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Lessons from the 2018 drought for management of local water supplies in upland areas: A tracer-based assessment

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Abstract

Climate change, combined with industrial growth and increasing demand, could result in serious future water shortages and related water quality and temperature issues, especially for upland and humid areas. The extreme 2018 drought that prevailed throughout Europe provided an opportunity to investigate conditions likely to become more frequent in the future. For an upland rural catchment utilised by the distilling industry in North-East Scotland, a tracer-based survey combined discharge, electrical conductivity, stable water isotopes and temperature measurements to understand the impacts of drought on dominant stream water and industry water sources, both in terms of water quantity and quality (temperature). Results showed that water types (groundwater, ephemeral stream water, perennial stream water and water from small dams) were spatially distinct and varied more in space than time. With regards to the drought conditions we found that streams were largely maintained by groundwater during low flows. This also buffered stream water temperatures. Water types with high young water fractions were less resilient, resulting in streams with an ephemeral nature. Although our results demonstrated the importance of groundwater for drought resilience, water balance data revealed these storage reserves were being depleted and only recovered towards the end of the following year because of above average rainfall in 2019. Increased storage depletion under continued trends of extreme drought and water abstraction could be addressed via informed (nature based) management strategies which focus on increasing recharge. This may improve resilience to droughts as well as floods, but site specific testing and modelling are required to understand their potential. Results could have implications for management of water volumes and temperature, particularly for the sustainability of an historic industry, balancing requirements of rural communities and the environment.

KEYWORDS

climate change, drought, environmental tracers, groundwater-resilience, water resources management, temperature

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1 | INTRODUCTION

Climate projections for regions across the world, including the UK, indicate more frequent periods of hydrological extremes, and relatively lower snow cover in winter (Chan, Falkner, Goldberg, & van Asselt, 2018; Kay, Crooks, Davies, & Reynard, 2014). Through drought periods, groundwater is often the sole source of streamflow (Blumstock, Tetzlaff, Malcolm, Nuetzmann, & Soulsby, 2015; Frisbee, Phillips, Campbell, Liu, & Sanchez, 2011; Gosling, 2014; Winter, 2007). Projected changes in climate could result in reductions to groundwater recharge, and thus lower contributions to streamflow (Cuthbert et al., 2019; Green et al., 2011; Isokangas, Ronkanen, Rossi, Marttila, & Kløve, 2019). This issue is of particular importance for rural upland catchments, such as in the Cairngorms (Scottish Highlands), where groundwater sources involve relatively small, near-surface aquifers (Marsh & Anderson, 2002; Scheliga, Tetzlaff, Nuetzmann, & Soulsby, 2017) which have less resilience to climate change (Hugman, Stigter, Monteiro, & Nunes, 2012; Wright & Novakowski, 2019). With mostly impermeable solid geology, groundwater stores are restricted to fractures and faults or superficial drift deposits (Robins, 2002; Scheliga, Tetzlaff, Nuetzmann, & Soulsby, 2018; Soulsby, Rodgers, Smart, Dawson, & Dunn, 2003).

Climate change predictions also indicate that water temperatures will increase (Capell, Tetzlaff, Essery, & Soulsby, 2014; Jyväsjärvi et al., 2015; Stocker et al., 2013). This will have significant impacts on aquatic and riparian habitats, and their associated organisms (Caissie, 2006; Lake, 2003; Loinaz, Kampp, Butts, & Bauer-gottwein, 2013; Wenger et al., 2011). This issue is even more acute when rural communities and distilling industries depend on such resources. This results in a situation where a balance must be established between potentially increased water demand and maintaining essential ecosystem services (Forzieri et al., 2014; Gosling, Zaidman, Wann, & Rodgers, 2012), in the face of a decreasing resource (Sample, Baber, & Badger, 2016). In this context, decisions made by water managers can have far-reaching consequences.

Various branches of the food and drink industry are reliant on substantial volumes of water meeting specific water quality standards (Crabtree, Macdonald, & Dunn, 2002). This is of particular importance for the distilling industry, as the distillate must be cooled, and changes in temperature of groundwater sources or stream water can have production implications (Freire-González, Decker, & Hall, 2017). Distilleries may use a mixture of private groundwater springs and riverine waters for high-quality process waters and relatively low-quality cooling waters. Abstractions from both surface and groundwater source types are usually regulated to ensure ecological standards are met and the stability of resources is maintained (Scottish Executive Environment Group, 2005), but these regulations often lack catchment-based long-term data and understanding (SEPA, 2019).

To ensure both future production and appropriate abstraction legislation under climate change projections, there is a need to understand (a) the relative role of different water sources (Isokangas et al., 2019), both in terms of water quantity and quality (specifically, temperature) (b) the resilience of these sources under different climatological conditions (Floriancic et al., 2018), and (c) the role of landscape properties such as geology, soils and land-use (Geris, Tetzlaff, & Soulsby, 2015; Zimmer, Bailey, McGuire, & Bullen, 2013; Zomlot, Verbeiren, Huysmans, & Batelaan, 2015).

Tracer-based methods to estimate water sources and flowpathways have been well established (Birkel, Tetzlaff, Dunn, & Soulsby, 2011; Borzi, Tanjal, Santucci, & Carol, 2019; Bowen & Good, 2015; McGlynn & McDonnell, 2003). Extensive research has been conducted using these methods in upland catchments, where logistics often preclude extensive instrumentation (Chiogna, Skrobanek, Narany, Ludwig, & Stumpp, 2018; Engel et al., 2019; Penna, van Meerveld, Zuecco, Dalla Fontana, & Borga, 2016). However, while collection of tracer data through key hydrological events has tended to focus on changing sources during storm events (Klaus & McDonnell, 2013; Litt, Gardner, Ogden, & Lyons, 2015; von Freyberg, Studer, & Kirchner, 2017), tracer studies focusing on droughts are still rare, with relatively few examples in the UK (Blumstock et al., 2015; Geris et al., 2015) and internationally (Chiogna et al., 2018; Floriancic et al., 2018; Marchina et al., 2015, 2017; Vanplantinga, Grossman, & Roark, 2017; Wu et al., 2018).

In the summer of 2018 a major drought occurred across much of Northern Europe, with wide-reaching consequences resulting from a prolonged period of below-average rainfall and above-average temperatures (Afzal & Ragab, 2019; Brunner, Liechti, & Zappa, 2019; Hänsel, Ustrnul, Łupikasza, & Skalak, 2019). Many rivers in Scotland experienced near-record low flows and high temperatures (CEH, 2019). These included the River Spey catchment which covers one of the most famous and economically important Scotch whisky production areas. Several distilleries were forced to halt production as a result. Nevertheless, the drought provided an opportunity to apply a tracer study to understand how water sources for distilleries are affected by drought conditions that may occur more frequently in the future as a result of climate change (Collet, Harrigan, Prudhomme, Formetta, & Beevers, 2018; Soulsby et al., 2003; Soulsby, Rodgers, et al., 2003; Spinoni, Vogt, Naumann, Barbosa, & Dosio, 2018).

Here, we investigated the impacts of the 2018 drought on the dominant water sources and their temperatures for a small headwater catchment of the River Spey. The site is typical for UK/Scottish headwaters and water is abstracted for the distilling industry. As such, the main aim was to evaluate an economically important small-scale upland water resource in the context of resilience to climate change and the threats it poses to the maintenance of water quantity and quality.

More specifically, our objectives were to:

1. Characterise the 2018 drought by exploring it within a long-term context
2. Use tracers to evaluate variation in catchment water sources and the impacts of drought
3. Investigate the impact of drought on water volumes and temperature

2 | METHODOLOGY

2.1 | Study site description

Blairfindy is a 0.9 km^2 sub-catchment of the river Livet, which is one of the main sites from which Glenlivet Distillery obtains cooling water. Adjacent to Blairfindy, the Castleton and Heather Cottage catchments also provide water to local residents as well as the distillery (Figure 1). The Blairfindy catchment is characteristic of the Cairngorms, dominated by crystalline bedrock with limited groundwater storage, whereby water movement is mostly restricted to shallow fractures and faults. A more permeable limestone member runs through the Blairfindy catchment (SW–NE) and may act as a deeper groundwater source. Active storage is also likely in shallow drift deposits of gravels, periglacial material and some in till in the valley bottom (Wilkinson, Stutter, & Gunter, 2016). Blairfindy has a mean elevation of 438 m.a.s.l. and is mostly north-facing with winter topographic shading. Mean annual precipitation is ~900 mm with most (93%) falling as rain during frequent (~60% rainy days per year) low intensity events, typically <10 mm/day. Due to the northerly location, low energy inputs and humid conditions, Potential Evapotranspiration (PET) is relatively low (~450 mm/year). Daily temperatures average 6.2°C; maximum average daily temperatures were 18.7°C for July and minimum −1.3°C for December.

On the upper slopes, the humus-iron podzol soils are relatively freely-draining. These support heather shrubs (*Calluna* and *Erica* spp), which dominate in the east, and grazed acidic grassland in the west, broken up by a small coniferous woodland (Wilkinson et al., 2016). The mid- to lower slopes are dominated by peaty podzols and peaty gleys with thicker peats where surface-water flow-pathways occur. Due to the generally wet soils and impermeable geology, streamflow responds rapidly to rainfall. In a preliminary study, Wilkinson et al. (2016) estimated a low Base Flow Index (BFI) of 0.31 and a storm runoff coefficient of ~50%, with flow paths dominated by overland or shallow subsurface stormflows. Estimated average annual runoff at Blairfindy is 450 mm/year (equivalent to mean daily runoff 0.013 m^3/s) before any abstraction. The main stream and most groundwater sources are perennial. Some surface water sources are ephemeral, becoming active during or after precipitation events and drying out through periods of low rainfall. This includes older drainage channels in the upper SW of the catchment, which result in the activation of ephemeral streams in wetter periods.

At G1 – 5 (Figure 1), groundwater from springs is abstracted by the distillery via concrete wells from which water is gravity-fed to surface-water reservoirs or to the distillery directly. Distillery abstractions are regular and average 750 m^3/day from all groundwaters, of which 234 m^3/day come from Blairfindy catchment (including G1 and G2; abstraction data 2009–2019; distillery abstractions diagram:

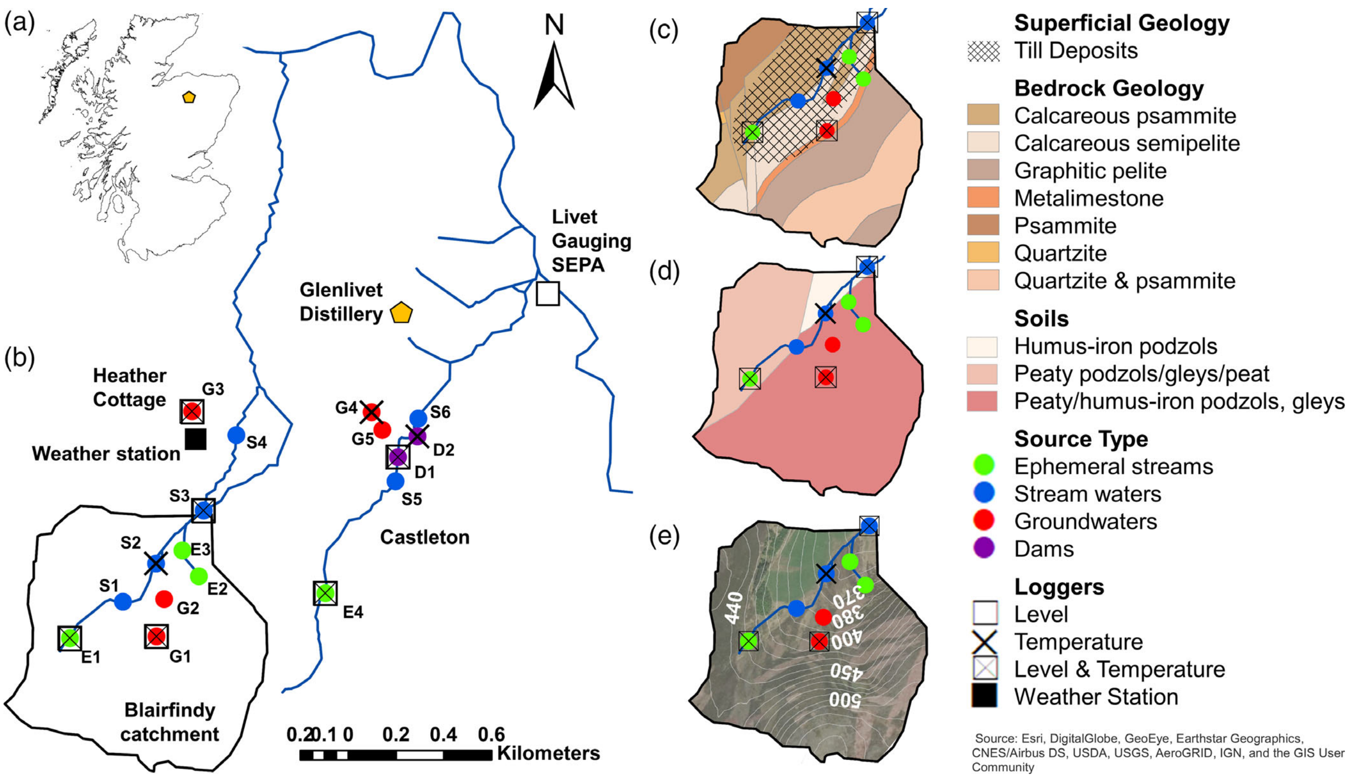

FIGURE 1 (a) Map of Scotland with Glenlivet Distillery marked. (b) Blairfindy catchment and surrounding area of Glenlivet distillery, with weather station, sampled Ephemeral streams (E1-4), stream waters (S1-6), groundwaters (G1-5) and dams (D1-2) with loggers installed for water temperatures and levels. (c) 50 K resolution bedrock geology and superficial till deposits in Blairfindy catchment (BGS, 2020). (d) Soil classification of Blairfindy catchment (Soil Survey of Scotland Staff, 1981). (e) Satellite imagery showing land use and 10 m contours of Blairfindy catchment (ESRI, 2020)

Supplementary Materials S1). For Blairfindy, total annual abstractions (i.e., from streams as well as groundwater wells) equate to approximately 10% of annual precipitation. There is a two-week “shutdown period” at the distillery coinciding with summer maintenance when flows are usually lower and air and stream temperatures higher.

2.2 | Data collection and analysis

2.2.1 | Long term data and analysis for context of study period

Precipitation data since 1950 at Ballindalloch (7.5 km north of Glenlivet at 145 m.a.s.l.; Met Office, 2019a) were analysed using the Standard Precipitation Index (SPI) method (Supplementary Materials S2). Briefly, SPI normalises rainfall deficits based on a long term data record for a specified period and data are transformed to normal distribution so that the mean SPI is zero (Tigkas, Vangelis, & Tsakiris, 2015; World Meteorological Organization, 2012). Here, a 4-month running mean of a 3-month SPI was chosen. This captured the impact of the 2018 drought and seasonal deficits over timescales relevant to the distillery which is dependent on groundwater, which may take time to recover from rainfall deficits (Barker, Hannaford, Chiverton, & Svensson, 2016; Svensson, Hannaford, & Prosdocimi, 2017; World Meteorological Organization, 2012).

Flow data from 1980–2019 at the River Livet (102 km^2) at Minmore gauging station (Figure 1b; SEPA) were used to contextualise the hydrological year 2017–2018 in terms of long-term river flow variability (National River Flow Archive, 2019). Similarly, air temperature data from 1980–2019 at Braemar, (the nearest station with complete data, 37 km south of Glenlivet, at a comparable altitude of 330 m.a.s.l.; Met Office, 2019b) allowed temperature extremes to be contextualised.

2.2.2 | Study period and hydrometric monitoring

The study's detailed monitoring period spanned from 01/12/2017 to 15/07/2019 to encompass pre- and post-drought periods. Hydroclimatic data (15-minute intervals) were collected from Glenlivet weather station from 24/05/2018 (Figure 1b). This included an Environmental Measurements ARG100 tipping bucket rain gauge (0.2 mm precision); temperature and relative humidity sensors; wind speed and direction; a Kipp and Zonen NR-Lite2 net radiometer (with wind speed correction); and air temperature measurement. Snowfall and melt were determined using a time-lapse camera (hourly recording at E1). Stream discharge was also obtained (from 24/02/2018 onwards) for Blairfindy catchment outlet (S3) using 15-minute stage data recorded by an In-situ Rugged TROLL100 level-logger in a rated section (50 discharge gaugings across the full range of stage observations).

F.A.O. grass reference crop evapotranspiration was derived from the weather station data using the Penman-Monteith equation (Allen, Pereira, Raes, & Smith, 1998). Data from local stations (Ballindalloch for precipitation; Aviemore for air temperatures/PET (30 km south west of Glenlivet, at a comparable altitude of 228 m.a.s.l.; Met Office, 2019b), and Livet flow data) were used to extend the Glenlivet hydrometric and climate data for the period prior to installation of the discharge monitoring and weather station on site (scaled, linear regression R^2 values Q = 0.51, PET = 0.86, P = 0.69, Air temperatures = 0.97). To understand temperature variability between water sources, temperature loggers (15-minute recording) were installed at most of the synoptic sampling sites (Figure 1b). Assuming water at these sites was well mixed, the loggers were positioned to be permanently submerged and capture representative temperatures (Folegot et al., 2018). This was of particular relevance for the small surface water reservoirs (maximum depth < 2 m) as their level varied depending on distillery usage and evaporation, as well as discharge.

2.2.3 | Water sampling

To characterise temporal changes in catchment water quality at high frequency, daily stream water samples were collected at Blairfindy outlet (S3; Figure 1b), and bulk daily precipitation samples at the weather station using ISCO 3700 auto-samplers. To characterise temporal changes in water quality at lower frequency, but at higher spatial resolution, synoptic sampling was conducted on a fortnightly basis throughout the study period, with all sites sampled on the same day (Figure 1b, Table 1). This routine sampling was complemented with opportunistic sampling of higher and lower flow events to capture a fuller range of hydro-climatological conditions.

Synoptic sampling captured the internal variability of water sources representing key sources used by the distillery within Blairfindy catchment and its surroundings, as well as other runoff sources (Figure 1b). Ephemeral streams (E1-4) form on hillslopes, drying out through prolonged periods of low precipitation, and become tributaries to the main stream in average to wet conditions. Stream waters (S1-6) are perennial. Farther downstream of monitoring sites, some of these are canalised and directed to cooling water ponds for use by the distillery. Groundwaters (G1-5) originate from different geological units of the catchment and are abstracted by the distillery. Finally, waters from small surface reservoirs with Dams (D1-2) were also sampled to assess some of the integrated sources feeding the distillery (Supplementary Materials S1).

2.2.4 | Laboratory analysis

Water samples were analysed for isotopes and a selection of water quality parameters, including temperature. Water stable isotopes of deuterium (δ^2H) and oxygen-18 (δ^{18}O) were measured for the daily (streamflow at site S3 and precipitation) and synoptic samples (all sites) using a Los Gatos DLT-100 laser isotope analyser following standard measurement protocols (precision of ±0.4 ‰ for δ^2H and 0.1 ‰ for δ^{18}O). These data are expressed in δ- notation [‰] relative to Vienna Standard Mean Ocean Water. The daily stream (S3) and

TABLE 1 Landscape characteristics and instrumentation for synoptic sampling sites

Type	Code	Depth (m)	Logger Temperature	Logger Level	Elevation (m.a.s.l.)	Soils	Geology Superficial	Geology Bedrock
Ephemeral streams	E1		Y	Y	428	Peaty podzols	Till	Calcareous psammite/semipelite
	E2				374	Peaty podzols		Graphitic pelite
	E3				342	Humus-iron podzols	Till	Metalimestone
	E4		Y	Y	378	Peaty podzols	Till	Graphitic pelite
Stream waters	S1				389	Peaty podzols	Till	Calcareous psammite/semipelite
	S2		Y		364	Peaty podzols	Till	Calcareous psammite/semipelite
	S3		Y	Y	340	Humus-iron podzols	Till	Calcareous psammite/semipelite
	S4				321	Humus-iron podzols		Calcareous psammite/semipelite
	S5				311	Humus-iron podzols	Till	Graphitic pelite
	S6				283	Humus-iron podzols	Till	Calcareous psammite/semipelite
Groundwaters	G1	2	Y	Y	398	Peaty podzols		Metalimestone
	G2	1.2			367	Peaty podzols		Calcareous semipelite
	G3	1.2	Y	Y	342	Humus-iron podzols	Till	Psammite
	G4	1.5	Y		293	Humus-iron podzols	Gravel, sand, silt	Graphitic pelite
	G5	1			303	Peaty podzols	Gravel, sand, silt	Graphitic pelite
Dams	D1		Y	Y	295	Humus-iron podzols	Gravel, sand, silt	Graphitic pelite
	D2		Y		289	Humus-iron podzols	Till	Graphitic pelite

Abbreviations: Code, synoptic sampling site code; Depth, depth to groundwater; Logger, logger type installed.

TABLE 2 Hydro-climatological conditions for 6 key sampling dates, and summary of sampling period

	Sampling date	Blairfindy Q (mm/day)	P (mm/day)	PET (mm/day)	Livet Q (mm/day)	T mean (°C)
Wet winter	05/12/2017	2.1	0	0.4	6.9	8.8
Dry summer	30/08/2018	0.2	0.2	1.8	0.6	11.7
Average winter	18/12/2018	0.3	0.8	0.3	3.9	6.7
Dry winter	08/01/2019	0.2	1.2	0.1	1.2	3
Snowmelt	04/04/2019	3.1	6	0.4	N/A	2.6
Wet summer	26/05/2019	7.6	18.6	0.8	N/A	8.4
Sampling period	Max	8.5	25	6.8	15.5	20.6
(01/12/17–15/07/19)	Min	0.2	0	0	0.6	-5
	Mean	0.7	2	1.2	2	6.9

Note: N/A denotes data not yet available.
Abbreviations: Blairfindy Q, Blairfindy outlet (S3) discharge; Livet Q, discharge from the River Livet; P, precipitation; PET, potential evapotranspiration; T, mean daily air temperature for Glenlivet weather station.

synoptic samples (all sites) were also analysed for Electrical Conductivity (EC) using a temperature corrected Jenway 4330 m. The fortnightly synoptic samples were analysed for pH and alkalinity using Gran titration to end points pH 4.5, 4.0 and 3.5 (Neal, Hill, Hill, & Reynolds, 1997), and on-site temperature measurements using a HANNA probe. From the synoptic surveys, six occasions were selected based on contrasting hydro-climatological conditions to represent the full variability of the conditions monitored. Table 2 shows this included: minimum discharge in summer (Dry Summer); close to minimum discharge in winter (Dry Winter); the maximum sampled discharge/biggest precipitation event (Wet Summer); a large snowmelt event (Snowmelt); and typical conditions during winter (Average Winter and Wet Winter). For these six sampling campaigns, all samples were analysed for major cations and anions (methods: Supplementary Materials S3; results Supplementary Materials S4).

2.2.5 | Data analysis

To understand how source water contribution in Blairfindy catchment changed during the drought, we combined hydrometric data with tracer analyses. First, we used two-component End-Member Mixing

Analysis (EMMA) (Soulsby, Rodgers, et al., 2003) to quantify relative proportions of groundwater and surface water sources in stream water (S3). Although there is well-known uncertainty associated with using EC as a tracer (Benettin & van Breukelen, 2017; Pelizardi, Bea, Carrera, & Vives, 2017), the distinct variation in EC between contributing sources (see below), typical for UK upland catchments (Jarvie et al., 2001), showed that average EC in Blairfindy ephemeral streams (E1) and groundwaters (S1, representing a groundwater source at the head of the stream network and G1 representing a hillslope groundwater source) for end-members would be suitable for a first approximation (Engel et al., 2016). Uncertainty bounds were derived from the upper and lower EC values from different groundwater sampling sites to account for differences between groundwater EC (Robins, 2002). This same method was repeated using alkalinity as a tracer, with similar results (not shown here). Second, we estimated the proportion of relatively young water (<∼3 months old; Young Water Fraction, YWF) in all sampled sources (except S4, G5 and D1, due to limitations in data availability). This approach used δ^2H as a tracer in precipitation and all sampling locations. Streams S2, S3 and S6 were flow-weighted using area-scaled discharge from Blairfindy (S3). This followed the method detailed by von Freyberg, Allen, Seeger, Weiler, and Kirchner (2018) whereby uncertainties were expressed as standard errors and used to calculate maximum and minimum estimates of YWFs (Jarvie et al., 2001; Kirchner, 2016a, 2016b; von Freyberg et al., 2018).

To investigate the impact of the drought on the water balance and temperature we used hydro-climatological data to track variations in catchment storage (set to 0 at the start of the monitoring period). This was approximated on a daily basis using the water balance approach, accounting daily precipitation inputs against evapotranspiration, discharge, and abstraction outputs from the catchment. The abstraction data was provided by the distillery (2009–2019), and used to approximate annual abstractions from within Blairfindy catchment upstream of the outlet (i.e., G1, G2 and associated groundwaters). Water budgets for hydrological years October 2017–2018 and 2018–2019 were derived for comparative purposes. Time series of temperature data from representative source types (E1, S3, G1, G3, G4, D2) were used, alongside synoptic sampling data to show differences between water types in terms of variability through different climatic conditions. The relationship between weekly mean air and water source temperature was also directly compared using linear regression (Arismendi, Safeeq, Dunham, & Johnson, 2014) over the time period where data for all sources was available (21/12/2018–15/07/2019).

3 | RESULTS

3.1 | The 2018 drought within a long-term context

Long-term data for the Glenlivet area shows that the period 2012–2019, particularly 2018, was unusually dry, which led to both meteorological and hydrological drought conditions (Hao & Singh, 2015). The SPI calculated over the 1950–2019 period at Ballindalloch showed an extended period of below average precipitation from around 2012 through to 2019 (Figure 2a). The longevity of this low rainfall period is comparable only to that of the UK-wide drought of the 1970s (Smith, 1977), with the 80s, 90s and 00s mainly consisting of positive anomalies. In more detail, through 2017–2018 the SPI became negative from January-18 and continued to decrease to −2.18 by late summer, characterised by the World Meteorological Association as "extreme dryness" (SPI < −2.0) with a return period of 1 in 50 years (2012) (Supplementary Materials S2). From October-19 onwards, the SPI slowly recovered but remained negative up until March 2019 (−0.89). Overall, the annual total of 575 mm at Glenlivet for 2017–2018 was very low compared to the average of 900 mm at Ballindalloch. During this period, precipitation was consistently lower than average in all seasons, ranging from 56% of average during April–June, to 79% of average during July–September.

The river Livet responded to these dry conditions with the lowest flows since complete records began in July 1980, in a prolonged low flow spell between June and October 2018 (Figure 2b). At the smaller scale of the Blairfindy outlet, prolonged periods of low flows lasted from April-18 to late January-19 (average = 0.24 mm/day). Discharge was 0.17 mm/day at its lowest during the dry summer. The winter was also relatively dry. For the period November 2018–February 2019, average discharge was 0.46 mm/day, less than 60% of the long-term (1980–2017) estimated average discharge for that period (0.80 mm/day; scaled from Livet; Figure 3a). Despite several precipitation events from October-18 onwards, there was very little response in the main stream until significant snowmelt occurred in February 2019.

Isotopically there was seasonal variation in precipitation (Figure 3d), with depletion of heavier isotopes in winter and enrichment through summer. Stream isotopes showed considerable damping by comparison and little response followed the small, frequent precipitation inputs through the dry period (Figure 3a,). The stream δ^2H standard deviation was low, though wetter periods resulted in greater changes (usually depression) following rainfall. EC in the stream showed more pronounced seasonal change, remaining >100 μS/cm through the summer, but abrupt reductions followed precipitation and a subsequent lower recovery.

Air temperatures through 2018 at Glenlivet were generally warmer than the long-term average (1980–2019) in Braemar for both daily maximum and minimum at similar altitude, despite Glenlivet being more sheltered (Figure 2c). The actual evapotranspiration (AET) of 485 mm was high in Glenlivet in 2017–2018 (Figure 3b), considering the low rainfall input and the long term average PET for the region (450 mm/year; Dunn & Mackay, 1995). For comparison, the AET in 2018–2019 was 405 mm, while precipitation was almost double of 2017–2018 during the following year.

3.2 | Variation in catchment water sources and the impact of drought

3.2.1 | Source water quality varied with source type and spatial location

We found distinct differences in the hydrochemistry of different water types. Also, within a water type group, there was generally

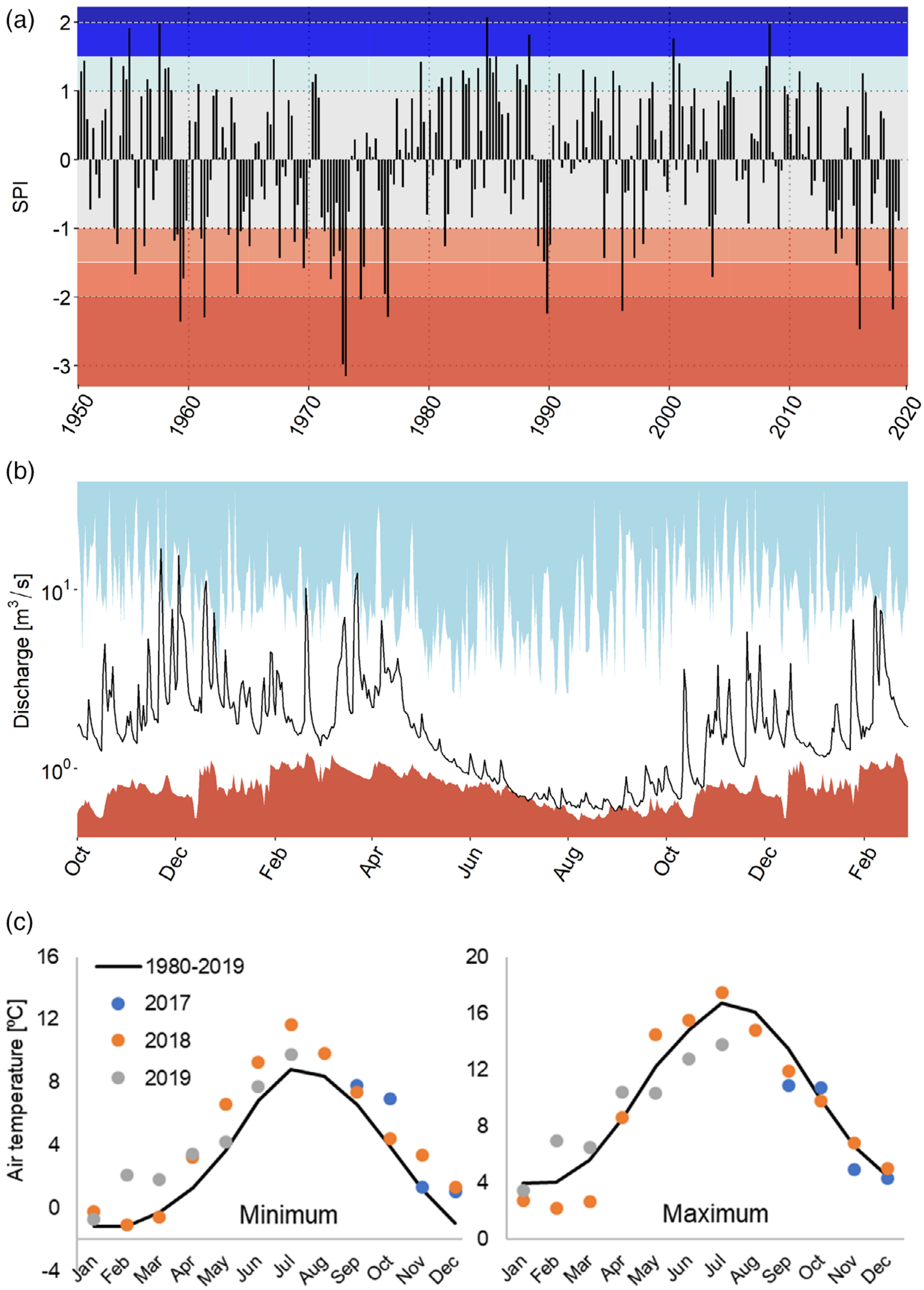

FIGURE 2 (a) Standard precipitation index (SPI) for Ballindalloch (7.5 km North Glenlivet), 1950–2019 using a 4 month running mean for a 3-month SPI. Band colours represent SPI values from −2 and below being "extremely dry" through to 2+ and above being "extremely wet" (World Meteorological Association, 2012). (b) River Livet at Minmore discharge from October 2017–February 2018 in black, with white band representing all flows recorded on each day from 1980–2019, with blue band delimiting highest flows recorded and red band delimiting lowest flows recorded through that period. (c) Black line shows long term mean (from Braemar 1980–2019) and study period years monthly air temperatures (for Glenlivet)

more variation in space than in time. Overall, differences in geology underlying the drainage of sampling locations and estimated residence times explained trends in EC, isotopes, alkalinity and pH.

EC was lower for ephemeral streams than for other water types, consistent with relatively short residence times, particularly E1 (Table 3). Groundwater average values were higher, although considerable variability reflected differences in geochemical composition of the geology and subsurface mixing of different sources (Table 3; Figure 4). This was particularly evident when comparing high EC of G4 (average = 390 μS/cm), consistent with underlying fractured bedrock connected to deeper, older, and more alkaline waters; to lower EC values of G1 and 2 (average = 67 and 83 μS/cm, respectively), likely shallower groundwaters associated with meta-limestone and calcareous semipelite skirting the edges of till deposits (Figure 1c; Tables 1 and 3). S1, at the head of the Blairfindy stream network, had higher values of EC (was more alkaline) than hillslope groundwaters (G1 and 2), suggesting that this was likely associated with a deep groundwater source. EC in streams represented time-varying mixtures of contributing sources up-stream and showed high variability in relation to the average values (*SD* = 8–40 μS/cm). This was also observed to a lesser extent in dammed surface reservoirs (D1 and 2), with larger water volumes in the reservoir buffering additional water inputs.

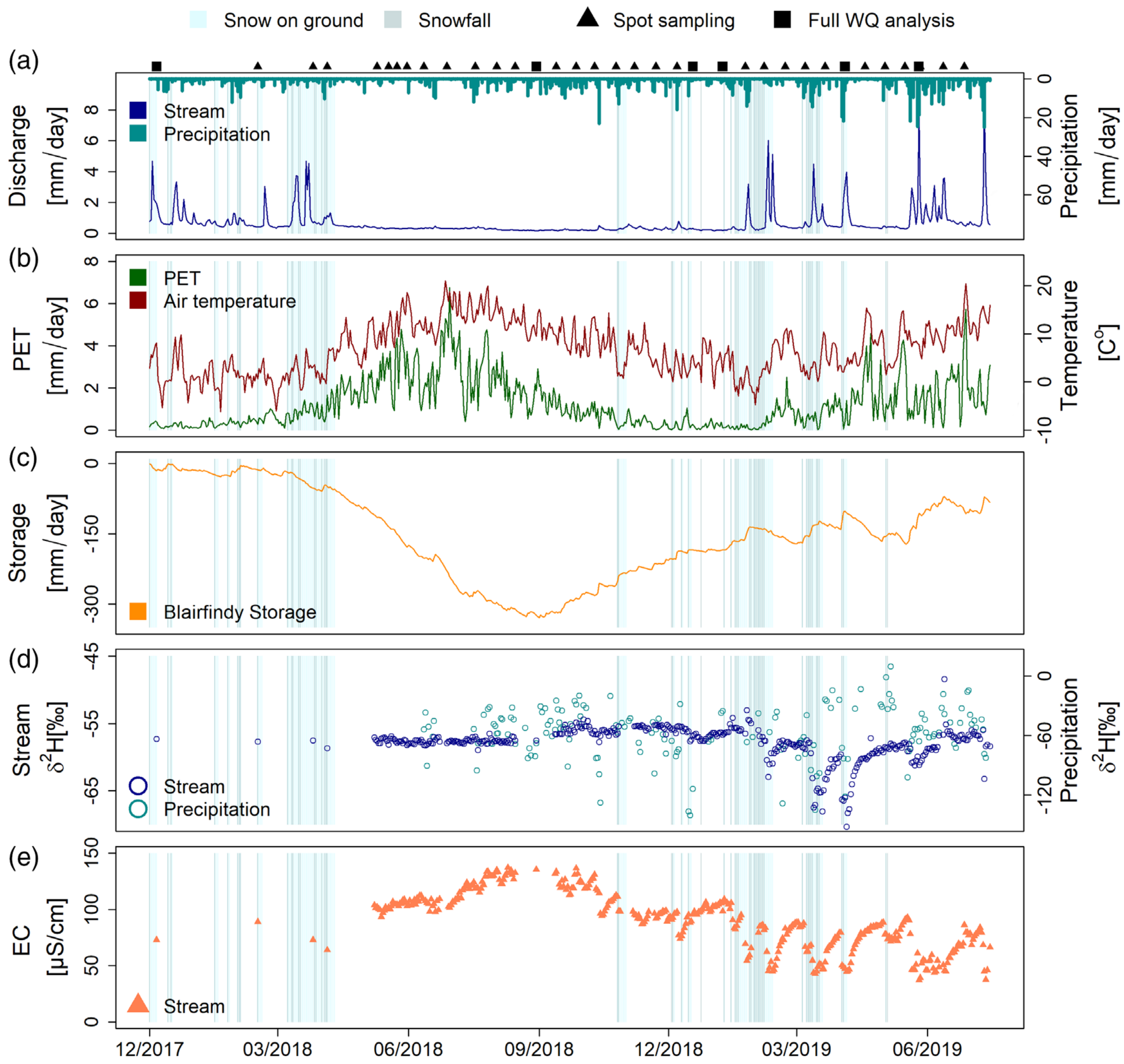

FIGURE 3 Sampling conditions (01/12/2017–15/07/2019) in Blairfindy catchment with timing of synoptic sampling campaigns; snowfall and snow on the ground. (a) Precipitation and discharge at Blairfindy stream outlet S3. (b) Air temperature and Potential Evapotranspiration calculated using weather station data and Penman-Monteith equation. (c) Blairfindy catchment storage displayed starting from 0 at start of sampling period. (d) δ^2H of precipitation and stream water at S3. (e) Electrical Conductivity of stream at S3

The pH and alkalinity of water types showed similar patterns to EC (Table 3). Spatial differences between sampling points of the same water type were again apparent, with data from individual groundwater sampling points reflecting different underlying geologies. For example, waters with high EC values in G4 were also high in alkalinity and high in pH. Mean values for groundwaters were on average higher, ephemeral streams the lowest and perennial streams intermediate (Figure 4; Table 3), and again S1 showed values in the range of deeper groundwaters. Overall, there were clear differences in relative variability between sources (for groundwaters this was low, for streams high, and dams intermediate).

Isotope data showed variation between water sampling type and spatial location, likely due to differences in subsurface mixing and residence times. Ephemeral streams, notably E1, showed the widest range in δ^2H, with a standard deviation much greater than for other water types, suggesting limited mixing and short travel times, although some mixing must occur as the range covered only 26% of δ^2H variability in precipitation (Figure 3d). Perennial streams varied less than ephemeral streams (Figure 4), while groundwater samples exhibited far narrower ranges in δ^2H, suggesting greater subsurface mixing (*SD* range = 0.4–0.5‰; Table 3). Between groundwater sampling sites, the means differed by 0.5 ‰ demonstrating limited spatial differences. Water in D1 and D2 reservoirs had δ^2H characteristics intermediate between streams and groundwaters and relatively low variability, thereby indicating mixing.

Isotopes were also analysed in dual isotope space; Figure 5a shows precipitation isotopes and all water sampling types combined. The Local Meteoric Water Line (LMWL) plots close to the Global Meteoric Water Line (GMWL). Ephemeral streams and stream waters plotted within a much narrower range than precipitation, and ephemeral streams varied more than stream waters (Table 3; Figure 5b). In comparison, groundwaters covered a very narrow isotopic range

TABLE 3 Summary of mean and standard deviation (*SD*) values for water quality parameters through total sampling period (01/12/17–15/07/19) for individual sources and mean vales for source type, with number of sampling occasions and % of which sources were dry or frozen (sample was not obtained)

Type	Code	Sample size (% dry/frozen)		pH		EC (μS/cm)		Alkalinity (μEq/L)		δ2H (‰)		Temperature (°C)		Young water fraction (%)		
				Mean	*SD*	Mean	*SD*	Mean	*SD*	Mean	*SD*	Mean	*SD*	Value	Min	Max
Ephemeral streams	E1	26	(42/4)	4.5	0.5	45	17	-5	55	−61.2	6.7	7.99	3.39	96.9	26.2	100
	E2	31	(39/3)	6.1	0.1	97	10	398	93	−58.1	0.3	7.04	0.86	1.7	0.5	2.1
	E3	35	(17/3)	7.0	0.4	90	15	313	110	−57.3	2.2	7.58	2.15	23.9	8.3	38.1
	E4	32	(22/0)	6.8	0.4	59	12	183	52	−57.7	1.9	8.02	3.56	8.8	2.7	12.1
	Mean	31		6.1	0.4	73	14	222	77	−58.6	2.8	7.65	2.49	32.8	9.4	43.1
Stream waters	S1	20		6.5	0.3	159	29	1551	336	−57.1	0.8	7.14	0.74	3.1	1	4.4
	S2	37		7.1	0.6	101	40	626	375	−58.1	2.3	8.92	4.24	36.9	11	50.4
	S3	37		7.1	0.5	95	29	519	271	−58.0	2.0	8.42	3.48	37.4	11.3	51.6
	S4	15		6.0	0.1	83	8	215	49	−60.4	0.4	10.55	1.90	N/A	N/A	N/A
	S5	36		6.5	0.4	60	13	183	105	−57.3	2.5	8.90	3.90	15.7	5	23
	S6	36		6.8	0.2	89	14	405	131	−59.0	1.1	8.36	2.77	16.4	3.9	17.8
	Mean	30		6.7	0.4	98	22	583	211	−58.3	1.5	8.71	2.84	21.9	6.4	29.4
Ground waters	G1	37		6.0	0.2	67	9	247	48	−58.7	0.4	8.21	1.65	3.1	1	4.5
	G2	35		5.9	0.2	83	12	402	67	−59.1	0.5	8.53	1.58	4.1	1.2	5.6
	G3	34		6.3	0.3	105	14	422	83	−59.2	0.5	9.66	1.94	1.3	0.2	1
	G4	36		7.9	0.3	390	33	4002	704	−58.8	0.4	9.46	1.83	3.3	0.9	4.3
	G5	14		6.0	0.2	125	38	339	81	−58.8	0.4	10.31	1.45	N/A	N/A	N/A
	Mean	31		6.4	0.2	154	21	1082	196	−58.9	0.4	9.23	1.69	3	0.8	3.9
Dam	D1	14		6.5	0.3	81	6	308	98	−58.3	1.0	14.18	5.03	N/A	N/A	N/A
	D2	37		6.5	0.3	85	12	373	120	−59.0	1.2	9.70	4.45	3.1	0.9	4.1
	Mean	26		6.5	0.3	83	9	341	109	−58.7	1.1	11.94	4.74	3.1	0.9	4.1

Note: Young water fraction (YWF) values are given as % with minimum and maximum calculated from standard error demonstrating uncertainty. N/A refers to samples where YWF analysis was not possible due to small sample size.

almost central to stream and ephemeral stream variability. This suggests that overall catchment groundwater was well-mixed, with relatively long flow pathways between surface and groundwater sources. There was little evidence of evaporative fractionation as most samples plotted close to both meteoric water lines.

Temperatures of contrasting water sampling methods and water types generally followed relatively similar patterns of variability to δ^2H being high in ephemeral streams and perennial streams, and low in groundwaters, although S1 at the head of Blairfindy stream and E2 were also particularly stable. D1 and D2 showed greatest variability and highest measured temperatures (Figure 4).

The major ion composition of all synoptic samples was plotted in a Piper diagram (Figure 6), which showed that the catchment was mostly dominated by calcium and sodium bicarbonate waters. Overlap of stream and dam water between ephemeral and groundwater demonstrated the mixing that occurs in stream waters. Further differences were also highlighted between sources types; ephemeral streams, generally more sodium-chloride dominated, varied in magnesium, calcium and bicarbonate depending on flow; groundwaters were dominant in calcium-bicarbonate; and streams mixed between the two depending on flow.

3.2.2 | Sources of streamflow

Taken together, the hydro-chemical data (alkalinity, EC, δ^2H and major ions) provided a consistent picture that streamflow at Blairfindy outlet (at S3) was a time-varying mix of groundwater and near-surface water as represented by the ephemeral stream waters. Daily EC values negatively correlated with flow at S3 (linear regression, R^2 = 0.7), with higher values associated with proportionally higher groundwater contributions (Figure 3e and 4). This enabled a simple hydrograph separation to disaggregate contribution of sources to S3 (Figure 7a). We used EC of ephemeral streams (E1; 31 μS/cm) and groundwaters including S1 as groundwater at the head of the stream (range = 115–195 μS/cm; Table 4) as the two end-members for EMMA. This showed that, unlike during rainfall events and wetter periods, S3 during the dry period was dominated (65–100%) by groundwater (Figure 7b). Uncertainty bands reflected the range of groundwater EC values, as groundwater mixing proportions were unknown. However, the absolute volumes of groundwater in S3 remained relatively constant through the year averaging 0.19 mm/day or around 30% of the average annual flow.

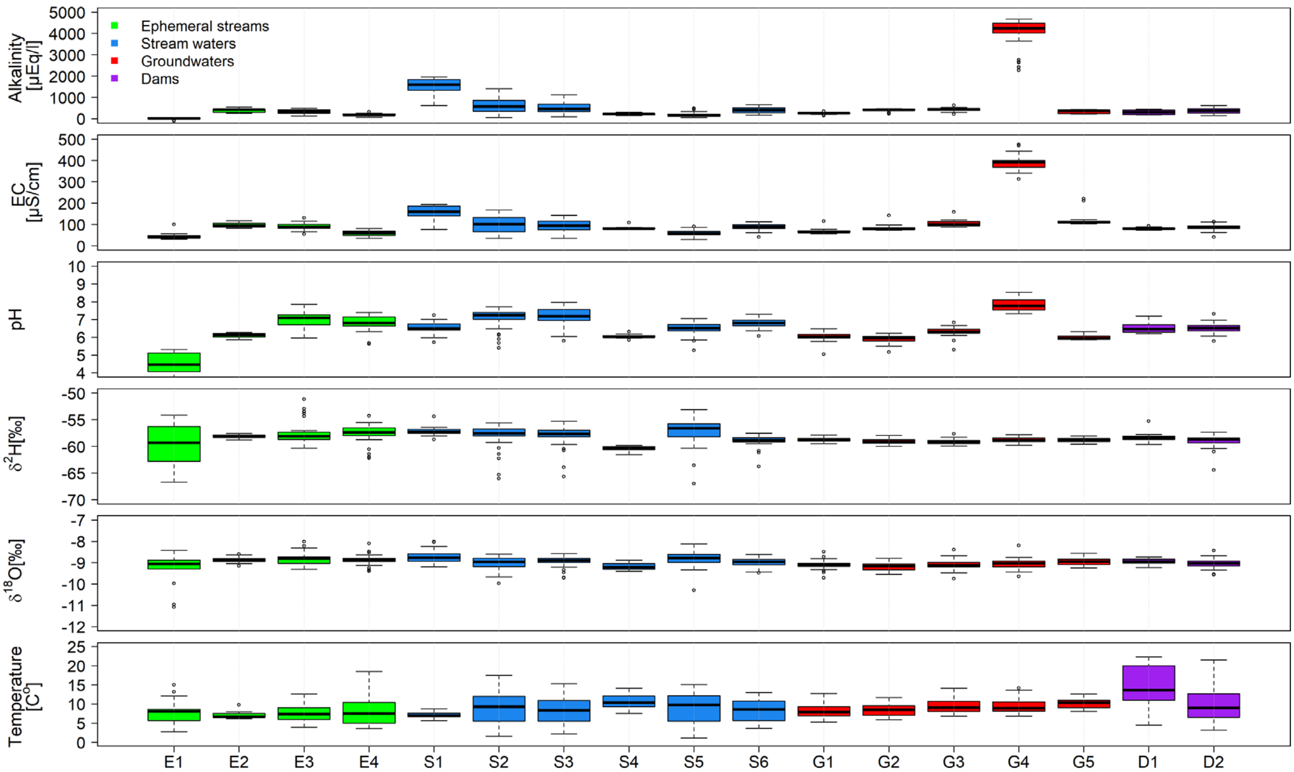

FIGURE 4 Boxplots showing water quality parameters for ephemeral streams, stream and ground-waters and dammed surface water reservoirs through sampling period

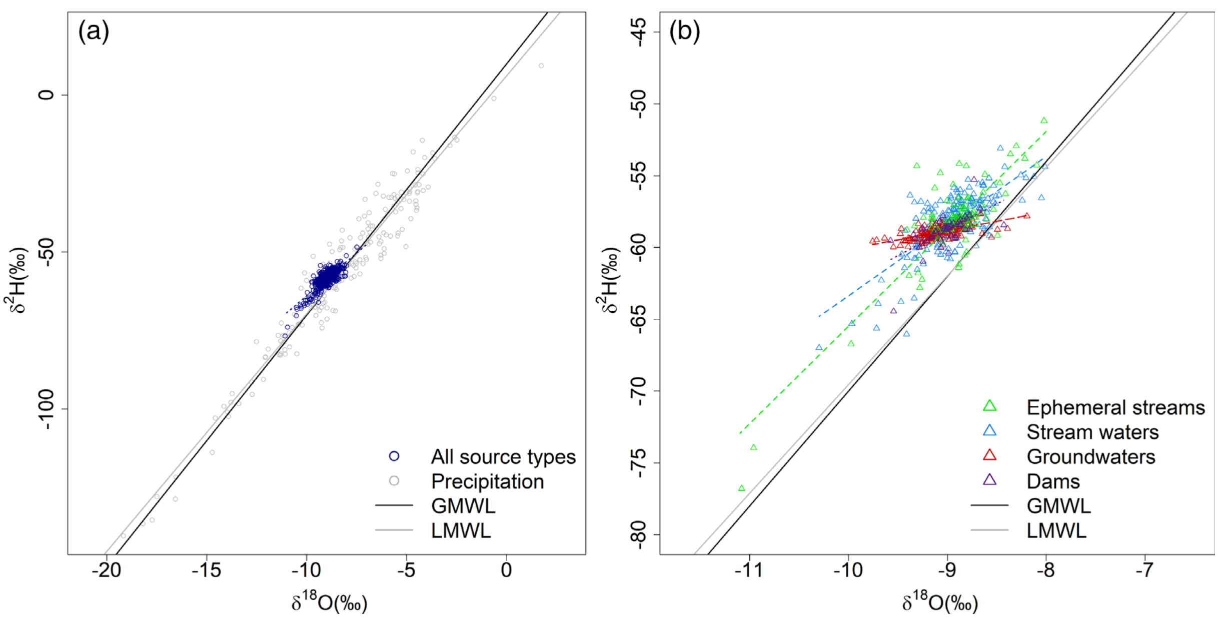

FIGURE 5 Dual isotope plot showing (a) all source types combined with precipitation and Global and Local Meteoric Water Lines (GMWL; LMWL) and (b) source types with GMWL and LMWL

Having highlighted the overall volumetric dominance of surface/near-surface runoff sources in S3 through EMMA, the variations of δ^2H in precipitation and source waters were used to estimate young water fractions (YWF) (i.e., proportion of water less than ~3 months old; Table 3). Ephemeral streams, especially E1, had the highest YWF, with 97% of water less than ~3 months old. YWFs at E2 and 4 were much lower (~2 and 9%), and in fact lower than stream flow, indicating longer residence times and more mixing (Table 3). Ephemeral streams were dry ~30% of sampling dates. The main streams in general had low YWFs but with a large range (~3–37%), which

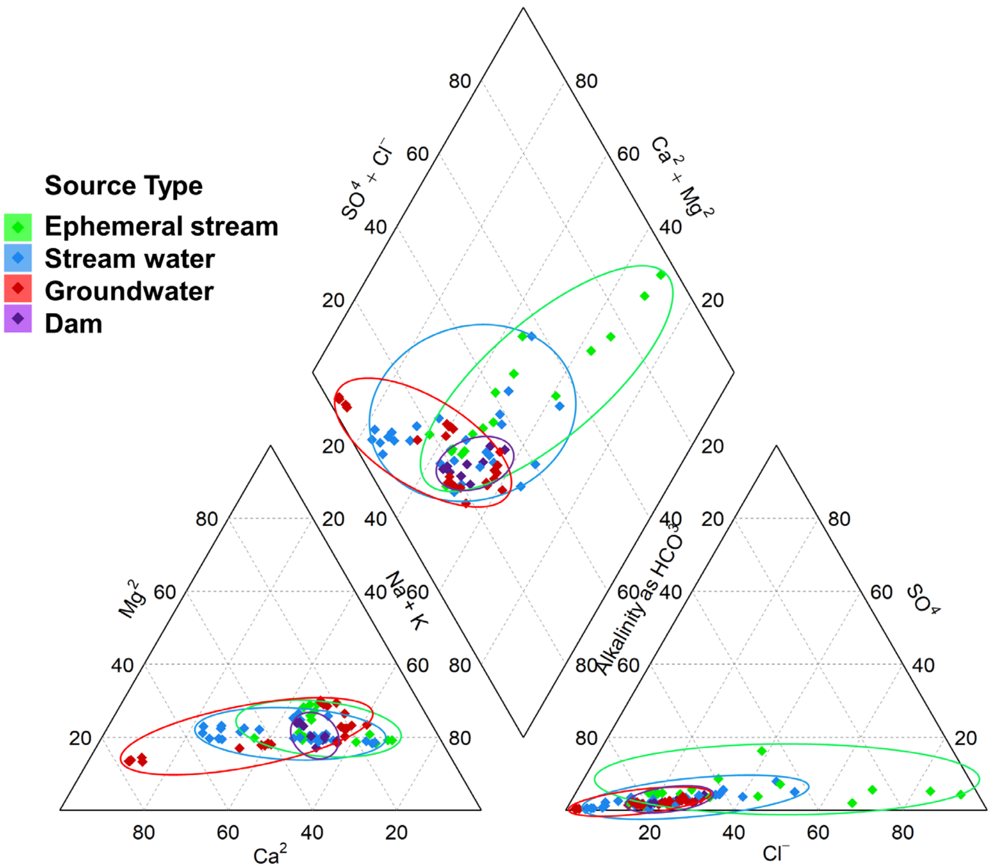

FIGURE 6 Piper plot of grouped sources sampled during the 6 key hydrological events, ellipses grouping each source type

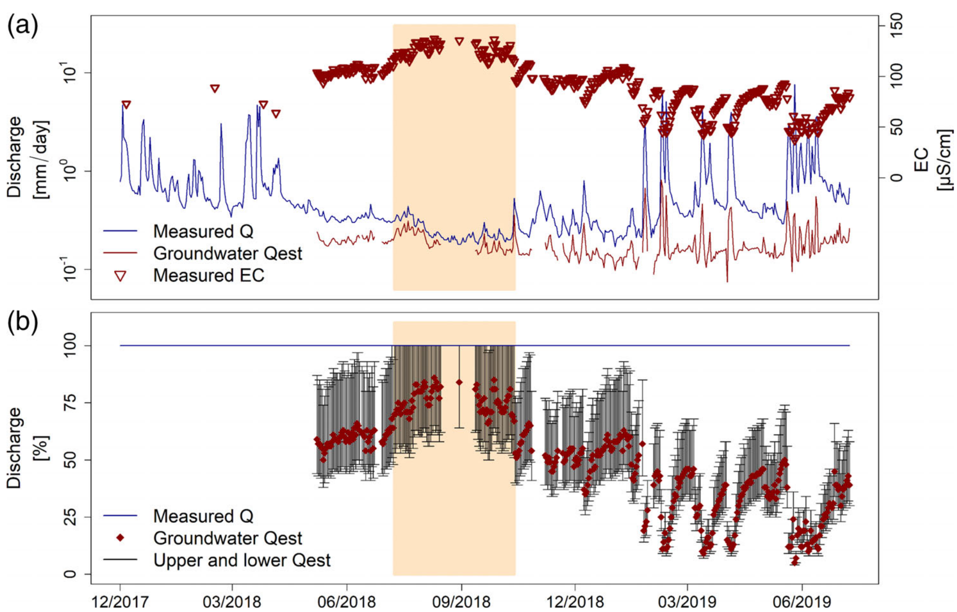

FIGURE 7 (a) Measured discharge (Q) and electrical conductivity (EC) through sampling period and estimated groundwater contribution to discharge (groundwater Qest) using End-Member mixing analysis (EMMA, using EC as end-members) from start of daily EC sampling at S3 Blairfindy stream outlet. (b) %groundwater Qest, with upper and lower groundwater Qest limits obtained from upper and lower range of groundwater end-members shown as error bars. Yellow band in both (a) and (b) denotes dry period where average groundwater Qest is greater than 65%

highlighted spatial variation, and how hydro-climatic conditions differentially affected stream sources (Table 3). Groundwaters all had very low YWFs with a small range (~1–4%). The very low YWF suggested longer residence times and mixing occurred within the till deposits and fractures in the bedrock. Dammed surface reservoirs also demonstrated mixing of contributing water sources, with similar YWFs to groundwaters (D2 average = 3.1%).

3.3 | The impact of the drought on the water balance and temperature

Basic water balance analyses revealed a storage deficit resulting from low precipitation and high evapotranspiration through the drought period (Figure 3a,b) and for the 2017–2018 hydrological year overall (Table 4). Above average precipitation in summer-19 enabled significant but not full

TABLE 4 Water balance for hydrological year 2017–2018 and 2018–2019

Start	End	P (mm)	PET (mm)	Q (mm)	A (mm)	ΔS (mm)
01/10/2017	01/10/2018	575	486	229	95	−234
01/10/2018	01/10/2019	1023	405	297	95	228

Abbreviations: A, distillery abstractions; P, precipitation; PET, potential evapotranspiration; Q, discharge; ΔS, change in storage.

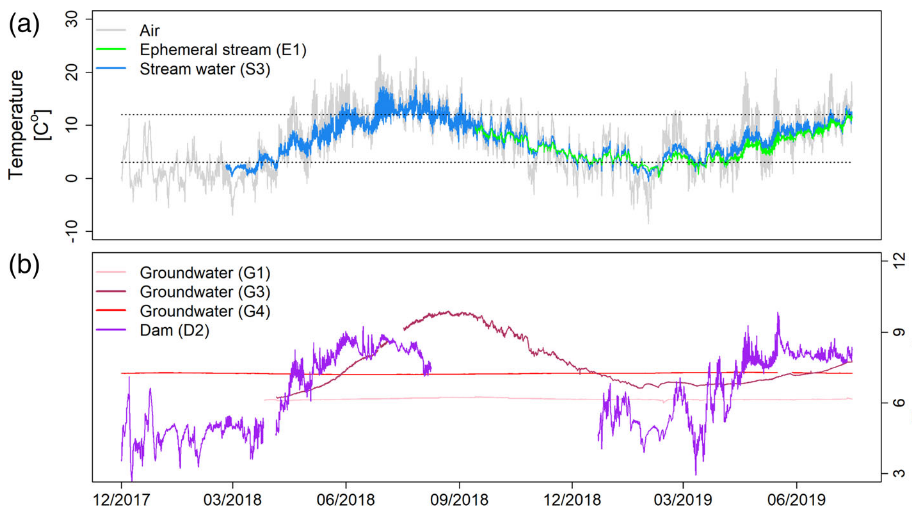

FIGURE 8 High frequency temperature recordings representative of each different source type monitored (a) air and streams, both perennial and ephemeral, with scale of (b) represented in (a) as black horizontal lines showing 3–12°C temperature range and (b) groundwaters and dammed surface water reservoirs. Gaps represent times when sources were drained (D2) or were dry (E1)

recovery of these storage deficits. Figure 3c shows that by the end of the study period (July-19) storage was still ~85 mm less than at the start (December-17). Nevertheless, when full hydrological years were considered, most of the storage deficits appeared to be replenished by October-19 (Table 4).

The impact of the drought conditions on water temperature varied with source type as observed in synoptic sampling (Figure 4; Table 3) and in the data from high temporal-frequency loggers, which captured the full extent of the temperature range and variability in sources (Figure 8). Continuous records showed stream (S3) and dammed surface water reservoir temperatures (D2) peaked in late July coinciding with heightened air temperatures. Shallower groundwater (G3) showed a delayed and damped response compared to streams, with highest temperatures recorded in September. However, shallower groundwater showed considerably more variability than deeper groundwaters, which remained stable throughout the full monitoring period (G1 and G5). This trend was also observed in the relationship between air and water temperatures of sources, where streams were more closely related to air temperatures (e.g., linear regression slope E1 = 0.58, R^2 = 0.83; S3 = 0.66, R^2 = 0.89) than dammed surface water reservoirs (D2 = 0.25, R^2 = 0.72), followed by shallow (slope G3 = 0.052, R^2 = 0.66) and deep groundwaters (G1 = 0.0013, R^2 = .16; G5 = 0.0006, R^2 = 0.054).

4 | DISCUSSION

4.1 | The 2018 drought and sampling period context

Compared to local 50-year records, the hydrological year 2017–2018 was characterised by below-average precipitation and above-average temperatures. This coincided with the River Livet experiencing some of the lowest July and August flows in 40 years of records (Figure 2b). In our SPI analysis, despite previous drought periods in the mid-1970s, 1990s, 2003 and 2015 (Hänsel et al., 2019; Spinoni, Naumann, Vogt, & Barbosa, 2015), only the drought in the mid-1970s was longer and more intense than the cumulative effects of two dry winters preceding the summer of 2018 (Smith, 1977). Whilst the other dry periods (SPI ranging between −2 and −1.5) had heatwaves, low river flows, and impacts on agriculture (Brunner et al., 2019; Fink et al., 2004; Ionita et al., 2017), they were considerably shorter. The 2018 summer period across Central and Northern Europe was a notably "hotter drought," and the UK was identified as one of the drought hotspots (Buras, Rammig, & Zang, 2019). In Scotland this drought also followed a prolonged period with frequent below-average rainfall anomalies (2012–2019), especially in the winters of 2016/17 and 2017/18. Therefore, the impact on and recovery of hydrological systems was more likely extended.

SPI is often a starting point for meteorological drought analysis (CEH, 2019; Hayes, Svoboda, Wall, & Widhalm, 2011; Svoboda & Fuchs, 2017). The resulting drought characteristics depend on averaging and return period choice, plus the length of study (Brunner et al., 2019). In addition, availability of long-term data for site-specific analysis can limit context (Barker et al., 2019). However local conditions at Glenlivet were assessed in context of the last 50 years. Although multivariate indicators using multiple drought characteristics have become popular (Hao & Singh, 2015), our simple method was well-suited to providing metrics closely associated with impacts of concern to stakeholders at the distillery (Svoboda & Fuchs, 2017). These included need of water of specific volumes and temperatures for processing and cooling (index focus on precipitation, temperature)

(Piggott, 2017) and production of discharge waters that need to be accommodated (index focus on River Livet flow) (SEPA, 2017). Additionally, in upland areas with limited catchment storage (such as the Cairngorms), the relationship between meteorological and hydrological drought is usually strongly correlated, with limited storage to buffer drought propagation (Barker et al., 2016; Haslinger, Koffler, Schöner, & Laaha, 2014).

4.2 | Use of tracers to evaluate variation in catchment water types

We found distinct differences in water quality characteristics between water source types, which reflected variations in storage and flow paths, both in space and time (Edmunds, Shand, Hart, & Ward, 2003; Hem, 1985). Most prominent differences were found between groundwater and ephemeral streams, while the perennial stream and reservoir water comprised mixtures of the two. Isotope analyses revealed that groundwaters were strongly mixed and had relatively low YWFs (i.e., most water was >2–3 months old) (Blumstock, Tetzlaff, Dick, Nuetzmann, & Soulsby, 2016; Kirchner, 2016b; Scheliga et al., 2017; von Freyberg et al., 2018), whereas signatures in ephemeral streams exhibited most variability and high YWFs (i.e., were dominated by water <2–3 months old) (Zimmer et al., 2013). This is consistent with the high alkalinity in groundwaters, which can be attributed to longer contact time in deeper storage, facilitating greater weathering and ion release (Cresser et al., 2000; Godsey, Kirchner, & Clow, 2009; Soulsby et al., 2007).

The role of soil and geological characteristics in determining these patterns has been shown in many tracer-based studies in Scotland (Blumstock et al., 2016; Geris et al., 2015; Scheliga et al., 2018; Soulsby et al., 1998), and elsewhere (Botter, Bertuzzo, & Rinaldo, 2011; Isokangas et al., 2019; Rinaldo et al., 2011; Tetzlaff et al., 2018; Zuecco, Penna, & Borga, 2018). Differences between sampling locations of the same water type can be explained by such variations. For example, S1 and G4 had much higher alkalinity than other stream and groundwater sites (Table 3), this is likely explained by geological heterogeneity (i.e., the limestone end-member associated with high alkalinity) and/or faults providing deeper groundwater sources (Edmunds et al., 2003). Similarly, variations in ephemeral stream water may also be related to sources, whereby E2 and E4 were more likely connected to shallow groundwater while E1 and E3 were more dominantly related to direct (near) surface runoff.

Overall, water types varied more in space than in time, with differences becoming more distinct during dry conditions as connectivity between end-members was reduced (Blumstock et al., 2015; Darling, Gooddy, Morris, & Peach, 2012). This enabled EMMA to be used as a first approximation assessment of groundwater contributions to S3. Identifying specific, distinct 'end-members' can be challenging (Barthold et al., 2011) as a result of geological heterogeneity, groundwater mixing, and non-conservative tracer behaviour (Benettin & van Breukelen, 2017; Pelizardi et al., 2017; Soulsby, Rodgers, et al., 2003). However, here use of multiple tracers helped to distinguish differences between sources and associated geological units, which explained results for groundwater contribution to the stream rather than different residence times (e.g., YWFs in groundwater are very similar). This ensured that, although EMMA is approximate, 'end-members' of source types delimited the range of values so that the mixing space of different source contributions to S3 could be understood (Abbott et al., 2016; Isokangas et al., 2019).

4.3 | Impacts of drought on catchment hydrological processes

Figure 9 conceptualises the key hydrological processes in Blairfindy catchment during drought conditions in comparison to average and wet states. The key flow paths are based on field observations and the tracer-based methods. Rainfall and snowmelt initiated transient runoff at all ephemeral stream sites, characterised by younger, low EC waters (light blue arrows) over relatively consistent well-mixed groundwater contribution to the main stream (dark blue arrows). Recharge to shallow, less alkaline groundwaters and deeper, highly alkaline sources in geological fractures was facilitated by infiltration on the more freely-draining upper hillslopes, although the stable composition suggested significant mixing along relatively long flow paths. More frequent (average) conditions occurred between precipitation events when drainage and ET led to unsaturated hillslope soils, and some ephemeral stream sites dried up (e.g., E1). Other ephemeral streams remained flowing in such conditions likely due to soil and (shallow) groundwater drainage, evidenced by higher EC, alkalinities and lower YWFs (notably E2, 4) than E1. All groundwaters sources continued to contribute. In drought conditions however, precipitation rapidly either evaporated or was transpired with no impact on the stream hydrograph. The other ephemeral streams (E2, 3, 4) also dried up, likely as groundwater levels dropped. Both high and low alkalinity groundwaters continued to contribute to flow. S1 at the head of the main Blairfindy stream also remained a source, and as surface water input decreased through the drought, S3 EC increased to values closer to those observed at S1 showing increasing dominance of groundwater from that of S1.

Our key findings are therefore that, through drought, groundwater (a) continued to maintain streamflow and (b) provided water at cool temperatures that buffered streamflow. This was encouraging in terms of maintaining ecosystem services and industrial abstractions (Folegot et al., 2018; Freire-González et al., 2017). End-member mixing showed that this groundwater contribution was 65–100% of streamflow throughout the drought, consistent with other Scottish studies, albeit not in such extreme droughts (Blumstock et al., 2015; Scheliga et al., 2018; Soulsby et al., 1998). This highlights: (a) the resilience of the streamflow quantity and quality (temperature) to the drought, (b) the importance of groundwater recharge to maintain this flow (Cuthbert et al., 2019; Haslinger et al., 2014; Segura et al., 2019), and (c) the fact that shallow sources with high YWFs are less resilient as they dried up (von Freyberg et al., 2018; Zimmer et al., 2013).

Although storage was sufficient to maintain streamflow throughout the drought, water balance data for 2017–2018 revealed a marked storage deficit. This was replenished due to above-average rainfall in 2018–2019, which included a wetter winter for the first time since 2015/16; though it was not until late summer/early autumn 2019 that storage deficits were replenished. However, the water balance estimates in the catchment are uncertain and previous investigation into the nature and role of deeper groundwater sources (e.g., G4) is also limited. One way to usefully complement our

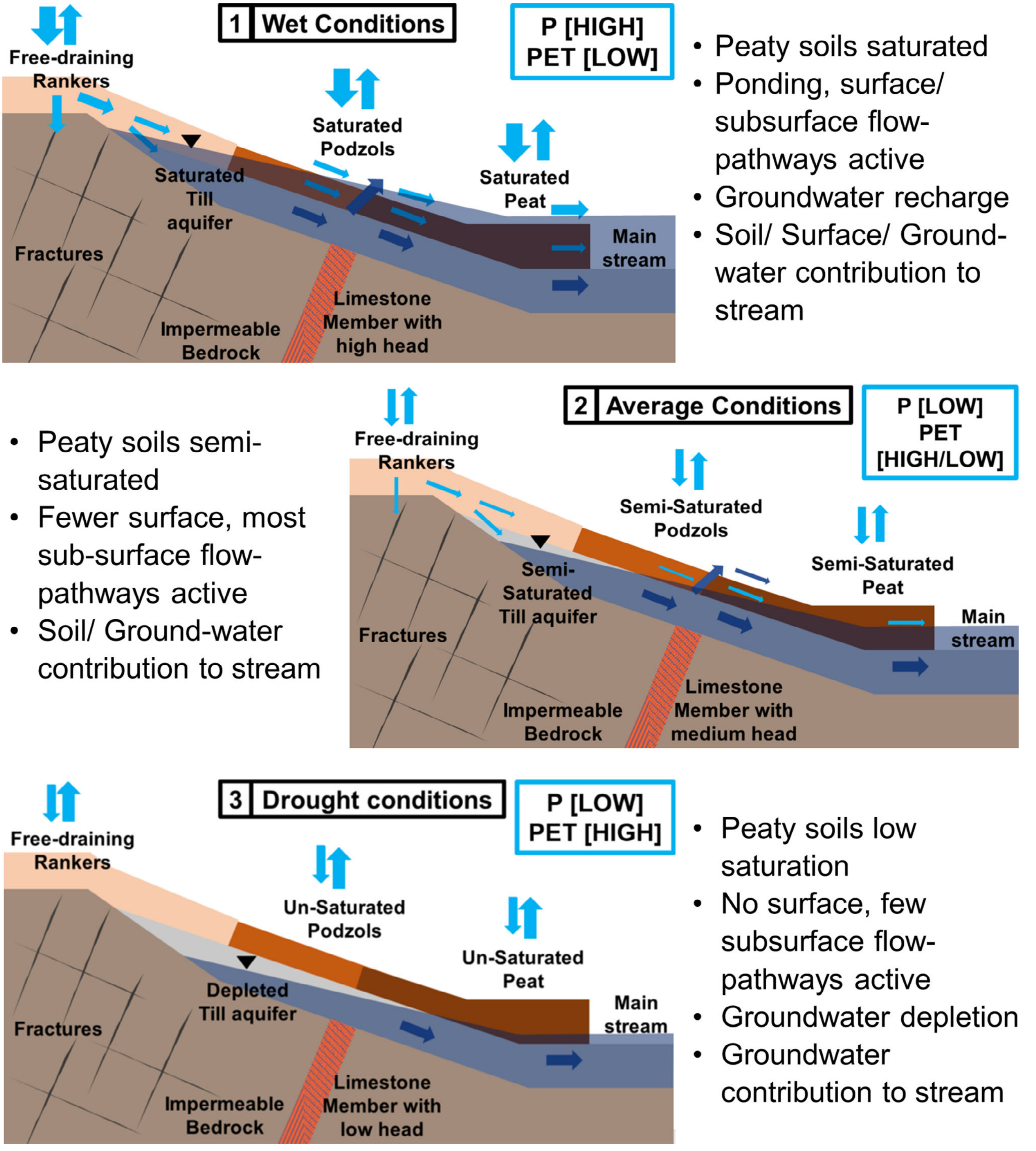

FIGURE 9 Conceptual Diagram derived from tracer-based methods and hydrological understanding of catchment, illustrating the presence and activation of flow pathways of different types of water as the catchment transitions from (1) a wet, surface-water dominated system to (2) an average, mixed surface/groundwater system to (3) drought conditions in groundwater dominated state. Light-blue arrows suggest younger water (precipitation/surface waters), darker blue arrows suggest older water (well-mixed groundwater). Black triangle denotes water table and Limestone member representative of different geological features with more/less permeable zones in crystalline geology enabling groundwater circulation with faults and groundwater springs to surface

preliminary work could be by geophysical surveys to characterise subsurface stores (Soulsby et al., 2016). Nevertheless, while our data suggested that the water balance for Blairfindy catchment at the end of the year 2017–2018 would have been negative regardless, this could have been mitigated with reduced industry abstractions. As drought events are likely to become more frequent, this suggests there is need for more informed water management, to improve sustainability of the resources available in future.

The high groundwater contributions (with stable temperatures) to the stream also helped buffer stream temperatures through the warm drought period (Snyder, Hitt, & Young, 2015). This was evident through comparison between datasets of high-frequency temperature recordings capturing full variability, rather than synoptic sampling, as this avoided issues with comparison between different approaches. The low variability in the stream compared to air temperatures, suggest that the narrow, short, north facing stream was relatively sheltered from radiation inputs (Dick, Tetzlaff, & Soulsby, 2015; Isokangas et al., 2019; MacDonald, Boon, Byrne, Robinson, & Rasmussen, 2014) allowing the imprint of groundwater temperatures to be maintained. This groundwater imprint was also observed in the 0.66 magnitude slope of the linear regression between air and S3 temperatures, which is relatively low given that this involves a small and shallow upland stream, and values up to 1 were observed by Arismendi et al. (2014) in Californian streams draining areas several orders of magnitude larger than Blairfindy at S3. However, a full assessment of the effects of factors such as altitude, aspect, vegetation shading, and riparian saturation (Dick et al., 2015) on stream temperatures would need energy balance modelling.

4.4 | Limitations

In this study we employed complementary hydrometric and temperature data with tracer analyses to investigate drought impacts on dominant water sources and their temperatures. However, certain limitations remain, though these are typical for most small-scale rural catchment studies (Soulsby, Tetzlaff, Dunn, & Waldron, 2006). Some of the issues relate to data availability. For example, we used hydrometeorological data from nearby stations to provide long-term context and to extend the time series pre-installation of the Glenlivet monitoring network. Although we scaled the data to consider variations in microclimate, topography and catchment size, some uncertainty remains, though the general picture is likely accurate. Additionally, despite the sampling throughout the period of interest being conducted at a relatively high temporal and spatial resolution; observations of all water balance components are inherently associated with uncertainty (Beven & Westerberg, 2011; McMillan, Krueger, & Freer, 2012). In particular, the abstraction data were obtained from aggregated distillery records, so that short-term variations in daily amounts were unknown. Even though this did not affect the overall annual water balance calculations, short-term uncertainty is likely. Higher frequency characterisation of variability could have been useful for several other parameters too. For example, high-frequency (i.e., sub-hourly) sampling has recently become more widely available for EC and isotopes (von Freyberg et al., 2017) and can provide additional insights, for example into source variations within an event (Kirchner, Feng, Neal, & Robson, 2004). However, as our aim was to capture the longer-term impacts and recovery of a catchment to drought, the longer study period was prioritised over high frequency sampling campaigns. We used EC as a tracer in EMMA with two groundwater end-members, which had relatively broad uncertainty bands. Nonetheless, the results from our complementary analyses have shown that this still provided a reasonable first approximation of groundwater contributions to the stream.

4.5 | Implications

The 2018 drought was an opportunity to study the implications of prolonged dry and warm conditions on water quantity, quality and temperature. Although we found that groundwater buffered stream flows and moderated water temperatures, projected trends to drier and warmer summers in Scotland may result in depletions of catchment storage becoming more common (Afzal & Ragab, 2019; Cuthbert et al., 2019; Gosling, 2014; Spinoni et al., 2018). Although these projections are also expected to be coupled with higher and more intense winter precipitation (Chan, Kahana, Kendon, & Fowler, 2018; Kay et al., 2014), the impacts of cumulative reoccurring dry periods could be significant, especially if groundwater recharge is affected (Cuthbert et al., 2019; Taylor et al., 2012; Van Lanen, Wanders, Tallaksen, & Van Loon, 2013). Generally, deeper aquifers take longer to respond to climatic fluctuations and may offer short-term resilience to low precipitation, maintaining streamflow with water of more stable temperatures longer than near-surface stores (Bovolo, Parkin, & Sophocleous, 2009; Shah, 2009). However, these longer response times also mean that groundwater takes longer to be replenished following droughts (Kundzewicz & Döll, 2009). In addition, if air temperatures continue to increase, particularly in the winter recharge period, groundwater temperatures may also increase (Figura, Livingstone, Hoehn, & Kipfer, 2011; Jyväsjärvi et al., 2015), reducing the potential for stream temperature buffering in future.

The gradual drying of catchments, climate change or anthropogenically induced, can affect water resource availability (Allen, Breshears, & McDowell, 2015; Buras et al., 2019; Seneviratne et al., 2010) and may lead to further cascades in ecological change and degradation (Allen et al., 2015; Stocker et al., 2019). Drought-induced soil deterioration such as the breakdown of organic matter in peat may lead to extensive carbon release (Cooper, Thoss, & Watson, 2007; Green et al., 2019; Worrall, Burt, & Adamson, 2006), and mobilisation to streams (Fenner & Freeman, 2011). Deterioration of soils may also affect physical properties linked to infiltration and recharge (Hueso, García, & Hernández, 2012; Toberman, Freeman, Evans, Fenner, & Artz, 2008). In drought, higher temperatures, changed biogeochemistry and longer hydraulic residence times (Figure 9), mean that deterioration of water quality is highly likely (Mosley, 2015). Aquatic ecosystem response to these multiple drivers can be difficult to predict (Brennan &

Collins, 2015), but such ecological changes need to be assessed conjunctively while maintaining water abstractions (Caissie, 2006; Ormerod, 2009). In regions such as Speyside in Scotland, environment and industry are highly dependent on maintaining these balances for the benefit of salmon and the production of whisky (Fabris, Malcolm, Buddendorf, & Soulsby, 2018), so likely changes need to be assessed to provide an evidence base for management (Isaak, Wollrab, Horan, & Chandler, 2012; Jackson, Fryer, Hannah, Millar, & Malcolm, 2018).

Abstraction and catchment management options have the potential to mitigate such impacts if designed to adapt to projected increases in precipitation, and reduce the impact of extended drought periods (Bouwer, 2002; Hewett, Wilkinson, Jonczyk, & Quinn, 2020). Storage and attenuation features, shown to affect flood peak discharge (Nicholson, O'Donnell, Wilkinson, & Quinn, 2020; Wilkinson, Quinn, & Welton, 2010) and groundwater recharge (Escalante et al., 2007), could help build resilience to flooding and droughts using both traditional engineering approaches (Volpi, Di Lazzaro, Bertola, Viglione, & Fiori, 2018) or nature-based solutions (NBS) (Nesshöver et al., 2017). NBS are "actions that aim to help societies address a variety of environmental, social and economic challenges in sustainable ways. They are actions which are inspired by, supported by or copied from nature" (Bauduceau et al., 2015). Applicable in rural or urban contexts, they are designed to address specific issues (e.g., fluvial or surface-water flooding) but often achieve multi-beneficial outcomes such as physical changes in water volumes, temperature, sediment and pollution control, but also improved water quality and biodiversity (Environment Agency, 2017; Calliari, Staccione, & Mysiak, 2019).

NBS in the UK focussed on water management have typically been applied in a flooding context (Lane, 2017; Nicholson, Wilkinson, O'Donnell, & Quinn, 2012; Wilkinson, Addy, Quinn, & Stutter, 2019) but may also be suitable for drought mitigation (Hartmann & Slavíková, 2019; Sapiano, Schembri, & Brincat, 2013). Measures may include the building of bunds or barriers to temporarily store water allowing time for infiltration, more commonly applied in small-scale managed aquifer recharge projects (Escalante, Sebastián Sauto, & Gil, 2019; Glendenning & Vervoort, 2011), or cross-slope planting to capture snow during winter (Kravčík et al., 2012). Furthermore, water temperature management may include localised riparian or hillslope planting (Jackson et al., 2018) offering shading of water sources. The efficacy of NBS however is likely very dependent on site conditions and location (Bouwer, 2002). Although the body of evidence is accumulating internationally for mitigation of drought (Calliari et al., 2019; Hartmann & Slavíková, 2019), particularly in India and Mediterranean countries (Sisoda, 2009; Stefan & Escalante, 2019), published work on the design and resultant impacts both in the short and long-term in the UK is not extensive (Environment Agency, 2018; Kabisch et al., 2016). As a result, further work developing the evidence base is required, both in terms of site-specific interventions and modelling studies and we aim to investigate these key requirements in future research.

At Blairfindy, NBS may provide one option for management and will benefit from the detailed knowledge of the role of soils, geology and hydrology reported here. With relatively responsive hydrology (Figure 9), opportunity for infiltration into soils lies mostly in the upper hillslopes (less peat, more rankers) where recharge is greatest. This could encourage further recharge into the active storage zone (till aquifer), and to connections with fractures and deeper groundwater sources. Our results have shown these sources provide a more stable, reliable contribution to streamflow. The water balance analysis has shown groundwater would benefit from informed management to improve resilience through drought periods. Hence, managing abstractions and potentially encouraging recharge to these sources through NBS could improve resilience to climate change in future.

5 | CONCLUSION

Drought conditions in summer 2018 had a significant impact on the quantity and quality (temperature) of water supplies to the Glenlivet distillery. Both in terms of duration and intensity, the drought provided an opportunity to investigate hydrological conditions that are likely to become more frequent in future. A tracer-based survey combined continuous measurements at the catchment outlet with synoptic sampling to identify dominant water sources and determine the impacts of the drought on them. Spatially distinct water sources could be identified with distinct hydrological and hydrochemical differences between water source types. End-member mixing analysis and young water fraction analyses showed that older groundwater maintained 65–100% of flow and more stable temperatures through the summer low flow period in the main stream.

Water balance analysis revealed that, although groundwater maintained streamflow throughout the drought, catchment storage was being depleted, and the hydrological year 2017–2018 ended with substantial water deficits. This was mostly replenished towards the end of 2018–2019 as a result of the years above average rainfall. With projected changes in climate suggesting more frequent drought events and warmer summer conditions, the effect on catchment storage dynamics will be a key relationship to maintaining water supplies.

Although our short study has limitations, it highlighted the importance of groundwater in maintaining resilient water supplies during an extreme drought period. Moreover, the results show potential opportunities to maintain water volumes and temperatures through informed groundwater management: this would involve combining storage and infiltration for groundwater recharge, through the use of NBS to improve resilience to hydro-climatic extremes. Although the evidence base for NBS is developing, pilot interventions and modelling are needed to develop site-specific evidence, to understand impacts at different locations and spatial scales. This research may be essential in improving the sustainability of industry in rural communities in upland catchments.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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REFERENCES

Abbott, B. W., Baranov, V., Mendoza-Lera, C., Nikolakopoulou, M., Harjung, A., Kolbe, T., … Pinay, G. (2016). Using multi-tracer inference to move beyond single-catchment ecohydrology. *Earth-Science Reviews*, *160*, 19–42. https://doi.org/10.1016/j.earscirev.2016.06.014

Afzal, M., & Ragab, R. (2019). Drought risk under climate and land use changes: Implication to water resource availability at catchment scale. *Water*, *11*(9), 1790. https://doi.org/10.3390/w11091790

Allen, C. D., Breshears, D. D., & McDowell, N. G. (2015). On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. *Ecosphere*, *6*(8), art129. https://doi.org/10.1890/ES15-00203.1

Allen R. G., Pereira L. S., Raes D., Smith M. 1998. Crop evapotranspiration: Guidelines for computing crop water requirements, FAO irrigation and drainage paper 56, Rome.

Arismendi, I., Safeeq, M., Dunham, J. B., & Johnson, S. L. (2014). Can air temperature be used to project influences of climate change on stream temperature? *Environmental Research Letters*, *9*(8), 084015. https://doi.org/10.1088/1748-9326/9/8/084015

Barker, L. J., Hannaford, J., Chiverton, A., & Svensson, C. (2016). From meteorological to hydrological drought using standardised indicators. *Hydrology and Earth System Sciences*, *20*(6), 2483–2505. https://doi.org/10.5194/hess-20-2483-2016

Barker, L. J., Hannaford, J., Parry, S., Smith, K. A., Tanguy, M., & Prudhomme, C. (2019). Historic hydrological droughts 1891-2015: Systematic characterisation for a diverse set of catchments across the UK. *Hydrology and Earth System Sciences*, *23*(11), 4583–4602. https://doi.org/10.5194/hess-23-4583-2019

Barthold, F. K., Tyralla, C., Schneider, K., Vaché, K. B., Frede, H.-G., & Breuer, L. (2011). How many tracers do we need for end member mixing analysis (EMMA)? A sensitivity analysis. *Water Resources Research*, *47*(8), 1–14. https://doi.org/10.1029/2011WR010604

Bauduceau, N., Berry, P., Cecchi, C., Elmqvist, T., Fernandez, M., Hartig, T., & Tack, J. (2015). *Towards an EU research and innovation policy agenda for nature-based solutions & re-naturing cities: Final report of the horizon 2020 expert group on' Nature-based solutions and re-naturing cities*, Luxembourg: Publications Office of the European Union. https://doi.org/10.2777/765301

Benettin, P., & van Breukelen, B. M. (2017). Decomposing the bulk electrical conductivity of Streamflow to recover individual solute concentrations at high frequency. *Environmental Science & Technology Letters*, *4*(12), 518–522. https://doi.org/10.1021/acs.estlett.7b00472

Beven, K., & Westerberg, I. (2011). On red herrings and real herrings: Disinformation and information in hydrological inference. *Hydrological Processes*, *25*(10), 1676–1680. https://doi.org/10.1002/hyp.7963

BGS. (2020). *BGS Geology 50K*. Retrieved from http://www.bgs.ac.uk/products/digitalmaps/DiGMapGB_50.html

Birkel, C., Tetzlaff, D., Dunn, S. M., & Soulsby, C. (2011). Using time domain and geographic source tracers to conceptualize streamflow generation processes in lumped rainfall-runoff models. *Water Resources Research*, *47*(2), W02515. https://doi.org/10.1029/2010WR009547

Blumstock, M., Tetzlaff, D., Dick, J. J., Nuetzmann, G., & Soulsby, C. (2016). Spatial organization of groundwater dynamics and streamflow response from different hydropedological units in a montane catchment. *Hydrological Processes*, *30*(21), 3735–3753. https://doi.org/10.1002/hyp.10848

Blumstock, M., Tetzlaff, D., Malcolm, I. A., Nuetzmann, G., & Soulsby, C. (2015). Baseflow dynamics: Multi-tracer surveys to assess variable groundwater contributions to montane streams under low flows. *Journal of Hydrology*, *527*, 1021–1033. https://doi.org/10.1016/j.jhydrol.2015.05.019

Borzi, G., Tanjal, C., Santucci, L., & Carol, E. (2019). Geochemical mechanisms controlling the isotopic and chemical composition of groundwater and surface water in a sector of the Pampean plain (Argentina). *Science of the Total Environment*, *683*, 455–469. https://doi.org/10.1016/j.scitotenv.2019.05.168

Botter, G., Bertuzzo, E., & Rinaldo, A. (2011). Catchment residence and travel time distributions: The master equation. *Geophysical Research Letters*, *38*(11), L11403. https://doi.org/10.1029/2011GL047666

Bouwer, H. (2002). Artificial recharge of groundwater: Hydrogeology and engineering. *Hydrogeology Journal*, *10*(1), 121–142. https://doi.org/10.1007/s10040-001-0182-4

Bovolo, C. I., Parkin, G., & Sophocleous, M. (2009). Groundwater resources, climate and vulnerability. *Environmental Research Letters*, *4*(3), 035001. https://doi.org/10.1088/1748-9326/4/3/035001

Bowen, G. J., & Good, S. P. (2015). Incorporating water isoscapes in hydrological and water resource investigations. *Wiley Interdisciplinary Reviews: Water*, *2*(2), 107–119. https://doi.org/10.1002/wat2.1069

Brennan, G., & Collins, S. (2015). Growth responses of a green alga to multiple environmental drivers. *Nature Climate Change*, *5*(9), 892–897. https://doi.org/10.1038/nclimate2682

Brunner, M. I., Liechti, K., & Zappa, M. (2019). Extremeness of recent drought events in Switzerland: Dependence on variable and return period choice. *Natural Hazards and Earth System Sciences*, *19*(10), 2311–2323. https://doi.org/10.5194/nhess-19-2311-2019

Buras, A., Rammig, A., & Zang, C. S. (2019). Quantifying impacts of the drought 2018 on European ecosystems in comparison to 2003. *Biogeosciences*, *17*(6), 1655–1672. https://doi.org/10.5194/bg-17-1655-2020

Caissie, D. (2006). The thermal regime of rivers: A review. *Freshwater Biology*, *51*(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Calliari, E., Staccione, A., & Mysiak, J. (2019). An assessment framework for climate-proof nature-based solutions. *Science of the Total Environment*, *656*, 691–700. https://doi.org/10.1016/j.scitotenv.2018.11.341

Capell, R., Tetzlaff, D., Essery, R., & Soulsby, C. (2014). Projecting climate change impacts on stream flow regimes with tracer-aided runoff models - preliminary assessment of heterogeneity at the mesoscale. *Hydrological Processes*, *28*(3), 545–558. https://doi.org/10.1002/hyp.9612

CEH. 2019. UK droughts tool. Droughts Tool. Retrieved February 26, 2020 from https://eip.ceh.ac.uk/droughts

Chan, S., Falkner, R., Goldberg, M., & van Asselt, H. (2018). Effective and geographically balanced? An output-based assessment of non-state climate actions. *Climate Policy*, *18*(1), 24–35. https://doi.org/10.1080/14693062.2016.1248343

Chan, S. C., Kahana, R., Kendon, E. J., & Fowler, H. J. (2018). Projected changes in extreme precipitation over Scotland and northern England using a high-resolution regional climate model. *Climate Dynamics*, *51*(9–10), 3559–3577. https://doi.org/10.1007/s00382-018-4096-4

Chiogna, G., Skrobanek, P., Narany, T. S., Ludwig, R., & Stumpp, C. (2018). Effects of the 2017 drought on isotopic and geochemical gradients in the Adige catchment, Italy. *Science of the Total Environment*, *645*, 924–936. https://doi.org/10.1016/j.scitotenv.2018.07.176

Collet, L., Harrigan, S., Prudhomme, C., Formetta, G., & Beevers, L. (2018). Future hot-spots for hydro-hazards in Great Britain: A probabilistic assessment. *Hydrology and Earth System Sciences*, *22*(10), 5387–5401. https://doi.org/10.5194/hess-22-5387-2018

Cooper, R., Thoss, V., & Watson, H. (2007). Factors influencing the release of dissolved organic carbon and dissolved forms of nitrogen from a small upland headwater during autumn runoff events. *Hydrological Processes*, *21*(5), 622–633. https://doi.org/10.1002/hyp.6261

Crabtree B., Macdonald D., Dunn S. 2002. Evaluating the economic impact of abstraction controls on high and medium volume water users in Scotland. Aberdeen.

Cresser, M. S., Smart, R., Billett, M. F., Soulsby, C., Neal, C., Wade, A., … Edwards, A. C. (2000). Modelling water chemistry for a major Scottish river from catchment attributes. *Journal of Applied Ecology*, *37*(s1), 171–184. https://doi.org/10.1046/j.1365-2664.2000.00525.x

Cuthbert, M. O., Gleeson, T., Moosdorf, N., Befus, K. M., Schneider, A., Hartmann, J., & Lehner, B. (2019). Global patterns and dynamics of climate–groundwater interactions. *Nature Climate Change*, *9*(2), 137–141. https://doi.org/10.1038/s41558-018-0386-4

Darling, W. G., Gooddy, D. C., Morris, B. L., & Peach, D. W. (2012). The hydrochemistry of a chalk aquifer during recovery from drought. *Quarterly Journal of Engineering Geology and Hydrogeology*, *45*(4), 473–486. https://doi.org/10.1144/qjegh2012-022

Dick, J. J., Tetzlaff, D., & Soulsby, C. (2015). Landscape influence on small-scale water temperature variations in a moorland catchment. *Hydrological Processes*, *29*(14), 3098–3111. https://doi.org/10.1002/hyp.10423

Dunn, S. M., & Mackay, R. (1995). Spatial variation in evapotranspiration and the influence of land use on catchment hydrology. *Journal of Hydrology*, *171*(1–2), 49–73. https://doi.org/10.1016/0022-1694(95)02733-6

Edmunds, W. M., Shand, P., Hart, P., & Ward, R. S. (2003). The natural (baseline) quality of groundwater: A UKpilot study. *Science of the Total Environment*, *310*(1–3), 25–35. https://doi.org/10.1016/S0048-9697(02)00620-4

Engel, M., Penna, D., Bertoldi, G., Dell'Agnese, A., Soulsby, C., & Comiti, F. (2016). Identifying run-off contributions during melt-induced run-off events in a glacierized alpine catchment. *Hydrological Processes*, *30*(3), 343–364. https://doi.org/10.1002/hyp.10577

Engel, M., Penna, D., Bertoldi, G., Vignoli, G., Tirler, W., & Comiti, F. (2019). Controls on spatial and temporal variability in streamflow and hydrochemistry in a glacierized catchment. *Hydrology and Earth System Sciences*, *23*(4), 2041–2063. https://doi.org/10.5194/hess-23-2041-2019

Environment Agency. 2017. Working with natural processes-the evidence base-project summary. Retrieved from https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/654429/Working_with_natural_processes_summary.pdf

Environment Agency. (2018). *Working with natural processes – Evidence directory*. Bristol: Environment Agency.

Escalante E., Prieto I., San M. A., González C., San Sebastián Sauto J. 2007. *DINA-MAR*. *project for management of aquifer recharge in the context of sustainable development*. *Lines of action and results*. ISMAR 7, Abu Dhabi.

Escalante, E. F., Sebastián Sauto, J. S., & Gil, R. C. (2019). Sites and indicators of MAR as a successful tool to mitigate climate change effects in Spain. *Water (Switzerland)*, *11*(9), 1943. https://doi.org/10.3390/w11091943

ESRI. (2020). *"World Imagery"*. Retrieved from http://www.arcgis.com/home/item.html?id=10df2279f9684e4a9f6a7f08febac2a9.

Fabris, L., Malcolm, I. A., Buddendorf, W. B., & Soulsby, C. (2018). Integrating process-based flow and temperature models to assess riparian forests and temperature amelioration in salmon streams. *Hydrological Processes*, *32*(6), 776–791. https://doi.org/10.1002/hyp.11454

Fenner, N., & Freeman, C. (2011). Drought-induced carbon loss in peatlands. *Nature Geoscience*, *4*(12), 895–900. https://doi.org/10.1038/ngeo1323

Figura, S., Livingstone, D. M., Hoehn, E., & Kipfer, R. (2011). Regime shift in groundwater temperature triggered by the Arctic oscillation. *Geophysical Research Letters*, *38*(23), L23401. https://doi.org/10.1029/2011GL049749

Fink, A. H., Brücher, T., Krüger, A., Leckebusch, G. C., Pinto, J. G., & Ulbrich, U. (2004). The 2003 European summer heatwaves and drought -synoptic diagnosis and impacts. *Weather*, *59*(8), 209–216. https://doi.org/10.1256/wea.73.04

Floriancic, M. G., Meerveld, I., Smoorenburg, M., Margreth, M., Naef, F., Kirchner, J. W., & Molnar, P. (2018). Spatio-temporal variability in contributions to low flows in the high alpine Poschiavino catchment. *Hydrological Processes*, *32*(26), 3938–3953. https://doi.org/10.1002/hyp.13302

Folegot, S., Hannah, D. M., Dugdale, S. J., Kurz, M. J., Drummond, J. D., Klaar, M. J., … Krause, S. (2018). Low flow controls on stream thermal dynamics. *Limnologica*, *68*(October 2017), 157–167. https://doi.org/10.1016/j.limno.2017.08.003

Forzieri, G., Feyen, L., Rojas, R., Flörke, M., Wimmer, F., & Bianchi, A. (2014). Ensemble projections of future streamflow droughts in Europe. *Hydrology and Earth System Sciences*, *18*(1), 85–108. https://doi.org/10.5194/hess-18-85-2014

Freire-González, J., Decker, C., & Hall, J. W. (2017). The economic impacts of droughts: A framework for analysis. *Ecological Economics*, *132*, 196–204. https://doi.org/10.1016/j.ecolecon.2016.11.005

Frisbee, M. D., Phillips, F. M., Campbell, A. R., Liu, F., & Sanchez, S. A. (2011). Streamflow generation in a large, alpine watershed in the southern Rocky Mountains of Colorado: Is streamflow generation simply the aggregation of hillslope runoff responses? *Water Resources Research*, *47*(6), 1–18. https://doi.org/10.1029/2010WR009391

Geris, J., Tetzlaff, D., & Soulsby, C. (2015). Resistance and resilience to droughts: Hydropedological controls on catchment storage and run-off response. *Hydrological Processes*, *29*(21), 4579–4593. https://doi.org/10.1002/hyp.10480

Glendenning, C. J., & Vervoort, R. W. (2011). Hydrological impacts of rainwater harvesting (RWH) in a case study catchment: The Arvari River, Rajasthan, India. Part 2. Catchment-scale impacts. *Agricultural Water Management*, *98*(4), 715–730. https://doi.org/10.1016/j.agwat.2010.11.010

Godsey, S. E., Kirchner, J. W., & Clow, D. W. (2009). Concentration-discharge relationships reflect chemostatic characteristics of US catchments. *Hydrological Processes*, *23*(13), 1844–1864. https://doi.org/10.1002/hyp.7315

Gosling, R. (2014). Assessing the impact of projected climate change on drought vulnerability in Scotland. *Hydrology Research*, *45*(6), 806–816. https://doi.org/10.2166/nh.2014.148

Gosling, R. D., Zaidman, M., Wann, M., & Rodgers, P. J. (2012). How low can you go? Using drought indices to protect environmental flows in Scottish rivers. *BHS Eleventh National Symposium, Hydrology for a changing world*, (1–6). Dundee: British Hydrological Society. Retrieved from: https://www.sepa.org.uk/media/163211/the-development-of-drought-indices.pdf

Green, J. K., Seneviratne, S. I., Berg, A. M., Findell, K. L., Hagemann, S., Lawrence, D. M., & Gentine, P. (2019). Large influence of soil moisture on long-term terrestrial carbon uptake. *Nature*, *565*(7740), 476–479. https://doi.org/10.1038/s41586-018-0848-x

Green, T. R., Taniguchi, M., Kooi, H., Gurdak, J. J., Allen, D. M., Hiscock, K. M., … Aureli, A. (2011). Beneath the surface of global change: Impacts of climate change on groundwater. *Journal of Hydrology*, *405*(3–4), 532–560. https://doi.org/10.1016/j.jhydrol.2011.05.002

Hänsel, S., Ustrnul, Z., Łupikasza, E., & Skalak, P. (2019). Assessing seasonal drought variations and trends over Central Europe. *Advances in Water Resources*, *127*, 53–75. https://doi.org/10.1016/j.advwatres.2019.03.005

Hao, Z., & Singh, V. P. (2015). Drought characterization from a multivariate perspective: A review. *Journal of Hydrology*, *527*, 668–678. https://doi.org/10.1016/j.jhydrol.2015.05.031

Hartmann, T., & Slavíková, L. (2019). In T. Hartmann, L. Slavíková, & S. McCarthy (Eds.), *Nature-based flood risk management on private land*. Cham: Springer International Publishing. https://doi.org/10.1007/978-3-030-23842-1

Haslinger, K., Koffler, D., Schöner, W., & Laaha, G. (2014). Exploring the link between meteorological drought and streamflow: Effects of climate-catchment interaction. *Water Resources Research*, *50*(3), 2468–2487. https://doi.org/10.1002/2013WR015051

Hayes, M., Svoboda, M., Wall, N., & Widhalm, M. (2011). The Lincoln declaration on drought indices: Universal meteorological drought index recommended. *Bulletin of the American Meteorological Society*, *92*(4), 485–488. https://doi.org/10.1175/2010BAMS3103.1

Hem, J. D. (1985). *Study and interpretation of the chemical characteristics of natural water* (Vol. 2254). Alexandria: Department of the Interior, US Geological Survey.

Hewett, C. J. M., Wilkinson, M. E., Jonczyk, J., & Quinn, P. F. (2020). Catchment systems engineering: An holistic approach to catchment management. *WIREs Water*, *7*(3), 1417. https://doi.org/10.1002/wat2.1417

Hueso, S., García, C., & Hernández, T. (2012). Severe drought conditions modify the microbial community structure, size and activity in amended and unamended soils. *Soil Biology and Biochemistry*, *50*, 167–173. https://doi.org/10.1016/j.soilbio.2012.03.026

Hugman, R., Stigter, T. Y., Monteiro, J. P., & Nunes, L. (2012). Influence of aquifer properties and the spatial and temporal distribution of recharge and abstraction on sustainable yields in semi-arid regions. *Hydrological Processes*, *26*(18), 2791–2801. https://doi.org/10.1002/hyp.8353

Ionita, M., Tallaksen, L. M., Kingston, D. G., Stagge, J. H., Laaha, G., Van Lanen, H. A. J., … Haslinger, K. (2017). The European 2015 drought from a climatological perspective. *Hydrology and Earth System Sciences*, *21*(3), 1397–1419. https://doi.org/10.5194/hess-21-1397-2017

Isaak, D. J., Wollrab, S., Horan, D., & Chandler, G. (2012). Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. *Climatic Change*, *113*(2), 499–524. https://doi.org/10.1007/s10584-011-0326-z

Isokangas, E., Ronkanen, A., Rossi, P. M., Marttila, H., & Kløve, B. (2019). A tracer-based method for classifying groundwater dependence in boreal headwater streams. *Journal of Hydrology*, *577*(May), 123762. https://doi.org/10.1016/j.jhydrol.2019.05.029

Jackson, F. L., Fryer, R. J., Hannah, D. M., Millar, C. P., & Malcolm, I. A. (2018). A spatio-temporal statistical model of maximum daily river temperatures to inform the management of Scotland's Atlantic salmon rivers under climate change. *Science of the Total Environment*, *612* (January), 1543–1558. https://doi.org/10.1016/j.scitotenv.2017.09.010

Jarvie, H. e. P., Neal, C. o., Smart, R. i., Owen, R. o., Fraser, D. e., Forbes, I. a., & Wade, A. n. (2001). Use of continuous water quality records for hydrograph separation and to assess short-term variability and extremes in acidity and dissolved carbon dioxide for the river Dee, Scotland. *Science of the Total Environment*, *265*(1–3), 85–98. https://doi.org/10.1016/S0048-9697(00)00651-3

Jyväsjärvi, J., Marttila, H., Rossi, P. M., Ala-Aho, P., Olofsson, B., Nisell, J., … Muotka, T. (2015). Climate-induced warming imposes a threat to north European spring ecosystems. *Global Change Biology*, *21*(12), 4561–4569. https://doi.org/10.1111/gcb.13067

Kabisch, N., Frantzeskaki, N., Pauleit, S., Naumann, S., Davis, M., Artmann, M., … Bonn, A. (2016). Nature-based solutions to climate change mitigation and adaptation in urban areas: Perspectives on indicators, knowledge gaps, barriers, and opportunities for action. *Ecology and Society*, *21*(2), art39. https://doi.org/10.5751/ES-08373-210239

Kay, A. L., Crooks, S. M., Davies, H. N., & Reynard, N. S. (2014). Probabilistic impacts of climate change on flood frequency using response surfaces II: Scotland. *Regional Environmental Change*, *14*(3), 1243–1255. https://doi.org/10.1007/s10113-013-0564-x

Kirchner, J. W. (2016a). Aggregation in environmental systems-part 2: Catchment mean transit times and young water fractions under hydrologic nonstationarity. *Hydrology and Earth System Sciences*, *20*(1), 299–328. https://doi.org/10.5194/hess-20-299-2016

Kirchner, J. W. (2016b). 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

Kirchner, J. W., Feng, X., Neal, C., & Robson, A. J. (2004). The fine structure of water-quality dynamics: The(high-frequency) wave of the future. *Hydrological Processes*, *18*(7), 1353–1359. https://doi.org/10.1002/hyp.5537

Klaus, J., & McDonnell, J. J. (2013). Hydrograph separation using stable isotopes: Review and evaluation. *Journal of Hydrology*, *505*, 47–64. https://doi.org/10.1016/j.jhydrol.2013.09.006

Kravčík, M., Kohutiar, J., Gazovic, M., Kovac, M., Hrib, M., Suty, P., & Kravcikova, D. (2012). *After us, the desert and the deluge?* Slovakia: Michal Kravík/NGO People and Water.

Kundzewicz, Z. W., & Döll, P. (2009). Will groundwater ease freshwater stress under climate change? *Hydrological Sciences Journal*, *54*(4), 665–675. https://doi.org/10.1623/hysj.54.4.665

Lake, P. S. (2003). Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology*, *48*(7), 1161–1172. https://doi.org/10.1046/j.1365-2427.2003.01086.x

Lane, S. N. (2017). Natural flood management. *Wiley Interdisciplinary Reviews: Water*, *4*(3), e1211. https://doi.org/10.1002/wat2.1211

Litt, G. F., Gardner, C. B., Ogden, F. L., & Lyons, W. B. (2015). Hydrologic tracers and thresholds: A comparison of geochemical techniques for event-based stream hydrograph separation and flowpath interpretation across multiple land covers in the Panama Canal watershed. *Applied Geochemistry*, *63*, 507–518. https://doi.org/10.1016/j.apgeochem.2015.04.003

Loinaz, M. C., Kampp, H., Butts, M., & Bauer-gottwein, P. (2013). Integrated flow and temperature modeling at the catchment scale. *Journal of Hydrology*, *495*, 238–251. https://doi.org/10.1016/j.jhydrol.2013.04.039

MacDonald, R. J., Boon, S., Byrne, J. M., Robinson, M. D., & Rasmussen, J. B. (2014). Potential future climate effects on mountain hydrology, stream temperature, and native salmonid life history. *Canadian Journal of Fisheries and Aquatic Sciences*, *71*(2), 189–202. https://doi.org/10.1139/cjfas-2013-0221

Marchina, C., Bianchini, G., Natali, C., Pennisi, M., Colombani, N., Tassinari, R., & Knoeller, K. (2015). The Po river water from the Alps to the Adriatic Sea (Italy): New insights from geochemical and isotopic (δ18O-δD) data. *Environmental Science and Pollution Research*, *22*(7), 5184–5203. https://doi.org/10.1007/s11356-014-3750-6

Marchina, C., Natali, C., Fazzini, M., Fusetti, M., Tassinari, R., & Bianchini, G. (2017). Extremely dry and warm conditions in northern Italy during the year 2015: Effects on the Po river water. *Rendiconti Lincei*, *28*(2), 281–290. https://doi.org/10.1007/s12210-017-0596-0

Marsh, T., & Anderson, J. (2002). Assessing the water resources of Scotland—Perspectives, progress and problems. *Science of the Total Environment*, *294*(1–3), 13–27. https://doi.org/10.1016/S0048-9697(02)00049-9

McGlynn, B. L., & McDonnell, J. J. (2003). Quantifying the relative contributions of riparian and hillslope zones to catchment runoff. *Water Resources Research*, *39*(11), 1310. https://doi.org/10.1029/2003WR002091

McMillan, H., Krueger, T., & Freer, J. (2012). Benchmarking observational uncertainties for hydrology: Rainfall, river discharge and water quality. *Hydrological Processes*, *26*(26), 4078–4111. https://doi.org/10.1002/hyp.9384

Met Office. 2019a. Dataset MIDAS open: UK daily rainfall data. NCAS British Atmospheric Data Centre.

Met Office. 2019b. Dataset MIDAS: UK daily temperature data. NCAS British Atmospheric Data Centre. Retrieved from http://catalogue.ceda.ac.uk/uuid/1bb479d3b1e38c339adb9c82c15579d8

Mosley, L. M. (2015). Drought impacts on the water quality of freshwater systems; review and integration. *Earth-Science Reviews*, *140*, 203–214. https://doi.org/10.1016/j.earscirev.2014.11.010

National River Flow Archive. 2019. Gauged daily flow data: Livet at Minmore. UK Centre for Ecology and Hydrology (UKCEH). Retrieved February 26, 2020 from https://nrfa.ceh.ac.uk

Neal, C., Hill, T., Hill, S., & Reynolds, B. (1997). Acid neutralization capacity measurements in surface and ground waters in the Upper River Severn, Plynlimon: From hydrograph splitting to water flow pathways. *Hydrology and Earth System Sciences*, *1*(3), 687–696. https://doi.org/10.5194/hess-1-687-1997

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., … Wittmer, H. (2017). The science, policy and practice of nature-based solutions: An interdisciplinary perspective. *Science of the Total Environment*, *579*, 1215–1227. https://doi.org/10.1016/j.scitotenv.2016.11.106

Nicholson, A. R., O'Donnell, G. M., Wilkinson, M. E., & Quinn, P. F. (2020). The potential of runoff attenuation features as a natural flood management approach. *Journal of Flood Risk Management*, *13*(S1), 1–14. https://doi.org/10.1111/jfr3.12565

Nicholson, A. R., Wilkinson, M. E., O'Donnell, G. M., & Quinn, P. F. (2012). Runoff attenuation features: A sustainable flood mitigation strategy in the Belford catchment, UK. *Area*, *44*(4), 463–469. https://doi.org/10.1111/j.1475-4762.2012.01099.x

Ormerod, S. J. (2009). Climate change, river conservation and the adaptation challenge. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *19*(6), 609–613. https://doi.org/10.1002/aqc.1062

Pelizardi, F., Bea, S. A., Carrera, J., & Vives, L. (2017). Identifying geochemical processes using end member mixing analysis to decouple chemical components for mixing ratio calculations. *Journal of Hydrology*, *550*, 144–156. https://doi.org/10.1016/j.jhydrol.2017.04.010

Penna, D., van Meerveld, H. J., Zuecco, G., Dalla Fontana, G., & Borga, M. (2016). Hydrological response of an alpine catchment to rainfall and snowmelt events. *Journal of Hydrology*, *537*, 382–397. https://doi.org/10.1016/j.jhydrol.2016.03.040

Piggott, J. R. (2017). Whisky. In *Current developments in biotechnology and bioengineering* (pp. 435–450). Amsterdam, Netherlands; Oxford, UK and Cambridge, MA: Elsevier.

Rinaldo, A., Beven, K. J., Bertuzzo, E., Nicotina, L., Davies, J., Fiori, A., … Botter, G. (2011). Catchment travel time distributions and water flow in soils. *Water Resources Research*, *47*(7), 1–13. https://doi.org/10.1029/2011WR010478

Robins, N. (2002). Groundwater quality in Scotland: Major ion chemistry of the key groundwater bodies. *Science of the Total Environment*, *294* (1–3), 41–56. https://doi.org/10.1016/S0048-9697(02)00051-7

Sample, J. E., Baber, I., & Badger, R. (2016). A spatially distributed risk screening tool to assess climate and land use change impacts on water-related ecosystem services. *Environmental Modelling and Software*, *83*, 12–26. https://doi.org/10.1016/j.envsoft.2016.05.011

Sapiano M., Schembri M., Brincat C. 2013. *Natural water retention measures*: *Aquifer recharge in Malta*. MEDIWAT final conference proceedingsEuropean Commission, Palermo. Retrieved January 15, 2020 from http://nwrm.eu/case-study/aquifer-recharge-malta

Scheliga, B., Tetzlaff, D., Nuetzmann, G., & Soulsby, C. (2017). Groundwater isoscapes in a montane headwater catchment show dominance of well-mixed storage. *Hydrological Processes*, *31*(20), 3504–3519. https://doi.org/10.1002/hyp.11271

Scheliga, B., Tetzlaff, D., Nuetzmann, G., & Soulsby, C. (2018). Groundwater dynamics at the hillslope–riparian interface in a year with extreme winter rainfall. *Journal of Hydrology*, *564*(July), 509–528. https://doi.org/10.1016/j.jhydrol.2018.06.082

Scottish Executive Environment Group. 2005. The water environment (controlled activities) (Scotland) regulations 2005: Polcy Statement and regulatory impact assessment.

Segura, C., Noone, D., Warren, D., Jones, J. A., Tenny, J., & Ganio, L. M. (2019). Climate, landforms, and geology affect Baseflow sources in a mountain catchment. *Water Resources Research*, *55*(7), 5238–5254. https://doi.org/10.1029/2018WR023551

Seneviratne, S. I., Corti, T., Davin, E. L., Hirschi, M., Jaeger, E. B., Lehner, I., … Teuling, A. J. (2010). Investigating soil moisture–climate interactions in a changing climate: A review. *Earth-Science Reviews*, *99*(3–4), 125–161. https://doi.org/10.1016/j.earscirev.2010.02.004

SEPA. 2017. CAR Regulations 2011 (as amended) A Practical Guide practical guide Briefing note. 2011 (June)

SEPA. (2019). *Significant water management issues for Scotland*. Stirling. SEPA.

Shah, T. (2009). Climate change and groundwater: India's opportunities for mitigation and adaptation. *Environmental Research Letters*, *4*(3), 035005. https://doi.org/10.1088/1748-9326/4/3/035005

Smith, K. (1977). Water resource management in Scotland. *Scottish Geographical Magazine*, *93*(2), 66–79. https://doi.org/10.1080/00369227708736364

Snyder, C. D., Hitt, N. P., & Young, J. A. (2015). Accounting for groundwater in stream fish thermal habitat responses to climate change. *Ecological Applications*, *25*(5), 1397–1419.

Soil Survey of Scotland Staff, (1981). *Soil maps of Scotland at a scale of 1: 250 000. Macaulay Institute for Soil Research, Aberdeen*.

Soulsby C, Bradford J, Dick J, P. McNamara J, Geris J, Lessels J, Blumstock M, Tetzlaff D. 2016. Using geophysical surveys to test tracer-based storage estimates in headwater catchments. *Hydrological Processes 30* (23): 4434–4445 DOI: https://doi.org/10.1002/hyp.10889

Soulsby, C., Chen, M., Ferrier, R. C., Helliwell, R. C., Jenkins, A., & Harriman, R. (1998). Hydrogeochemistry of shallow groundwater in an upland Scottish catchment. *Hydrological Processes*, *12*(7), 1111–1127. https://doi.org/10.1002/(SICI)1099-1085(19980615)12:7<1111::AID-HYP633>3.0.CO;2-2

Soulsby, C., Petry, J., Brewer, M. J., Dunn, S. M., Ott, B., & Malcolm, I. A. (2003). Identifying and assessing uncertainty in hydrological pathways: A novel approach to end member mixing in a Scottish agricultural catchment. *Journal of Hydrology*, *274*(1–4), 109–128. https://doi.org/10.1016/S0022-1694(02)00398-0

Soulsby, C., Rodgers, P., Smart, R., Dawson, J., & Dunn, S. (2003). A tracer-based assessment of hydrological pathways at different spatial scales in a mesoscale Scottish catchment. *Hydrological Processes*, *17*(4), 759–777. https://doi.org/10.1002/hyp.1163

Soulsby, C., Tetzlaff, D., Dunn, S. M., & Waldron, S. (2006). Scaling up and out in runoff process understanding: Insights from nested experimental catchment studies. *Hydrological Processes*, *20*(11), 2461–2465. https://doi.org/10.1002/hyp.6338

Soulsby, C., Tetzlaff, D., van den Bedem, N., Malcolm, I. A., Bacon, P. J., & Youngson, A. F. (2007). Inferring groundwater influences on surface water in montane catchments from hydrochemical surveys of springs and streamwaters. *Journal of Hydrology*, *333*(2–4), 199–213. https://doi.org/10.1016/j.jhydrol.2006.08.016

Spinoni, J., Naumann, G., Vogt, J. V., & Barbosa, P. (2015). The biggest drought events in Europe from 1950 to 2012. *Journal of Hydrology: Regional Studies*, *3*, 509–524. https://doi.org/10.1016/j.ejrh.2015.01.001

Spinoni, J., Vogt, J. V., Naumann, G., Barbosa, P., & Dosio, A. (2018). Will drought events become more frequent and severe in Europe? *International Journal of Climatology*, *38*(4), 1718–1736. https://doi.org/10.1002/joc.5291

Stefan C., Escalante E. F. 2019. *A whole day on MAR managed aquifer recharge as a real climate change adaptation mechanism*. *Examples and indicators from five countries*. Accelerating action to tackle water pollution and enhance EU preparedness to water-related climate change impacts Zaragoza, Spain. pp. 27–31.

Stocker, B. D., Zscheischler, J., Keenan, T. F., Prentice, I. C., Seneviratne, S. I., & Peñuelas, J. (2019). Drought impacts on terrestrial primary production underestimated by satellite monitoring. *Nature Geoscience*, *12*(4), 264–270. https://doi.org/10.1038/s41561-019-0318-6

Stocker, T. F., Dahe, Q., Plattner, G.-K., Alexander, L. V., Allen, S. K., Bindoff, N. L., et al. (2013). Technical summary. In *Climate Change 2013-The Physical Science Basis, intergovernmental panel on Climate Change* (pp. 31–116). Cambridge: Cambridge University Press.

Svensson, C., Hannaford, J., & Prosdocimi, I. (2017). Statistical distributions for monthly aggregations of precipitation and streamflow in drought indicator applications. *Water Resources Research*, *53*(2), 999–1018. https://doi.org/10.1002/2016WR019276

Svoboda, M., & Fuchs, B. (2017). *Handbook of drought indicators and indices* (pp. 155–208). Geneva, Switzerland: World Meteorological Organization. https://doi.org/10.1201/9781315265551-12

Tarun Bharat Sangh, (2009). *25 years of evolution: Restoring life and hope to a barren land*. Alwar, Rajasthan: Tarun Bharat Sangh.

Taylor, R. G., Scanlon, B., Döll, P., Rodell, M., van Beek, R., Wada, Y., … Treidel, H. (2012). Ground water and climate change. *Nature Climate Change*, *3*(4), 322–329. https://doi.org/10.1038/nclimate1744

Tetzlaff, D., Piovano, T., Ala-Aho, P., Smith, A., Carey, S. K., Marsh, P., … Soulsby, C. (2018). Using stable isotopes to estimate travel times in a data-sparse Arctic catchment: Challenges and possible solutions. *Hydrological Processes*, *32*(12), 1936–1952. https://doi.org/10.1002/hyp.13146

Tigkas, D., Vangelis, H., & Tsakiris, G. (2015). DrinC: A software for drought analysis based on drought indices. *Earth Science Informatics*, *8*(3), 697–709. https://doi.org/10.1007/s12145-014-0178-y

Toberman, H., Freeman, C., Evans, C., Fenner, N., & Artz, R. R. E. (2008). Summer drought decreases soil fungal diversity and associated phenol oxidase activity in upland Calluna heathland soil. *FEMS Microbiology Ecology*, *66*(2), 426–436. https://doi.org/10.1111/j.1574-6941.2008.00560.x

Van Lanen, H. A. J., Wanders, N., Tallaksen, L. M., & Van Loon, A. F. (2013). Hydrological drought across the world: Impact of climate and physical catchment structure. *Hydrology and Earth System Sciences*, *17*(5), 1715–1732. https://doi.org/10.5194/hess-17-1715-2013

Vanplantinga, A. A., Grossman, E. L., & Roark, E. B. (2017). Chemical and isotopic tracer evaluation of water mixing and evaporation in a dammed Texas River during drought. *River Research and Applications*, *33*(3), 450–460. https://doi.org/10.1002/rra.3080

Volpi, E., Di Lazzaro, M., Bertola, M., Viglione, A., & Fiori, A. (2018). Reservoir effects on flood peak discharge at the catchment scale. *Water Resources Research*, *54*(11), 9623–9636. https://doi.org/10.1029/2018WR023866

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

von Freyberg, J., Studer, B., & Kirchner, J. W. (2017). A lab in the field: High-frequency analysis of water quality and stable isotopes in stream water and precipitation. *Hydrology and Earth System Sciences*, *21*(3), 1721–1739. https://doi.org/10.5194/hess-21-1721-2017

Wenger, S. J., Isaak, D. J., Luce, C. H., Neville, H. M., Fausch, K. D., Dunham, J. B., … Williams, J. E. (2011). Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. *Proceedings of the National Academy of Sciences*, *108*(34), 14175–14180. https://doi.org/10.1073/pnas.1103097108

Wilkinson, M., Stutter, M., & Gunter, D. (2016). *Feasibility assessment on land management for spring source optimisation at Glenlivet distillery*. Aberdeen, Dundee.James Hutton Institute.

Wilkinson, M. E., Addy, S., Quinn, P. F., & Stutter, M. (2019). Natural flood management: Small-scale progress and larger-scale challenges. *Scottish Geographical Journal*, *135*(1–2), 23–32. https://doi.org/10.1080/14702541.2019.1610571

Wilkinson, M. E., Quinn, P. F., & Welton, P. (2010). Runoff management during the September 2008 floods in the Belford catchment, Northumberland. *Journal of Flood Risk Management*, *3*(4), 285–295. https://doi.org/10.1111/j.1753-318X.2010.01078.x

Winter, T. C. (2007). The role of ground water in generating Streamflow in headwater areas and in Maintaining Base Flow1. *JAWRA Journal of the American Water Resources Association*, *43*(1), 15–25. https://doi.org/10.1111/j.1752-1688.2007.00003.x

World Meteorological Organization (2012). In M. Svoboda, M. Hayes, & D. Wood (Eds.), *Standardized precipitation index: User guide*. Geneva: World Meteorological Organization.

Worrall, F., Burt, T. P., & Adamson, J. K. (2006). Trends in drought frequency – The fate of DOC export from British Peatlands. *Climatic Change*, *76*(3–4), 339–359. https://doi.org/10.1007/s10584-006-9069-7

Wright, S. N., & Novakowski, K. S. (2019). Groundwater recharge, flow and stable isotope attenuation in sedimentary and crystalline fractured rocks: Spatiotemporal monitoring from multi-level wells. *Journal of Hydrology*, *571*(January), 178–192. https://doi.org/10.1016/j.jhydrol.2019.01.028

Wu, H., Li, J., Song, F., Zhang, Y., Zhang, H., Zhang, C., & He, B. (2018). Spatial and temporal patterns of stable water isotopes along the Yangtze River during two drought years. *Hydrological Processes*, *32*(1), 4–16. https://doi.org/10.1002/hyp.11382

Zimmer, M. A., Bailey, S. W., McGuire, K. J., & Bullen, T. D. (2013). Fine scale variations of surface water chemistry in an ephemeral to perennial drainage network. *Hydrological Processes*, *27*(24), 3438–3451. https://doi.org/10.1002/hyp.9449

Zomlot, Z., Verbeiren, B., Huysmans, M., & Batelaan, O. (2015). Spatial distribution of groundwater recharge and base flow: Assessment of controlling factors. *Journal of Hydrology: Regional Studies*, *4*, 349–368. https://doi.org/10.1016/j.ejrh.2015.07.005

Zuecco, G., Penna, D., & Borga, M. (2018). Runoff generation in mountain catchments: Long-term hydrological monitoring in the Rio Vauz catchment, Italy. *Cuadernos de Investigación Geográfica*, *44*(2), 397. https://doi.org/10.18172/cig.3327

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