# Impact of drought hazards on flow regimes in anthropogenically impacted streams: an isotopic perspective on climate stress

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**Abstract**

Flow regimes are increasingly impacted by more extreme natural hazards of droughts and floods as a result of climate change, compounded by anthropogenic influences in both urban and intensively managed rural catchments. However, the characteristics of sustainable flow regimes that are needed to maintain or restore hydrologic, biogeochemical and ecological function under rapid global change remain unclear and contested. We conducted an inter-comparison of two streams in the Berlin-Brandenburg region of NE Germany, which are both mesoscale sub-catchments of the River Spree; an intermittent rural agricultural stream (the Demnitzer Millcreek) and a heavily anthropogenically impacted urban stream (the Panke). Through tracer-based analyses using stable water isotopes, we identified the dominant physical processes (runoff sources, flowpaths and age characteristics) sustaining streamflow over multiple years (2018-2023), including three major drought years (2018-20, 2021-22). In the urban stream, low flows are regulated through artificially increased baseflow from treated waste water effluent (by up to 80%), whilst storm drainage drives rapid, transient high flow and runoff responses (up to 80%) to intense convective summer rainfall. The intermittent groundwater-dominated rural stream experienced extended no-flow periods during drought years (~ 60% of the year), and only moderate storm runoff coefficients (<10%) in winter along near-surface flows paths after heavy rainfall. In both streams, groundwater dominance with young water influence prevails, with low water ages in the urban stream (<10%) despite significant urban runoff, and higher ones in the rural stream (~15%). Urban cover resulted in mean transit time of ~4 years compared to arable land with ~3 years, highlighting the interlinkages of landuse and catchment properties on catchment transit times. Understanding seasonal and interannual variability in streamflow generation through a tracer-based hydrological template, has potential for assessing the impacts of natural hazards on the sustainability of future baseflow management, including wider water quality and ecological implications across anthropogenically impacted environments.

## 1 Introduction

Urbanization and anthropogenic alterations to hydrological pathways, drainage networks and flow regimes have progressively changed the water balance and dynamics of contemporary streams and rivers, increasing their sensitivity and impacts to climatological and hydro-meteorological hazards (Bonneau et al., 2018; Soulsby et al., 2014; Stewardson et al., 2017). Along with the well-established impacts of anthropic changes imposed on urban freshwaters, many other areas, including peri-urban and rural agricultural environments, are also experiencing dramatic alterations to natural flow regimes and hydrologic processes due to increase drought frequency and intensity (Döll & Schmied, 2012; Yang et al., 2011). These changes are propagated by the persistent reorganization of surface and subsurface hydrological flowpaths, widespread landuse changes and stream network alterations, as well as increasing baseflow manipulations (Bonneau et al., 2018; Marx et al., 2021; Oswald et al., 2023; Soulsby et al., 2014).

In recent years the frequency and intensity of hazards such as floods and severe multi-year droughts have contributed a paradigm shift in future streamflow management needs and the recognition of a persistent lack of understanding of essential hydrologic processes in urban and other anthropogenically impacted systems (Arthington et al., 2006; Oswald et al., 2023). Despite the importance of natural flow variability (Poff et al., 1997; Stewardson et al., 2017) and numerous studies demonstrating the effects of changes in natural flow regimes on hydrological and ecological function (Arthington et al., 2006; Bhaskar et al., 2016; Olden & Poff, 2003; Poff & Zimmerman, 2010; Tetzlaff et al., 2005), there is still a distinct lack of understanding of anthropogenically impacted flow regimes and their evolution in the face of rapid global change. This provides a weak evidence base for managers wanting to maintain or restore a baseline of natural flow regime characteristics that supports the hydrologic, biogeochemical and ecological functionality of freshwater systems that provide important ecosystem services (Acreman et al., 2014; Arthington et al., 2006).

Concerns over water stress and drought as drivers of rapid hydrological change have intensified both in cities (Kuhlemann et al., 2020; Paton et al., 2021), and lowland agricultural catchments (Kleine, Tetzlaff, et al., 2021; O'Briain, 2019; Wu et al., 2021). During recent severe drought years (2018-2020) in northern and central Europe, where significant shifts of streamflow from perennial to intermittent were widely observed, and the probability and longevity of intermittency is likely to increase with projected increases of temperatures across Europe (Kleine et al., 2020; Lobanova et al., 2018; Sarremejane et al., 2022; Tramblay et al., 2021). In addition, ongoing urban densification contributes to increasingly flashy hydrographs, deteriorating water quality and increased influence of waste water discharges, causing flow regimes to increasingly deviate from the "natural flow paradigm" with a seasonal succession of high and low flows (Bhaskar et al., 2016; Bonneau et al., 2018; Marx et al., 2021; Soulsby et al., 2015). Especially biodiversity and the health of aquatic and wetland ecosystems hinge on ecological processes, which are dependent on the natural occurrence and variability of high and low flows, while baseflows exert critical controls on habitat maintenance and the survival of different aquatic species, as well as the moderation of water temperatures, water quality, oxygen levels, nutrient loads and vegetation growth (Arthington et al., 2006; Poff & Zimmerman, 2010; Stewardson et al., 2017). Biodiversity

For stream management, a landscape scale understanding of controls on streamflow regimes requires an integrative approach that captures ecologically and hydrological meaningful characteristics of seasonal flow dynamics (Arthington et al., 2006; Tetzlaff et al., 2005; Tonkin et al., 2021). Environmental tracers, such as stable water isotopes, can be useful for characterizing complex hydrological systems in order to understand hydrological functioning across multiple scales (Ehleringer et al., 2016; Jasechko, 2019; Kendall & McDonnell, 1998; Stevenson et al., 2022). Tracer applications and tracer-based models can provide insight into controls on streamflow generation across different climatic and geographic scales (Bonneau et al., 2018; Stevenson et al., 2022; Von Freyberg et al., 2018). The conservative behaviour of stable water isotope ratios of water ($\delta^{18}O$, $\delta^{2}H$) and their ability to integrate hydrological processes make them useful indicators of water sources and flowpaths (Ehleringer et al., 2016; Marx et al., 2021; Von Freyberg et al., 2018). This can help quantify drought effects (Kleine et al., 2020; Kuhlemann et al., 2020; Smith et al., 2020), mean transit times and water ages (Birkel et al., 2016; Hrachowitz et al., 2010; Soulsby et al., 2015; Tetzlaff et al., 2015), as well as groundwater-surface water interactions and recharge (Wallace et al., 2021; Ying et al., 2024) across a range of temporal and spatial scales.

As streamflow generation and intermittency is becoming an increasingly important issue under advancing climate change in the Berlin/Brandenburg region (Kleine, Tetzlaff, et al., 2021; Luo et al., 2024; Ying et al., 2024), a more integrated understanding of a catchments' ability to capture and release water is invaluable. We focus on two contrasting catchments - one urbanized and one rural agricultural catchment, both within the Berlin-Brandenburg region, Both catchments are tributaries of the river Spree, which serves as a major water provider to the city of Berlin and experienced extreme hydroclimatic conditions between 2018-2020, including a major drought period (Creutzfeldt et al., 2021). Using a long-term dataset of daily stable water isotopes and hydrological analyses of discharge dynamics, we sought to understand how drought and intensive storm events have influenced the hydrological and physical functioning of streamflow regimes in two contrasting anthropogenically impacted catchments.

We specifically aim to assess i) how hydrological functioning such as water partitioning, runoff sources, transit times and water ages varies between two contrasting over a 5-year period, ii) to characterize streamflow persistence and resilience during the extreme drought period of 2018-2020, and in response to selected intensive storm events as exemplary hazards and iii) to understand implications for flow regime changes under projected future hydroclimate perturbations. We focused on characterizing the contrasting streamflow responses between urban systems with large urban storm drain effects and artificially increased baseflow, vs. rural groundwater-fed intermittent systems with agricultural drainage within the Berlin/Brandenburg region, where drought hazards and water-related risk affect the availability and distribution of fresh water. In doing so, we sought to provide the hydrological context and evidence basis to support environmental decision makers in establishing sustainable environmental flow targets, maintain biodiversity and ecological integrity (Poff & Zimmerman, 2010; Tonkin et al., 2021).

## 2 Study Catchments

In this study, Demnitz Millcreek - a rural agriculturally influenced stream in the State of Brandenburg, and the river Panke – a heavily urbanized stream in Berlin were compared (Fig. 1). Both catchments (hereafter referred to as the rural and urban catchments) are situated within 100km of each other and are tributaries of the river Spree, which has a catchment size of > 10.000 $km^2$ and is the major water provider to the city of Berlin. Both catchments are situated in one of the driest parts of NE Germany with 577 mm of annual precipitation (1981-2010 average) distributed throughout the year as frequent, lower-intensity frontal winter rains and infrequent heavy convective storms in summer (DWD, 2023). However, regional climate differences exist between the dense urban metropolitan area of Berlin and the drier lowland Brandenburg region, where potential evapotranspiration generally exceeds annual precipitation inputs, thus increasing drought sensitivity (~700mm/yr) (DWD, 2023).

The rural catchment (total catchment area ~66 $km^2$) is strongly influenced by the regional groundwater system of the Spree valley. The catchment has a mixed landuse, including forest, wetland, cereal crops and pastures, with only small, distributed villages giving 2% urbanized area (Fig. 1b). A full table of landuse distribution for both catchments is provided in Supplementary Materials (Table S1). The site is part of a long-term ecohydrological observatory and extensive meteorological and ecohydrological data has been measured at multiple temporal and spatial scales since 2018 (Tetzlaff et al., 2023). Arable non-irrigated landuse is highest in the northern part of the catchment (~68%). Agricultural crops primarily include water-demanding cereal crops, such as winter wheat, barley, and maize, occupying the higher quality soils, whereas in the lower part of the catchment mixed forest and wetlands dominate, which are primarily used as pastureland (Fig. 1d) (Smith et al., 2021).

As the topography is flat and characteristic of a lowland landscape, elevation gradients are negligible across the catchment. This reflects the strong influence of glaciation, resulting in characteristics sections of unmixed sediments. While soils in the agricultural areas comprise silty brown-earths, the lower catchment is dominated by sandy soils with low water retention. The river channel network is fringed by peaty soil, particularly in the central wetlands (Fig 1f). The upper catchment is underlain by unconsolidated sediments of a ground moraine, with a moderately permeable unconfined aquifer and shallow groundwater levels within a few meters of the surface (Fig. 1f) (Ying et al., 2024). The stream network has been significantly altered by historical agricultural management, resulting in a high density of drainage ditches and artificial channels, affecting nutrient transport and water quality. To increase nutrient retention and regulate water quality, a nature-based solution approach was adopted by restoring the central wetlands by partially blocking drains (starting in 2000), installing a weir and shallowing wetland channels, which increased the water retention in the area and led to a subsequent recolonization by beavers (around 2007) (Smith et al., 2020). The rural catchment is highly drought sensitive, with an intermittent groundwater-dominated flow regime (with the stream being dry most years for 3-4 months) and overall low run-off coefficients even during wet periods (<10%) due to high evaporative losses (Kleine et al., 2020). Groundwater recharge primarily occurs during the cold/wet season, driving seasonal streamflow generation.

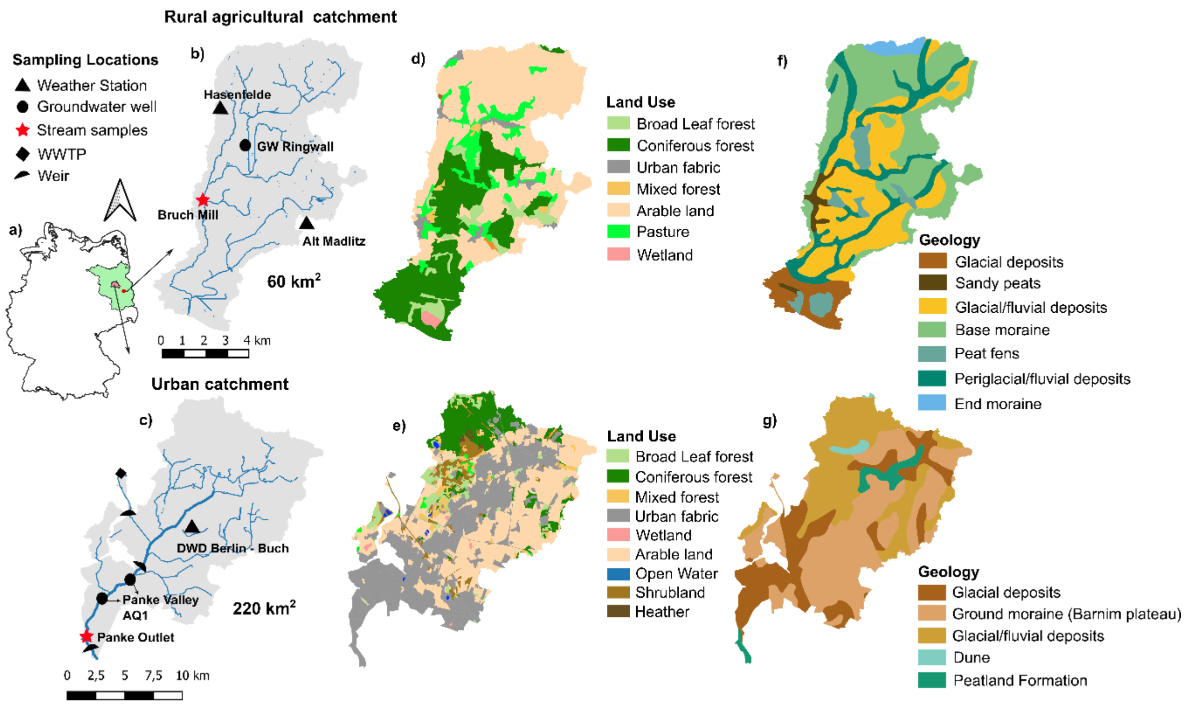



**Figure 1:** *a) Location of catchments within the Berlin-Brandenburg region. b, c) Overview of sampling locations in the rural and urban catchment respectively, including stream sampling locations (red) of regular stream isotopic samples, groundwater monitoring wells, weather stations and in stream weirs; d, e) Distribution of landuse and f, g) Map of the geology in both catchments. (Basemaps: LGB (Landesvermessung und Geobasis Information Brandenburg) 2020; Umweltatlas Berlin/ALKIS, 2020)*

The urban catchment (~220 km$^2$) is drained by the highly regulated Panke river, also a tributary of the Spree, draining a densely urbanized landscape (Fig. 1c). The headwaters are predominantly rural, lying on the northern edge of the Warsaw-Berlin glacial spillway (Fig. 1g). The main geological unit of the shallow aquifer (AQ1) that feeds the catchment is partially confined in the East by an overlying ground moraine, and unconfined in the lower catchment with layers of sands and gravels overlying an aquitard of glacial till. Like the rural catchment, the stream network drains silty soils in the headwaters and sandy soils in the south. While inthe North of the catchment ~22% is covered by urban fabric (see Table S1 in Supplementary Material) the lower catchment is more densely urbanized (~40%). Upstream, streamflow is generally groundwater-dominated, with seasonally varying inflows from agricultural and forested areas, in addition to the impact of urban storm drains, with some stretches of the river dry in summer (Marx et al., 2021) (Fig. 1e). In the lower catchment the dominant sources of runoff are effluent releases, which generally vary in response to season, with an increased volume of wastewater flowing into the river (up to 80%) during the drier summer periods to enhance baseflow. Conversely, during wetter periods and in case of heavy summer-storm events, peak flows are  diverted in to the neighbouring Tegeler catchment and the Nordgraben (Marx et al., 2023). This has led to a highly artificial flow regime with no clear seasonal variation between high and low flows. In recent years the catchment has been subject to targeted stream restoration to improve ecological conditions and water quality, as well as flood mitigation by rainwater management (SenUVK, 2009). The urban catchment, while different in size, land use, geology and water management, did resemble the rural catchment prior to the advanced urbanization and loss of agricultural and forest area. Therefore, the comparison between both catchments elucidates baseflow responses following anthropogenic impact and extensive management.

## 3 Data and Methods (Kleine et al., 2020; Kleine, Smith, et al., 2021; Marx et al., 2023)

### 3.1 Climate and hydrological data

In the rural catchment, hourly meteorological data were obtained for the period 2018-2023 from automatic weather stations (AWS) at locations in Hasenfelde (WLV, Environmental Measurement Limited, UK) and Alt-Madlitz (Campbell Scientific, USA) (Fig. 1b15 min discharge was measured at Bruch Mill (catchment area: 42 $km^2$) from water level measurements by pressure sensors (AquiLite ATP 10, Aquitronic Umweltmeßtechnik GmbH, Kirchheim/Teck, Germany) and transformed through an established rating curve (Smith et al., 2020). Hourly meteorological data (2018-2023) for the urban catchment were obtained from a climate station (Buch) of the German Weather Service (DWD, 2023) (Fig. 1c). 15-min as well as daily discharge data for a station in the lower Panke catchment (Bürgerpark). Both weather stations have been previously used in studies of each catchment (Kleine et al., 2020; Kleine, Smith, et al., 2021; Marx et al., 2023).  Groundwater levels in the urban catchment was taken from wells in the unconfined Panketal aquifer (AQ1), provided as publicly available data from the Berlin Senate (SenUVK, 2023). Groundwater in the rural catchment was monitored from a location in a forest in the central catchment (GW Ringwall, screened 2-4 m below surface), with groundwater levels logged every 4hrs (AquiLite ATP10, AquiTronic Umwelmeßtechnik, GmbH, Kirchheim/Teck, Germany).

Hydrological and climate data was separated into hydrological years (October 1$^{st}$ – September 30$^{th}$), which were used to calculate different metrics for each stream, such as total runoff ($Q_{Ro}$), storm totals and intensity, storm duration as well as annual and seasonal precipitation totals and magnitudes (minimum, maximum) of discharge (see Table S2 for a summary of parameters)Discharge was normalized to catchment areas (urban: 220 $km^2$; rural 42 $km^2$) and flow duration and double-mass curves derived to characterize interannual variability of discharge responses. Hydrograph separation of stream discharge into baseflow ($Q_B$) and stormflow ($Q_S$) was achieved using HydRun, a MATLAB-based toolbox (Tang & Carey, 2017), which is based on a recursive digital filter technique developed by Nathan & McMahon (1990). The filter coefficients (fc) and number of filter passes ranged between 0.7-0.99 and 0-10 respectively. Flashiness, as the rate of change in streamflow, was estimated through the Richard Baker Flashiness index (Baker et al., 2004), using daily stormflow data.

To examine storm activity and streamflow responses, precipitation data was separated into summer (June-September) and winter data (October -May). Hourly precipitation characterized total event precipitation ($P_{total}$, mm), mean precipitation intensity ($P_{int}$, mm $h^{-1}$), streamflow peak ($Q_{peak}$, $ls^{-1}km^2$), maximum precipitation over 1 h ($P_{max}$, mm $h^{-1}$) and rainfall duration (T, hours). Storm events were identified automatically, whereby precipitation events exceeding one hour were aggregated; for multiple consecutive events with <5hr breaks or multi-day events, the event precipitation was summarized into one total storm amount. Statistical differences in storm characteristics were assessed through Kolmogorov-Smirnov tests and Spearman rank correlations, using a p-threshold of at least 0.05 (95% confidence level). The annual runoff coefficient was calculated as the ratio between stormflow and precipitation using annual runoff and total annual precipitation ($Q_{Ro}/P$), while total annual runoff was estimated using annual discharge and baseflow totals ($Q_{Ro} = Q_S - Q_B$). The baseflow index (BFI) was calculated as a ratio of the total baseflow volume to the total runoff volume for each hydrological year, to assess the proportion of stream runoff derived from stored sources (e.g. groundwater).

### 3.2 Stable water isotopes

Daily precipitation isotopes were collected at Hasenfelde (rural) from 2018, and at the Steglitz Urban Ecohydrological Observatory (SUEO) (urban) in SE Berlin from 2019, using a modified ISCO 3700 (Teledyne Isco Lincoln, USA) automated samplers. Samples were protected from evaporation by a paraffin layer (thickness > 0.5mm, IAEA/GNIP precipitation sampling guide V2.02 September 2014). At Bruch Mill (rural), daily stream water isotopes were sampled from 2018, at 16:00 each day, using an automated ISCO 3700 (also protected from evaporation by paraffin). In the urban catchment, daily stable water isotope samples were collected manually from October 2019 until December 2022, and weekly isotopes from 2023 onwards, near the most downstream gauging station (Fig. 1c). Monthly groundwater isotope samples were collected from multiple wells in the AQ1 aquifer as part of a measurement campaign in 2020/2021 (Marx et al., 2021). All samples were filtered (0.2 µm cellulose acetate) into 1.5ml vials and analyzed for $\delta^{18}O$ and $\delta^{2}H$ using a Picarro L-2130-I cavity ring-down water isotope analyzer (Picarro Inc., Santa Clara, CA, USA) in reference to the Vienna Standard Mean Ocean Water (VSMOW). Relationships between daily discharge and streamwater isotopes were assed using Kolmogorov-Smirnov and Spearman's rank correlation coefficients.

Local Meteoric water lines (LMWL) were derived using daily precipitation isotope values from Steglitz (February 2019 – September 2023) and Hasenfelde (July 2018- July 2023) for the urban and rural catchments, respectively, by weighting respective precipitation inputs. To assess evaporation effects on stream water isotopic composition we also calculated the line-conditioned excess (lc-exc), which defines residuals from the LMWL (Landwehr & Coplen, 2006). For each catchment, lc-exc was estimated as:

Urban:

$$lcexc = \delta^2 H - 7.8 * \delta^{18} O - 7.1 \; (R^2 = 0.98, p < 0.001) \quad (1)$$

Rural:

$$lcexc = \delta^2 H - 7.68 * \delta^{18} O - 7.68 \; (R^2 = 0.98, p < 0.001) \quad (2)$$

### 3.3 Water ages and mean transit time estimations

To assess the fraction of stream water that fell as recent precipitation and use it as a metric of the age of stream water, young water contributions ($F_{yw}$) were estimated using the open-access code of von Freyberg et al. (2018). This is based on an iteratively re-weighted least square fitted sine-wave method, using observed precipitation and stream water isotopes to estimate the fraction of stream water that fell as precipitation within previous 2-3 months as an indicator of catchment function. We compared sine-wave fit amplitudes of daily amounts of weighted precipitation $\delta^2$H and $\delta^{18}$O to stream water isotopes in both streams. For simplicity only results for $\delta^{18}$O are shown in subsequent plots.

For inter-comparison between the rural and urban stream functioning, mean transit times (MTT) were estimated as another metric of hydrologic response. We used daily amount-weighted precipitation isotope data from SUEO (urban) and AWS Hasenfelde (rural), and daily stream water isotopes and applied two different lumped convolution integral models - the three parameter Two Parallel Linear Reservoir Model (TPLR) (Weiler et al., 2003) and the two parameter Gamma Model (Hrachowitz et al., 2010), to estimate transit time distributions (TTDs) (see Supplementary Table S3 for details). The TPLR model accounts for fast ($\tau_f$) and slow flow reservoirs ($\tau_s$), approximating younger and older water contributions. The reservoirs are partitioned by the $\varphi$ parameter, ranging from 0-1, and which separates the rapid and slow responding flows from surface and subsurface sources. In the urban area we based $\varphi$ on the percentage of impervious area, driving fast urban drainages to the stream. In the rural catchment we used non-irrigated arable land area, as rapid runoff is more likely contributed due to more compacted soils and agricultural drainage networks.

The Gamma Model is defined by the shape parameter $\alpha$ (-) and the scale parameter $\beta$ (days), with the MTT calculated as the product of the two. The parameter ranges for the TPLR ($\tau_f$ , $\tau_s$ ) and Gamma model ($\alpha$,$\beta$) were sampled from pre-defined parameter ranges using Monte Carlo realizations, in order to find the best fit estimates. To avoid influence of evaporative fractionation on model estimates, a lc-excess filter was applied to stream water isotopes, whereby samples with strong evaporative fractionation were excluded from calibration. We used different lc-excess filters for urban and rural streams (rural: lc-excess < -2.5‰ (i.e. more enriched); urban: lc-excess < -4‰ (more depleted)), as greater evaporative fractionation effects were observed in the rural stream. Model fits were assessed using Nash Sutcliffe efficiency (NSE) (Nash & Sutcliffe, 1970), root-mean-square-error (RMSE), and coefficient of determination ($R^2$). Due to the limitations of the stable water isotopes to detect transit times longer than 5 years (Stewart et al., 2010), the scale parameter $\beta$ of the gamma model and the $\tau_s$ parameter in the TPLR model, were limited to 1825 days.

## 4 Results

### 4.1 Rainfall-runoff characteristics and storm responses

The sampling period (2018 – 2023) was strongly marked by the intensive drought between 2018 – 2020, resulting in below average precipitation, and marked discharge responses in both catchments. Annual discharge statistics for both catchments were summarized for each hydrological year in Table 1. During WY 2019-20 and 2021-22 precipitation totals in the Brandenburg region were ~20-30% below the long-term average, while in the metropolitan area they were reduced by ~10-20%.

Several heavy convectional rainstorms during the summers of 2021 and 2023 resulted in above average annual totals and distinct discharge responses in both catchments. However, regional differences become apparent most notably during summer 2019, where a large convectional event over Berlin produced almost 46 mm/hr and an urban discharge peak of 5.3 $ls^{-1}km^2$ (Fig. 2c, d), while in rural Brandenburg no rain was recorded during the same period. The extremely sandy soil of the rural catchment and large soil moisture deficits generally limited streamflow response to summer storm events, despite the size and intensity of some of events (i.e. June 2021) (Fig. 2a, b). Only minimal transient flow, or very small streamflow peaks (i.e. summer 2020) were detected, likely originating from saturated areas fringing the channel bed or the wetter riparian areas upstream, and receding quickly to isolated pools of water, whereas significant discharge responses could be seen in the urban stream (Fig. 2d). This is in line with a noted strong positive correlation between storm discharge ($Q_S$) and total event precipitation ($R^2$= 0.54, $p<0.0001$) in the urban stream, which was less evident in the rural stream ($R^2 = 0.18$, $p<0.001$).

Due to the nature of the urban catchment functioning and management, distinct and large step changes with differences of up to 10 $ls^{-1}km^2$ were also observed following the most intense events. Hydrograph separation showed that the large streamflow peaks after events typically subsided within < 1hr. Conversely during low flow and drought periods, step changes by up to 1 $m^3/s$ (from 0.5 to 1.5 $m^3/s$) could be observed (i.e. April 2020), a clear response to additional discharge of wastewater. Characteristic of such intensely managed urban streams, a higher specific discharge was observed compared to the rural stream, with mean daily discharge ranging between 0.2-0.45 $l\ s^{-1}\ km^2$, and peak flows of up to 3.4 $l\ s^{-1}\ km^2$ (Fig. 2d). This is a clear effect of urban water management, keeping baseflow at a minimum level (~ 0.29 $l\ s^{-1}\ km^2$) through the consistent influence of wastewater, which contributed up to 80% of annual discharge. In contrast, the rural stream was characterized by clear seasonal intermittency. with the onset of the flowing and fully connected phase strongly tied the onset of autumn/winter rainfall and increasing groundwater levels. The lack of management and additional water sources to increase streamflow in the rural catchment resulted in relatively low mean discharge rates (0.15 $l\ s^{-1}\ km^2$), with groundwater supporting 50-70% of baseflow (BFI: 0.5 – 0.7; Table 1), High flows could reach up to 0.5 $l\ s^{-1}\ km^2$ (i.e. in spring 2022), which is consistent with elevated groundwater levels during the same winter/spring period (Fig. 2b).

While annual runoff was characteristically low in the rural catchment, with runoff coefficients between 0.1 to 0.26 (Table 1), owing to the high evapotranspiration (up to 80%), they were distinctly higher in the urban catchment (0.8-1.5), generally exceeding the annual precipitation input, and thus reflecting the substantial water subsidy from waste water effluent and inter-basin water transfers. Water abstraction occurs in the catchment to support the city’s water supply, with large amounts of water imported through inter-basin water transfers (from the Spree and Havel) from bank filtration.. Groundwater levels in the rural catchment showed strong seasonality, with relatively shallow depths of around 2.4 mbgl (± 0.3m), and highest levels in April and lowest in autumn. A slight trend in recovery has been noted in 2022 and 2023, following a wetter year (Fig. 2b) As part of the Barnim aquifer, groundwater in the confined urban aquifer was also relatively shallow (2-3 mbgl), showing only minimal seasonal variability but a notable overall decline of summer levels in recent years due to lack of winter recharge.

**Table 1: Summarized annual discharge statistics for the rural and urban stream per water year; total annual precipitation (mm), specificdischarge ($Q_{mean}$), maximum and minimum daily discharge ($Q_{max}$,$Q_{min}$ ) (all in $ls^{-1}km^{2}$) and the 95$^{th}$ and 5$^{th}$ discharge percentiles ($Q_{95}$, $Q_{5}$), as well as total annual runoff ($Q_{RO}$ in mm), annual baseflow index (BFI) and annual runoff coefficient (Q/P). and**

| Water Year | Annual P (mm/wy) | $Q_{max}$ ($ls^{-1}km^{-2}$) | $Q_{min}$ ($ls^{-1}km^{-2}$) | $Q_{mean}$ ($ls^{-1}km^{-2}$) | $Q_{5}$ ($ls^{-1}km^{-2}$) | $Q_{95}$ ($ls^{-1}km^{-2}$) | $Q_{Ro}$ (mm/wy) | *BFI* *(-)* | *Q/P* (-) |
|---|---|---|---|---|---|---|---|---|---|
| **Rural** | | | | | | | | | |
| 2019 | 388.6 | 0.23 | 0 | 0.01 | 0.06 | <0.01 | 44.6 | 0.54 | 0.11 |
| 2020 | 494.4 | 0.34 | 0 | 0.03 | 0.13 | <0.01 | 52.5 | 0.72 | 0.11 |
| 2021 | 534.8 | 0.42 | 0 | 0.06 | 0.18 | <0.01 | 133.7 | 0.58 | 0.25 |
| 2022 | 434.1 | 0.40 | 0 | 0.06 | 0.18 | <0.01 | 113.1 | 0.64 | 0.26 |
| 2023 | 535.2 | 0.48 | 0 | 0.05 | 0.22 | <0.01 | 115.3 | 0.56 | 0.22 |
| **Urban** | | | | | | | | | |
| 2019 | 546.4 | 0.75 | 0.16 | 0.45 | 0.70 | 0.2 | 819.4 | 0.76 | 1.49 |
| 2020 | 527 | 0.64 | 0.01 | 0.27 | 0.54 | 0.07 | 670.5 | 0.72 | 1.27 |
| 2021 | 600.5 | 0.73 | 0.16 | 0.45 | 0.67 | 0.19 | 872.8 | 0.78 | 1.45 |
| 2022 | 461.4 | 0.9 | 0.1 | 0.35 | 0.65 | 0.14 | 447.4 | 0.86 | 0.96 |
| 2023 | 624.0 | 0.76 | 0.02 | 0.33 | 0.62 | 0.165 | 523.0 | 0.82 | 0.84 |

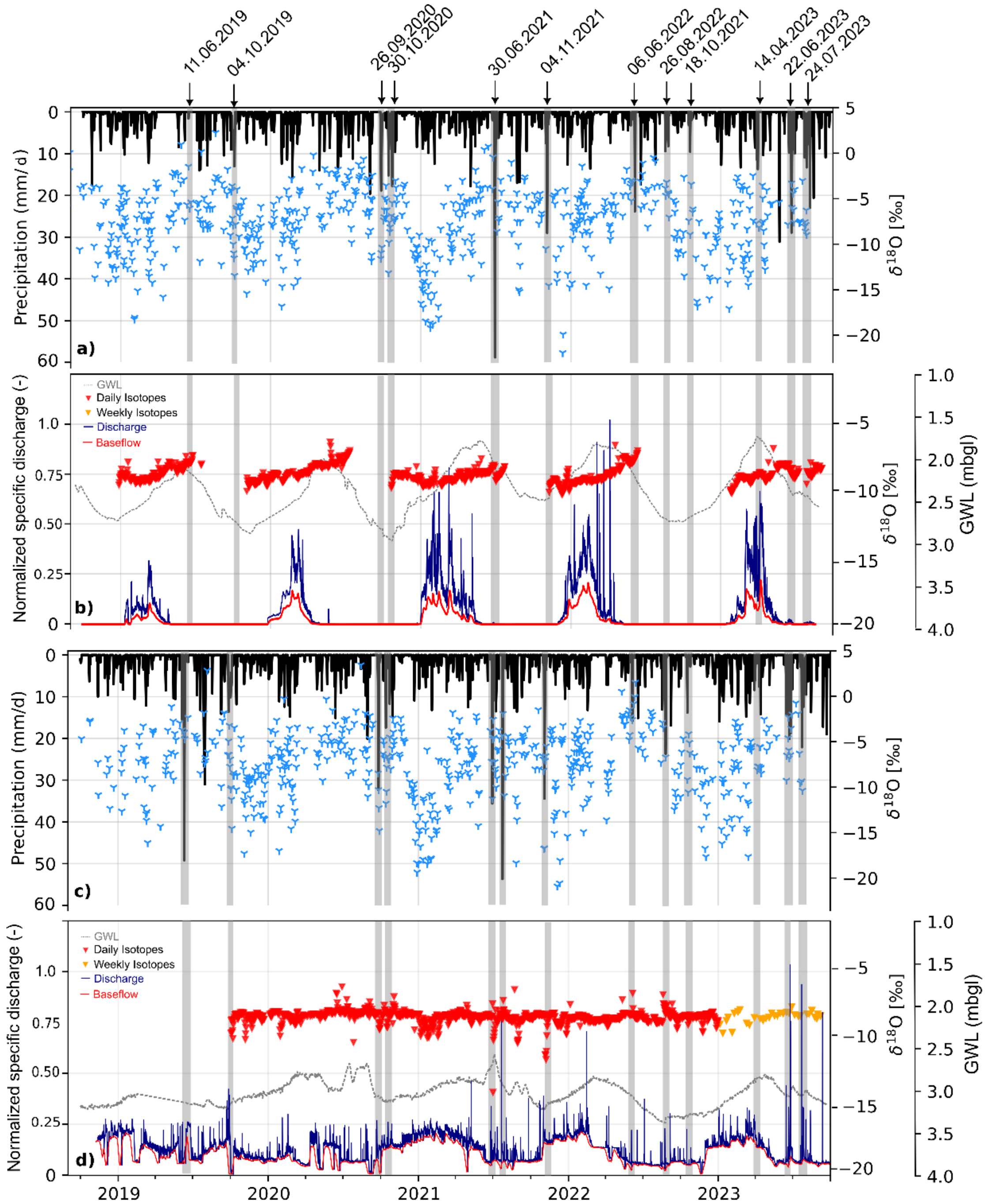



**Figure 2: Timeseries of daily precipitation and precipitation $\delta^{18}O$ isotopes in the a) rural and c) urban stream as well as timeseries of hourly discharge and daily streamflow $\delta^{18}O$ isotopes in the b) rural and d) urban stream. Baseflow is indicated in red as part of the discharge timeseries. Groundwater levels are indicated as meters below ground level (mbgl) in grey. Weekly isotopes in the urban stream (orange) are indicated for 2023 (d). Grey shading indicates selected storm events.**

### 4.2 Differences in seasonal flow regimes

Annual flow duration curves (FDCs) and double- mass curves clearly illustrated the different catchment sensitivities of high and low flow conditions to cumulative annual precipitation and hydroclimate (Fig. 3a, b). In the rural stream, the higher slope of flow duration curves highlighted its intermittent nature and low baseflow component, with most years showing zero flow periods for an average of 60% of the year. The effects of the drought are reflected in the flat FDCs of WY 2018-19 and 2019-20, where low or no flow conditions ($Q_{95}$) were exceeded almost 70% of the year and double- mass curves show the lack of response in streamflow. In contrast, high flow conditions ($Q_5$) were greatest during 2022-23 following a wet spring, and flows continued for almost 65% of the time. The mean flashiness index for the rural stream was relatively low at 0.07 (Table 1). In the rural catchment, the imbalance between annual discharge and precipitation becomes evident, with the majority of annual precipitation lost to the high evaporative demand (~80%), as seen in the low runoff coefficients (Table 2).

In contrast, the effect of urban water management on seasonal streamflow generation become apparent in the urban stream, with cumulative discharge often exceeding cumulative annual precipitation (Fig. 3b) due to additional inflow of water from sources outside the catchment. The higher discharge variability and stronger baseflow component, as well as marked responses to precipitation events as evidenced also in the runoff coefficients (Table 1) were visible in the individual FDCs. Similarly, to the rural stream, the effects of the drought became most visible in evident in WY 2019 and WY 2021, as low flows ($<Q_{95}$) were uncharacteristically high due to additional contributions of increased effluent discharge, resulting in observed higher annual baseflow ($Q_B \sim 0.35$ $ls^{-1}km^2$). The flashy streamflow responses to summer convectional events during WY 2022 and 2023 were also visible in the steeper curves of the medium flow segment (0.2-0.7 flow exceedance probabilities), while the flatter mid-segment in WY 2020 hints at a more sustained groundwater flow contributions (Fig. 3a), corresponding to the increased groundwater levels observed in the Panketal Aquifer (AQ 1.2) in those years (Fig. 2 d).

Relationships between daily isotopic variations (indexed by $\delta^{18}O$) and specific discharge provides further evidence of time-variant source contributions to streamflow in each catchment (Fig. 4b, c). While in the rural stream, $\delta^{18}O$ showed a positive correlation with discharge ($r= 0.41$, $p <0.01$), despite overall modest variations in daily $\delta^{18}O$ (SD= 0.6‰), in the urban stream, $\delta^{18}O$ and discharge were negatively correlated ($r= -0.25$, $p<0.01$) with a clear seasonal distinction between high flows in winter and low flows in summer. The intermittent streamflow regime of the rural stream and marked seasonality in stable water isotopes, illustrated the time-variant contributions from mixed sources (e.g. precipitation and groundwater) during winter and spring flow periods, while in summer, as streamflow starts to decline, the remaining stream water was increasingly subject to evaporative fractionation effects. Particularly as groundwater levels recede and the channel network becomes disconnected, only isolated open water surfaces remain in the streambed, which are affected by evaporative fractionation (Fig. 4c).

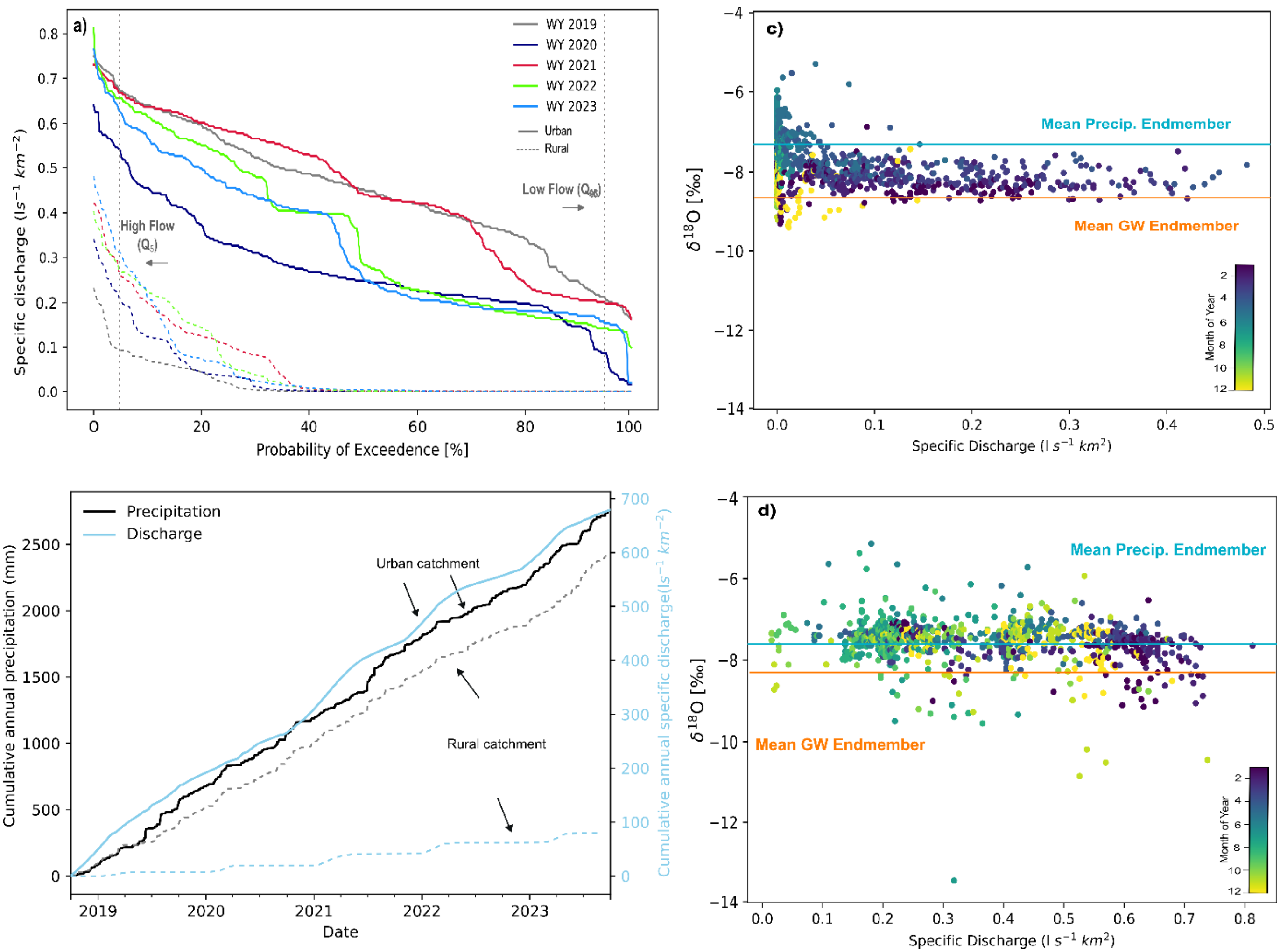



**Figure 3: a) Annual flow duration curves per water year using daily streamflow data for the rural (dotted lines) and urban (solid lines) streams. The stream in the rural catchment shows a characteristic intermittency with much higher likelihood of drying, while the urban catchments clearly perennial with only ephemeral high flows. b) Double-mass curves of cumulative precipitation and specific discharge over the study period for both streams and c) and d) Relationships between daily stream water $\delta^{18}O$ and specific discharge for the rural and urban stream respectively. Horizontal lines indicated mean measured groundwater (red) and mean precipitation (blue) end member $\delta^{18}O$.**

### 4.3 Spatial variability of precipitation and streamflow isotopes

The isotopic signatures of rainfall in both catchments was highly seasonal (rural: mean $\delta^{18}O$ = -6.03‰, SD=3.27‰ and mean $\delta^{2}H$ = -42.12‰, SD=21.59‰; urban: mean $\delta^{18}O$ = -6.09‰, SD=3.15‰ and mean $\delta^{2}H$ = -42.11‰, SD=21.97‰), with rainfall being more depleted in winter and more enriched in summer (Fig.2a, c). Occasionally highly enriched signatures can be seen following larger summer convective storms. The differences in catchment characteristics (runoff, climate) and urban water management were reflected in the daily isotopic signatures of the streams (Fig. 2, 4). While in the rural stream, distinct effects of evaporative fractionation were evident in early summer samples from ponded water in riparian areas and wetlands, as well as near beaver dams after intense storm events (Fig. 4a), the majority of the urban stream samples plotted below the GMWL, indicating minimal influence of evaporation (Fig. 4b). Still, the streamflow signature in the urban stream was more variable and less damped than in the rural stream, indicating occasionally pronounced responses urban storm drains, which is as important as the limited temporal variability (Fig. 4b)

The most notable seasonal differences were evident in early streamflow samples from the rural stream between December and January (mean $\delta^{18}O$ = -8.36‰), which resembled groundwater (mean $\delta^{18}O$ = -8.7‰, SD = 0.18‰; mean $\delta^{2}H$ = -58.9‰, SD=0.9‰), while later in the season the influence of recent precipitation was more pronounced (mean $\delta^{18}O$ = -7.2‰) (Fig. 3a, also see Supplementary Figure S1). Lc-excess in the rural stream frequently showed negative values and a strong seasonality (0.8 to -11‰), a clear sign of the seasonal fractionation from late spring – early summer during and prior to streamflow cessation. Because of the overwhelming influence of effluent, more enriched signals in the stream could be seen in summer (mean $\delta^{18}O$ = -7.4‰), which are likely attributable to an increased inflow of enriched effluent. Because the upper urban catchment also receives water from a north bank tributary which drains forested and wetland areas, the variable inflow of differently fractioned water sources was visible in the lc-excess signal which showed strong variability (-7.9 to 4.3 ‰), and lower lc-excess values estimated in winter - reflecting the inflow of more fractionated sources (e.g. waste water, lake water), and only occasional positive lc-excess values following rain events (Fig. 2d, see also Supplementary Figure S3).

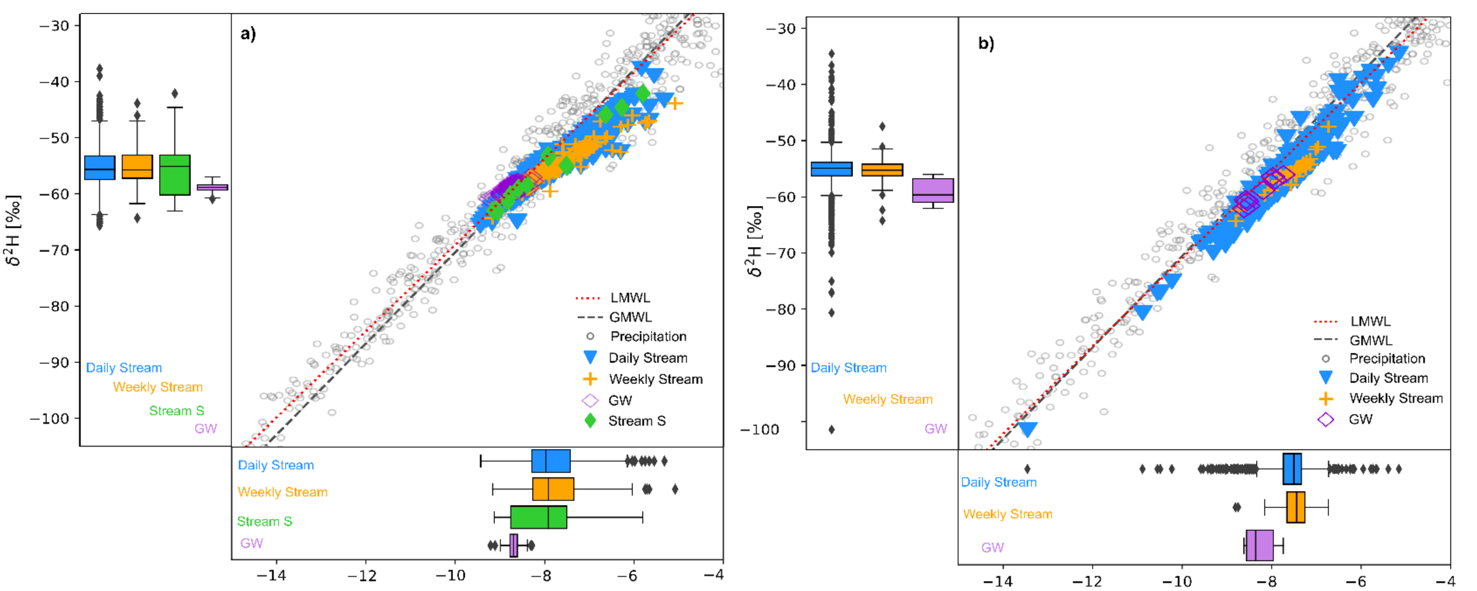



**Figure 4: Dual isotope (centre) and boxplots (left, bottom) showing the isotopic composition of daily precipitation (grey), daily streamflow (blue, weekly streamflow (orange) and groundwater isotopes (purple) in the a) rural and b) the urban catchment. Additional isotope samples (Stream S; green) from isolated pools in the stream are shown for the rural catchment. The Global Meteoric Water Line (GMWL, black) and amount-weighted local meteoric water lines (LMWL, red) from precipitation signals are shown for reference.**

### 4.4 Inferring stream water age and transit times from isotope data

Estimates of water ages and average young water fractions ($F_{yw}$) (in this case % of water younger than 2-3 months) are shown in Figure 5. In the rural catchment, the average young water fraction was ~ 15%, ($p<2.2e-16$, $R^2_{adj}=0.41$, RMSE=0.25) (Fig. 5a), while in the urban stream young water estimations were relatively low, averaging around 7% ($p<2.2e-16$, $R^2_{adj}=0.11$, RMSE=0.28), despite the significant amounts of urban storm drainage (Fig. 5b). For individual water years, the model suggests some interannual variability in the rural catchment between wetter and drier years, with young water contributions ranging from ~ 10 up to ~35% while the generally limited interannual variability in the urban stream, with estimates only between 5% to 15% were a clear indication of the time-variant contributions of younger water that includes urban water sources (incl. waste water discharge and storm drains) as well as precipitation (Table 2). Notably, the different catchment responses to precipitation can be seen in the negative correlation of mean annual young water fractions to annual precipitation (r= 0.67, $p<0.01$) and annual discharge (r= 0.61, $p<0.01$) in the rural catchment, while they were positively correlated with annual precipitation totals (r=0.47, $p<0.01$) and negatively correlated with annual discharge (r =0.49, $p<0.01$) in the urban catchment. This is broadly consistent with the higher impact of precipitation events during the winter and spring season, and greater urban drainage and fast flowpaths, which are activated in response to intense summer precipitation, as seen in the highest $F_{yw}$ (~15 %) in 2023 (Table 2).

Most notably for WY 2019, despite low runoff and overall low precipitation in the rural Brandenburg region, $F_{yw}$ was surprisingly high at around 37%, consistent with a low BFI (~0.5) (Table 1). At the same time for WY 2023, $F_{yw}$ was also estimated to be around 30%, which would be more consistent with increased contributions of precipitation to streamflow, during the wet spring and intense summer convective events. Similarly, the rain-intensive period of early 2023 and subsequent intense summer convective events, also resulted in an increased young water contribution of up to 15% in the urban catchment, while $F_{yw}$ were lowest in WY 2021 and 2022, with estimates of around 5%, and also consistent with higher BFI values (~ 0.8) (Table 1).

**Table 2: Young water fractions over the entire study period and per water year for the rural and urban stream obtained from sine-wave fitting, including coefficient of determination ($R^2$), p-value and residual standard errors over the entire study period.**

| WY | *$\delta^{18}O$ Rural* | | | | *$\delta^{18}O$ Urban* | | | |
|---|---|---|---|---|---|---|---|---|
| | *$F_{yw}$* | *$R^2_{adj}$* | *p-value* | *RSE* | *$F_{yw}$* | *$R^2_{adj}$* | *p-value* | *RSE* |
| **2019** | 0.37 | 0.83 | <2.2e-16 | 0.14 | - | - | - | - |
| **2020** | 0.22 | 0.70 | <2.2e-16 | 0.25 | 0.12 | 0.35 | 3.82e-13 | 0.22 |
| **2021** | 0.09 | 0.58 | <2.2e-16 | 0.17 | 0.05 | 0.05 | 0.0002 | 0.33 |
| **2022** | 0.24 | 0.62 | <2.2e-16 | 0.15 | 0.05 | 0.08 | 3.67e-07 | 0.23 |
| **2023** | 0.28 | 0.37 | <2.2e-16 | 0.23 | 0.15 | 0.21 | 0.0002 | 0.24 |

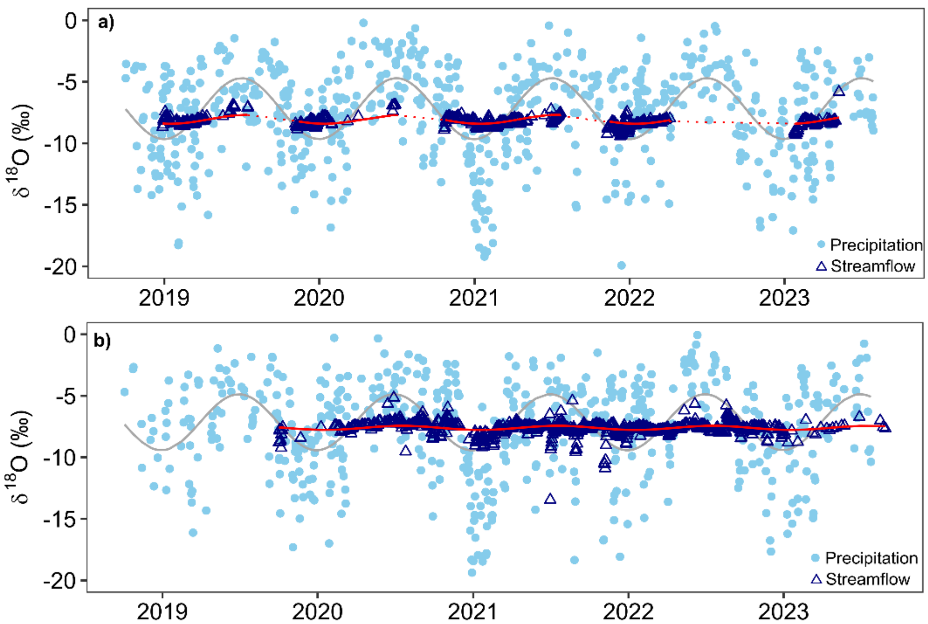
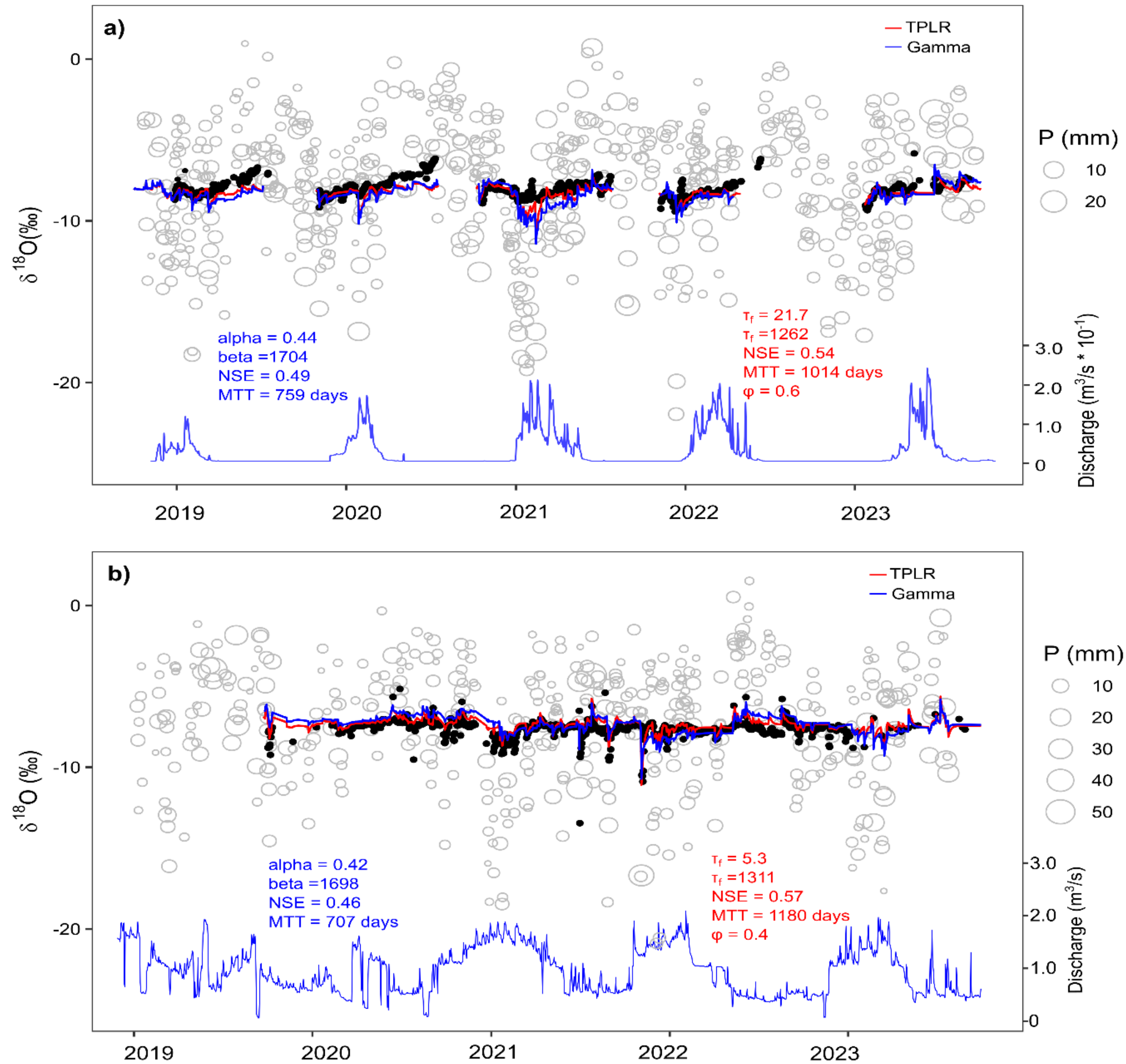



**Figure 5: Hydrologic and isotopic seasonality of precipitation (light blue) and streamflow (dark blue) for the a) rural and b) urban stream, respectively. Sinusoidal cycles (red) were fitted to daily stream isotope data using IRLS (after von Freyberg et al., 2018) for estimates of recent water contributions (<2-3 months). Periods of now flow in the rural catchment are noted with dotted lines in the sinusoidal fitting.**

The TTDs from the TPLR and gamma models were fitted successfully to both streams (Fig. 6, see Supplementary Table S4 for further model results). The TPLR model gave slightly better fits for modelled $\delta^{18}O$ values than the Gamma Model in terms of NS statistics and $R^2$ (Table 4). Estimated MTTs were distinctly higher for the TPLR, ranging between 3 to 4 years for the rural and urban stream, respectively, while MTTs with the gamma model were only estimated to be less than 2 years for both streams. The median fast and slow transit times ($\tau_f$ /$\tau_s$) were 22/1262 days for the rural stream and 5/1311 days for the urban stream. Despite inherent limitations in the detectability of stream water $\delta^{18}O$ signatures older than 5 years, the TPLR model was useful in capturing the rapidly responding flowpaths through the fast component, which drives streamflow flashiness in both streams. The fast flow contributions were also similar to the range of $F_{yw}$ estimates (rural: $\tau_f$ ~ 22d, $F_{yw}$ ~ 15%; urban: $\tau_f$ ~ 5d, $F_{yw}$ ~ 7%). Nevertheless, especially in the rural stream, both models fail to capture the streamflow signatures towards summer, especially during the 2019 period, but appear to work well for the wet periods of 2023. Furthermore, in early 2021, the influence of the more depleted winter precipitation on streamflow was overestimated. The more damped isotopic dynamics of the urban stream were generally well captured in both models, with only the most shortterm responses to the severely depleted high intensity convectional events (i.e. summer 2021) and the occasional inflow of more enriched sources (e.g. effluent), not fully captured.



420 **Figure 6: Fitted transit time distributions expressed as cumulative distribution functions using the TPLR (red) and Gamma model (blue) for the a) rural and b) urban catchment. Precipitation amount (in mm) is denoted by the size of the open circles, while isotopic values of precipitation inputs correspond with the y-axis (‰). Periods of no flow in the urban catchment have been indicated with a gap in the simulations. Daily discharge is shown in pink ($m^3/s$).**

## 5 Discussion

### 5.1 Seasonal and event flow responses

Recent severe drought hazards across central Europe have highlighted the vulnerability of streams to the compound effects of drought on frequency of intermittency and longevity of zero-flow periods (Creutzfeldt et al., 2021; Sarremejane et al., 2022; Tramblay et al., 2021). In Berlin – Brandenburg, the 2018 drought and sustained negative rainfall anomalies in subsequent years, exacerbated a decade-long trend of declining groundwater levels, causing increasing dis-connectivity between groundwater and surface water and raising concerns regarding future water availability, especially in less managed, agricultural areas but also in cities like Berlin (Creutzfeldt et al., 2021; DWD, 2019). Propagation of precipitation deficits into groundwater droughts is particularly detrimental in areas where groundwater is the main contributor to baseflow, as in the case of the rural stream (up to 70%), but also in cities where groundwater and bank filtration are the main sources of water supply, as is the case in Berlin. Consequently, flow regimes in similar drought-sensitive lowland areas are exposed to hydroclimate hazards and landuse change impacts, (Luo et al., 2024; Paredes del Puerto et al., 2024; Van Loon & Laaha, 2015; Wunsch et al., 2022).

Compared to the long-term mean discharge of the rural catchment (0.11 $m^3/s$; 2001-2017, Ying et al., 2024), daily discharge has declined by up to 60% following the 2018 drought. Due to the limited replenishment of moisture during winter in 2018-19, the observed low discharge and longer no-flow periods defined the 2018-19 and subsequent hydrological years. The lack of summer streamflow response to even the largest precipitation events (e.g. June 2021), coincided with observed soil moisture deficits as well as limited groundwater recharge, resulting in limited surface water connectivity and streamflow generation (Landgraf et al., 2022; Ying et al., 2024). This also affected nutrient transport and chemodynamic behaviour, as low flows decreased transport capacity and dilution effects (Wu et al., 2021).

Through daily isotopes the close link to seasonal groundwater and the relative elasticity in streamflow responses timing of precipitation became evident (for more detail see Supplementary Figure S1, S2). The importance of seasonal precipitation in maintaining streamflow connectivity and the groundwater for generating winter streamflow generation could be confirmed through the seasonal isotopic signal. Nevertheless, increasing ET and projected precipitation decreases may compromise the catchments' ability to maintain green and blue water fluxes throughout the year in areas with lower water retention. Especially in agricultural catchments, the dominance of green water fluxes from water intensive crops such as maize and other cereals, can propagate inter-year drought memory effects from reduced blue water fluxes to groundwater recharge and streamflow generation (Ihinegbu & Ogunwumi, 2022; Orth & Seneviratne, 2013), creating uncertainty over the resilience of agricultural production and landuse (Beillouin et al., 2020; McNamara et al., 2024)

The contrasting urban streamflow responses to drought and seasonal precipitation not only illustrated the effects of urban water management but also the impacts of progressive urbanization and land use changes on streamflow dynamics. While baseflow in the urban stream was only slightly lower during drier years, overall discharge was defined by the increased discharge of WWTP effluents, which can regularly account for up to 90% of total discharge (Marx et al., 202). The moderated drought response of streamflow in WY 2019 and 2020, underscored the crucial role of effluents in sustaining baseflows during extreme drought periods (Luthy et al., 2015) and the impact of baseflow regulation and runoff following storm events. The influence of urban water management on rapid flowpaths, inter-basin water transfers and streamflow generation defines many urban systems, with urban groundwater recharge and baseflow being shaped by urban infrastructure and development (Bhaskar et al., 2016). At the same time natural hazards such as floods and droughts, increasingly challenge not only water availability resilience, but managers' ability to tailor resource management approaches to changes in local climate conditions (Hale et al., 2016). The broader interaction of the urban stream network with the surrounding environment and different urban water sources (i.e. wetlands, floodplains, built environment) clearly shape the downstream propagation of a flood peak through the urban catchment (Johnson et al., 2022; Oswald et al., 2023). In times of increased drought and flood hazards, urban catchemtn responses to natural hazards need to be strengthened even more by preserving and enhancing wetland areas in peri-urban and rural areas, by allowing infiltration and storage of excess surface water through and optimizing urban water management within the built environment through more nature-based solutions (Davis & Naumann, 2017; Green et al., 2021).

### 5.2 Transit times and water ages

Estimates of young water fractions, MTTs and TTDs provided a rough approximation of complex age and transit time distributions, due to limitations of the $F_{yw}$ approach (Kirchner, 2016; Seeger & Weiler, 2014) and the inability of lumped convolution models to account for evaporative fractionation and mixed water sources (McGuire & McDonnell, 2006). The relatively low young water fractions of ~ 7 % in the urban stream, despite significant urban runoff, further highlight the complicating issue of constraining water ages in urban streams, where an overwhelming dominance of mixed urban sources weakens the influence and differentiation of recent precipitation (Bonneau et al., 2018; Soulsby et al., 2014). This is a particular impediment in closed urban water management systems, such as Berlin, where there is overlap in the isotopic composition of surface waters, groundwater and wastewater contributions (Massman et al., 2008).

In the rural catchment, the longer, continuous timeseries of daily stream and precipitation isotopes over 5 years allowed for a more systematic filtering of fractionated signals of early summer streamflow, as previous estimates from only <2 years of daily data indicated high levels of uncertainty in MTT and $F_{yw}$ estimates (Kleine et al., 2021). Our results not only revealed inter-annual differences in young water contributions related to annual precipitation, but also improved young water fraction estimates for different years with lower values between 0.1 and 0.37 (mean 0.15) compared to previous estimates (mean 0.37, Kleine et al., 2021). These values are more consistent with estimates from other lowland catchments in central Germany (Lutz et al., 2018) and only slightly below the median young water fraction of 26% for European catchments (Jasechko et al., 2016). The larger $F_{yw}$ in the rural stream compared to the urban, may be attributed to the smaller catchment size, whereby the predominantly agricultural landuse (drainage and compaction) likely facilitates rapid runoff responses via the artificial drainage pipes and channels, thus driving the release of young water contributions to the stream from relatively small areas in the catchment (Lutz et al., 2018; Von Freyberg et al., 2018). However, the higher Fyw in the rural stream during the drought may well also be an artefact of the method, as the data collection period was also limited due to the stream's intermittency.

Fractionation effects or difficulties in constraining water sources, invariably increase the uncertainties associated with the non-stationarity of TTDs over different time scales in response to hydroclimatic conditions (Hrachowitz et al., 2010). As a result, particularly during the drought this may produce a less robust seasonal cycle coefficient for each individual hydrological period, as evidenced by the overall higher uncertainty attached to $F_{yw}$ estimates in drier years (Table 2).

Regarding TTDs and estimated MTTs, our estimates for the rural catchment between 2-4 years (730 days – 1622 days), with the higher estimate from the TPLR model, are considered more likely and within the range of tritium estimates of water age of local groundwater of around 5 years (Ying et al., 2024) and stream water ages of ~7 years (Smith et al., 2021). Still, similar to previously noted limitations in MTT estimations in such lowland areas (Tetzlaff et al., 2011), the pervasive influence of older groundwater in the rural stream likely weakens the influence of precipitation intensity on MTTs, as evidenced by the occasional overprediction of isotopic response in the stream during early flow periods in winter (e.g. early 2021) (Fig. 6a). The better performance of the TPLR compared to the Gamma model in the urban catchemnt underlines the particular suitability of the two-reservoir model in an urban environment, where runoff generation follows a more binary distribution (Soulsby et al., 2014). Nevertheless, a caveat to these conclusions is again the extreme damping of isotope signatures in the urban stream due to the overwhelming influence of effluent, which limits the ability of both models to constrain the hydro-demographics of different water sources (e.g. groundwater, precipitation etc.) in a conclusive manner (Sprenger et al., 2019)

**5.3 Wider Implications**

With the projected changes for Europe suggesting a greater seasonal divergence of lower and higher precipitation amounts in summer and winter respectively, the seasonal synchronicity of groundwater-baseflow responses we found in our study may widen even further, as the longer time scales for groundwater recovery after extended precipitation deficits usually lags behind several years despite the return to wetter conditions (Hellwig et al., 2020; Smith et al., 2022). The effects of such hydroclimate changes may be even more severe in urban areas, where urbanization has directly been linked with the intensification of extreme rainfall (Singh et al., 2020). Urban "plumbing", a high level of imperviousness and lack of urban green space, can reduce recharge and groundwater storage to the point where they are unable to buffer natural climate variability, requiring even more extensive water management (Bonneau et al., 2018; Marx et al., 2021; Oswald et al., 2023).

The differential impacts of drought and extreme events on streamflow generation are conceptualized in Figure 7, illustrating different seasonal controls on flow permanence and magnitude of associated hydrological processes in contrasting environments. During drought periods with reduced rainfall and high ET, severe soil moisture deficits and low recharge increase the disconnection between surface and groundwater, leading to long-lasting groundwater droughts, reduced agricultural production and increased streamflow intermittency. Conversely, extensive water management can moderate drought impacts through the supply of waste water effluent or other continuous sources, but which in turn can lead to increased nutrient concentration and negative impacts on freshwater quality and microbial diversity (Numberger et al., 2022; Warter et al., 2024). In the context of natural hazards and climate change, more frequent and intense rain events and flooding, can challenge urban infrastructure and inner-city drainage systems, as greater quantities of water also increase organic pollutant loads and threaten aquatic habitats and biodiversity (Creutzfeldt et al., 2021; Haase et al., 2023).

Nevertheless, there is a need to reconcile different water use requirements and management approaches, that preserve the hydrological and ecological integrity, is inherently more difficult with insufficient information on hydrological process dynamics and landuse influences. Especially in urban systems, which encompass engineered management systems as well as natural systems, a conjoint understanding of basic hydrologic processes is urgently needed (Gessner et al., 2014). Emerging questions whether declining baseflows should be seasonally augmented by treated wastewater or inter-basin water transfer are not only a matter of social choice but also require consideration of the hydrological, ecological and chemical impacts of increased baseflow contributions on instream habitat, biodiversity targets and water quality (Numberger et al., 2022; Warter et al., 2024). Designing flow regimes to achieve specific ecological and hydrological restoration goals may become the norm in modified and managed rivers where a return to natural, pre-anthropic conditions is no longer feasible and a maintenance of certain flow levels is governed by different targets (Acreman et al., 2014; Stewardson et al., 2017). As such, identifying thresholds at which important hydrological changes occur, requires a thorough understanding of how water moves through catchment systems, for which tracer studies are an invaluable tool to understand water resource systems and their vulnerability to hydroclimate changes (Ehleringer et al., 2016) .

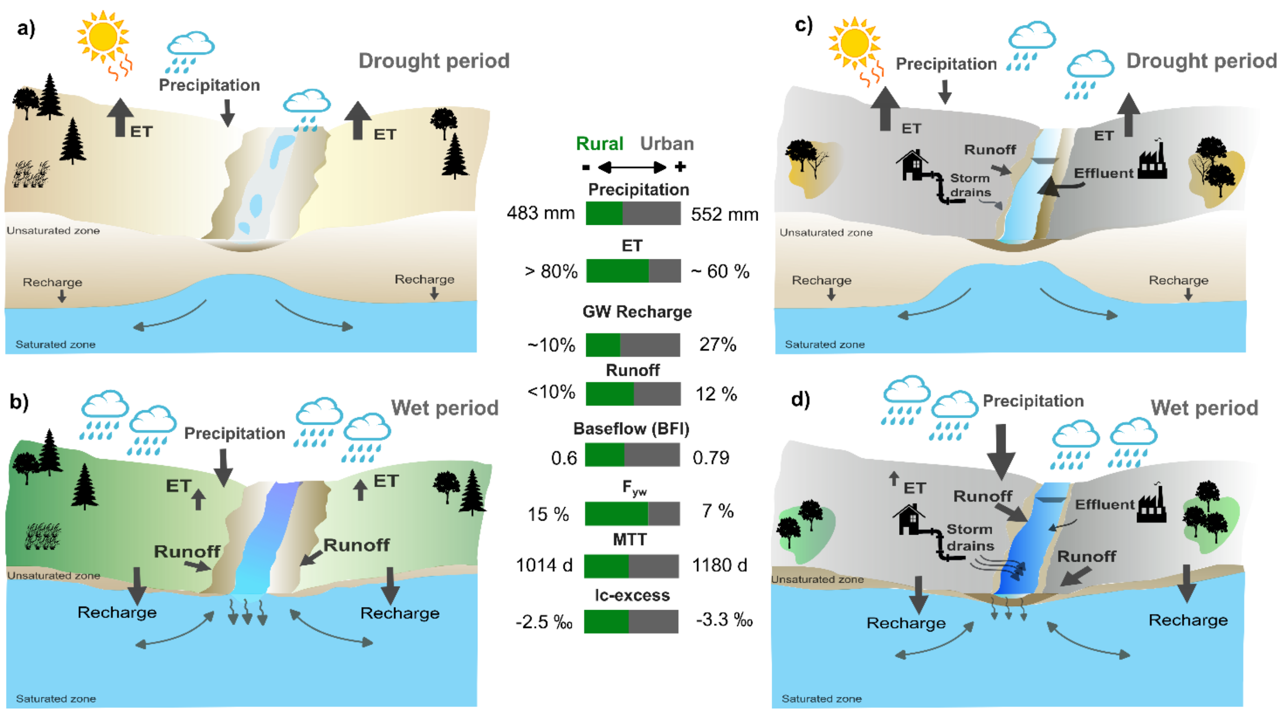



**Figure 7: Conceptual summary of dynamics in hydrological processes and metrics in the rural agricultural and urban catchment during a) and c) drought periods and b) and d) wet periods. Bars in the middle represent general magnitude of fluxes in each catchment in comparison.**

## 6. Conclusion

Inter-catchment comparisons between urban and rural stream systems using multi-year tracer-based assessments are still rare but – as demonstrated here - very insightful and much needed. From the strong responses to drought and an increasing hydroclimatic variability, landuse (i.e. drainage, vegetation, wetland restoration) showed to be important in water partitioning of groundwater-surface water interactions and streamflow generation in these anthropogenically impacted streams Our results not only highlighted the continued importance and value of high-resolution long-term tracer data to develop a synoptic understanding of the principal hydrologic mechanisms by which flow regimes directly and indirectly respond to climate perturbations, especially in understudied urban environments. They provided immediate evidence of contrasting catchment functioning and streamflow generation in different geographical settings, which will be useful for the identification of future environmental flow assessments in similar urban and lowland catchments where a return to pre-anthropic natural conditions may be no longer attainable. The limitations in the detection of water ages and source contributions in the urban stream highlighted the need for long-term tracer-based assessments of urban hydrological fluxes, using either stable isotopes or other reactive tracers, to better constrain current and future inter- and intra-annual variability and to mitigate the effects of hazards such as floods and droughts. Although the challenges associated with sustained monitoring often limit long-term observations over broader scales, the benefits and value of long-term observations are crucial for hydrologists, ecologists and urban planners and local stakeholders interested in protecting and maintaining ecosystem function and manage future water resources in the most sustainable and integrated way that reduces environmental impact and economic costs.

**Data Availability Statement:**

The data used in this study is available on the Leibniz Freshwater and Environmental Database (FRED) under the following doi: 10.18728/igb-fred-865.0. Data will be made available upon publication. Public discharge data is available from https://wasserportal.berlin.de/start.php. Climate Data from the German Weather Service (DWD) can be downloaded from https://www.dwd.de/DE/leistungen/cdc_portal/cdc_portal.html?nn=17626.

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The authors declare that they have no conflict of interest.

**CRediT author contribution statement**

Conceptualization: DT, CS, MMW; Methodology: MMW, CM, DT, CS; Investigation: CM, MMW; Formal Analysis: MMW; Data Curation: MMW, CM; Writing-Original Draft: MMW; Writing – Review & Editing: DT, CS; Visualization: MMW; Supervision: DT, CS; Funding acquisition: DT, CS

## References


Acreman, M., Arthington, A. H., Colloff, M. J., Couch, C., Crossman, N. D., Dyer, F., Overton, I., Pollino, C. A., Stewardson, M. J., & Young, W. (2014). Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. *Frontiers in Ecology and the Environment*, *12*(8), 466–473. https://doi.org/10.1890/130134

Arthington, A. H., Bunn, S. E., Poff, N. L. R., & Naiman, R. J. (2006). The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications*, *16*(4), 1311–1318. https://doi.org/10.1890/1051-0761(2006)016[1311:TCOPEF]2.0.CO;2

Baker, D. B., Richards, R. P., Loftus, T. T., & Kramer, J. W. (2004). A new flashiness index: Characteristics and applications to Midwestern rivers and streams. *Journal of the American Water Resources Association*, *40*(2), 503–522. https://doi.org/10.1111/j.1752-1688.2004.tb01046.x

Beillouin, D., Schauberger, B., Bastos, A., Ciais, P., & Makowski, D. (2020). Impact of extreme weather conditions on European crop production in 2018: Random forest - Yield anomalies. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *375*(1810). https://doi.org/10.1098/rstb.2019.0510

Bhaskar, A. S., Beesley, L., Burns, M. J., Fletcher, T. D., Hamel, P., Oldham, C. E., & Roy, A. H. (2016). Will it rise or will it fall? Managing the complex effects of urbanization on base flow. *Freshwater Science*, *35*(1), 293–310. https://doi.org/10.1086/685084

Birkel, C., Geris, J., Molina, M. J., Mendez, C., Arce, R., Dick, J., Tetzlaff, D., & Soulsby, C. (2016). Hydroclimatic controls on non-stationary stream water ages in humid tropical catchments. *Journal of Hydrology*, *542*, 231–240. https://doi.org/10.1016/j.jhydrol.2016.09.006

Bonneau, J., Burns, M. J., Fletcher, T. D., Witt, R., Drysdale, R. N., & Costelloe, J. F. (2018). The impact of urbanization on subsurface flow paths – A paired-catchment isotopic study. *Journal of Hydrology*, *561*(January), 413–426. https://doi.org/10.1016/j.jhydrol.2018.04.022

Creutzfeldt, B., Pohle, I., Zeilfelder, S., Birner, J., Köhler, A., Von Seggern, D., & Rehfeld-Klein, M. (2021). *Die Niedrigwasserjahre 2018, 2019 und 2020 - Analysen und Auswirkungen für das Land Berlin*. www.berlin.de/sen/uvk

Davis, M., & Naumann, S. (2017). *Making the Case for Sustainable Urban Drainage Systems as a Nature-Based Solution to Urban Flooding*. https://doi.org/10.1007/978-3-319-56091-5_8

Döll, P., & Schmied, H. M. (2012). How is the impact of climate change on river flow regimes related to the impact on mean annual runoff? A global-scale analysis. *Environmental Research Letters*, *7*(1). https://doi.org/10.1088/1748-9326/7/1/014037

DWD. (2019). Klimareport Brandenburg. *Deutscher Wetterdienst*, *1*, 40.

DWD, D. W. D. (2023). *No Title*. Climate Data Center (CDC). https://opendata.dwd.de/climate_environment/CDC/

Ehleringer, J. R., Barnette, J. E., Jameel, Y., Tipple, B. J., Bowen, J., Ehleringer, J. R., Barnette, J. E., Jameel, Y., & Tipple, B. J. (2016). *Urban water – a new frontier in isotope hydrology*. *6016*(May). https://doi.org/10.1080/10256016.2016.1171217

Gessner, M. O., Hinkelmann, R., Nützmann, G., Jekel, M., Singer, G., Lewandowski, J., Nehls, T., & Barjenbruch, M. (2014). Urban water interfaces. *Journal of Hydrology*, *514*, 226–232. https://doi.org/10.1016/j.jhydrol.2014.04.021

Green, D., Donnell, E. O., Slater, L., Chan, F. K. S., Thorne, C., Li, L., Boothroyd, R. J., & Stirling, R. (2021). *Green infrastructure : The future of urban flood risk management ? October 2020*, 1–18. https://doi.org/10.1002/wat2.1560

Haase, P., Bowler, D. E., Baker, N. J., Bonada, N., Domisch, S., Garcia Marquez, J. R., Heino, J., Hering, D., Jähnig, S. C., Schmidt-Kloiber, A., Stubbington, R., Altermatt, F., Álvarez-Cabria, M., Amatulli, G., Angeler, D. G., Archambaud-Suard, G., Jorrín, I. A., Aspin, T., Azpiroz, I., … Welti, E. A. R. (2023). The recovery of European freshwater biodiversity has come to a halt. *Nature*, *620*(7974), 582–588. https://doi.org/10.1038/s41586-023-06400-1

Hale, R. L., Scoggins, M., Smucker, N. J., & Suchy, A. (2016). Effects of climate on the expression of the urban stream syndrome. *Freshwater Science*, *35*(1), 421–428. https://doi.org/10.1086/684594

Hellwig, J., de Graaf, I. E. M., Weiler, M., & Stahl, K. (2020). Large-Scale Assessment of Delayed Groundwater Responses to Drought. *Water Resources Research*, *56*(2). https://doi.org/10.1029/2019WR025441

Hrachowitz, M., Soulsby, C., Tetzlaff, D., Malcolm, I. A., & Schoups, G. (2010). Gamma distribution models for transit time estimation in catchments: Physical interpretation of parameters and implications for time-variant transit time assessment. *Water Resources Research*, *46*(10). https://doi.org/10.1029/2010WR009148

Ihinegbu, C., & Ogunwumi, T. (2022). Multi-criteria modelling of drought: a study of Brandenburg Federal State, Germany. *Modeling Earth Systems and Environment*, *8*(2), 2035–2049. https://doi.org/10.1007/s40808-021-01197-2

Jasechko, S. (2019). Global Isotope Hydrogeology―Review. *Reviews of Geophysics*, *57*(3), 835–965. https://doi.org/10.1029/2018RG000627

Jasechko, S., Kirchner, J. W., Welker, J. M., & McDonnell, J. J. (2016). Substantial proportion of global streamflow less than three months old. *Nature Geoscience*, *9*(2), 126–129. https://doi.org/10.1038/ngeo2636

Johnson, B. G., Morris, C. S., Mase, H. L., Whitehouse, P. S., & Paradise, C. J. (2022). Seasonal flashiness and high frequency discharge events in headwater streams in the North Carolina Piedmont (United States). *Hydrological Processes*, *36*(3), 1–19. https://doi.org/10.1002/hyp.14550

Kendall, C., & McDonnell, J. J. (1998). *Isotope Tracers in Catchment Hydrology*. Elsevier.

Kirchner, J. W. (2016). Aggregation in environmental systems-Part 1: Seasonal tracer cycles quantify young water fractions, but not mean transit times, in spatially heterogeneous catchments. *Hydrology and Earth System Sciences*, *20*(1), 279–297. https://doi.org/10.5194/hess-20-279-2016

Kleine, L., Smith, A., & Goldhammer, T. (2021). Using isotopes to understand landscape-scale connectivity in a groundwater-dominated , lowland catchment under drought conditions. *Hydrological Processes*, *September 2020*, 1–20. https://doi.org/10.1002/hyp.14197

Kleine, L., Tetzlaff, D., Smith, A., Dubbert, M., & Soulsby, C. (2021). Modelling ecohydrological feedbacks in forest and grassland plots under a prolonged drought anomaly in Central Europe 2018–2020. *Hydrological Processes*, *35*(8), 1–20. https://doi.org/10.1002/hyp.14325

Kleine, L., Tetzlaff, D., Smith, A., Wang, H., & Soulsby, C. (2020). Using water stable isotopes to understand evaporation, moisture stress, and re-wetting in catchment forest and grassland soils of the summer drought of 2018. *Hydrology and Earth System Sciences*, *24*(7), 3737–3752. https://doi.org/10.5194/hess-24-3737-2020

Kuhlemann, L. M., Tetzlaff, D., & Soulsby, C. (2020). Urban water systems under climate stress: An isotopic perspective from Berlin, Germany. *Hydrological Processes*, *34*(18), 3758–3776. https://doi.org/10.1002/hyp.13850

Landgraf, J., Tetzlaff, D., Freymüller, J., & Soulsby, C. (2022). *Using stable water isotopes to understand ecohydrological partitioning under contrasting land uses in a drought-sensitive rural , lowland catchment*.

*December*, 1–20. https://doi.org/10.1002/hyp.14779

Landwehr, J. M., & Coplen, T. . (2006). Line-conditioned excess: a new method for characterizing stable hydrogen and oxygen isotope ratios in hydrologic systems (IAEA-CN-118/56). In *Isotopes in Environmental Studies* (IAEA-CSP-2, Issue October, pp. 132–135). IAEA.

Lobanova, A., Liersch, S., Nunes, J. P., Didovets, I., Stagl, J., Huang, S., Koch, H., Rivas López, M. del R., Maule, C. F., Hattermann, F., & Krysanova, V. (2018). Hydrological impacts of moderate and high-end climate change across European river basins. *Journal of Hydrology: Regional Studies*, *18*(May), 15–30. https://doi.org/10.1016/j.ejrh.2018.05.003

Luo, S., Tetzlaff, D., Smith, A., & Soulsby, C. (2024). Assessing impacts of alternative land management strategies on water partitioning, storage and age in drought-sensitive catchments using tracer-aided ecohydrological models. *Hydrological Processes*. https://doi.org/In Press

Luthy, R. G., Sedlak, D. L., Plumlee, M. H., Austin, D., & Resh, V. H. (2015). Wastewater-effluent-dominated streams as ecosystem-management tools in a drier climate. *Frontiers in Ecology and the Environment*, *13*(9), 477–485. https://doi.org/10.1890/150038

Lutz, S. R., Krieg, R., Müller, C., Zink, M., Knöller, K., Samaniego, L., & Merz, R. (2018). Spatial Patterns of Water Age: Using Young Water Fractions to Improve the Characterization of Transit Times in Contrasting Catchments. *Water Resources Research*, *54*(7), 4767–4784. https://doi.org/10.1029/2017WR022216

Marx, C., Tetzlaff, D., Hinkelmann, R., & Soulsby, C. (2021). Isotope hydrology and water sources in a heavily urbanized stream. *Hydrological Processes*, *35*(10), 1–20. https://doi.org/10.1002/hyp.14377

Marx, C., Tetzlaff, D., Hinkelmann, R., & Soulsby, C. (2023). Effects of 66 years of water management and hydroclimatic change on the urban hydrology and water quality of the Panke catchment, Berlin, Germany. *Science of the Total Environment*, *900*(July), 165764. https://doi.org/10.1016/j.scitotenv.2023.165764

Massman, G., Sültenfuß, J., Dünnbier, U., Knappe, A., Taute, T., & Pekdeger, A. (2008). Investigation of groundwater residence times during bank filtration in Berlin: a multi-tracer approach. *Hydrological Processes*, *22*, 788–801. https://doi.org/10.1002/hyp.6649

McGuire, K. J., & McDonnell, J. J. (2006). A review and evaluation of catchment transit time modeling. *Journal of Hydrology*, *330*(3–4), 543–563. https://doi.org/10.1016/j.jhydrol.2006.04.020

McNamara, I., Flörke, M., Uschan, T., Baez-Villanueva, O. M., & Herrmann, F. (2024). Estimates of irrigation requirements throughout Germany under varying climatic conditions. *Agricultural Water Management*, *291*(December 2023), 108641. https://doi.org/10.1016/j.agwat.2023.108641

Nash, J. E., & Sutcliffe, J. V. (1970). River flow forecasting through conceptual models part I — A discussion of principles. *Journal of Hydrology*, *10*(3), 282–290. https://doi.org/https://doi.org/10.1016/0022-1694(70)90255-6

Nathan, R. J., & McMahon, T. A. (1990). Evaluation of automated techniques for base flow and recession analyses. *Water Resources Research*, *26*(7), 1465–1473. https://doi.org/https://doi.org/10.1029/WR026i007p01465

Numberger, D., Zoccarato, L., Woodhouse, J., Ganzert, L., Sauer, S., Márquez, J. R. G., Domisch, S., Grossart, H. P., & Greenwood, A. D. (2022). Urbanization promotes specific bacteria in freshwater microbiomes including potential pathogens. *Science of the Total Environment*, *845*(July). https://doi.org/10.1016/j.scitotenv.2022.157321

O'Briain, R. (2019). Climate change and European rivers: An eco-hydromorphological perspective. *Ecohydrology*, *12*(5). https://doi.org/10.1002/eco.2099

Olden, J. D., & Poff, N. L. (2003). Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications*, *19*(2), 101–121. https://doi.org/10.1002/rra.700

Orth, R., & Seneviratne, S. I. (2013). Propagation of soil moisture memory to streamflow and evapotranspiration in Europe. *Hydrology and Earth System Sciences*, *17*(10), 3895–3911. https://doi.org/10.5194/hess-17-3895-2013

Oswald, C. J., Kelleher, C., Ledford, S. H., Hopkins, K. G., Sytsma, A., Tetzlaff, D., Toran, L., & Voter, C. (2023). Integrating urban water fluxes and moving beyond impervious surface cover: A review. *Journal of Hydrology*, *618*, 129188. https://doi.org/https://doi.org/10.1016/j.jhydrol.2023.129188

Paredes del Puerto, J. M., Sathicq, M. B., Altieri, P., Nicolosi Gelis, M. M., Paracampo, A., Pazos, R. S., Tarda, A. S., Gómez, N., & Colautti, D. (2024). Extreme drought conditions interact with urbanisation, affecting hydrological regimes and water quality in temperate lowland streams. *Aquatic Sciences*, *86*(1), 1–14. https://doi.org/10.1007/s00027-023-01031-0

Paton, E., Vogel, J., Kluge, B., & Nehls, T. (2021). Ausmaß, Trend und Extreme von Dürren im urbanen Raum. *Hydrologie Und Wasserbewirtschaftung*, *65*. https://doi.org/10.5675/HyWa

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E., & Stromberg, J. C. (1997). The Natural Flow Regime. *BioScience*, *47*(11), 769–784. https://doi.org/10.2307/1313099

Poff, N. L., & Zimmerman, J. K. H. (2010). Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. *Freshwater Biology*, *55*(1), 194–205. https://doi.org/10.1111/j.1365-2427.2009.02272.x

Sarremejane, R., Messager, M. L., & Datry, T. (2022). Drought in intermittent river and ephemeral stream networks. *Ecohydrology*, *15*(5). https://doi.org/10.1002/eco.2390

Seeger, S., & Weiler, M. (2014). Reevaluation of transit time distributions, mean transit times and their relation to catchment topography. *Hydrology and Earth System Sciences*, *18*(12), 4751–4771. https://doi.org/10.5194/hess-18-4751-2014

SenUVK. (2009). *Panke 2015: Ein Bach wird naturnah*.

SenUVK. (2023). *Wasserportal Berlin*. https://wasserportal.berlin.de/stationen_start.php

Singh, J., Karmakar, S., Paimazumder, D., Ghosh, S., & Niyogi, D. (2020). Urbanization alters rainfall extremes over the contiguous United States. *Environmental Research Letters*, *15*(7). https://doi.org/10.1088/1748-9326/ab8980

Smith, A. A., Tetzlaff, D., Maneta, M., & Soulsby, C. (2022). Critical Zone Response Times and Water Age Relationships Under Variable Catchment Wetness States: Insights Using a Tracer-Aided Ecohydrological Model. *Water Resources Research*, *58*(4). https://doi.org/10.1029/2021WR030584

Smith, A., Tetzlaff, D., Gelbrecht, J., Kleine, L., & Soulsby, C. (2020). Riparian wetland rehabilitation and beaver re-colonization impacts on hydrological processes and water quality in a lowland agricultural catchment. *Science of The Total Environment*, *699*, 134302. https://doi.org/https://doi.org/10.1016/j.scitotenv.2019.134302

Smith, A., Tetzlaff, D., Gelbrecht, J., L., K., & Soulsby, C. (2020). Riparian wetland rehabibiltation and beaver re-colonisation impacts on hydrological processes and water quality in a lowland agricultural catchment. *Science of the Total Environment*, *699*.

Smith, A., Tetzlaff, D., Kleine, L., Maneta, M., & Soulsby, C. (2021). Quantifying the effects of land use and model scale on water partitioning and water ages using tracer-aided ecohydrological models. *Hydrology and Earth System Sciences*, *25*, 2239–2259. https://doi.org/10.5194/hess-25-2239-2021

Soulsby, C., Birkel, C., Geris, J., & Tetzlaff, D. (2015). Spatial aggregation of time-variant stream water ages in urbanizing catchments. *Hydrological Processes*, *29*(13), 3038–3050. https://doi.org/10.1002/hyp.10500

Soulsby, C., Birkel, C., & Tetzlaff, D. (2014). Assessing urbanization impacts on catchment transit times. *Geophysical Research Letters*. https://doi.org/10.1002/2013GL058716.Received

Sprenger, M., Stumpp, C., Weiler, M., Aeschbach, W., Allen, S. T., Benettin, P., Dubbert, M., Hartmann, A., Hrachowitz, M., Kirchner, J. W., McDonnell, J. J., Orlowski, N., Penna, D., Pfahl, S., Rinderer, M., Rodriguez, N., Schmidt, M., & Werner, C. (2019). The Demographics of Water: A Review of Water Ages in the Critical Zone. *Reviews of Geophysics*, *57*(3), 800–834. https://doi.org/10.1029/2018RG000633

Stevenson, J. L., Geris, J., Birkel, C., Tetzlaff, D., & Soulsby, C. (2022). Assessing land use influences on isotopic variability and stream water ages in urbanising rural catchments. *Isotopes in Environmental and Health Studies*, *58*(3), 277–300. https://doi.org/10.1080/10256016.2022.2070615

Stewardson, M. J., Acreman, M., Costelloe, J. F., Fletcher, T. D., Fowler, K. J. A., Horne, A. C., Liu, G., McClain, M. E., & Peel, M. C. (2017). Chapter 3 - Understanding Hydrological Alteration. In A. C. Horne, J. A. Webb, M. J. Stewardson, B. Richter, & M. Acreman (Eds.), *Water for the Environment* (pp. 37–64). Academic Press. https://doi.org/https://doi.org/10.1016/B978-0-12-803907-6.00003-6

Stewart, M. K., Morgenstern, U., & McDonnell, J. J. (2010). Truncation of stream residence time: How the use of stable isotopes has skewed our concept of streamwater age and origin. *Hydrological Processes*, *24*(12), 1646–1659. https://doi.org/10.1002/hyp.7576

Tang, W., & Carey, S. K. (2017). HydRun: A MATLAB toolbox for rainfall–runoff analysis. *Hydrological Processes*, *31*(15), 2670–2682. https://doi.org/10.1002/hyp.11185

Tetzlaff, D., Buttle, J., Carey, S. K., Mcguire, K., Laudon, H., & Soulsby, C. (2015). Tracer-based assessment of flow paths, storage and runoff generation in northern catchments: A review. *Hydrological Processes*, *29*(16), 3475–3490. https://doi.org/10.1002/hyp.10412

Tetzlaff, D., Grottker, M., & Leibundgut, C. (2005). Hydrological criteria to assess changes of flow dynamic in urban impacted catchments. *Physics and Chemistry of the Earth*, *30*(6-7 SPEC. ISS.), 426–431. https://doi.org/10.1016/j.pce.2005.06.008

Tetzlaff, D., Smith, A., Kleine, L., Dubbert, D., & Freymuller, J. (2023). Integrated ecohydrological hydrometric and stable water isotope data of a drought-sensitive mixed land use lowland catchment. *Earth System Science Data*, *15*, 1543–1554. https://doi.org/doi.org/10.5194/essd-15-1543-2023

Tetzlaff, D., Soulsby, C., Hrachowitz, M., & Speed, M. (2011). Relative influence of upland and lowland headwaters on the isotope hydrology and transit times of larger catchments. *Journal of Hydrology*, *400*(3–4), 438–447. https://doi.org/10.1016/j.jhydrol.2011.01.053

Tonkin, J. D., Olden, J. D., Merritt, D. M., Reynolds, L. V., Rogosch, J. S., & Lytle, D. A. (2021). Designing flow regimes to support entire river ecosystems. *Frontiers in Ecology and the Environment*, *19*(6), 326–333. https://doi.org/10.1002/fee.2348

Tramblay, Y., Rutkowska, A., Sauquet, E., Sefton, C., Laaha, G., Osuch, M., Albuquerque, T., Alves, M. H., Banasik, K., Beaufort, A., Brocca, L., Camici, S., Csabai, Z., Dakhlaoui, H., DeGirolamo, A. M., Dörflinger, G., Gallart, F., Gauster, T., Hanich, L., … Datry, T. (2021). Trends in flow intermittence for European rivers. *Hydrological Sciences Journal*, *66*(1), 37–49. https://doi.org/10.1080/02626667.2020.1849708

Van Loon, A. F., & Laaha, G. (2015). Hydrological drought severity explained by climate and catchment characteristics. *Journal of Hydrology*, *526*, 3–14. https://doi.org/10.1016/j.jhydrol.2014.10.059

Von Freyberg, J., Allen, S. T., Seeger, S., Weiler, M., & Kirchner, J. W. (2018). Sensitivity of young water fractions to hydro-climatic forcing and landscape properties across 22 Swiss catchments. *Hydrology and Earth System Sciences*, *22*(7), 3841–3861. https://doi.org/10.5194/hess-22-3841-2018

Wallace, S., Biggs, T., Lai, C. T., & McMillan, H. (2021). Tracing sources of stormflow and groundwater recharge in an urban, semi-arid watershed using stable isotopes. *Journal of Hydrology: Regional Studies*, *34*(March), 100806. https://doi.org/10.1016/j.ejrh.2021.100806

Warter, M. M., Tetzlaff, D., Ring, A. M., Christopher, J., Kissener, H. L., Funke, E., Sparmann, S., Mbedi, S., Soulsby, C., & Monaghan, M. T. (2024). Environmental DNA, hydrochemistry and stable water isotopes as integrative tracers of urban ecohydrology. *Water Research*, *250*(December 2023). https://doi.org/10.1016/j.watres.2023.121065

Weiler, M., McGlynn, B. L., McGuire, K. J., & McDonnell, J. J. (2003). How does rainfall become runoff? A combined tracer and runoff transfer function approach. *Water Resources Research*, *39*(11), 1–13. https://doi.org/10.1029/2003WR002331

Wu, S., Tetzlaff, D., Goldhammer, T., & Soulsby, C. (2021). Hydroclimatic variability and riparian wetland restoration control the hydrology and nutrient fluxes in a lowland agricultural catchment. *Journal of Hydrology*, *603*(PB), 126904. https://doi.org/10.1016/j.jhydrol.2021.126904

Wunsch, A., Liesch, T., & Broda, S. (2022). Deep learning shows declining groundwater levels in Germany until 2100 due to climate change. *Nature Communications*, *13*(1), 1–13. https://doi.org/10.1038/s41467-022-28770-2

Yang, G., Bowling, L. C., Cherkauer, K. A., & Pijanowski, B. C. (2011). The impact of urban development on hydrologic regime from catchment to basin scales. *Landscape and Urban Planning*, *103*(2), 237–247. https://doi.org/10.1016/j.landurbplan.2011.08.003

Ying, Z., Tetzlaff, D., Freymueller, J., Comte, J. C., Goldhammer, T., Schmidt, A., & Soulsby, C. (2024). Developing a conceptual model of groundwater – Surface water interactions in a drought sensitive lowland catchment using multi-proxy data. *Journal of Hydrology*, *628*(December 2023), 130550. https://doi.org/10.1016/j.jhydrol.2023.130550