Riparian wetland rehabilitation and beaver re-colonisation impacts on hydrological processes and water quality in a lowland agricultural catchment

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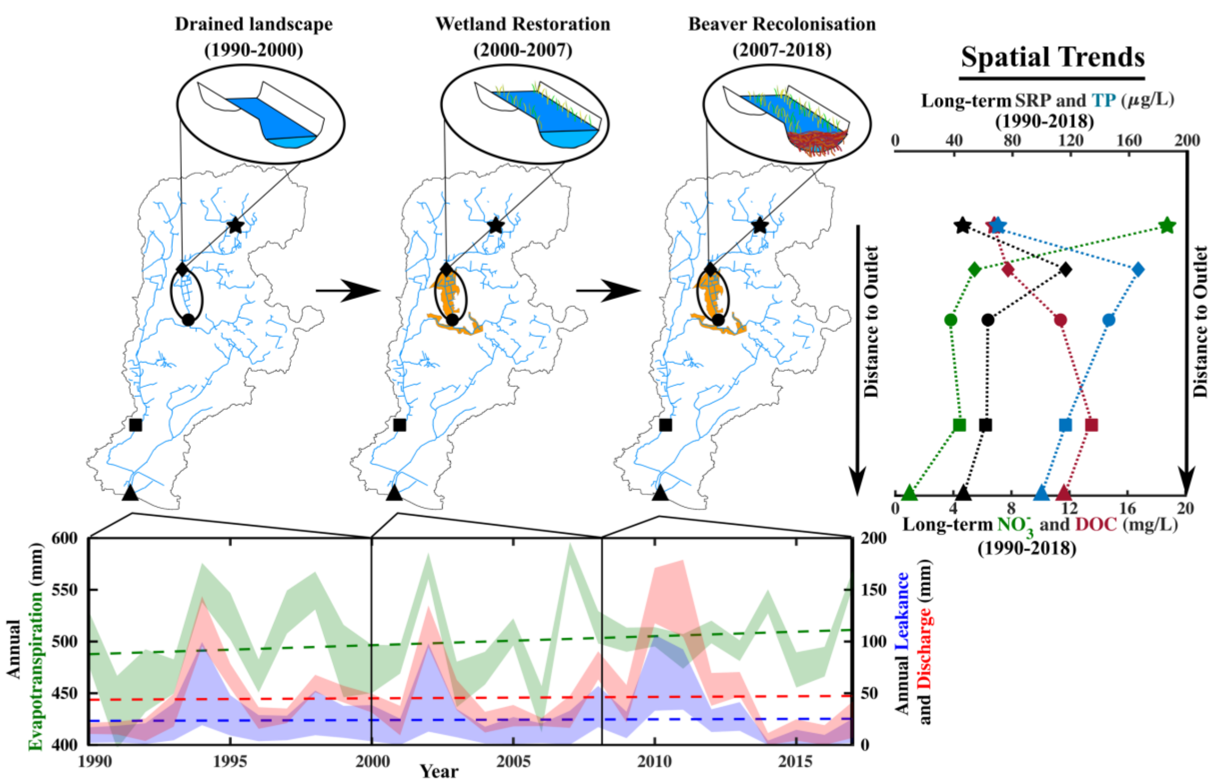
Graphical Abstract



Abstract

Quantifying the catchment water balance and the characterization of its water quality changes are effective tools for establishing the response of catchments to shifting land management practices. Here we assess long-term hydrological partitioning and stream water chemistry over a 30-year period in a rural mixed land use catchment in northern Germany undergoing riparian wetlands and widespread re-colonisation by beavers (*Castor fiber*) along the river network. We used long-term spatially distributed stream discharge, groundwater levels and surface water quality data with a simple monthly water balance model, changes in the variability in discharge measurements, and statistical analysis of spatio-temporal changes in stream water quality to assess long-term changes. Water balance estimates indicated high proportions of evapotranspiration loss (~90% of total precipitation) and relatively low groundwater recharge (<5% of total precipitation) prior to riparian rehabilitation in 2000. Increasing groundwater levels from 2000-2017 and the relatively linear nature of the catchment storage – discharge relationship, indicate a gradual increase in groundwater recharge (buts still <10% of total precipitation). Wetland rehabilitation, greatly enhanced by increasing beaver populations, resulted in longer water transit times in the stream network, less linear storage-discharge relationship and a loss of daily stream variability, increased DOC concentrations, isotopic evaporative enrichment downstream, and moderated stream temperatures. There was limited long-term water quality improvements from wetland rehabilitation on either nitrate or total phosphorus concentrations, with unchanged seasonal summer and winter peak concentrations for phosphorus and nitrate, respectively. This likely reflects the long-term legacy of fertilizer use on nutrient reservoirs in the catchment's soils, aquifers, and stream network. These long-term changes in hydrology and stream chemistry resulting from riparian rehabilitation and changes in agricultural management practices provide invaluable insights into catchment functioning and an evidence base for future planning in relation to long-term climatic changes.

1. Introduction

Lowland continental regions in the temperate zone are key areas for agricultural production, often sustaining large populations. Such areas are subject to increasing legislation and water management practices to maintain agricultural productivity whilst sustaining the ecological status of river systems (e.g. Gutzler et al., 2015; Holman et al., 2017). Many lowland continental agricultural regions are dependent on groundwater for irrigation; however, over-exploitation of aquifers has led to declining groundwater storage and deterioration in water quality from fertilizers and other chemicals (Gleeson et al., 2012). These issues are currently of critical importance in European agriculturally-dominated catchments, where traditional patterns of crop production and relatively high evapotranspiration (ET) rates have resulted in a legacy of high nutrient levels (Dezsi et al., 2018). Intensifying concerns of water stress, long-term temperature trends indicate a higher regional rate of increase in temperature relative to the global average (Lahmer and Pfützner, 2003), with a corresponding decline in annual precipitation and a shift in seasonality accentuating the difference between high ET in summer months to low-energy, high-runoff conditions in winter (Bronstert, 2003). The anticipated changes in hydroclimatic conditions has led to studies highlighting the sensitivity and adaptations needed in land management to build landscape resilience and develop sustainable agricultural strategies (Holsten et al., 2009).

The quantification of "blue" water storage and fluxes (groundwater and stream water), and "green" water storage and fluxes (interception and evapotranspiration) is essential for the interpretation of a catchments hydrological processes and how they will change due to anthropogenic effects (e.g. changes in irrigation strategies) or climatic effects (e.g. droughts). In many European agricultural regions, the dominance of "green" water fluxes (Rost et al., 2005) increase the sensitivity and drought susceptibility to land management or vegetation changes (Wattenbach et al., 2007; Teuling et al., 2013). These agricultural regions generally have a long legacy of extensive land drainage, with widespread channelization, and drainage of riparian wetlands (groundwater-fed fens) and kettle hole lakes which reduces "blue" water storages compared to natural conditions (Germer et al., 2011). Such agricultural catchments can have a “memory effect”, with decreases in “blue” water storages showing lasting effects of drought periods on discharge responses for months or years (Orth & Senevirantne 2013; Thomas et al., 2015) leading to uncertainty over long-term water security and availability of water for crop growth and other societal needs.

Long-term changes in water availability also have implications for stream water quality and nutrient transport, which are closely coupled to the hydrological functioning of catchments, and have obvious implications for aquatic ecology. Compounded with long-term water balance changes, long-term increases in fertilizer have often resulted in a significant nitrogen and phosphorus legacies (eg. Van Meter et al., 2016). This has tended to accelerate nutrient leaching to groundwater and streams (Swank 1988; Foley et al., 2005), which in some places may be further exacerbated by atmospheric deposition of nutrients from fossil fuel burning (Tipping et al., 2014). The importance of wetlands in retaining nutrients and regulating water quality has become increasingly demonstrated over the past few decades (Fisher and Acreman, 2004; Reinhardt et al., 2005), with changes in agricultural policies incentivised to reduce downstream nutrient and sediment loadings in runoff from farmlands (Jordan et al., 2003; Daneshvar et al., 2017; Vymazal et al., 2018).

Along with such policy changes, the re-establishment of beaver populations (*Castor fiber*) across many European river systems in recent decades, has resulted in the re-establishment of natural wetlands (Halley and Rosell, 2002). Beaver dams have high potential for effective nutrient and sediment removal by increasing water storage and flow attenuation (John & Klein 2004; Puttock et al., 2017, 2018), with a simultaneous effect on the interactions of groundwater and surface runoff (Woo and Waddington, 1990; Westbrook et al., 2006; Martinez-Martinez et al., 2014).

This study focuses on a long-term monitoring site at an agricultural catchment in eastern Germany, which is representative of many other agricultural catchments in northern Europe. The catchment is dominated by agricultural areas that have been drained for cultivation over the past few centuries with numerous long-term datasets collected for studies on stream nutrient transport (including nitrogen and phosphorus) into the larger Spree River system (Gelbrecht et al., 1996; 2005). Since 1990 the catchment has undergone significant change to some riparian areas due to wetland rehabilitation (circa 2000) and beaver dams during beaver recolonization (circa 2007). The main objective of this study was the evaluation of the hydrology and nutrient transport in the catchment in relation to wetland rehabilitation and beaver recolonization. The specific objectives were: 1) quantify the dominant hydrological processes of evapotranspiration, groundwater storage, and discharge, and discharge responsiveness, 2) investigate the long-term temporal and spatial trends of water quality since the beginning of stream water chemistry in 1990, and 3) identify future management options and research needs in the context of long-term changes of catchment hydrology and stream water quality. Hydrologic and stream chemistry changes in the catchment (pre-2000, 2000 – 2007, and post 2007) were assessed through modelling and statistical testing for measured datasets. The influence of these changes on catchment function provides a basis for long-term water security and management in the area and availability of water for crop growth and other societal needs.

2. Site Background

2.1 Site Characteristics

The Demnitzer Millcreek (DMC) catchment is a 66 km^2 long-term study site, 55 km east of Berlin, in Brandenburg, Germany (52^{o}23'N, 14^{o}15'E) (Figure 1). Agriculture is the dominant land use, with more than 60 % of the catchment used for arable farming or pasture for livestock grazing (Table 1), especially in the northern part of the catchment. Forestry (broadleaf, conifer, and mixed forest) is the other major land use, accounting for more than 36 % of the area, predominantly in the southern parts of the catchment (Figure 1d, Table 1). Several small dispersed urban settlements cover approximately 2 % of the catchment, and sustain a low population of approximately 5,000 residents, with limited impervious areas (SBB, 2019). A single wastewater treatment facility functions for the urban settlements in the north-western region of the catchment; however, the low population yields very low treatment plant effluent or nutrient inputs to stream water.

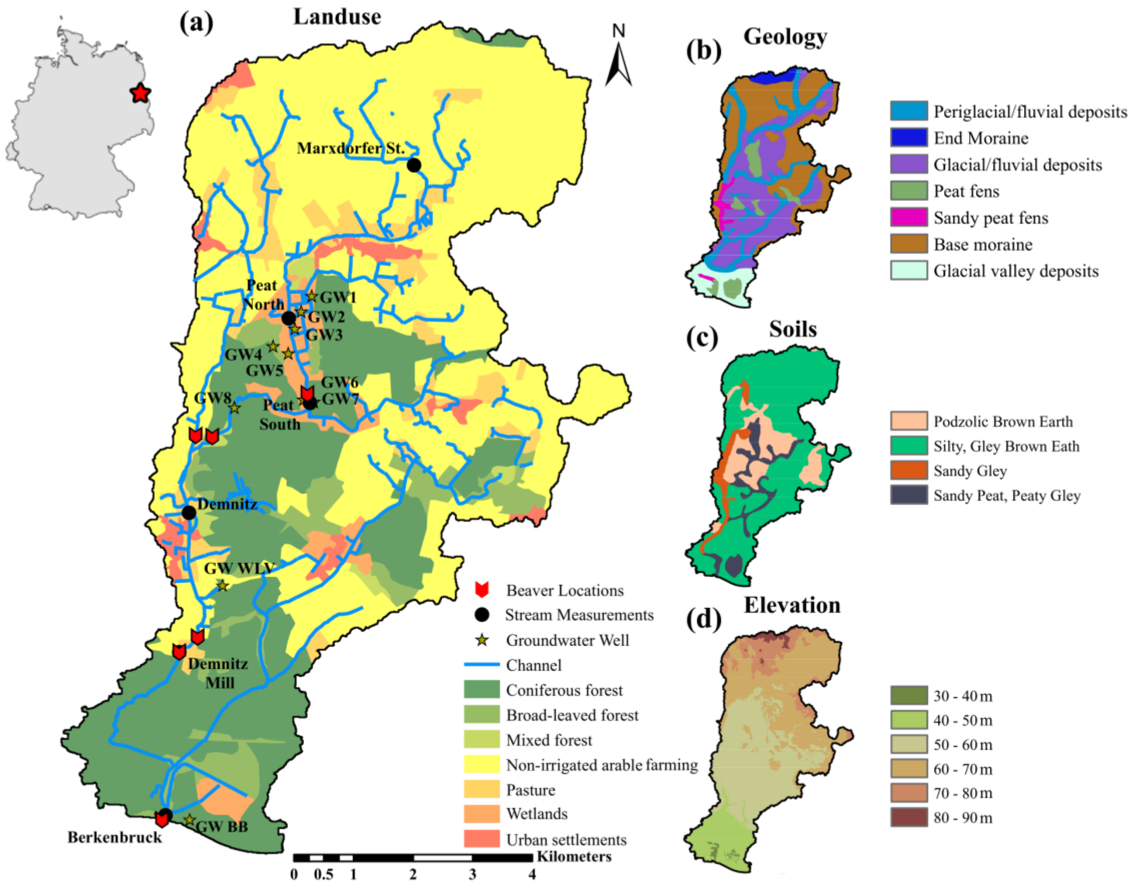


Figure 1: Landuse map of the Demnitzer Millcreek catchment (DMC) and locations of bi-weekly stream chemistry and weekly isotope (since 2018) measurement locations and groundwater wells. Subset figures show the location of the DMC catchment with respect to Germany (in grey), (b) the catchment geology, (c) surficial soils, and (d) elevation (LfU, 2019).

The relatively flat catchment (average slope of 2 %; Table 1) and surrounding landscape bear the strong imprint of the last glaciation, with significant cover of glacial deposits and moraine material throughout the catchment (Figure 1). Glacial depressions and regions of low topographic relief formed numerous kettle hole lakes and wetlands (Figure 1a). Within the DMC catchment, the river network generally follows fluvial/periglacial deposits, incised into surrounding basal tills, with intermittent riparian peat fens in the mid- and southern parts of the catchment. Glacial valley deposits located near the outlet of the catchment result in convergence with a regional (east to west) groundwater flow system in the main valley of the River Spree.

Catchment soils have generally formed as unmixed sediments (Gelbrecht et al., 2005). The dominant agricultural soils are categorized as silty brown earths, and cover the headwaters of the catchment upstream of the channel network (Figure 1c). The stream network is generally fringed with sandy soils, both sandy gleys and peat soils in the riparian areas. Brown podzolic soils are present within the mid-lowland regions of the catchment, downstream of the largest settlement in the catchment (Arensdorf). The podzolic region also coincides with a land use shift towards forests and pasture lands (Figure 1 c and d).

Table 1: Demnitzer Millcreek (DMC) catchment characteristics, topographic relief, slope and land use (land use information for 2018) (LfU, 2019)

		Stream location ID				
		Marxdorfer St.	**Peat North**	**Peat South**	**Demnitz Mill**	**Berkenbrück**
Area (km^2)		2.99	18.47	21.27	43.60	66.39
Topographic Relief (m)		23.39	32.23	33.52	41.48	50.23
Mean Slope (%)		2.48	2.51	2.26	1.81	1.98
Land use (%)	Non-irrigated arable farming	100	82.4	71.7	59.8	50.4
	Urban settlements	0	2.1	1.9	2.9	2.5
	Pasture land	0	11.2	9.7	8.9	6.9
	Broadleaf forest	0	0.1	1.0	3.2	6.0
	Conifer forests	0	3.3	9.5	20.2	29.2
	Mixed forest	0	0.9	0.8	1.0	1.0
	Wetlands	0	0	8.1	4.0	4.0

The area experiences a humid continental climate (Köppen index) with a modest annual precipitation (569 mm·year^{-1}, Table 2), and a moderate energy input, reflected in the annual potential evapotranspiration (PET) of 650 – 700 mm·year^{-1} (UFZ, 2019). Daily precipitation has been measured for over a century by Deutscher Wetterdienst (DWD) at locations ranging from 1 to 20 km surrounding the catchment (Lindenberg, and Müncheberg). Summer rainfall is dominated by convective events, with sporadic rainfall intensities of more than 40 mm·day^{-1} at precipitation locations surrounding the catchment (Figure 2a). Snowfall is a relatively small proportion of the total precipitation, generally less than 10%. Annual precipitation has slightly increased from 560 mm·year^{-1} between 1990 – 2007, to 580 mm·year^{-1} between 2008 – 2018, (DWD, 2019). Over the past 30 years, a slight increase in the mean annual temperature (MAT) occurred after 1999 (9.1^{o}C between 1990 – 1999), where the MAT was 9.6 and 9.5^{o}C between 2000 – 2007 and 2008 – 2018, respectively. Annual relative humidity remains near 80 %, with higher humidity during the wetter winter months.

Table 2: Monthly mean, minimum, and maximum precipitation, snowfall to precipitation ratio, temperature, and relative humidity between1990-2018 (DWD, 2019)

		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Precipitation (mm)	Mean	43	34	39	31	54	59	78	61	47	38	42	43	569
	Maximum	95	82	94	92	128	128	235	195	134	98	107	110	778
	Minimum	2	6	13	1	12	8	16	18	4	7	1	11	366
Snowfall/ Precipitation	Mean	0.32	0.42	0.23	0.05	0.01	0.00	0.00	0.00	0.00	0.01	0.11	0.32	0.9
Temperature (^{o}C)	Mean	0.2	1.5	4.3	9.2	14.1	17.0	19.1	18.5	14.1	9.2	4.5	1.3	9.4
	Maximum	5.2	6.1	7.5	12.3	16.2	19.6	23.4	21.7	17.6	12.7	7.3	6.6	10.4
	Minimum	-5.8	-3.5	-1.6	6.5	11.1	15.1	16.2	16.1	11.0	5.3	-0.8	-5.4	7.1
Relative Humidity (%)		87	83	78	70	69	70	70	71	78	83	89	89	78

2.2 Catchment Discharge Regulation and Wetland Rehabilitation

To improve drainage and recapture water from agricultural fields, the channel network was historically incised and straightened, wetlands were artificial drained, and tile drainage was installed in agricultural fields. These drainage practices were discontinued in 1990, resulting in a gradual reduction in the effectiveness of tile drainage and re-naturalisation of channel shape (B. Bösel, personal communication, Nov 6, 2018). The discharge regime usually has distinct seasonality, with winter maxima and summer minima. The highest flows are associated with storm events, driven by transient, rapid surface runoff from sealed and compacted agricultural soils. Discharge is often lower at Berkenbrück than Demnitz Mill due to leakage into groundwater in the more freely draining glacial valley deposits south of Demnitz Mill (Figure 1b). Closer to Berkenbrück, increased groundwater contribution to the stream results in an effluent stream. In extreme summers the river network can become fragmented during significant lengths of dry channel.

Rehabilitation of riparian wetlands in the central part of the catchment in 2000 involved excavation of backwaters connected to the main channel, and installation of in-channel bunds to raise the water level. The incised channels of the wetlands were shallowed, a small weir was installed at the wetland outlet to reduce discharge and retain water. The weir did not result in any inundation of the surrounding area. Vegetation growth since the rehabilitation primarily occurred on the in-channel bunds, aiding in the restriction of flow. In the rehabilitated wetland, vegetation is managed with bi-annual cutting and harvesting. Groundwater levels surrounding the wetland rehabilitation were monitored with the installation of four groundwater wells.

Following rehabilitation, the continued expansion of beavers (*Castor fiber*) throughout Europe (Halley et al., 2012; Mai et al., 2018), was observed in the catchment in 2007 – 2008. The beavers progressed from the catchment outlet to the rehabilitation area in approximately a one-year period (Figure 1a). The beaver dams are generally constructed in the deeper incised channels, with a channel width ranging from 1 – 3 m. *Castor fiber* is a protected species; therefore, hunting is prohibited. Each of the primary beaver sites (six locations, Figure 1a) contains a cluster of dams ranging from 1 to 3 dams, with multiple in-stream ponded sections at each site. Weak construction and high winter flows have led to continuous breach and failure of the dams, followed by continuous re-building at, or nearby the original dam location (Figure 1a). The wetland rehabilitation (in-channel vegetation, bunds, and weir) and creation of beaver dams worked to naturally attenuate flow and rapidly regenerate the wetland regions of the catchment.

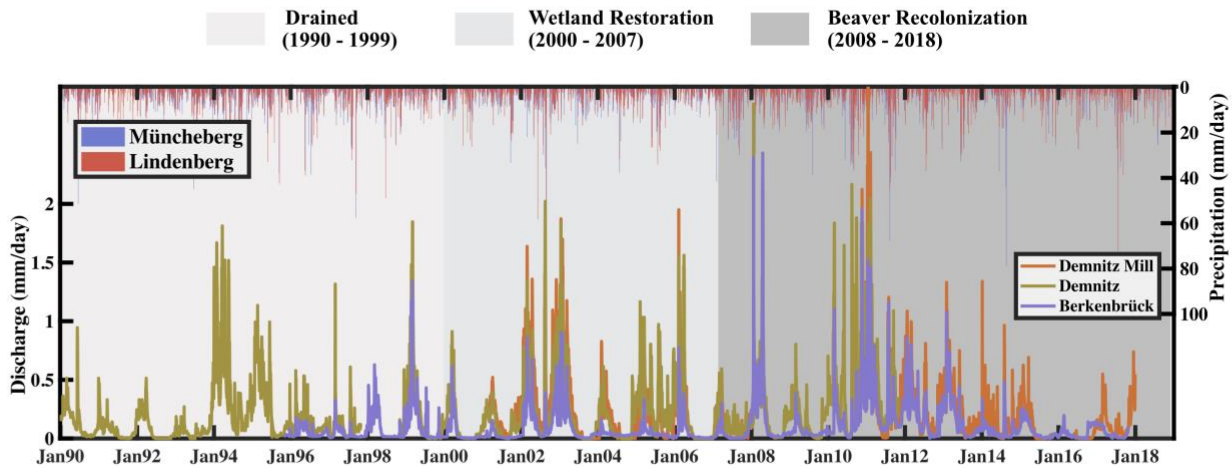


Figure 2: (a) Long-term daily precipitation amounts from stations Müncheberg and Lindenberg, and daily discharge per unit area from stations Demnitz Mill, Demnitz, and Berkenbrück of the Demniter Millcreek catchment.

3. Methods

3.1 Hydrological evaluation

3.1.1 Hydrological measurements

Daily water level data at Berkenbrück (est. 1982) and Demnitz (est. 1986) were collected by the Landesamt für Umwelt (LfU, 2019) using pressure transducers; however, water level measurements by LfU at Demnitz ended in 2011 (Figure 1a, Table 3). In 2001, a pressure transducer was installed at Demnitz Mill by the Leibniz-Institut für Gewässerökologie und Binnenfischerei (IGB) because the site provided a more defined channel geometry than at Demnitz. Measurements at Demnitz Mill were recorded with hourly water levels until 2007, and three-hourly after 2011 (Table 3). Water level at each site was translated to discharge with a rating-curve developed using multiple point measurements with a current meter. Daily groundwater level measurements in the catchment were recorded at nine wells beginning in 2001 (Table 3, Figure 1a).

Table 3:Temporal measurement periods in the Demnitzer Millcreek (DMC) catchment of hydrological measurements and modelling of stream water level and groundwater level

Hydrological Measurement				
	Location	Years	Measurement Resolution	Equipment
Stream Water Level	Berkenbrück	1982 – present	Daily	Aquitronic AquiLite ATP10
	Demnitz	1986 – 2011	Daily	
	Demnitz Mill	2001 – 2007	Hourly	
		2011 – present	3-hourly	
Groundwater	GW4, GW5, GW7, GW8, GW WLV	2001 - present	Daily	Aquitronic AquiLite ATP10
	GW1, GW2, GW3, GW6	2001 - 2004	Daily	
Hydrological Modelling				
Method	Location	Years	Resolution	
Storage – Discharge	Demnitz Mill	2001 – 2007	Hourly discharge	
		2011 – 2018	Hourly discharge estimated with spline interpolation of 3 hourly measurements	
Diel Stream Variability		2001 – 2007	Hourly discharge	
		2011 – 2018	Hourly discharge estimated with spline interpolation of 3 hourly measurements	
Thornthwaite Model	Demnitz Mill	1990 - 2018	Monthly discharge and groundwater levels	
	Berkenbrück	1990 - 2018		

3.1.2 Assessment of catchment responsiveness

The responsiveness of the DMC was assessed with storage – discharge relationships and discharge variability using sub-daily discharge to examine the dynamic storage used to derive surface water flow. Sub-daily discharge data are required for the storage – discharge relationships to examine periods when ET has a limited effect on catchment storage, and to capture the diel stream variability. Therefore, catchment responsiveness was assessed in the wetland rehabilitation period (hourly from 2001 – 2007) and the beaver recolonization period (three-hourly from 2011 – 2017) at Demnitz Mill (sub-daily measurements, Table 3).

To conduct the storage – discharge relationship with hourly time-steps in the beaver recolonization period (2011 – 2017), spline interpolation was used to derived hourly time-series. A direct comparison of the use of hourly data to three-hourly data for the storage – discharge relationship was additionally examined to infer the suitability of the spline interpolation (Appendix A). The storage – discharge relationship was assessed by correlating the non-ET influenced with a linear or non-linear regression of the log-log relationships of discharge (Q) and recession (-dQ/dt) (Kirchner, 2009). Regression of the log-log relationships yielded power functions for linear functions ($\ln(-dQ/dt) = b \cdot \ln(Q) + c$),

$$\frac{-dQ}{dt} = C \cdot Q^{b} \tag{1}$$

and non-linear power functions ($\ln(-dQ/dt) = a \cdot \ln(Q)_2 + b \cdot \ln(Q) + c$),

$$\frac{-dQ}{dt} = C \cdot Q^{a \cdot \ln(Q) + b} \tag{2}$$

where b and a are the first and second order coefficient from the log-log regression, and C is the exponential of the log-log regression intercept ($C = \exp(c)$). The solution of the power functions (Equations 1 & 2) to estimate the storage and discharge was conducted with the method shown in Kirchner (2009).

The responsiveness of the catchment with daily discharge variability was evaluated with the peaks and troughs of daily normalized hourly discharge measurements during the wetland rehabilitation (2001 – 2007) and beaver re-colonization periods (2011 – 2018, Table 3). Hourly data during the beaver re-colonization period was derived from a spline interpolation of 3 hourly data. The influence of spline interpolation on diel variability was examined in Appendix B. The normalisation of hourly discharge measurements was conducted daily,

$$N(Q,t) = (Q(t) - \mu(Q(t)))/\sigma(Q(t)) \tag{3}$$

where $N(Q,t)$ is normalized discharge, $Q(t)$ is hourly discharge, $\mu(Q)$ is the mean discharge on day t, and $\sigma(Q)$ is the standard deviation on day t. Values of normalized discharge greater and less than 0 indicate higher and lower than average discharge, respectively.

3.1.3 Water balance estimation using a Thornthwaite-based model

To quantify water partitioning within the catchment (precipitation, soil storage dynamics, discharge, and ET), a simple monthly hydrologic model was developed based on the Thornthwaite approach (Thornthwaite, 1948). The model uses only monthly precipitation inputs (P) and temperature (T_a) as model forcing data. Winter snowfall (35% of precipitation from December – February, Table 2) was estimated by partitioning incoming precipitation with a simple temperature threshold,

$$F_s = \begin{cases} 1 & T_a < T_{min} \\ \dfrac{T_a - T_{min}}{T_{max} - T_{min}} & T_{min} < T_a < T_{max} \\ 0 & T_a > T_{max} \end{cases} \tag{4}$$

where F_s is the snowfall fraction of precipitation, T_{min} is the minimum temperature threshold, and T_{max} is the maximum temperature threshold. The snow fraction of precipitation was added to the snowpack each month. Snowpack depth ($D_s(t)$) was estimated using F_s and the previous months snowpack depth ($D_s(t-1)$) ($D_s(t) = \left(P \cdot (1 - F_s)^2 + D_s(t-1) \cdot (1 - F_s)\right)$). Snowmelt ($M$) was estimated from snowpack depth and snowfall fraction ($M = D_s(t) \cdot F_s$). The PET was estimated using a simple, lumped conceptualisation using the Hamon method (Hamon 1963),

$$\mathrm{PET}(t) = 924 \cdot DL \cdot 0.611 \cdot \frac{\exp(17.3 \cdot T_a(t)/(T_a(t) + 237.3))}{(T_a(t) - 273.2)} \tag{5}$$

where DL is the average monthly daylight length (from catchment latitude), and T_a is the monthly average air temperature. Monthly soil storage was estimated by using the hydrologic input (rain and snowmelt, $W = P\cdot(1\text{-}F_s) + M$) and the PET,

$$S(t) = \begin{cases} \min(dW + S(t-1), S_{max}) & \mathrm{W} > \mathrm{PET} \\ S(t-1)\cdot\exp\left(-\frac{\left(\mathrm{PET}(t) - W(t)\right)}{S_{max}}\right) & \mathrm{W} \leq \mathrm{PET} \end{cases} \tag{6}$$

where dW is the net potential hydrologic input ($dW = W$ - PET), S_{max} is the maximum soil storage, $S(t)$ is the current soil storage, and $S(t\text{-}1)$ is the soil storage in the previous month. Overland/rapid flow ($O(t)$) occurs only when W is larger than PET and the net water input (dW) with antecedent storage ($S(t\text{-}1)$) exceeds S_{max} ($O(t) = (dW + \mathrm{S}(t\text{-}1)) - \mathrm{S}_{\mathrm{max}}$). The actual evapotranspiration (AET) was estimated using soil storage,

$$\mathrm{AET}(t) = \begin{cases} \mathrm{PET} & dW > \mathrm{PET} \\ W(t) + S(t) - S(t-1) & dW < \mathrm{PET} \end{cases} \tag{7}$$

Following the estimation of AET(t), groundwater leakance from the soil storage (L, subsurface flow out of the catchment) and the soil water contribution to the stream discharge (GQ) were estimated using the soil storage. Both fluxes were estimated using a simple linear relationship to soil storage ($L(t) = S(t)\cdot GW_p$ and $GQ(t) = S(t)\cdot GW_Q$ for groundwater leakance and groundwater contribution to discharge, respectively) with a single parameter indicating the flux as a fraction of soil storage (value 0 – 1). The monthly surface discharge was calculated as the sum of groundwater contribution ($GQ(t)$) and overland/rapid flow ($O(t)$).

Multi-criteria calibration of discharge and soil storage was used for the sites with long-term sub-daily or daily discharge only (Demnitz Mill and Berkenbrück, 1990 – 2017) which helps to minimize volumetric errors of total monthly water volumes (mm·month^{-1}). The modelled soil storage was calibrated using the changes in groundwater levels (2001 – 2017) using a specific yield of unconsolidated glacial silty sands (0.16, Dingman, 2002) to estimate volumetric changes.

The Kling-Gupta efficiency (KGE, Kling et al., 2012) metric was used as an objective function for discharge and soil storage to capture the temporal variability. Calibration was conducted using 200,000 Monte Carlo simulations retaining the best 1% (2000 simulations) for analysis. The model was calibrated independently at Berkenbrück and Demnitz Mill, as the region between represents a significant geological divide (Figure 1b). The regional groundwater recharge in the glacial valley gravels may influence the primary hydrological flow paths. The model was also calibrated for each time-frame, the original drained landscape (1990 – 1999), the wetland rehabilitation (2000 – 2007), and the beaver re-colonisation (2008 – 2017).

3.2 Stream isotope and water quality measurements and analysis

3.2.1 Stream isotope sample measurements

The stable isotopes of deuterium (δ_2H) and oxygen-18 (δ_{18}O) were measured in precipitation taken as bulk daily samples from a precipitation gauge near the Demnitz Mill site. Paraffin was added to the gauge to prevent evaporation. Stream water samples were taken nested along the main stream and incoming tributaries at five locations (Marxdorfer St., Peat North, Peat South, Demnitz Mill, and Berkenbrück; Figure 1) every two weeks beginning in January 2018 (when water was flowing). Samples were taken as grab samples from the centre of the stream 10 cm below the stream surface and kept cool until analysed. Groundwater samples were taken at five locations (GW4, GW6, GW8, GW WLV, and GW BB; Figure 1a) throughout the catchment beginning in late summer of 2018. Stable isotope samples were analysed with an off-axis Integrated Cavity Output Spectroscopy (OA-ICOS) (Triple Water-Vapor Isotope Analyzer TWIA-45-EP, Model#: 912-0032-000 Los Gatos Research, Inc., USA) in liquid analysed mode. Samples were run six times, using the average of the last three samples. The analytical uncertainty of the analyser is ±

0.6 and ± 0.3 ‰ for δ^2H and $\delta^{18}O$, respectively. If the measured standard deviation of the of the injected sample water vapour exceeds 400 ppm of the total injected water sample through the designated injection plateau, the analysis is re-conducted.

3.2.2 Stream chemistry sample measurements

Stream samples for chemical analysis were usually collected as grab samples every two weeks from a series of five monitoring locations (Marxdorfer St., Peat North, Peat South, Demnitz Mill, and Berkenbrück; Figure 1a) between 1991 – 2018. Intermittent data gaps of water quality parameters occurred due to changes in research objectives and funding availability, with gaps predominantly between 2003 – 2013 (time-series in Appendix C). Rainfall oriented sampling occurred predominantly between 2000 – 2003, during the wetland rehabilitation period, with an increased sampling frequency of up to 6 samples per month. Measured stream chemistry includes: electrical conductivity (EC), pH, water temperature (T_w), dissolved oxygen (O_2), soluble reactive phosphorus (SRP), total phosphorus (TP), nitrate (NO_3^-), ammonium (NH_4^+), sulphate (SO_4^{2-}), chloride (Cl^-), dissolved organic carbon (DOC), calcium (Ca^{2+}), and magnesium (Mg^{2+}) (Figure 1a). Samples were collected in 1000 ml bottles and were analysed upon returning to the lab and filtered through 0.45 µm pore size filters (Gelbrecht et al., 2005). EC, pH, T_w and O_2 were measured by hand-held electrodes directly in the field. Standard analytical methods with membrane-filtered samples, as outlined by the German Institute for standardization (DIN, 2019), were used for: Cl^- and SO_4^{2-} by ion chromatography (precision ~ 2%), Ca^{2+} and Mg^{2+} by inductively-coupled plasma optical emission spectroscopy (precision ~2%), SRP by molybenum blue spectroscopy (precision ~3%), NO_3^- and NH_4^+ by flow-segmented analysis (both precision ~3%), DOC by infrared spectrometry after high temperature oxidation (precision~3%). TP was determined in homogenized original samples by molybdenum blue spectroscopy after wet digestion with H_2O_2/H_2SO_4 (precision ~5%).

3.2.3 Spatial and temporal analysis of stream water quality

Stream chemistry data were analysed to identify long-term spatial patterns and intra- and inter-annual temporal trends using data from the five sites (Marxdorfer St., Peat North, Peat South, Demnitz Mill, and Berkenbrück; Figure 1a). Long-term spatial differences between sites were assessed using a two-sided Students t-test and a Wilcox rank sum test. The Students t-test assesses the significant difference of the mean values (assuming a normal distribution), and the Wilcox rank sum evaluates the significant difference of the median values (does not assume a distribution). Significant spatial differences were assessed only against the next downstream site (e.g. between Marxdorfer St. and Peat North). To ensure that total sample numbers or sampling timing did not bias the statistical tests, statistical tests were performed only with similar sample dates between sites. Long-term temporal trends of stream chemistry were assessed using the average stream chemistry of each year when more than 12 samples were available (minimum one per month). The time-series was divided into the three distinct periods, pre-wetland rehabilitation, post-wetland rehabilitation, and post-beaver recolonization. Long-term changes in stream chemistry were evaluated with the significant difference of mean stream chemistry between each period (two-sided Students t-test).

To further assess the seasonal dynamics of management practices and biological influences on the steam water chemistry, intra-annual variability of the stream water samples of each site were assessed. Stream water quality parameters of particular interest and without a notable long-term term (pH, O_2, SRP, TP, NO_3^-, and NH_4^+) were binned for winter (Dec-Feb), spring (March – May), summer (June – August), and autumn (Sept – Nov) and with a mean value for each location

(25^{th} and 75^{th} quartiles with the time-series shown in Appendix C). Similar to the long-term analysis, significant spatial differences for each next downstream site for each season were assessed using a two-sided Students t-test. Temporal significance differences at each site was examined between each season with a two-sided Students t-test. The significance of the temporal and spatial differences was only conducted when a minimum of 15 samples of each dataset were available.

4. Results

4.1 Spatio-temporal variability in discharge responsiveness

Using the sub-daily discharge measurements indicated distinctly linear catchment dynamics throughout the wetland rehabilitation and beaver recolonization periods (2001 – 2007 and 2011 – 2018, respectively; Figure 3a and b). Through all years the discharge was very responsive to changes in storage, with most of the flow variability occurring across a very narrow range of storage change of less than 10 mm (discharge rate against storage, Figure 3a). However, there is a significant difference ($p \ll 0.05$, 95% coefficient range on Figure 3c and d) between the storage – discharge relationship from the end of the hourly stream level measurements (2007) and the establishment of three-hourly stream measurements (2011). More notably, the linear storage – discharge relationship during wetland rehabilitation (2001 – 2007, shown with linear regression on Figure 3c) was inadequate for estimating the storage-discharge relationship during the beaver recolonization (2008 – 2018); this could only be approximated with a non-linear relationship (Figure 3d). For each period and respective model, the slope of the storage discharge relationship was consistent between individual years. Beaver dams significantly changed the dynamics of catchment storage, with lower discharges for equivalent storage relative to the wetland rehabilitation (e.g. at 17.5 mm storage is 0.1 $mm \cdot hr^{-1}$ discharge during the wetland rehabilitation and 0.05 mm·hr^{-1} discharge during beaver recolonization, Figure 3a). The change in the dynamic storage – discharge relationship resulted in an increase in the mean hydrograph recession time of 30 hours during beaver recolonization (dashed lines, Figure 3b). The increase in hydrograph recession time may infer an increase in both transit and residence times of the stream network.

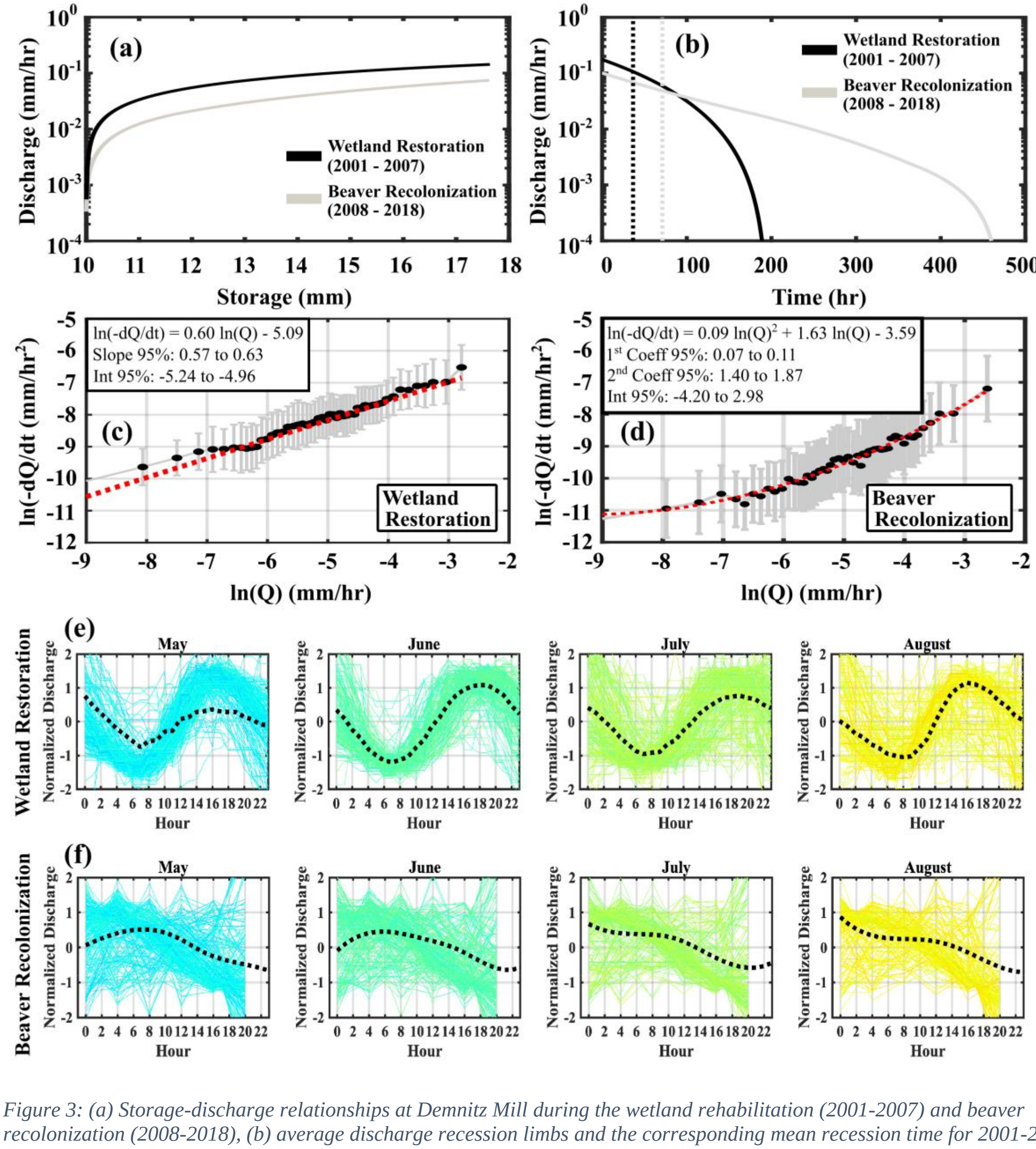


Figure 3: (a) Storage-discharge relationships at Demnitz Mill during the wetland rehabilitation (2001-2007) and beaver recolonization (2008-2018), (b) average discharge recession limbs and the corresponding mean recession time for 2001-2007 and 2008-2018. Linear regression on log-log space of discharge and discharge recession in (c) 2001-2007, and (d) 2008-2018. Hourly normalized discharge from May to August from (e) 2001-2007, and (f) 2008-2018.

Complimenting the storage – discharge relationships, the normalised daily discharge showed the catchment responsiveness. Significant differences in the normalized discharge (p-value $\ll$ 0.05) were visible in the variability during the wetland rehabilitation (2001 – 2007) (Figure 3e) compared to the beaver recolonization (2008 – 2018) (Figure 3f) throughout the growing season. Only a few hourly periods were not significantly different (14:00 in April, May and August, and 00:00 in July and August). Diel variability during the wetland rehabilitation (2001 – 2007) showed distinctive sinuosity in discharge variability with peak summer discharge in the late afternoon or early evening (16 – 19hr). The sinuosity of the normalized discharge was greatly dampened during higher flows and months with less ET (November – March, not shown). Following beaver recolonization (2008 – 2018), diel discharge variability was greatly reduced (Figure 3f). Unlike the diel sinuosity in the wetland rehabilitation, the diel variability during the beaver recolonization was broadly consistent throughout the year with the highest discharge near midnight but also experienced much more uncertainty (Figure 3f).

4.2 Catchment Hydrological Processes: Water balance and estimation of storage and fluxes

The Thornthwaite-based water balance model was effective at estimating the monthly variation of discharge, catchment storage, and ET at both Demnitz Mill and Berkenbrück (Figure 4). The mean KGE for discharge and soil storage simulations through all three periods was 0.75 and 0.70, respectively. A slight under-estimation of discharge occurred during months of highest rainfall intensity (e.g summer 2002 and 2012, Figure 4b) with over-estimation of discharge potentially due to under-estimation of ET or leakance in wetter years (2010 and 2011). Rapid hydrological flow paths were not found to be significant for monthly estimations (no simulated monthly occurrences), though they clearly have a role in short-term storm event responses (Figure 2a). The estimation of ET shows that the annual loss of precipitation to ET ranged from 65 % in high precipitation years to 119 % in low precipitation years. The average simulated annual loss to ET was generally between 82 – 94 % of the total precipitation (25_{th} - 75_{th} simulated interquartile range). Slight gradual increases in AET occurred throughout the simulation, from 491 to 505 mm·year$_{-1}$ for Demnitz Mill from the rehabilitation (2000 – 2007) to the beaver recolonization (2008 – 2018), respectively. AET at Berkenbrück also slightly increased (499 to 506 mm·year$_{-1}$) from the rehabilitation to the beaver recolonization, respectively. Slightly higher AET was estimated at Berkenbrück than Demnitz Mill, corresponding to the large forested regions in Berkenbrück but south of the Demnitz Mill sub-catchment (Table 1), and the higher estimated storage at Berkenbruck than Demnitz Mill (Figure 4g – i).

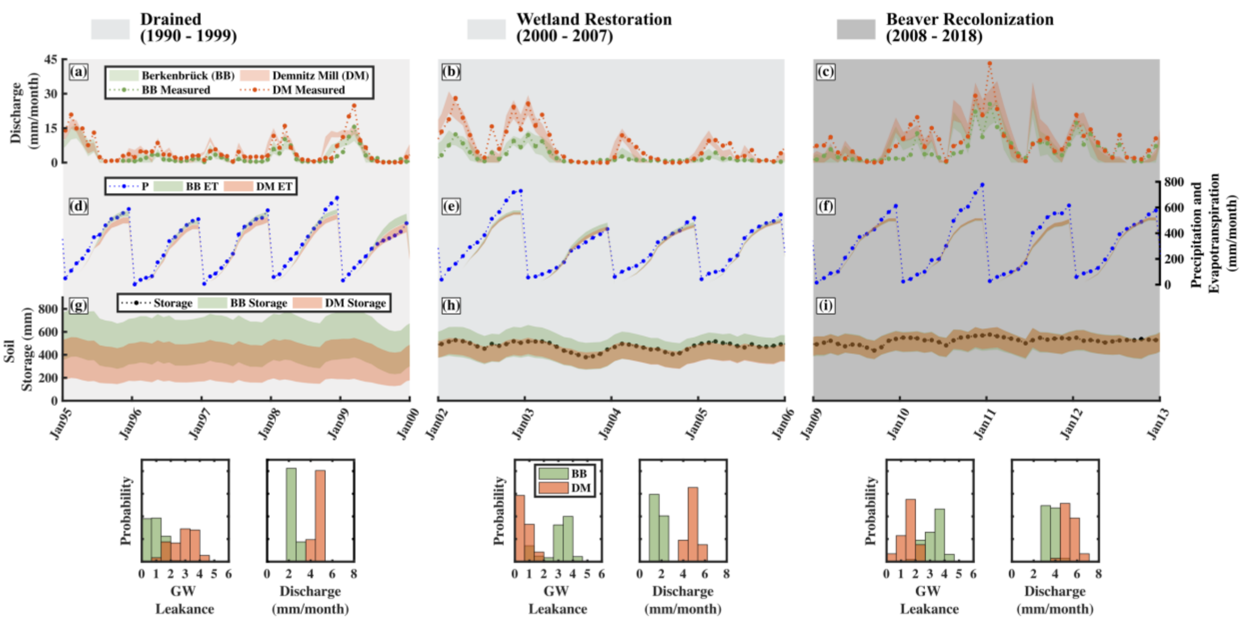


Figure 4: Thornthwaite estimations for discharge (a-c), ET (d-f), and soil storage (g-i) at Demnitz Mill and Berkenbrück. Also shown are the changes in the proportion of water loss groundwater leakance rate and discharge rate for each site during the drained, rehabilitation, and beaver recolonization time-periods.

Simulation results also showed notable differences between Demnitz Mill and Berkenbrück for discharge and groundwater leakance despite similarities of the soil storage (Figure 4a – c). The lack of groundwater level data during the drained period (pre – 2000) result in higher uncertainty and less constrained soil storage and groundwater leakance estimations (Figure 4g). However, during the wetland rehabilitation (2000 – 2007), simulations of higher groundwater leakance at Berkenbrück relative to Demnitz Mill are consistent with the increased channel water loss due to the glacial valley deposits (Figure 1b). Groundwater levels (and subsequently groundwater leakance) also increased following the beaver recolonization (2008 – 2018).

4.3 Catchment stable water isotopes

Isotopes in catchment precipitation correspond quite closely to the global meteoric water line (GMWL) with a slope of 7.63 (95% confidence bounds range 7.37 to 7.89) and an intercept of 6.45 (95% confidence bounds range 4.05 to 8.86), and isotopic compositions ranging from values of -95 to 0 ‰ in δ_2H throughout the year (Figure 5a). The isotopes in stream water reveal notable spatial variability broadly corresponding to the arable land dominating Marxdorfer St. and Peat North; the large wetland area between Peat North and Peat South; the numerous tributaries contributing between Peat South and Demnitz Mill and the geological divide between Demnitz Mill and Berkenbrück (Figure 1). Stream isotopes (δ_2H and δ_{18}O) at Marxdorfer St. (median -56.62 and -8.12 ‰ for δ_2H and δ_{18}O, respectively) and Peat North (median -56.86 and -8.07 ‰ for δ_2H and δ_{18}O, respectively) were statistically similar (Wilcox Rank sum test, $p < 0.05$), with small deviations from groundwater isotopes (median -58.73 and -8.50 ‰ for δ_2H and δ_{18}O, respectively) (Figure 5b). The isotopic composition of stream water at Peat South (median -53.56 and -7.12 ‰ for δ_2H and δ_{18}O, respectively) and Demnitz Mill (median -54.25 and -7.53 ‰ for δ_2H and δ_{18}O, respectively) were significantly similar (95% confidence) and showed significant enrichment during the summer of 2018, plotting below the GMWL. The evaporative enrichment between Peat North and Peat South resulted in a significant difference between the isotopic composition of stream water, with a gradual decrease in stream enrichment from Peat South to Demnitz Mill. Isotopic enrichment was not evident at Berkenbrück (median -57.10 and -8.03 ‰ for δ^{2}H and δ^{18}O, respectively) through most of 2018 and was significantly different to the isotopic composition measured at Demnitz Mill (95% confidence, Figure 5b). Stream isotopes at Berkenbrück were statistically similar to Marxdorfer St. and Peat North in both δ^{2}H and δ^{18}O, with isotopic composition shifting towards groundwater isotopes. The shift towards groundwater is consistent with the re-emergence of regional groundwater in the lower stream network during summer (Figure 4).

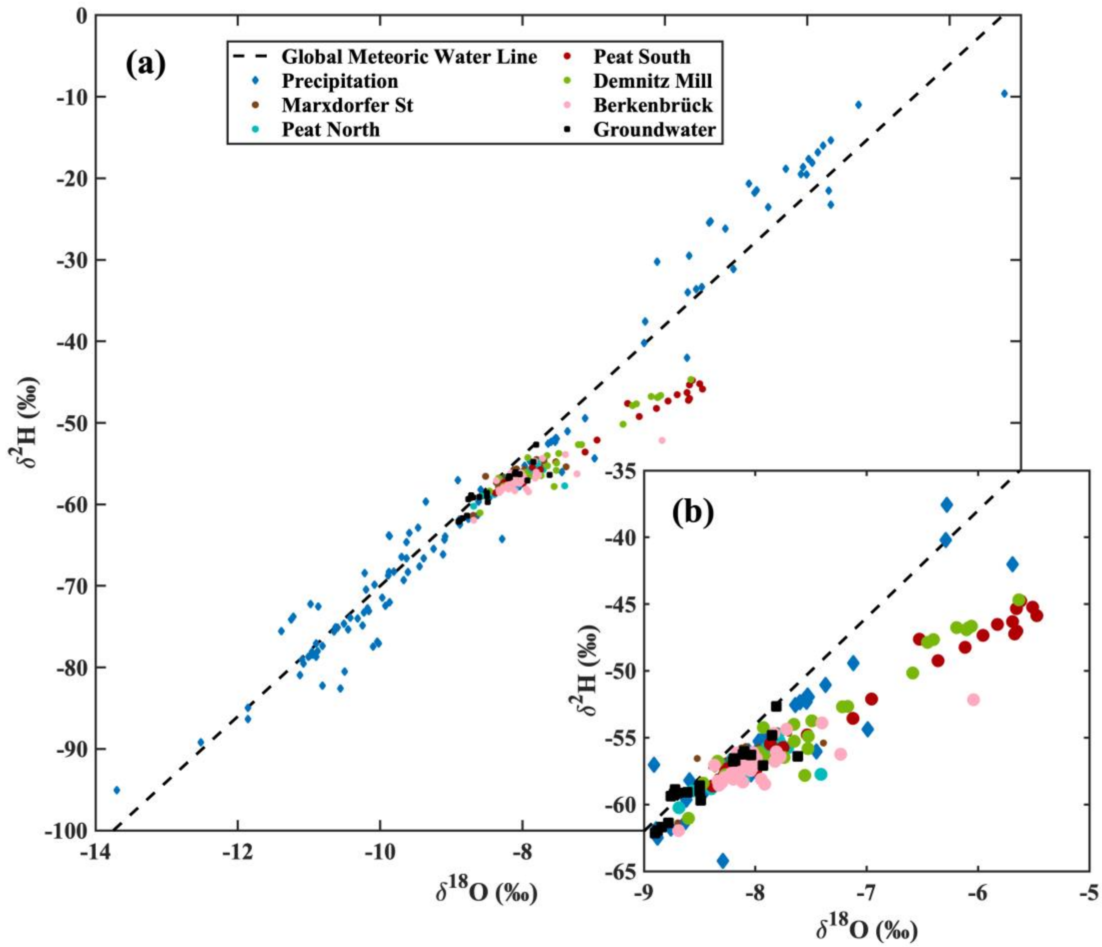


Figure 5: (a) $\delta^{2}H$ vs $\delta^{18}O$ for precipitation, stream water, and groundwater in relation to the Global Meteoric Water Line. (b) Expanded $\delta^{2}H$ vs $\delta^{18}O$ for stream isotopes at Marxdorfer St., Peat North, Peat South, Demnitz Mill, and Berkenbrück

4.4 Spatial variability in long-term stream water chemistry

The long-term median stream chemistry revealed circum-neutral waters that have a quite high ionic strength (though this tended to decrease downstream) and are enriched in nitrogen (NO_3^-) and phosphorus (TP and SRP) (Table 1). Persistent spatial patterns in the stream water quality were evident, with varying changes in different chemical parameters that are consistent with contrasts in soils, geological features, and land use. Similar to the stream isotopes, four broad regions sub-dividing the catchment explain the main changes in water quality. Significant differences in most stream chemistry parameters occurred between the small catchment area at Marxdorfer St., and Peat North. Concentrations significantly increased for SRP, TP, NH_4^+, Cl^-, and DOC, while pH, O_2, and NO_3^- significantly decreased. The large areas of agricultural land contributing to Peat North (Table 1) generally result in high concentrations of fertilizer-based nutrients (SRP, TP, and NO_3^-) in the stream water relative to other catchment streams (Table 4). The extensive wetlands between Peat North and Peat South, only caused DOC to significantly increase in both median and mean values at Peat South from Peat North. Although not significant for both mean and median values, NH_4^+, SO_4^{2-}, and Ca^{2+} showed a tendency to increase in concentration. A significant decrease in key stream chemistry parameters, SRP and NO_{32}^- was observed from Peat North to Peat South (Table 4).

*Table 4: Stream water chemistry with the median values, number of samples (n), standard deviation, and skewness. The superscripts * and † indicate a significant difference in the mean and median, respectively, of 95% to the next downstream site. Colours progress from the highest (red) to lowest (green) concentrations.*

	Marxdorfer St.		**Peat North**		**Peat South**		**Demnitz Mill**		**Berkenbrück**	
	Median (n)	**Max (Min)**	**Median (n)**	**Max (Min)**	**Median (n)**	**Max (Min)**	**Median (n)**	**Max (Min)**	**Median (n)**	**Max (Min)**
EC (µS/cm)	1082 (70)	1351 (846)	1063*† (393)	1422 (256)	1039*† (367)	1290 (208)	954*† (887)	2070 (220)	907 (404)	1429 (450)
pH	7.9*† (69)	8.2 (7.3)	7.8*† (388)	8.4 (6.7)	7.6*† (363)	8.2 (7.1)	7.9*† (875)	8.9 (7.1)	7.5 (417)	9.1 (6.8)
Water Temperature (°C)	6.6 (71)	17.5 (0.8)	8.9 (382)	20.1 (-0.2)	9.4 (355)	22.7 (-0.1)	8.1 (815)	22 (-0.9)	8.5 (394)	81.0 (-0.8)
O_2 (mg/L)	11.6*† (69)	16.6 (5.4)	8.9*† (378)	19.5 (1.8)	6.1*† (351)	13.4 (0.1)	9.8*† (824)	49 (0.1)	6.8 (376)	13.8 (1.0)
SRP (µg/L)	46*† (72)	174 (16)	117.5*† (404)	610 (16)	63.5* (370)	871 (9)	62*† (900)	722 (5)	45 (423)	525 (9)
Total P (µg/L)	66*† (70)	292 (57)	167† (399)	1176 (57)	146*† (364)	1008 (52)	117*† (898)	2934 (24)	100 (419)	782 (24)
NO_3^- (mg/L - N)	18.8*† (70)	27.7 (0.1)	5.4*† (392)	20.0 (0.1)	3.8 (344)	15.8 (0.0)	4.5*† (874)	25.0 (0.0)	1.0 (407)	19.5 (0.0)
NH_4^+ (mg/L -N)	0.06*† (66)	0.2 (0.0)	0.09† (222)	3.7 (0.0)	0.14*† (218)	1.8 (0.0)	0.08† (498)	1.9 (0.0)	0.11 (333)	1.3 (0.0)
SO_4^{2-} (mg/L)	166 (58)	218 (10)	188 (147)	321 (10)	196 (141)	314 (19)	197 (340)	665 (33)	210 (251)	433 (69)
Cl^- (mg/L)	43*† (58)	58 (24)	58*† (145)	90 (31)	52*† (141)	69 (20)	44* (342)	110 (20)	44 (282)	85 (18)

DOC (mg/L)	6.8*† (44)	12.7 (5.1)	7.8*† (205)	17.7 (4.5)	11.4*† (171)	29.0 (6.7)	13.5*† (528)	27.0 (6.8)	11.6 (351)	32.9 (5.5)
Ca^{+} (mg/L)	169 (21)	242 (150)	191 (130)	372 (41)	195*† (131)	604 (42)	175*† (170)	479 (49)	157 (108)	244 (59.0)
Mg^{2+} (mg/L)	16.5 (21)	18.3 (14.4)	17.2*† (130)	24.7 (4.5)	15.9*† (131)	23.3 (3.4)	15.5*† (158)	23.0 (4.4)	14.1 (76)	21.0 (5.4)

A significant difference in the median long-term concentrations of EC, pH, O_2, total P, NH_4^{+}, SO_4^{2-} DOC and Ca^{+} was observed between Peat South and Demnitz Mill (Table 4). The median concentrations generally declined from Peat South to Demnitz Mill (consistent with the increased forest cover), though pH, O_2, and DOC increased. The increase of pH between Peat South and Demnitz Mill marked the only increase in pH through the stream network. Long-term median stream water chemistry at Berkenbrück was markedly different than the long-term median stream water chemistry through other areas of the catchment. Long-term median stream water chemistry generally reached the lowest concentration in the catchment (EC, pH, SRP, NO_3^{-}, Mg^{2+}) or were observed to have low concentration with respect to the of stream measurement locations (O_2, total P, and Cl^{-}). Significant spatial decreases in the long-term median concentration from Demnitz Mill to Berkenbrück were observed for 10 parameters (including NO_3^{-}, SRP and total P), with only NH_4^{+}, SO_4^{2-}, and water temperature showing no significant change (Table 4).

Assessment of the seasonal variability of water quality, and associated spatial patterns, give higher resolution insight into how catchment land use and dominant hydrological flow paths interact. Due to the relatively low number of summer and autumn samples at Marxdorfer St. the significance of spatio-temporal changes was not considered for these seasons at this site. Most parameters showed some significant seasonal changes, with significant differences observed on average in three of the four seasons (Table 5). The stream generally showed highest TP and SRP concentrations during the summer months, declining gradually during the higher flows of autumn and winter to the lowest concentrations in spring. In contrast, NO_3^- and O_2 both showed the highest concentration during winter and spring. Concentrations of NH_4^+ and pH showed similar but inverse trends, with highest NH_4^+ concentrations in winter and summer (lowest pH in the winter and summer), and lowest NH_4^+ concentrations in the spring and autumn (highest pH in the spring and autumn). Spatially, the water quality patterns in each season were similar to the long-term median values, with the highest concentrations of TP, SRP, and NO_3^- observed throughout the year at Peat North, and the highest concentration of NH_4^+ observed at throughout the year Peat South (Table 4, Table 5).

*Table 5: Seasonal average stream water chemistry for Marxdorfer St., Peat North, Peat South, Demnitz Mill, and Berkenbrück, shown with the total number of samples (in parentheses). Significant (95% two-sided t-test with unequal variance) differences between locations downstream are indicated with *, and significant temporal differences between season are shown with †.*

		Winter (n)	Spring (n)	Summer (n)	Autumn (n)
pH	Marxdorfer St.	7.8 (22) †	7.9 (30)	8.0 (12)	7.9 (5)
	Peat North	7.8 (95) *†	7.9 (109) *†	7.7 (96) *	7.8 (88) *
	Peat South	7.7 (86) *	7.7 (100) *†	7.5 (92) *†	7.6 (85) *†
	Demnitz Mill	7.9 (246) *†	8.0 (244) *†	7.7 (204) *†	7.8 (181) *†
	Berkenbrück	7.5 (96) †	7.7 (120) †	7.5 (109)	7.4 (92)
O_2 (mg/L)	Marxdorfer St.	12.0 (22) *	11.3 (31)	7.8 (12)	10.4 (4)
	Peat North	10.8 (89) *	10.5 (110) *†	6.3 (93) *†	8.2 (86) *†
	Peat South	9.1 (79) *	7.9 (100) *	2.4 (89) *	4.7 (83) *
	Demnitz Mill	11.8 (225) *†	11.0 (234) *†	6.1 (189) †	8.0 (176) *†
	Berkenbrück	8.7 (80)	9.0 (113) †	5.6 (98)	5.1 (85) †
SRP (µg/L)	Marxdorfer St.	48 (23) *	40 (32) *	102 (12)	54.4 (5)
	Peat North	103 (98) *†	82 (117) *†	219 (100) *†	174 (89) *†
	Peat South	72 (88) †	44 (107) †	141 (87) †	104 (82) †
	Demnitz Mill	67 (252) *†	43 (255) *†	128 (207) *†	91 (186) †
	Berkenbrück	43 (98)	33 (121) †	85 (110)	77 (95) †
TP (µg/L)	Marxdorfer St.	63 (22) *	72 (31) *	198 (12)	72 (5)
	Peat North	153 (98)	133 (115) *†	296 (100) *†	219 (85) †
	Peat South	157 (87) †	119 (105) †	250 (82) †	207 (78) *†
	Demnitz Mill	129 (250) *†	109 (255) *†	223 (207) *†	150 (186) *
	Berkenbrück	85 (95)	94 (120) †	162 (109) †	122 (95) †
NO_3^- (mg/L-N)	Marxdorfer St.	18.8 (22) *	19.6 (32) *	11.9 (11)	12.9 (5)
	Peat North	9.1 (94) *	9.0 (115) *†	3.8 (96) *	4.7 (87) *†
	Peat South	7.4 (82)	6.9 (107) †	1.6 (87) †	3.2 (80) †
	Demnitz Mill	8.0 (239) *†	6.8 (250) *†	1.7 (202) *†	3.6 (184) *†
	Berkenbrück	4.5 (90)	4.9 (117) †	0.9 (106)	1.2 (94) †
NH_4^+ (mg/	Marxdorfer St.	0.05 (22) *†	0.07 (29) *	0.12 (10)	0.05 (5)
	Peat North	0.27 (55)	0.13 (64)	0.21 (52) †	0.10 (51) *†
	Peat South	0.32 (54) †	0.18 (61) *	0.22 (54) *†	0.14 (49) *†

	Demnitz Mill	0.23 (129) †	0.11 (150)	0.13 (107)	0.09 (112) †
	Berkenbrück	0.24 (72) †	0.11 (108) †	0.15 (81) †	0.09 72) †

4.5 Long-term changes in stream water quality

Some long-term changes in water quality were detected at each site throughout the study, aligning with both physical changes in the catchment (e.g. wetland rehabilitation and beaver dams), and atmospheric changes (e.g. increasing air temperatures). DOC increased by 50 % at most stream locations after the wetland rehabilitation (2000 – 2007), except for Marxdorfer St. and Peat North (Figure 6a). With the primary focus of the wetland rehabilitation (with beaver recolonization) on the relatively short stream segment between Peat North and Peat South, the increase in DOC of stream water at Peat South and downstream, relative to Peat North, suggests some effectiveness of increased water times. Annual average stream water temperatures also showed a slight increase after the beaver recolonization (2008 – 2018), though these changes were not significant (Figure 6b). However, significant increases in stream temperature following beaver recolonization were observed in the spring, summer, and autumn months (not shown). In the spring, significant increases in long-term temperature were observed at Marxdorfer St. and Demnitz Mill. In the summer and autumn months, significant long-term increases in water temperature were observed at Peat North and Peat South. Increases in summer water temperature are consistent with a longer water retention times due to stream rehabilitation and beaver dams. Beaver dams also likely influenced the water temperatures at Demnitz Mill (significant in the spring) but were limited to only early season increases in temperature (summer increase at Demnitz was significant to 5.1%). The water loss due to the subsurface between Demnitz Mill and Berkenbrück was likely the reason for no significant change in the long-term water temperature at Berkenbrück.

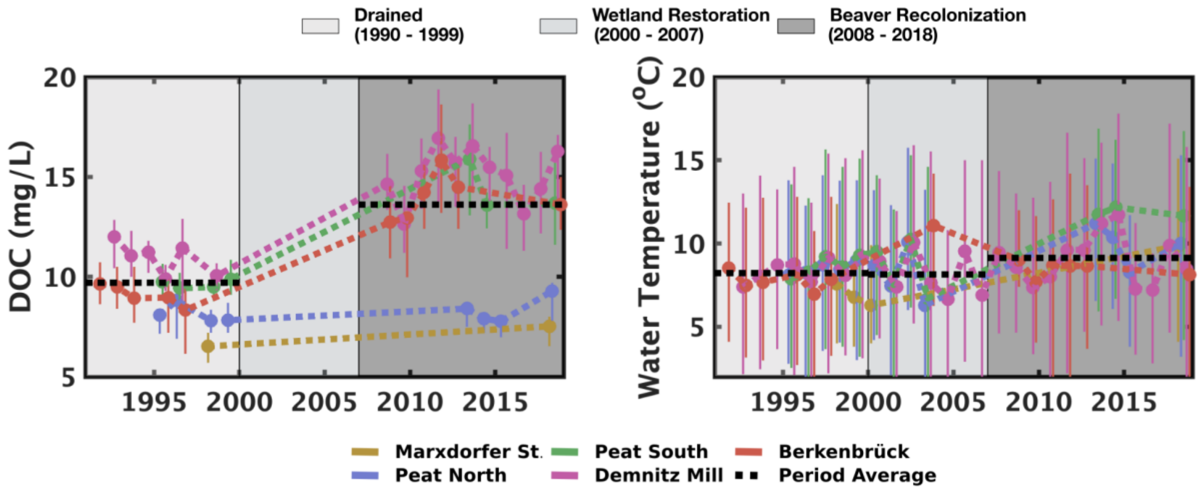


Figure 6: Long-term dynamics in stream chemistry. Circles indicate the annual average value with the corresponding 25th and 75th percentiles of the year shown with vertical lines. Horizontal black dashed lines for each period indicate the annual average for the period.

5. Discussion

5.1 Long-term catchment hydrological processes and responsiveness and hydrological partitioning and effects of long-term wetland change

The similarities of the DMC catchment to many lowland north-eastern European catchments presents an opportunity to explore these long-term changes for an improved understanding of the current and future functionality of the hydrology of such regions. Hydrological, isotope, and stream water chemistry all showed high groundwater influence in the stream water, consistent with the high baseflow contributions to the discharge (>75%) throughout much of the North German plain (Jankiewicz et al., 2005; Döll and Fiedler 2008) where sandy and loamy soils facilitate rapid groundwater recharge (Wittenberg et al., 2019). The significant changes in stream water chemistry and stream isotopes between Demnitz Mill and Berkenbrück suggests an associated change in the source of water driving the stream discharge. Significant interactions of the sub-surface blue water fluxes and surface waters in the greater northern German plains (Krause et al., 2007; Lewandowski et al., 2009), combined with the glacial till geology of the southern region of the catchment suggests increased interaction of the groundwater to the surface water relative to the northern regions of the catchment. Similar to other agriculturally influenced catchments, stream isotopic compositions at sites in the upper catchment dominated by agricultural land-use on siltier soils (Marxdorfer St and Peat North) were comparable to the groundwater isotopic compositions (Orlowski et al., 2016). Although there is a high estimated AET by the Thornthwaite model, the stream water showed very limited fractionation effects. The only fractionation effects were evident immediately downstream of a riparian wetland, similar to isotopic changes observed in riparian wetlands even in wetter, lower energy environments (in Scotland, Sprenger et al., 2017). However, the dominance of some of the processes and catchment responsiveness were clearly not temporally static.

The long-term increasing groundwater levels have had surprisingly small influence on the catchment discharge, which hasn't notably increased through either the wetland rehabilitation or following the re-colonization of beaver populations (Figure 5). Given the dominance of groundwater for driving discharge, this trend was unanticipated through the wetland rehabilitation. However, the limited effect may be in part due to the relatively small wetland area (1.7 km^2) may limit the effectiveness of the wetland to greatly change the average discharge (Martinez-Martinez et al. 2014). While the average discharge did not greatly change after the beaver re-colonization (after 2007), the increasing and more stable groundwater level (Figure 4) suggests that the beaver dams may have resulted in increased surface water leakage to groundwater fluxes as observed in other catchments (Hill & Duval, 2009). This change is further supported by the storage – discharge, diel variability and stream water temperature from before to after the beaver re-colonization (Figure 2, Figure 6). The longer responsiveness (increase of 30 hours to mean stream recession time) of storage to discharge may be indicative of a slower, smoothed release of impounded water downstream of beaver dams (Woo and Waddington, 1990). Pre-beaver diel variability is likely a combination of the ET and the attenuation of the stream channel network (Fonley et al., 2016). Following the re-colonization of the beaver, loss of diel variability coincides with the smoother release of water observed with the storage – discharge relationship and suggests that the loss of variability is due to a disruption of the stream network. While only three-hourly data (with spline interpolation to hourly data) were available for the diel variability analysis, the hours available provided sufficient information to inform on small daily changes (Appendix B). Although the sub-daily water level measurement gap (2008 – 2010) may inform on the transition of the catchment responsiveness due to the beaver re-colonization, the gap provides a distinct effect of the beavers rather than highly variable conditions. Along with the long-term increase in groundwater level, the estimated AET has increased in the catchment since 1990. However, with the limited size of the wetland rehabilitation (1.7 km^2) and additional effect of the beaver dams, it is likely that the increase in AET is due to a combination of higher groundwater levels and the long-term increases in temperature.

5.2 How are seasonal and long-term water quality affected by changes in the riparian wetland landscape?

The extensive water quality data provided significant insights into the spatial patterns and short (seasonal) and longer (inter-annual) term temporal variability due to changes (wetland rehabilitation and creation of beaver dams) in some riparian regions of the catchment. Despite the remediation efforts, the move towards ecological farming, and added benefits of the migration of beaver populations into the catchment, there have not been clear unequivocal improvements in stream water quality. The seasonal trends in TP and SRP indicated a distinct pattern of high-concentration in the spring/summer months (Table 5). The high concentration of TP and SRP in the spring and summer months may be due to many factors, including high runoff periods due to late winter rainfall/snowmelt processes, fertilizer application timing, evapoconcentration, and desorption from the channel sediments. While spatial differences in SRP and DOC show reduction downstream of the wetlands, the consistent seasonal trends at each site and negligible change in TP downstream of wetland rehabilitation or beaver recolonization suggests limited effectiveness. This limited effectiveness of the TP from the stream could be a result a limited extent of the wetlands (2.5% of catchment area), relative to the catchment size. (Lee et al., 2009). However, the longer retention in the wetland rehabilitation may be effective for biological decomposition of peat (Mann and Wetzel, 1995), resulting in increased DOC, similar to other studies (Pinney et al, 2000; Bossio et al., 2005).

Conversely for NO_{3^-} and NH_{4^+}, lowest concentrations occurred during the spring/summer months. Increased vegetation during the growing season is likely the result of decreasing concentration of NO_{3^-} and increasing concentration of NH_{4^+} during the spring and summer months. While spatial patterns showed that NO_{3^-} reduced between Peat North and Peat South, the negligible temporal decrease of NO_{3^-} at Peat South after the wetland rehabilitation or beaver recolonization may be a result of the gradual increase in the groundwater table. The increasing potential for groundwater reconnecting with surface ponded waters and releasing legacy NO_{3^-} from groundwater could contribute to maintaining high stream NO_{3^-} concentrations (Taylor et al., 2005; Van Meter et al., 2016). Over extended periods, the connection of the surface-groundwater recharge interactions, combined with the high connectivity between the groundwater and stream, can be beneficial for reducing catchment nutrient loss. The longer groundwater travel times allow for greater biogeochemical processing which may immobilize nutrients in stored sediments (in the case of P) or lead to reducing conditions and atmospheric losses (in the case of N). In addition, longer transit times of nutrients facilitate uptake by wetland vegetation which can further reduce downstream nutrient fluxes (Griebler and Avramov, 2015). While a legacy effect of groundwater to the channel may result in a stabilization of the NO_3^{2-}, the limited connectivity of the surface water to the groundwater outside of the channel could be a significant reason for a lack of NO_3^{2-} reduction through the wetland rehabilitation. The change in water temperature similarly suggests a very limited increase in surface area, with a relatively small, but increasing water temperature following the beaver recolonization is consistent with observations of beaver dams at other sites (Burchsted et al., 2010; Majerova et al., 2015).

5.3 Implications for future management and ecohydrological research needs.

The catchment is representative of a landscape that covers a much larger area of the North German plain. Insights into the changing hydrological function of the catchment can thus be extrapolated to a larger region where climate change predictions are likely to increase drought-sensitivity in future. The long-term study sites like the DMC are therefore important resources for focusing new work to provide an evidence base to guide future land management to protect water resources. The increasing temperature trends, both in stream water (Figure 6) and in air temperature (UBA, 2005) will likely extent the duration of optimal growing temperature and increase PET in catchments, such as the DMC, already dominated by green water fluxes (>90% of precipitation) (Bormann, 2011). Increasing PET may not have an equivalent effect on crop production, as projected precipitation decreases result in an estimation of declines in total production of wheat and maize, which are primary crops in the DMC (Asseng et al., 2015; Moore and Lobell, 2015; Hatfield and Prueger, 2015). While other vegetation may increase the green water fluxes of ET, these green water fluxes likely influence future droughts less than blue water flux decreases (Orth and Desouni 2018). However, potentially increased green water fluxes with decreasing precipitation, has the potential to further reduce already sensitive blue water fluxes to groundwater recharge and discharge generation (Berghuijs et al., 2016). Consequently, we need to move forward from this initial assessment to improve our understanding of how vegetation and riparian wetlands influences the partitioning of green and blue water fluxes. This imperative is further underlined by the likelihood that declining precipitation and a prolonged growing season will likely be accompanied by increased frequency and intensity of droughts, potentially further reducing blue water fluxes (Klove et al., 2014; Dezsi et al., 2018). These changes have implications for land management, in terms of the appropriate mix of forestry and agriculture as well as conservation of riparian habitats and species such as beavers. Furthermore, within the agricultural sector, consideration of the choice of crop species is needed, and a likely move towards more drought resilient species which may limit green water fluxes. These decisions are more difficult without sufficient information on the water usage of specific vegetation, which may be resolved through further exploration within the catchment using stable isotopes in the critical zone, including in soil water and vegetation (Brooks et al., 2015; Tetzlaff et al., 2015). The observed spatial heterogeneity of the catchment additionally warrants further exploration of how ecohydrological processes and flow paths change with scale.

6. Conclusion

Agricultural regions across Europe are key areas for long-term hydrological assessment due to possible effects of land management and climatic change on water resources, and in-stream nutrients. In this study, we assessed the long-term hydrological processes and stream water chemistry under the ecohydrological implications of catchment re-naturalization on water partitioning. Long-term effects of wetland rehabilitation and stream damming by beavers had notable impacts on the catchment storage, increasing the total ET loss from the catchment which is already ~90% of the catchment water loss. Stable isotopes helped to quantify the importance of groundwater contribution to the stream network and evaporative fractionation in wetlands. The 30-year stream water quality record indicated significant increases in stream water temperature and DOC, with minimal changes in other stream chemistry including NH_4, NO_3, TP. Retention in the wetlands in the middle of the catchment provided a significant decrease in the concentrations of some nutrients. The integrated examination of long-term stream discharge, stable isotope data, stream water quality, and simple water balance approaches provided valuable insights into the observed changes due to both wetland rehabilitation and recolonization of beaver populations, as well as an improved understanding of the key areas for partitioning blue and green water fluxes in the landscape.

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