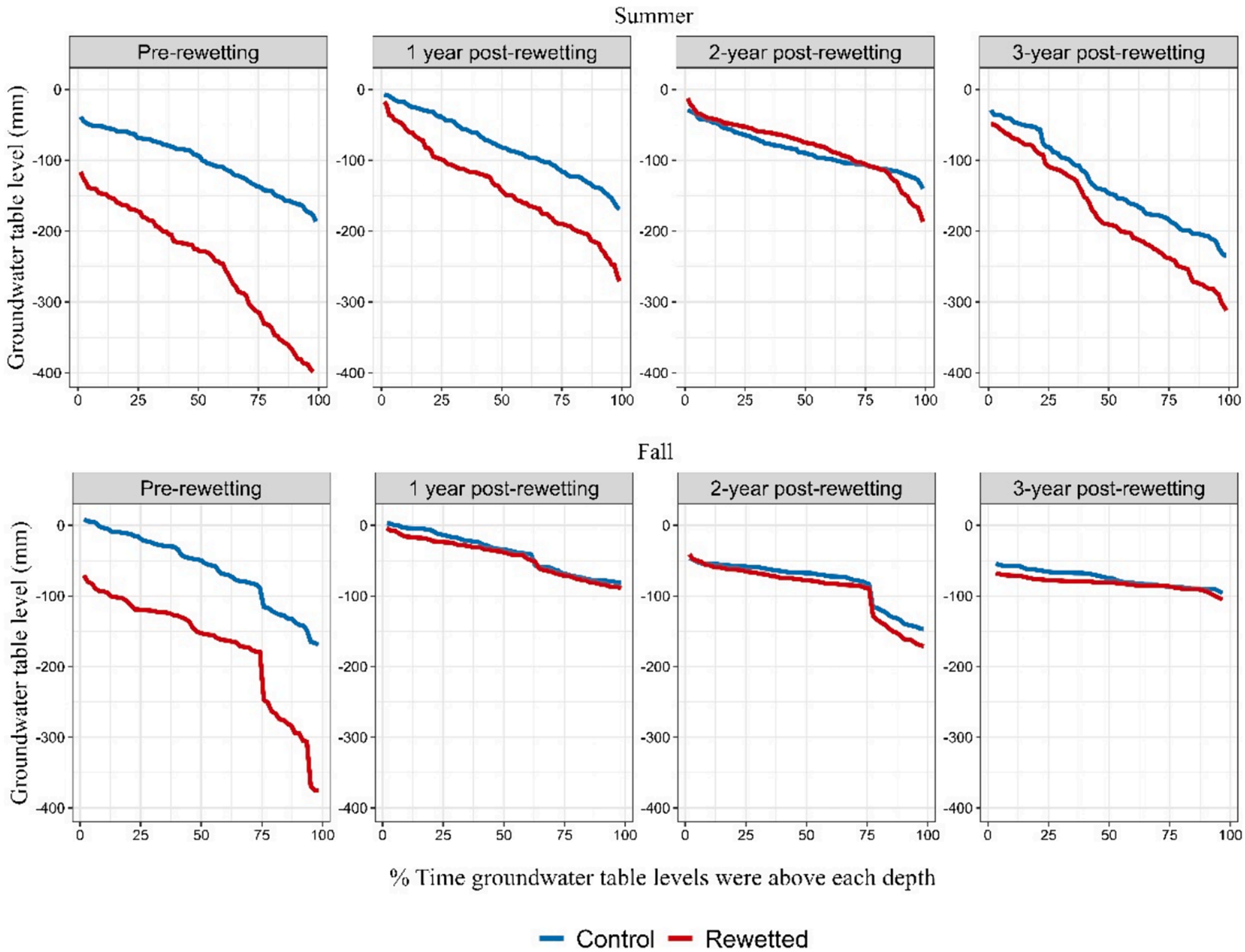

Fig. 5. Groundwater table level (mm) fluctuation duration curves during summer (June to August) and fall (September and October) for the control (blue line) and rewetted site (red line) for the pre-rewetting year (2020) and the three post-rewetting years (2021, 2022, and 2023). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

surrounding peat, and that the rise of GWL is generally rapid after restoration (Shantz and Price, 2006; Wilson et al., 2010; Haapalehto et al., 2011; Ketcheson and Price, 2011; Menberu et al., 2016; D'Acunha et al., 2018; Menberu et al., 2018), we also expand the general knowledge by addressing a notable gap in the existing literature. While reports of successful restoration efforts have mainly focused on studies in Finland, Canada, and the UK where the climatic and/or drainage activity has been substantially different from a Swedish context (Laudon et al., 2023), there is also a scarcity of data and studies specifically focused on peatland restoration outcomes from boreal Sweden (Bring et al., 2022). Especially compared to the same latitudes in Finland, drainage in Sweden was primarily done much earlier, dug less deep, and in a more unsystematic manner.

Comparing our results to other peatland restoration studies, GWL rise has been frequently reported, although the extent of recovery varies. For example, Menberu et al. (2016) conducted a study examining the impact of rewetting on different peatland types, including spruce mires, pine mires, and fens. Their findings revealed that a substantial proportion of previously drained sites exhibited higher GWL following the restoration measures. Before rewetting, the average differences in GWL ranged from −31 to −484 mm, whereas post-rewetting, these differences spanned from −177 to 357 mm. Menberu et al. (2016) also categorized the success of restoration by comparing the mean GWL position of the restored sites to that of pristine control sites. Notably, all poor fen sites exhibited successful restoration (well-restored), while three spruce mires were identified as being in an over-restored condition. Moreover, Ketcheson and Price (2011) examined block-cut bogs in Quebec, Canada, two years before and one year after the site was rewetted. Their findings indicated that the restoration led to an increase in the average site GWL, shifting from approximately −440 mm two years before restoration to −100 mm one year after restoration. However, their study lacked control catchments, introducing uncertainties regarding the net effect of rewetting, compared to changes in year to year variability in precipitation.

Similarly, Haapalehto et al. (2011) studied the effect of rewetting on an ombrotrophic bog and a minerotrophic fen in southern Finland. Their results showed that GWL rose soon after rewetting at both peatlands and remained higher than the control site, rising from approximately −450 to −100 mm, even 10 years after the restoration. However, the difference in the GWL between their rewetted and drained sites decreased over time (10 years post-rewetting), which could be due to the potential irreversible alterations in physical properties. These alterations, such as peat subsidence induced by prolonged drainage, could have contributed to the observed changes. Additionally, the declining efficacy of peat dams over time may have facilitated easier water permeation through the dam material (Holden and Burt, 2003).

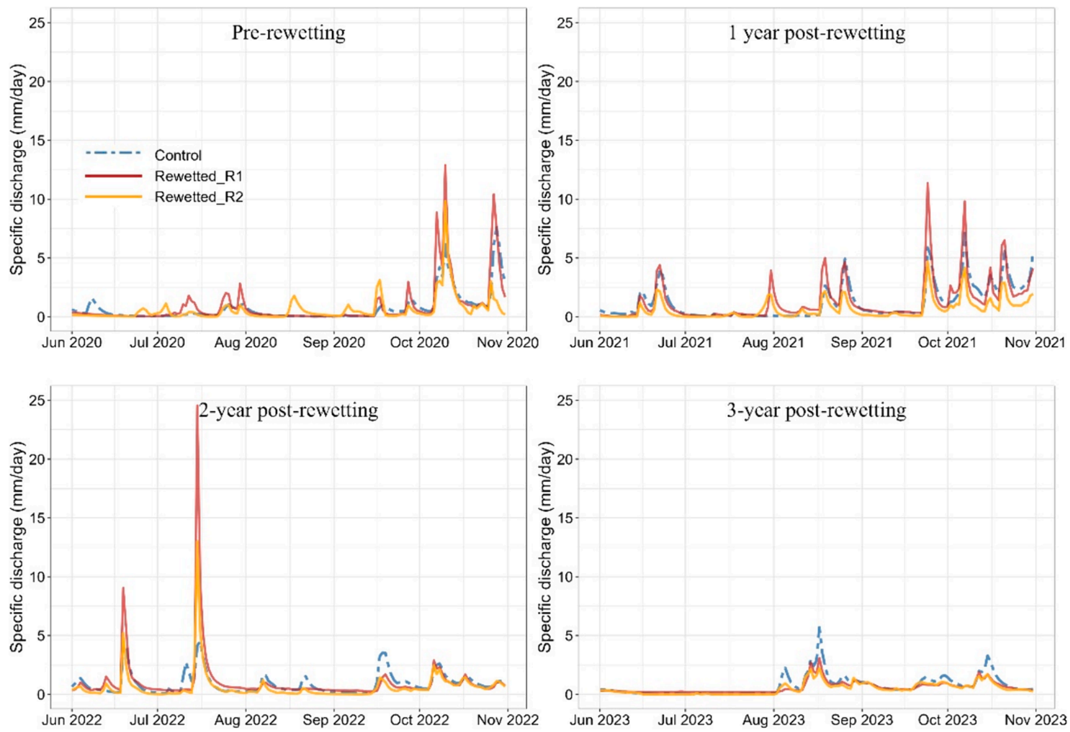

Fig. 6. Hydrographs of control and rewetted sites (Rewetted_R1 and Rewetted_R2) for pre- and post-rewetting years.

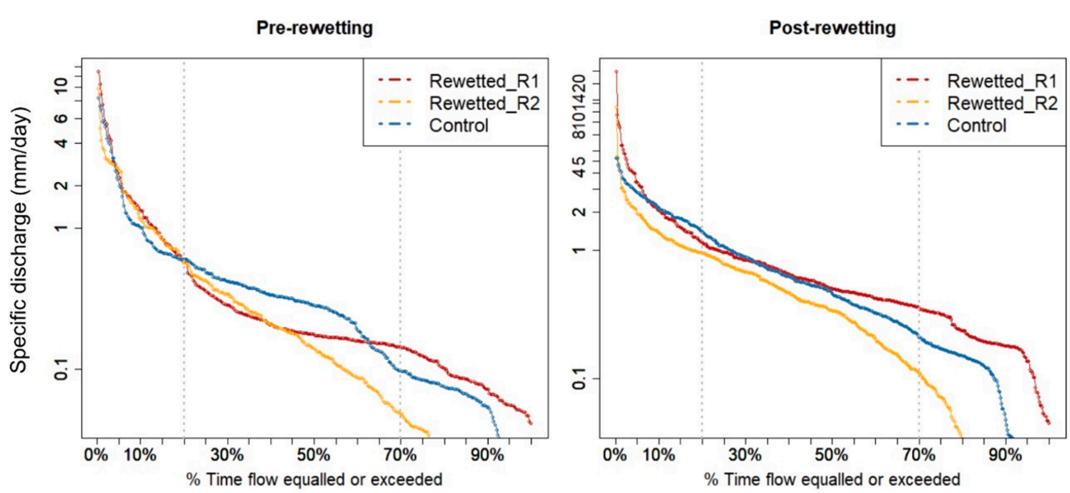

Fig. 7. Flow duration curve showing exceedance probability of specific discharge for rewetted (R1 and R2) and control sites during the pre- (2020) and post-rewetting years (2021–2023). The vertical dashed lines represent thresholds between high flow (0–20%), mid flow (20–70%) and low flow segments (70–90%).

However, a few studies have also reported that rewetting had no or only small effects on GWL. For instance, Wilson et al. (2019) studied the effect of rewetting on a blanket bog in the UK. Although the rewetted bog exhibited a slow recovery, characterized by reduced dry areas and increased surface water occurrence, the rise in GWL was relatively small, estimated at around 2 cm. This is possibly due to the steeper slopes of the blanket peat bog on hillslopes. Similarly, Holden et al. (2011) investigated GWL dynamics on intact, drained, and restored peatland slopes in a blanket peat in northern England. Surprisingly, even after several years of management intervention, they observed no significant effect on GWL. This could be attributed to the inherent nature of bogs as enclosed systems, primarily reliant on precipitation for water input. In contrast, fens are sustained by both groundwater and precipitation. Consequently, during periods of low rainfall, peat dams may prove less effective in maintaining GWL up to the natural peatland. Moreover, Holden et al. (2017) examined the impacts of ditch blocking on blanket peatlands in North Wales over a four-year period using a similar BACI design as our study. According to their findings, the ditch blocking methods did not have a significant impact on GWL in relation to the peat surface over the entire rewetted period, using a strict statistical approach (Time-weighted mean effect < 20 mm). This lack of impact on GWL was attributed to the already shallow GWL in their peatland, due to high precipitation levels, coupled with ineffective drainage from existing ditches, which may have contributed to stabilizing the GWL. Additionally, the ditches in their study were situated on steeper downslope locations, further diminishing the efficacy of ditch blocking.

Various factors may explain the diverse GWL outcomes of peatland restoration studies. GWL response is influenced by different factors, including site topography (Holden et al., 2006), block position (Holden, 2005), spacing and distance from the ditch (Dunn and Mackay, 1996), as

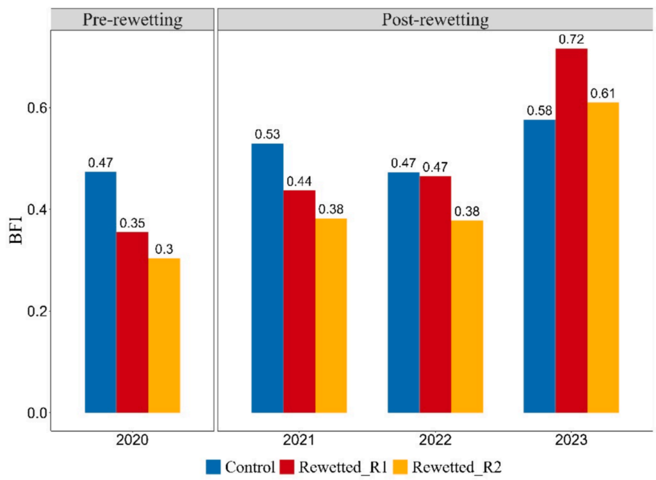

Fig. 8. Baseflow index (BFI) of control and rewetted sites during the pre-(2020) and post-rewetting years (2021–2023).

well as the peat structure (Holden et al., 2006; Sherwood et al., 2013). For example, Ketcheson and Price (2011) showed that the extent of GWL increase at a specific site was significantly impacted by the variability in topography and the placement of peat blocks. Furthermore, the distance between the measurement points and the rewetted site is a critical factor that may account for the variations observed in GWL. Wilson et al. (2010) demonstrated that the vertical and horizontal distance from the drains also play a significant role in shaping the behavior of GWL following rewetting. Furthermore, to assess the success of any restoration programs, it is important to evaluate peat structures. For example, ditch blocking may raise the GWL immediately; however, it could also lead to flashier GWL due to changes in the structure of the peat, by mechanisms related to subsidence, shrinkage, and oxidation resulting from historical drainage practices.

Furthermore, the rate of GWL recovery may vary significantly, as evidenced by contrasting reports. Wilson et al. (2019) reported a slow response, while Haapalehto et al. (2011) observed a rapid recovery of GWL. Therefore, changes in peat hydrology may not occur uniformly, leading to a delayed response of GWL to rewetting. For instance, Holden et al. (2011) found that six years after restoration, GWL and hydrological behavior were not similar to those of a comparative undisturbed site. These delayed responses can be attributed to changes in peat structure, hydrological conductivity, and vegetation following drainage, which impact the hydrological characteristics of peat (Holden et al., 2006; Wallage and Holden, 2011; Ramchunder et al., 2012; Ballard et al., 2012). While our observation of a rapid rise and stability in GWL at our rewetted site is similar to the level of the control, it is worth noting that this process may not be fully complete due to the above-stated factors.

4.3. *Effect of rewetting on baseflow*

The flow duration curve analysis provided support for the second hypothesis, indicating that the rewetting has resulted in a discernible increase in baseflow. Overall, the most notable changes in streamflow patterns occurred during low-flow periods following rewetting. In particular, the relative increase in the low-flow threshold compared to high-flows was greater at the two rewetted catchments (2.5 and 2.2 times higher at R1 and R2, respectively) in contrast to the control site (1.8 times). This is supported by the results illustrated in Fig. 8, demonstrating a doubling of the increase in BFI at the two rewetted catchments three years post-rewetting. These results suggest that rewetting had a substantial impact on maintaining baseflow and preventing streams from drying up during drought. The noticeable increase in baseflow at the rewetted site can be attributed to the improved water storage of the peatland. Following rewetting, the peatland is likely to have better retention of spring snowmelt, which is later released during the summer months. These results are consistent with other studies that have shown an increase in baseflow regulated by peatland restoration (Holden et al., 2017; Howson et al., 2021; Norbury et al., 2021). However, there are more studies supporting the impact of peatland rewetting on peak flow attenuation compared to those focusing on low flow

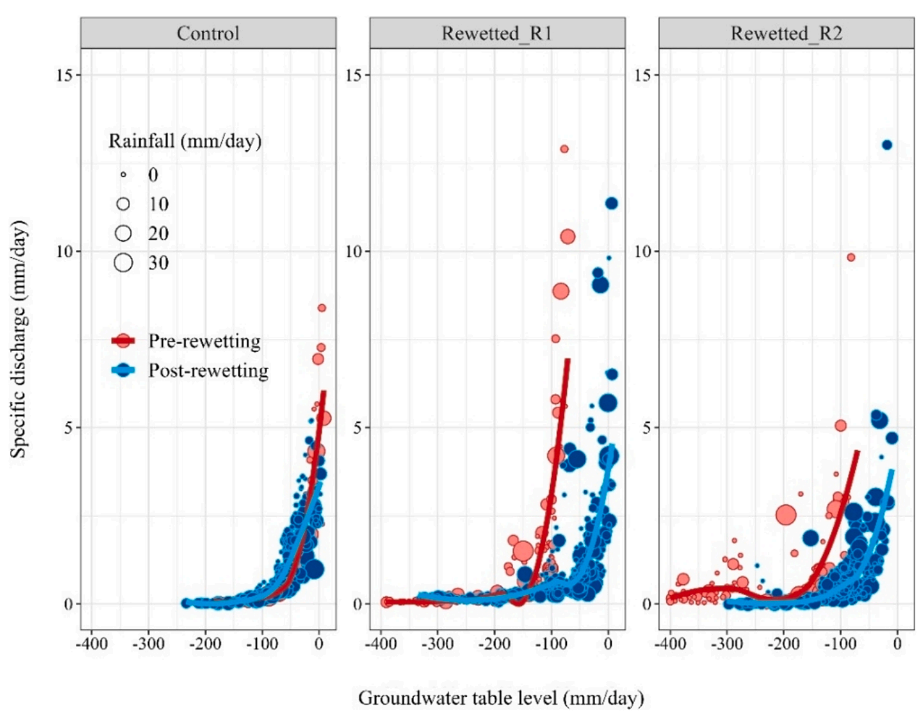

Fig. 9. Scatter plots of daily specific discharge (mm/day) versus groundwater table level (mm/day) during 2020 (Pre-rewetting) and 2021–2023 (Post-rewetting) for rewetted and control sites. Smoothed lines are provided for visual aid only. The varying sizes of circles denote the corresponding amounts of rainfall recorded on each respective day.

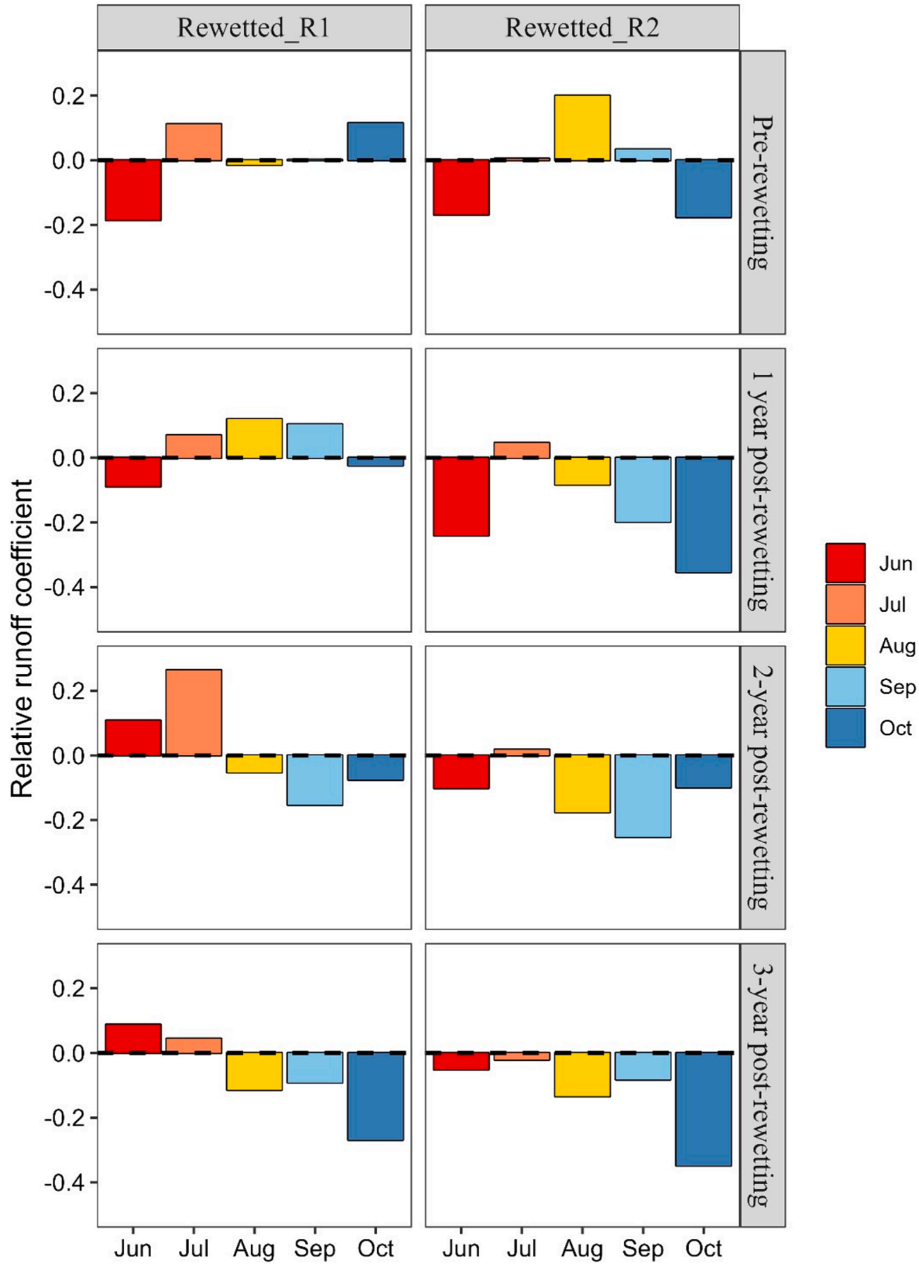

Fig. 10. Relative monthly runoff coefficients at the two rewetted catchments (R1 and R2) for the pre-rewetting year (2020) and the three post-rewetting years (2021, 2022, and 2023). Positive values indicate that the metric is greater at the rewetted sites than at the control site, while negative values indicate the opposite.

augmentation (Wilson et al., 2010; Shuttleworth et al., 2019; Gatis et al., 2023). Thus, further studies are needed in order to address contradictory findings regarding rewetting impacts on baseflow.

Our results also support the hypothesis that peatland rewetting increased water storage and hydrological buffer capacity. This is demonstrated by the distinct threshold behavior in the relationship between the GWL and discharge (see Fig. 9). Before rewetting, the initiation of flow occurred in the drainage ditch when the GWL was approximately within 100 mm of the peat surface. However, after rewetting, flow generation was observed when the GWL was much closer to the surface, typically within 20 mm of the peat. This increase is presumably linked to the positive impact of restoration activities on the peatland water storage and hydrological buffer capacity, as GWL thresholds play a crucial role in regulating a catchment's ability to transfer water to its outlet and thereby regulating hydrological stream processes (Spence, 2007). Identifying these relationships and thresholds is particularly important in peatland ecosystems, given the proximity of the water level to the surface, where any increase may result in flood events (Evans et al, 1999). Our results agree with other findings that underline the impact of peatland rewetting on the increase in water storage and hydrological buffer capacity. Ahmad et al. (2020) demonstrated this by calculating the ratio of precipitation event size to GWL rise as a proxy for specific storage capacity, which was higher in their rewetted fen study. A recent study by Stachowicz et al. (2022) quantified the benefits of peatland rewetting, specifically, the possibility of changes in water storage capacity within the Neman River Basin within the Baltic Sea Region. Based on their findings, rewetting can be an efficient management technique for enhancing the storage capacity of the basin. Moreover, a study by Ketcheson and Price (2011) investigating the impact of ditch blocking on the hydrology of an abandoned cutover peatland, found that peatland rewetting resulted in an increased storage capacity a year after rewetting, and the peatland responded to precipitation inputs only after a critical storage threshold had been reached.

While many studies have focused on the potential decrease in water storage capacity that may occur after peatland rewetting due to the increase in the GWL (Shuttleworth et al., 2019), there is another mechanism related to the unique physical property of peat soils that may contribute to an increase in water storage and hydrological buffer capacity. This mechanism is known as 'mire breathing,' which is the ability of peat to swell and shrink resulting from wetting and drying processes (Ingram, 1983; Anderson and Burt, 1990; Evans et al., 1999; Camporese et al., 2006; Rezanezhad et al., 2016). The GWL directly influences mire breathing, with a higher GWL leading to increased peat height (Morton and Heinemeyer, 2019; Nijp et al., 2019). However, it is crucial to note that a portion of peat may undergo irreversible consolidation due to prolonged drainage (Balliston and Price, 2023). Upon rewetting peatlands, the elevated GWL results in increased moisture levels within the peat. This causes the peat to swell, enlarging the pore sizes and enhancing the peat's ability to buffer hydrological changes. As a result, swollen peat can retain more water during heavy rainfalls (Howie and Hebda, 2018).

Additionally, through our BACI design, which assesses deviations from the control, we have demonstrated that the rewetted sites generated much lower runoff coefficients compared to the control site after rewetting. Specifically, three years post-rewetting, during the wet month of October, the runoff produced by the control site was 1.3 and 1.5 times higher than that of R1 and R2, respectively (Supplementary Table 2). We also observed higher runoff production during the dry months of June and July, particularly at rewetted catchment R1. The transition from a lower to a higher runoff coefficient observed at R1 in August and September during the first post-rewetting year, was likely attributed to the abrupt rise in the GWL. The catchment reached a saturated state, thereby increasing the potential for runoff generation compared to the pre-rewetting year. However, two and three years after restoration, likely with the development of peat properties and vegetation, there was a discernible decrease in the overall production of runoff. These findings also confirm the hypothesis that rewetting increases water storage and hydrological buffer capacity, allowing the catchment to store more water during snowmelt and wet periods, and release it slowly during dry months. Furthermore, our results align with Shantz and Price (2006), who studied the effect of ditch-blocking on a bog previously drained for peat extraction in Quebec. Their findings demonstrated a decrease in runoff coefficient at the rewetted site, indicating the beneficial impact of restoration methods for storing water within the site.

The data and analysis presented in this study provide empirical evidence that peatland rewetting immediately increased GWL and baseflow, and also enhanced the water storage and hydrological buffer capacity within the rewetted site. Our study highlights the importance of ongoing monitoring and evaluation to ensure the long-term effects of restoration efforts and the sustained success of peatland rewetting projects. Furthermore, a crucial aspect to follow up in rewetting projects is vegetation dynamics. Shuttleworth et al. (2019) demonstrated in their study that vegetation, due to its surface roughness, can significantly influence peatland runoff response. Degradation, such as drainage, has a direct impact on vegetation dynamics; therefore, vegetation recovery and changes are important variables to consider in assessing long-term consequences and planning future studies. The findings of this study can be utilized to support and guide decision-makers by valuable insights into the anticipated initial ecosystem responses in the context of restoring wetlands in boreal climates following rewetting activities. This is particularly important in Sweden, where such information is generally lacking.

5. Conclusions

In our study of a Swedish boreal peatland, we investigated the impact of rewetting on the GWL and runoff dynamics during ice- and snow-free periods. Using two nearby natural peatlands as controls, we observed significant positive effects on the hydrological functioning of the rewetted site. Rewetting significantly shifted the GWL position toward the ground surface, resulting in a more stable GWL condition that more closely resembled that of the control sites. The study also highlighted a substantial increase in baseflow at the rewetted site, particularly during low-flow periods. Moreover, rewetting appeared to enhance water storage and hydrological buffer capacity, as indicated by a higher GWL threshold for runoff initiation. Additionally, there was a reduction in the runoff coefficient, generally considered a good indicator of peatland water storage and hydrological buffer capacity, especially during wet periods.

CRediT authorship contribution statement

Shirin Karimi: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Eliza Maher Hasselquist:** Writing – review & editing, Validation, Supervision, Resources, Investigation, Funding acquisition, Data curation, Conceptualization. **Shokoufeh Salimi:** Writing – review & editing, Conceptualization. **Järvi Järveoja:** Conceptualization, Data curation, Methodology, Resources, Supervision, Writing – review & editing. **Hjalmar Laudon:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2024.131729.

References

Ågren, A.M., Larson, J., Paul, S.S., Laudon, H., Lidberg, W., 2021. Use of multiple LIDAR-derived digital terrain indices and machine learning for high-resolution national-

scale soil moisture mapping of the Swedish forest landscape. Geoderma 404, 115280.
Ahmad, S., Liu, H., Günther, A., Couwenberg, J., Lennartz, B., 2020. Long-term rewetting of degraded peatlands restores hydrological buffer function. Sci. Total Environ. 749, 141571.
Allott, T.E.H., Evans, M.G., Lindsay, J.B., Agnew, C.T., Freer, J.E., Jones, A., Parnell, M., 2009. Water tables in Peak District blanket peatlands. Moors for the Future Reports.
Anderson, M.G. and Burt, T.P., 1990. Process studies in hillslope hydrology: an overview. In Process studies in hillslope hydrology (pp. 1-8).
Aronson, J., Goodwin, N., Orlando, L., Eisenberg, C., Cross, A.T., 2020. A world of possibilities: six restoration strategies to support the United Nation's Decade on Ecosystem Restoration. Restor. Ecol. 28 (4), 730–736.
Ballard, C.E., McIntyre, N., Wheater, H.S., 2012. Effects of peatland drainage management on peak flows. Hydrol. Earth Syst. Sci. 16 (7), 2299–2310.
Balliston, N.E., Price, J.S., 2023. Aquifer depressurization and water table lowering induces landscape scale subsidence and hydrophysical change in peatlands of the Hudson Bay Lowlands. Sci. Total Environ. 855, 158837.
Bring, A., Rosén, L., Thorslund, J., Tonderski, K., Åberg, C., Envall, I., Laudon, H., 2020. Groundwater storage effects from restoring, constructing or draining wetlands in temperate and boreal climates: a systematic review protocol. Environmental Evidence 9, 1–11.
Bring, A., Thorslund, J., Rosén, L., Tonderski, K., Åberg, C., Envall, I., Laudon, H., 2022. Effects on groundwater storage of restoring, constructing or draining wetlands in temperate and boreal climates: a systematic review. Environmental Evidence 11 (1), 38.
Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. Hydrol. Earth Syst. Sci. 7 (3), 358–389.
Camporese, M., Ferraris, S., Putti, M., Salandin, P., Teatini, P., 2006. Hydrological modeling in swelling/shrinking peat soils. Water Resour. Res. 42 (6).
Casselgård, M., 2020. Effects of 100 years of drainage on peat properties in a drained peatland forests in northern Sweden.
D'Acunha, B., Lee, S.C., Johnson, M.S., 2018. Ecohydrological responses to rewetting of a highly impacted raised bog ecosystem. Ecohydrology 11 (1), e1922.
Dinsmore, K.J., Skiba, U.M., Billett, M.F., Rees, R.M., 2009. Effect of water table on greenhouse gas emissions from peatland mesocosms. Plant and Soil 318, 229–242.
Erwin, K.L., 2009. Wetlands and global climate change: the role of wetland restoration in a changing world. Wetl. Ecol. Manag. 17 (1), 71–84.
Evans, M.G., Burt, T.P., Holden, J., Adamson, J.K., 1999. Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. J. Hydrol. 221 (3–4), 141–160.
Fluet-Chouinard, E., Stocker, B.D., Zhang, Z., Malhotra, A., Melton, J.R., Poulter, B., Kaplan, J.O., Goldewijk, K.K., Siebert, S., Minayeva, T., Hugelius, G., 2023. Extensive global wetland loss over the past three centuries. Nature 614 (7947), 281–286.
Frolking, S., Talbot, J., Jones, M.C., Treat, C.C., Kauffman, J.B., Tuittila, E.S., Roulet, N., 2011. Peatlands in the Earth's 21st century climate system. Environ. Rev. 19 (NA), 371–396.
Haapalehto, T.O., Vasander, H., Jauhiainen, S., Tahvanainen, T., Kotiaho, J.S., 2011. The effects of peatland restoration on water-table depth, elemental concentrations, and vegetation: 10 years of changes. Restor. Ecol. 19 (5), 587–598.
Härkönen, L.H., Lepistö, A., Sarkkola, S., Kortelainen, P., Räike, A., 2023. Reviewing peatland forestry: Implications and mitigation measures for freshwater ecosystem browning. For. Ecol. Manage. 531, 120776.
Harris, L.I., Moore, T.R., Roulet, N.T., Pinsonneault, A.J., 2020. Limited effect of drainage on peat properties, porewater chemistry, and peat decomposition proxies in a boreal peatland. Biogeochemistry 151, 43–62.
Helbig, M., Waddington, J.M., Alekseychik, P., Amiro, B.D., Aurela, M., Barr, A.G., Black, T.A., Blanken, P.D., Carey, S.K., Chen, J., Chi, J., 2020. Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. Nat. Clim. Chang. 10 (6), 555–560.
Holden, J., 2005. Peatland hydrology and carbon release: why small-scale process matters. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 363 (1837), 2891–2913.
Holden, J., Burt, T.P., 2003. Hydraulic conductivity in upland blanket peat: measurement and variability. Hydrol. Process. 17 (6), 1227–1237.
Holden, J., Chapman, P.J., Labadz, J.C., 2004. Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. Prog. Phys. Geogr. 28 (1), 95–123.
Holden, J., Evans, M.G., Burt, T.P., Horton, M., 2006. Impact of land drainage on peatland hydrology. J. Environ. Qual. 35 (5), 1764–1778.
Holden, J., Wallage, Z.E., Lane, S.N., McDonald, A.T., 2011. Water table dynamics in undisturbed, drained and restored blanket peat. J. Hydrol. 402 (1–2), 103–114.
Holden, J., Green, S.M., Baird, A.J., Grayson, R.P., Dooling, G.P., Chapman, P.J., Evans, C.D., Peacock, M., Swindles, G., 2017. The impact of ditch blocking on the hydrological functioning of blanket peatlands. Hydrol. Process. 31 (3), 525–539.
Howie, S.A., Hebda, R.J., 2018. Bog surface oscillation (mire breathing): A useful measure in raised bog restoration. Hydrol. Process. 32 (11), 1518–1530.
Howson, T., Chapman, P.J., Shah, N., Anderson, R., Holden, J., 2021. The effect of forest-to-bog restoration on the hydrological functioning of raised and blanket bogs. Ecohydrology 14 (7), e2334.
Ingram, H.A.P., 1983. Hydrology. Mires: swamp, bog, fen and moor-General studies.
Kadykalo, A.N., Findlay, C.S., 2016. The flow regulation services of wetlands. Ecosyst. Serv. 20, 91–103.
Ketcheson, S.J., Price, J.S., 2011. The impact of peatland restoration on the site hydrology of an abandoned block-cut bog. Wetlands 31, 1263–1274.
Kettenring, K.M., Tarsa, E.E., 2020. Need to seed? Ecological, genetic, and evolutionary keys to seed-based wetland restoration. Front. Environ. Sci. 8, 109.
Kwon, M.J., Ballantyne, A., Ciais, P., Qiu, C., Salmon, E., Raoult, N., Guenet, B., Göckede, M., Euskirchen, E.S., Nykänen, H., Schuur, E.A., 2022. Lowering water table reduces carbon sink strength and carbon stocks in northern peatlands. Glob. Chang. Biol. 28 (22), 6752–6770.
Ladson, A.R., Brown, R., Neal, B., Nathan, R., 2013. A standard approach to baseflow separation using the Lyne and Hollick filter. Australasian Journal of Water Resources 17 (1), 25–34.
Laudon, H., Taberman, I., Ågren, A., Futter, M., Ottosson-Löfvenius, M., Bishop, K., 2013. The Krycklan Catchment Study—A flagship infrastructure for hydrology, biogeochemistry, and climate research in the boreal landscape. Water Resour. Res. 49 (10), 7154–7158.
Laudon, H., Hasselquist, E.M., Peichl, M., Lindgren, K., Sponseller, R., Lidman, F., Kuglerova, L., Hasselquist, N.J., Bishop, K., Nilsson, M.B., Ågren, A.M., 2021. Northern landscapes in transition: Evidence, approach and ways forward using the Krycklan Catchment Study. Hydrol. Process. 35 (4), e14170.
Laudon, H., Lidberg, W., Sponseller, R.A., Maher Hasselquist, E., Westphal, F., Östlund, L., Sandström, C., Järveoja, J., Peichl, M., Ågren, A.M., 2022. Emerging technology can guide ecosystem restoration for future water security. Hydrol. Process. 36 (10).
Laudon, H., Mosquera, V., Eklöf, K., Järveoja, J., Karimi, S., Krasnova, A., Peichl, M., Pinkwart, A., Tong, C.H.M., Wallin, M.B., Zannella, A., 2023. Consequences of rewetting and ditch cleaning on hydrology, water quality and greenhouse gas balance in a drained northern landscape. Sci. Rep. 13 (1), 20218.
Leach, J.A., Larsson, A., Wallin, M.B., Nilsson, M.B., Laudon, H., 2016. Twelve year interannual and seasonal variability of stream carbon export from a boreal peatland catchment. J. Geophys. Res. Biogeo. 121 (7), 1851–1866.
Leifeld, J., Wüst-Galley, C., Page, S., 2019. Intact and managed peatland soils as a source and sink of GHGs from 1850 to 2100. Nat. Clim. Chang. 9 (12), 945–947.
Liu, H., Lennartz, B., 2019. Hydraulic properties of peat soils along a bulk density gradient—A meta study. Hydrol. Process. 33 (1), 101–114.
Liu, H., Rezanezhad, F., Lennartz, B., 2022. Impact of land management on available water capacity and water storage of peatlands. Geoderma 406, 115521.
Locky, D.A., Bayley, S.E., 2006. Plant diversity, composition, and rarity in the southern boreal peatlands of Manitoba. Canada. Botany 84 (6), 940–955.
Loisel, J., Gallego-Sala, A., 2022. Ecological resilience of restored peatlands to climate change. Communications Earth & Environment 3 (1), 208.
Lyne, V., Hollick, M., 1979, September.. Stochastic time-variable rainfall-runoff modelling. In: Institute of Engineers Australia National Conference, Vol. 79(10. Institute of Engineers Australia, Barton, Australia, pp. 89–93.
Malmström, C., 1923. Degerö stormyr: en botanisk, hydrologisk och utvecklingshistorisk undersökning över ett nordsvenskt myrkomplex (Vol. 20, No. 1). Centraltryckeriet.
Menberu, M.W., Tahvanainen, T., Marttila, H., Irannezhad, M., Ronkanen, A.K., Penttinen, J., Kløve, B., 2016. Water-table-dependent hydrological changes following peatland forestry drainage and restoration: Analysis of restoration success. Water Resour. Res. 52 (5), 3742–3760.
Menberu, M.W., Haghighi, A.T., Ronkanen, A.K., Marttila, H., Kløve, B., 2018. Effects of drainage and subsequent restoration on peatland hydrological processes at catchment scale. Water Resour. Res. 54 (7), 4479–4497.
Morton, P.A., Heinemeyer, A., 2019. Bog breathing: the extent of peat shrinkage and expansion on blanket bogs in relation to water table, heather management and dominant vegetation and its implications for carbon stock assessments. Wetl. Ecol. Manag. 27 (4), 467–482.
Nijp, J.J., Metselaar, K., Limpens, J., Bartholomeus, H.M., Nilsson, M.B., Berendse, F., van der Zee, S.E., 2019. High-resolution peat volume change in a northern peatland: Spatial variability, main drivers, and impact on ecohydrology. Ecohydrology 12 (6), e2114.
Nilsson, M., Sagerfors, J., Buffam, I., Laudon, H., Eriksson, T., Grelle, A., Klemedtsson, L., Weslien, P.E.R., Lindroth, A., 2008. Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire–a significant sink after accounting for all C-fluxes. Glob. Chang. Biol. 14 (10), 2317–2332.
Norbury, M., Phillips, H., Macdonald, N., Brown, D., Boothroyd, R., Wilson, C., Quinn, P., Shaw, D., 2021. Quantifying the hydrological implications of pre-and post-installation willowed engineered log jams in the Pennine Uplands, NW England. J. Hydrol. 603, 126855.
Noumonvi, K.D., Ågren, A.M., Ratcliffe, J.L., Öquist, M.G., Ericson, L., Tong, C.H.M., Järveoja, J., Zhu, W., Osterwalder, S., Peng, H., Erefur, C., 2023. The Kulbäcksliden research infrastructure: a unique setting for northern peatland studies. Front. Earth Sci. 11, 1194749.
Peichl, M., Martínez-García, E., Fransson, J.E., Wallerman, J., Laudon, H., Lundmark, T., Nilsson, M.B., 2023. Landscape-variability of the carbon balance across managed boreal forests. Glob. Chang. Biol. 29 (4), 1119–1132.
Price, J.S., Heathwaite, A.L., Baird, A.J., 2003. Hydrological processes in abandoned and restored peatlands: an overview of management approaches. Wetl. Ecol. Manag. 11, 65–83.
R Core Team, 2023. _R: A Language and Environment for Statistical Computing_. R Foundation for Statistical Computing, Vienna, Austria https://www.R-project.org/.
Ramchunder, S.J., Brown, L.E., Holden, J., 2012. Catchment-scale peatland restoration benefits stream ecosystem biodiversity. J. Appl. Ecol. 49 (1), 182–191.
Regan, S., Flynn, R., Gill, L., Naughton, O., Johnston, P., 2019. Impacts of groundwater drainage on peatland subsidence and its ecological implications on an Atlantic raised bog. Water Resour. Res. 55 (7), 6153–6168.
Rezanezhad, F., Price, J.S., Quinton, W.L., Lennartz, B., Milojevic, T., Van Cappellen, P., 2016. Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. Chem. Geol. 429, 75–84.

Schimelpfenig, D.W., Cooper, D.J., Chimner, R.A., 2014. Effectiveness of ditch blockage for restoring hydrologic and soil processes in mountain peatlands. Restor. Ecol. 22 (2), 257–265.
Schrautzer, J., Sival, F., Breuer, M., Runhaar, H., Fichtner, A., 2013. Characterizing and evaluating successional pathways of fen degradation and restoration. Ecol. Ind. 25, 108–120.
Shantz, M.A., Price, J.S., 2006. Characterization of surface storage and runoff patterns following peatland restoration, Quebec, Canada. Hydrological Processes: an International Journal 20 (18), 3799–3814.
Sherwood, J.H., Kettridge, N., Thompson, D.K., Morris, P.J., Silins, U., Waddington, J.M., 2013. Effect of drainage and wildfire on peat hydrophysical properties. Hydrol. Process. 27 (13), 1866–1874.
Shuttleworth, E.L., Evans, M.G., Pilkington, M., Spencer, T., Walker, J., Milledge, D., Allott, T.E., 2019. Restoration of blanket peat moorland delays stormflow from hillslopes and reduces peak discharge. Journal of Hydrology X 2, 100006.
Silins, U., Rothwell, R.L., 1998. Forest peatland drainage and subsidence affect soil water retention and transport properties in an Alberta peatland. Soil Sci. Soc. Am. J. 62 (4), 1048–1056.
Smakhtin, V.U., 2001. Low flow hydrology: a review. J. Hydrol. 240 (3–4), 147–186.
Spence, C., 2007. On the relation between dynamic storage and runoff: A discussion on thresholds, efficiency, and function. Water Resour. Res. 43 (12).
Stachowicz, M., Manton, M., Abramchuk, M., Banaszuk, P., Jarašius, L., Kamocki, A., Povilaitis, A., Samerkhanova, A., Schäfer, A., Sendžikaitė, J., Wichtmann, W., 2022. To store or to drain—To lose or to gain? Rewetting drained peatlands as a measure for increasing water storage in the transboundary Neman River Basin. Sci. Total Environ. 829, 154560.
Swindles, G.T., Morris, P.J., Mullan, D.J., Payne, R.J., Roland, T.P., Amesbury, M.J., Lamentowicz, M., Turner, T.E., Gallego-Sala, A., Sim, T., Barr, I.D., 2019. Widespread drying of European peatlands in recent centuries. Nat. Geosci. 12 (11), 922–928.
Temmerman, S., Meire, P., Bouma, T.J., Herman, P.M., Ysebaert, T., De Vriend, H.J., 2013. Ecosystem-based coastal defence in the face of global change. Nature 504 (7478), 79–83.
Waddington, J.M., Quinton, W.L., Price, J.S., Lafleur, P.M., 2009. Advances in Canadian peatland hydrology, 2003–2007. Canadian Water Resources Journal 34 (2), 139–148.
Waltham, N.J., Elliott, M., Lee, S.Y., Lovelock, C., Duarte, C.M., Buelow, C., Simenstad, C., Nagelkerken, I., Claassens, L., Wen, C.K. and Barletta, M., 2020. UN decade on ecosystem restoration 2021–2030—what chance for success in restoring coastal ecosystems?. Frontiers in Marine Science, p.71.
Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A., Morris, M., 2010. Recovery of water tables in Welsh blanket bog after drain blocking: discharge rates, time scales and the influence of local conditions. J. Hydrol. 391 (3–4), 377–386.
Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A., Morris, M., 2011. The impact of drain blocking on an upland blanket bog during storm and drought events, and the importance of sampling-scale. J. Hydrol. 404 (3–4), 198–208.
Zedler, J.B., 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Front. Ecol. Environ. 1 (2), 65–72.