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## Assessing the role of location and scale of Nature Based Solutions for the enhancement of low flows

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### ABSTRACT

Water resources management during drought is a significant challenge worldwide, particularly for upland areas. Additionally, variations in water availability are becoming more extreme with climate change. Nature Based Solutions (NBS) e.g. Runoff Attenuation Features (RAFTs) could provide an alternative to hard-engineering. Using more natural processes, flow pathways are intercepted and attenuated in features during wet periods, increasing infiltration opportunity and thus water availability for use later. NBS research has primarily focused on flood mitigation, but little is known about low flow impacts; knowledge is required on where and at what scale to implement NBS. To explore these questions, we used a physically-based catchment model (MIKE SHE) integrated with a hydraulic river model (MIKE 11) to evaluate scenarios with varying RAFT volumes and locations. We applied this to an intensively monitored upland Scottish catchment (0.9 km<sup>2</sup>) where 40 RAFTs (~2m<sup>3</sup> storage each) were installed for low flow enhancement. Model results showed installed RAFTs increase recharge (~0.1%), groundwater contribution to streamflow (~4%) and low flows (~1%) and reduce high (~5%) and mean flows (~2%), suggesting RAFTs could be used to mitigate extreme flows. The scenarios revealed that RAFT location (primarily soil type) and scale (total storage volume and spread of features) were both important. Doubling installed RAFT volumes increased impact on low flows by ~25% and high flows by ~40%, although lower additional benefits were predicted with further storage increases. RAFTs had greater impact in freely-draining soils than poorly-draining, however distributing the same storage volume across many smaller RAFTs over greater areas (both soil types) provided the largest effect. Absolute changes observed were relatively small, and given model uncertainty, should be treated with caution. Nevertheless, the direction of change was clear and given ecological systems and water supply rely on small margins of change, even slight increases in low flows will likely be beneficial.

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### KEYWORDS

Drought management; low flows; water resources management; groundwater recharge; Runoff Attenuation Features; MIKE SHE

## 1. Introduction

Management of water resources through drought can be a significant challenge, often with wide-reaching impacts (Stahl et al., 2016; Svoboda & Fuchs, 2017). Even under relatively humid climate conditions, such as in the UK, many areas are now considered ‘water-stressed’. Upland rural areas in particular are forced to balance water supply between local communities, the environment and food and drink industries (Visser-Quinn et al., 2021). In 2018 a serious drought occurred throughout Europe, with river flows amongst the lowest on record (Fennell et al., 2020; Soulsby et al., 2021). These conditions suggest water management solutions are needed (Visser-Quinn et al., 2021) and with climate projections for increased frequency of extremes (floods and droughts) and less snow (Spinoni et al., 2018; Stocker et al., 2013; Werritty & Sugden, 2012), this becomes an ever more pressing issue.

Historically, a broad range of approaches have been implemented to improve drought-preparedness and resilience, e.g. through early warning systems (Hund et al., 2018), reservoirs (Brunner et al., 2019), flood water harvesting (Kowsar, 2008) and as recognized more recently, Nature Based Solutions (NBS) (Cohen-Shacham et al., 2016; Neshhöver et al., 2017; OECD, 2020; Somers et al., 2018; UNCCD, 2019).

NBS are defined by the International Union for Conservation of Nature as ‘actions to protect, sustainably manage and restore natural and modified ecosystems in ways that address societal challenges effectively and adaptively, to provide both human well-being and biodiversity benefits (Cohen-Shacham et al., 2016). There are three key concepts of NBS for flow management: (1) reducing rapid runoff generation; (2) enhancing storage/retention of water or (3) reducing conveyance (Lane, 2017). Examples include afforestation, buffer strips, river restoration, and Runoff Attenuation Features (RAFTs) (Burgess-Gamble et al., 2017). RAFTs are soft-engineering approaches in the landscape, such as leaky barriers, designed to intercept and attenuate runoff, whilst increasing potential for infiltration and sub-surface storage, thereby addressing the three key NBS flow management concepts (Hewett et al., 2020; Metcalfe et al., 2017a; Wilkinson et al., 2010).

Although the knowledge base has developed in the last 10–20 years, NBS for water resources management is still a relatively immature research field with many remaining questions. Consequently, there is an urgent need for more evidence from empirical, modelling, and integrated studies in diverse locations (Burgess-Gamble et al., 2017; Cooper, 2020; Whelchel et al., 2018). One key knowledge gap involves NBS impacts on recharge and low flows (Sahani et al., 2019);

Simpson et al., 2016). Research on NBS for water management has so far primarily focussed on flood mitigation as Natural Flood Management (NFM), but the application to low flow management is now increasingly being considered (Burgess-Gamble et al., 2017; European Commission, 2020; Sahani et al., 2019; Simpson et al., 2016). Some examples of drought management approaches exist globally for water-stressed areas with dry climates, although not always under the guise of NBS (Standen et al., 2020). These studies have successfully shown increases in recharge and streamflow from small-scale Rain Water Harvesting (RWH) features similar to RAFs, through to large-scale reservoir recharge approaches (Dashora et al., 2018; Kravčík et al., 2012; Sisodia, 2009; Van Steenberg et al., 2011; Wilkinson, 2019). These technologies have evolved over millennia (Ochoa-Tocachi et al., 2019), are adapted to different settings and classed as those designed to attenuate or store water at the ground surface (e.g. RWH), or in the sub-surface (e.g. Managed Aquifer Recharge) (Dillon, 2005; Pavelic et al., 2012). Knowledge on applications for wetter climates, where NBS could potentially address both flood and drought problems, is lacking.

A second key knowledge gap relates to the optimal placement and scale of implementation (Reaney, 2022; Wilkinson et al., 2019), i.e. where to locate features and whether it is best to have many small or few large features. In addition to the storage potential of features, the spatial scale of NBS also relates to the spread of implementation within the catchment and the proportional area affected. Overall, better understanding is required on the relative roles of different landscape characteristics, and the location, size and distribution of implementations in relation to their impact on catchment processes. Such knowledge is both important for cost–benefit analysis for broader uptake and for optimal performance (Cohen-Shacham et al., 2019; Nelson et al., 2020). Consequences of poor placement can then be avoided e.g. increased flood peaks from synchronization in a high flow mitigation context (Hankin et al., 2017; Wilkinson et al., 2019), or greater evaporative losses from poor infiltration in a low flow mitigation context (Kumar et al., 2006; Shanafield & Cook, 2014; Somers et al., 2018).

Geophysics, modelling or tracer-based methods can be used to locate sites for optimal NBS placement (Shivanna et al., 2008; von Freyberg et al., 2015), and predict the direction of change expected (Glendenning & Vervoort, 2008, 2011; Parimalarenganayaki & Elango, 2015; Stiefel et al., 2009). There are uncertainties associated with these methods which must be considered (von Freyberg et al., 2015). Modelling in particular, can provide answers for different NBS scenarios, but can also have large uncertainties in parameterization particularly in data-limited catchments (Beven & Westerberg, 2011). Uncertainty can be reduced by using empirical data to inform the model, however there are relatively few examples of this; Nicholson et al. (2020) being one of the few for NFM. The limitations of each approach need to be communicated carefully, as often a number of different stakeholders are involved (Sowińska-Świerkosz & García, 2021).

The potential use of NBS for both high and particularly low flow management therefore requires further investigation, for application at different scales and in different locations. Here we used a fully-distributed coupled hydrological/hydraulic model, to investigate the impacts of RAFs on recharge and flows. We applied this to the Blairfindy

catchment, in the Scottish Highlands, where RAFs were recently installed to increase subsurface recharge and enhance summer low flows to maintain industrial water supplies used in a distillery producing ‘The Glenlivet’, a popular malt whisky worldwide. The site benefits from extensive empirical data during and after the 2018 drought, which showed higher resilience of groundwater-fed streams (which remained flowing throughout) than those fed by surface water (occasionally dried out), and how this related to the soils and geology (Fennell et al., 2020). This conceptual understanding was used to inform the modelling, which then allowed testing of the present and a range of other NBS scenarios. These were designed to explore the impacts of location (defined by the hydrological properties of the soil/vegetation types and placement within the catchment), and scale of RAFs (defined by the size of RAFs i.e. volume they can hold, and the number of features i.e. the size of the total area affected), on groundwater recharge, contributions to streamflow and in particular low flows (indexed by the  $Q_{95}$ ) in the Blairfindy catchment.

The overall main aim was to assess the impact of location and scale on the effectiveness of RAFs in management of low flows using a locally-informed hydrological model. More specifically the objectives were: (a) to evaluate the conceptual understanding of spatial and temporal variations in storage and flow of water sources in Blairfindy catchment; (b) to quantify the impact of RAFs on recharge and potential to mitigate low flows and (c) to assess the importance of location and scale on the impacts of RAFs for low flow mitigation.

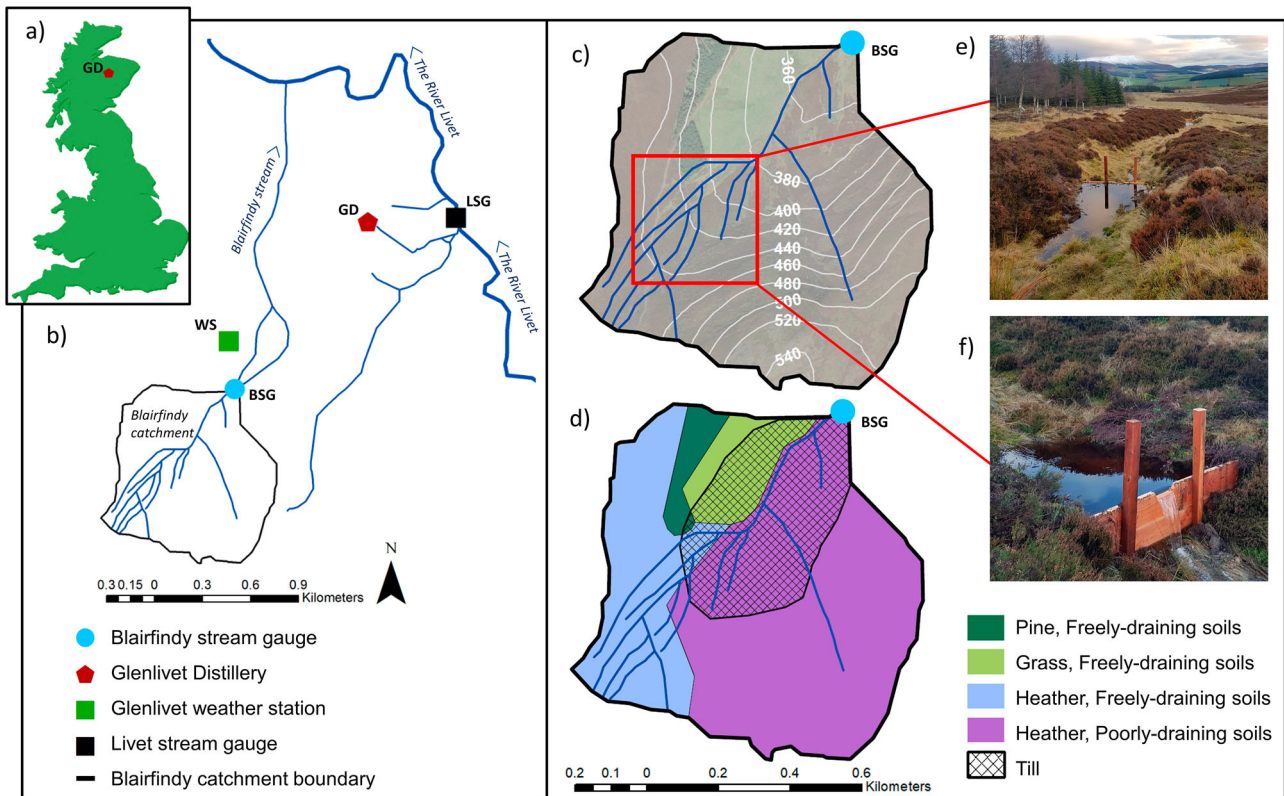
## 2. Data and methods

### 2.1. Study site

Blairfindy is a 0.9 km<sup>2</sup> sub-catchment of the river Livet, in Speyside Scotland (Figure 1(a,b)), and one of the main sites from which a key whisky producer obtains water for distilling. Detailed descriptions of the site are provided by Wilkinson et al. (2016) and Fennell et al. (2020); a brief summary is provided below. The study catchment has a mean elevation of 438 m.a.s.l. and is mostly north-facing with winter topographic shading. Mean annual precipitation (P) is ~900 mm (~7% as snow), Potential Evapotranspiration (PET) is relatively low ~450 mm/year and daily air temperatures (T) average 6.2°C; with maximum average daily T 18.7°C for July and minimum –1.3°C for December.

The geology is dominated by crystalline bedrock which offers relatively little groundwater storage, though an interspersed limestone member and faults and fractures in the bedrock provide sources of deep groundwater. Water storage is also provided by the periglacial, shallow gravel drift deposits and till in the valley bottom (Wilkinson et al., 2016). In the west of the catchment, the humus-iron/iron podzol soils are relatively freely-draining, with high infiltration potential and available storage. These support heather shrubs (*Erica spp* and *Calluna*), grazed acidic grassland and a small coniferous woodland (Figure 1(d); Wilkinson et al., 2016). The slopes in the east are dominated by peaty gleys and thicker peats covered by heather shrubs (*Erica spp*) and moss. The soils in these areas are poorly-draining, remain close to saturation and have lower water storage capacity (Tetzlaff et al., 2014).

The main Blairfindy stream is perennial; long term estimated annual discharge before abstraction is 450 mm



**Figure 1.** Blairfindy (a) location in Speyside, NE of Scotland (b) schematic of catchment area (0.9 km<sup>2</sup>) to the river Livet (102 km<sup>2</sup>), including local monitoring equipment, (c) Satellite imagery overlay with the river network and 20 m contours show change in elevation (ESRI and Digital Globe, 2012) with red box surrounding area of river network into which RAFs installed (d) Vegetation and soils distinguished by drainage properties, and superficial geology (BGS, 2020; Soil Survey of Scotland Staff, 1987) (e) RAFs installed in ephemeral stream channel (f) Close-up of RAF showing wooden construction, including notch in the centre of the top plank.

(Wilkinson et al., 2016). Some smaller streams in the headwaters are ephemeral, especially in the upper south-west of the catchment. These become active during or after precipitation events and dry out through periods of low rainfall. Stream water and groundwater are abstracted by the distillery throughout the year (~95 mm/year from Blairfindy) except for a two- to four-week ‘shutdown period’ for maintenance.

RAFs were installed by the distillery in December 2020 in the ephemeral stream channels (Figure 1(c,e,f)) with the intention to intercept flow pathways and create temporary storage of runoff (Hewett et al., 2020). The rationale for this was to increase the potential for infiltration and recharge of groundwater resources and thereby increase resilience to drought periods by enhancing summer low flows. This is particularly important in catchments where groundwater maintains streamflow through summer (Fennell et al., 2020; Isokangas et al., 2019). Although the installed RAFs are classed as ‘leaky barriers’, they were designed mainly to hold water to allow infiltration (leakage) into the subsurface, with relatively little leakage downstream through the barrier (Figure 1(e,f)). The RAFs were constructed from local timber or till overlain with peat. They were installed across soil/vegetation types with different properties (Figure 1(d)), which could affect RAF functioning in terms of those properties (i.e. freely draining versus poorly draining).

## 2.2. Data

15-minute meteorological data were collected from the Blairfindy catchment weather station between 24/05/2018–17/12/2020 (Figure 1(b)). P was summed, and T and PET

were averaged for 6 hourly model inputs. Prior to 24/05/2018, hourly T and PET, and daily P were available from long-term local weather stations (01/01/2000–31/01/2020). These were scaled for Blairfindy following Fennell et al. (2020) and averaged (T; PET) or spread evenly (P) for use on the 6-hourly time-step.

Stream discharge data were obtained from 23/02/2018 onwards for Blairfindy stream at the catchment outlet (Figure 1(b)) using 15 min stage data recorded by an In-situ Rugged TROLL100 level-logger in a rated section (described in Fennell et al. (2020)). Distillery abstractions were averaged for the period where data were available (2009–2020). These were added evenly to the discharge data to estimate streamflow without the abstractions. Soil volumetric water content data (VWC) in grass, freely-draining soils were obtained at the location of the weather station (Figure 1(b)). Daily estimates for groundwater contribution to discharge were available from Fennell et al. (2020) using End-Member Mixing Analysis.

## 2.3. Model set-up

To address our objectives, we required a coupled hydrological/hydraulic model and selected MIKE SHE - MIKE 11 for our approach. MIKE SHE is a physically-based, deterministic, fully-distributed 3D catchment model which simulates the land-based phase of the hydrological cycle (Abbott et al., 1986). When dynamically-coupled with the MIKE 11 1D hydraulic model, this enables: detailed river network modelling with an integrated module for structures (e.g. RAFs); overland flow to - and out-of-bank flooding from - the river network; and river - baseflow reservoir exchange

(Butts & Graham, 2005). Thus overflow from RAFs and their impact on different flow pathways (Fennell et al., 2020) could be simulated. MIKE SHE – MIKE 11 has been applied on scales ranging from  $<10\text{ km}^2$  to nationwide (Al-Khudhairi et al., 1999; Henriksen et al., 2003), for varied purposes, e.g. investigating stream temperatures (Fabris et al., 2018), water conservation structures (Ramteke et al., 2020), river and floodplain restoration (Clilverd et al., 2016) and climate change impacts (Thompson et al., 2017).

First, we set up the model and simulated the baseline (i.e. before RAF installation) conditions in the Blairfindy catchment, then implemented the recently implemented, and a range of alternative RAF scenarios to explore the impacts on recharge and flow. The model structure was informed by previous conceptual understanding of catchment functioning ((Fennell et al., 2020); Figure 2). The upper layers of land use and unsaturated soil zones were fully distributed so that processes most affected by RAFs, such as overland flow and infiltration, would be modelled in detail. Deeper interflow and baseflow reservoirs were conceptual, and represented the saturated soil zones and geology (Wang et al., 2012). This was appropriate given the level of detail available from maps and field data, and also reduced model run time (Butts & Graham, 2005).

The catchment was discretized using a grid cell mesh of  $15 \times 15\text{ m}$ , which balanced fine enough resolution to capture small flow paths, and computational time. The Digital Terrain Model (DTM) was input at a 5 m resolution (ESRI and Digital Globe, 2012), which was then averaged in MIKE SHE over 15 m grid cells for the model topography. Four different soil/vegetation types were represented to account for the impact vegetation types have on soil properties (Geris et al., 2015). The first three units involved heather, pine and grass, all on freely-draining soils; and the fourth was heather on poorly-draining soils (Figure 2). Vegetation types were based on satellite imagery and land cover maps (ESRI and Digital Globe, 2012; Rowland et al., 2017). Soils were grouped using UK Hydrology of Soil Types (HOST) into those which drain relatively freely (podzols), and those which drain poorly (gleyed soils or peat) (Boorman et al., 1995). Soil surveys of the area determined soil depths (Wilkinson et al., 2016). The different characteristics of the four soil/vegetation types were represented in the unsaturated soil water movement module, and in the upper, interflow module of the linear reservoirs for saturated soil water movement. Characteristics and depths of the two lower baseflow reservoirs were obtained from earlier research and geology maps (BGS, 2020; Ó Dochartaigh et al., 2015; Fennell et al., 2020). These two distinguished between relatively faster-moving groundwater associated with shallower glacial till and the slow-moving groundwater associated with deep, fractured bedrock.

The river network for MIKE 11 was derived from the DTM using flow direction and accumulation tools in ESRI ArcGIS. For the ephemeral streams, detailed land survey data were available, and these were superimposed on the river network derived from the elevation model. Field measurements of the channel dimensions were taken at  $\sim 100\text{ m}$  intervals or more frequently when channel shape notably changed. These cross-sections were input to MIKE 11 river segments at 1 in 10 m, replicating those where channel shape was similar and adding new cross-sections if different. Each river segment represented a flow pathway so that the ephemeral streams,

hillslope surface flow pathways and the main stream joined to form the river network (Figure 1(b)).

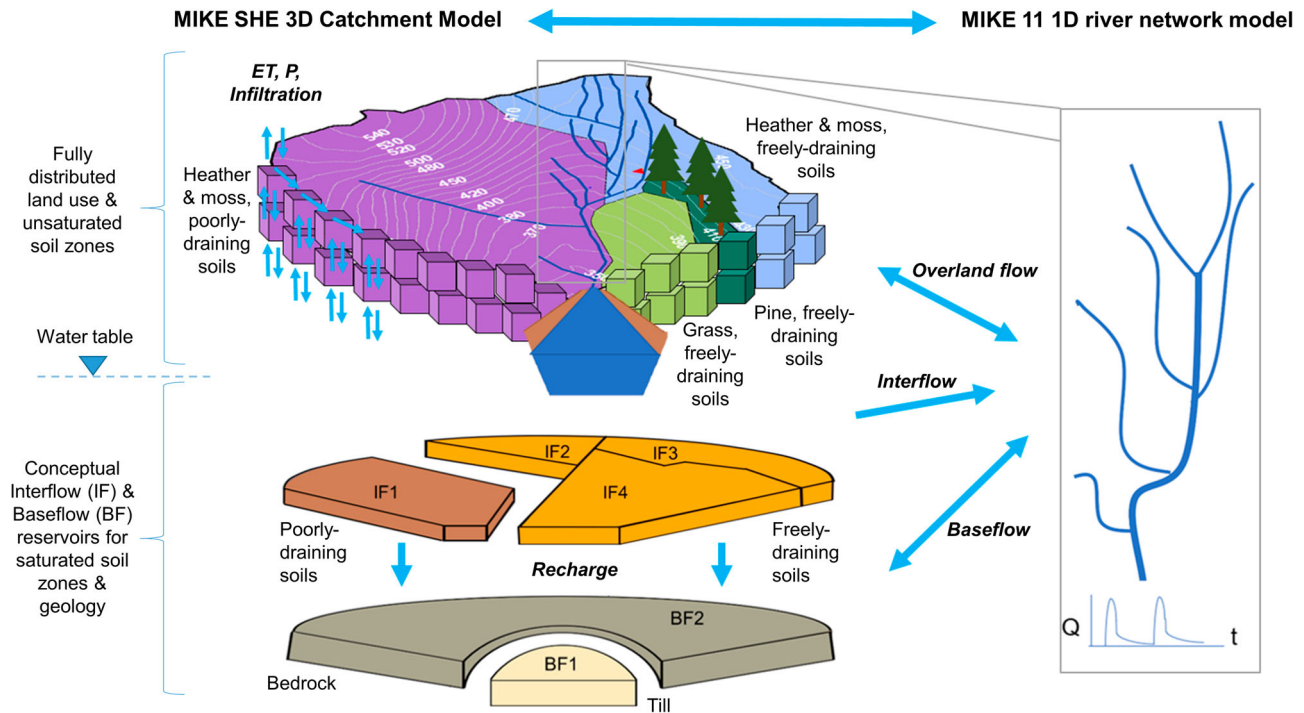
P, T and PET inputs drove the model, which simulated processes dependent on model parameterization. Parameterization was based on an initial sensitivity analysis, where parameters were manually varied individually and the most sensitive model parameters were calibrated (Ma et al., 2016). This led to 32 out of the total 60 parameters being fixed and 28 being calibrated (Table 1). Parameter values were taken from literature and field data (Table 1). P fell as rainfall or snow based on T data and climatic parameter values from local catchments. Actual Evapotranspiration (AET) was simulated using the Kristensen and Jensen (1975) method and leaf area index, root depth and Manning's M (proportional to Strickler roughness coefficient) were set for each vegetation type. Overland flow was modelled using the finite difference method and 2D diffusive wave approximation. Unsaturated soil processes were simulated using gravity flow and Green and Ampt (1911) infiltration (as in Fabris et al. 2018). Each soil/vegetation type was assigned ranges for the van Genuchten (1980) parameters ( $\alpha$ ,  $\eta$ ), with saturated water content and saturated hydraulic conductivity ranges verified by field data, all subject to calibration (except residual water content, which was set). For saturated flow the linear reservoir method was used (conceptual alternative to Darcy's equation described in Wang et al. 2012). For interflow reservoirs, each soil/vegetation type was assigned a range for specific yield, interflow and percolation time constant parameters, and for each baseflow reservoir specific yield and baseflow time constant ranges were required, all obtained from the literature (Table 1).

MIKE 11 channel flow was simulated by 1D approximations of Saint-Venant equations (Abbott et al., 1986). Based on observations through the drought (Fennell et al., 2020), river segments shown to dry out were assigned a 'leakage coefficient' in the model (subject to calibration) and lost water to baseflow reservoirs, whereas those maintained by groundwater were 'gaining' streams and received baseflow from the baseflow reservoir.

#### 2.4. Model calibration, evaluation and validation

The baseline model was run on a 6 hourly time step with a spin-up period 01/01/2015–22/02/2018 to ensure that the soils, interflow and baseflow reservoir levels were stabilized. The calibration and evaluation period ran from 23/02/2018 to 01/07/2020 and a validation period from 02/07/2020 to 17/12/2020. These time periods were chosen so that, given the limited available data, the calibration period captured the relatively extreme conditions experienced (Fennell et al., 2020), and was as long as possible, whilst ensuring the validation period was also long enough to capture both high and low flow periods.

A Latin Hypercube sampling approach was taken to generate 20,000 parameter sets, which were used in a Monte Carlo analysis. This approach was taken to achieve a balance between sampling the full parameter space and maintaining reasonable model run times (Fabris et al., 2018). The parameters were assumed to have uniform distributions except for those covering several orders of magnitude, which were assumed log-transformed uniform distributions (Beven & Binley, 1992). These 20,000 parameter sets were run and baseline model performance was calibrated against observed



**Figure 2.** Conceptual diagram of the model structure, with orientation facing upstream from catchment outlet. The left-hand side represents the MIKE SHE 3D catchment model set-up, with a fully distributed surface layer for infiltration, evapotranspiration and overland flow processes, and conceptual lower interflow and baseflow reservoirs. The outputs of all of MIKE SHE 3D catchment model reservoirs are inputs to the MIKE 11 river network (right). The vertical thickness of reservoirs and size of grid cells are not to scale

stream discharge using a combined objective function (COF) of equally weighted Kling-Gupta Efficiency (KGE) (Gupta et al., 2009) and Volumetric Error (VE). KGE was chosen to capture both high and low flow dynamics, as opposed to Nash-Sutcliffe efficiency (NSE, Nash & Sutcliffe, 1970), which often over-emphasizes the high to medium flows (Krause et al., 2005; Legates & McCabe, 1999). VE assesses the model ability to accurately represent the overall water balance, key to our initial objectives (Janssen & Heuberger, 1995; Mizukami et al., 2019). KGE values greater than  $-0.41$  improve on the mean flow benchmark (Knoben et al., 2019), but only values between  $0.4$ - $0.6$  are deemed acceptable and above  $0.6$  generally are deemed as 'good' (Gupta et al., 2009; Tunaley et al., 2017). VE ranges from  $0$  to  $1$ , with accepted values from  $0.5$  and above (Birkel et al., 2014; Criss & Winston, 2008).

The model was calibrated based on performance relating to the COF, and the top 5% of parameter sets were selected. Out of these parameter sets, the top 20 were chosen so that the internal dynamics of the model were most realistically representative of the conditions observed in the catchment (Beven, 2018). This evaluation involved saturated hydraulic conductivity for heather, freely-draining soils to be greater than for heather, poorly-draining soils (Kuppel et al., 2018; Rezanezhad et al., 2016), and average VWC for soils beneath pines to be less than for heather and grass (Geris et al., 2015). The final 20 runs from the selected parameter sets were then assessed for the validation period with COF and if within similar range to calibration period, used in the NBS scenarios (Booij & Krol, 2010).

## 2.5. Nature Based Solution scenarios – RAFs

To explore the potential impact of RAFs on recharge and mitigation of low flows, and the role of location and scale

within this, we simulated the effects of 12 RAFs scenarios in Blairfindy catchment (Table 2; Figure 3). Here by scenario, we mean a unique combination specifying location and scale of RAFs, with RAF scenarios made up of 4 intervention zones (A–D) and 3 total volumes (increasing from volumes 1–3). In this context, location is defined by the hydrological properties of the soil/vegetation types and the orientation of placement within the catchment; scale is defined by the size of the RAFs (i.e. volume they can hold, increasing with height of RAFs) and the number of features (i.e. the size of the total area affected). The design and placement of RAFs in the model was initially informed by the existing RAFs in the catchment (Figure 1(e,f); scenario C1), and realistic scenarios were then developed further for our objectives (Table 2; Figure 3).

To test the role of location, the same number of RAFs were modelled with the same total potential volumes in different parts of the catchment, specifically with placement on freely-draining versus poorly-draining soils (e.g. scenario A1 vs B1, volume 1 =  $\sim 40 \text{ m}^3$ ). To test the role of scale in terms of storage volume, the same number of RAFs were modelled with double the storage volume at the same location (e.g. for intervention zone A, scenario A1 vs A2 and A2 vs A3; volumes 1–3 =  $\sim 40, 80, 160 \text{ m}^3$ ). Finally, to test the impact of scale in terms of total area affected, the same total storage volume was spread over double the number of RAFs, so storage was distributed over a wider area on the same combination of soil types (scenario C2 vs D1, volumes both  $\sim 160 \text{ m}^3$ ).

RAFs were modelled within the MIKE 11 network as structures between cross sections, so height (from  $0.5 \text{ m}$  up to bank height to increase storage volumes), width (width of channel) and notch size ( $1/5$ th of width) were specified (see in Figure 1(f)). RAF storage volumes were calculated using RAF dimensions, river network and elevation data.

**Table 1.** Calibration parameters, initial and final ranges in brackets for those calibrated, and fixed values without brackets, all based on field data or literature values (references in final row).

	Initial	Final							Refs
P lapse rate (%/100m)		5.4							a
T lapse rate (C/100m)		-0.649							
Tmelt (deg.C)	(-1-1)	(-0.94-0.85)							b
Degree day (mm/C/d)	(1-5)	(1.15-4.84)							
Detention storage (mm)	(0-3)	(0.15-2.89)							
Losing streams leakage	(0.01-1)	(0.02-0.92)							
	Heather, freely draining soils		Grass, freely draining soils		Scots pine, freely draining soils		Heather, poorly draining soils		
Land Use	Initial	Final	Initial	Final	Initial	Final	Initial	Final	
Root Depth (mm)		150		50		500		150	c, d
Leaf Area Index		1.7		1		2.5		1.7	e, f
Manning's M		20		33		10		20	g
Soils									
Sat. hydraulic conductivity (m/s)	(3.6e <sup>-7</sup> -1.2e <sup>-3</sup> )	(3e <sup>-6</sup> -1.1e <sup>-3</sup> )	(3.6e <sup>-7</sup> -4e <sup>-5</sup> )	(3.8e <sup>-7</sup> -3.1e <sup>-5</sup> )	(5.6e <sup>-8</sup> -6.6e <sup>-5</sup> )	(5.1e <sup>-7</sup> -6e <sup>-5</sup> )	(1e <sup>-7</sup> -2.7e <sup>-4</sup> )	(3.5e <sup>-7</sup> -1.9e <sup>-4</sup> )	c, e, l, h, j, k, l, m
Sat. water content	(0.6-0.75)	(0.60-0.74)	(0.5-0.6)	(0.52-0.59)	(0.6-0.75)	(0.60-0.74)	(0.75-0.95)	(0.75-0.95)	
Residual water content		0.04		0.04		0.04		0.1	
alpha (cm <sup>-1</sup> )	(0.02-0.1)	(0.02-0.1)	(0.02-0.1)	(0.02-0.10)	(0.02-0.1)	(0.03-0.10)	(0.04-0.25)	(0.06-0.24)	
N	(1.2-1.8)	(1.20-1.78)	(1.2-1.9)	(1.25-1.81)	(1.2-1.8)	(1.47-1.78)	(1.15-1.69)	(1.17-1.49)	
Interflow specific yield	(0.2-0.35)	(0.20-0.35)	(0.1-0.2)	(0.10-0.20)	(0.2-0.3)	(0.20-0.29)	(0.25-0.35)	(0.26-0.34)	n, o
Interflow t (days)	(0.1-5)	(0.26-4.93)	(0.1-5)	(0.11-4.04)	(0.1-5)	(0.15-4.70)	(0.1-5)	(0.32-4.97)	
Percolation (days)	(0.1-5)	(0.13-0.94)	(0.1-5)	(0.23-4.35)	(0.1-5)	(0.17-4.64)	(0.1-5)	(0.21-0.99)	
	Till		Bedrock						
Baseflow	Initial	Final	Initial	Final					
Baseflow specific yield	(0.1-0.45)	(0.13-0.44)	(0.08-0.35)	(0.12-0.35)	p, q, r, s				
Baseflow t (days)	(60-730)	(180-682)	(1850-20,000)	(9988-19,234)					

Note: (a) (Ala-aho et al., 2017) (b) (Soulsby et al., 2011) (c) (Sprenger et al., 2018) (d) (Humphreys et al., 2018) (e) (Wang et al., 2018) (f) (Byrne et al., 2005) (g) (Te Chow, 1959) (h) (Archer et al., 2016) (i) (Geris et al., 2015) (j) (Kuppel et al., 2018) (k) (Rezanezhad et al., 2016) (l) (Weiss et al., 1998) (m) (Roberts et al., unpublished data) (n) (Refsgaard et al., 2010) (o) (Wang et al., 2012) (p) (Jie et al., 2011) (q) (Robins & Misstear, 2000) (r) (Ó Dochartaigh et al., 2015) (s) (Johnson, 1967).

**Table 2.** Location and scale-related variables which, when combined, form the 12 Nature based solutions scenarios A1-3, B1-3, C1-3, D1-3. For example, 20 RAFs placed in intervention zone A, with starting total combined volume of 41m<sup>3</sup> gives scenario A1.

Intervention zone (see Figure 3)	No. of RAFs	Total volume stored (m <sup>3</sup> )		
		Volume 1	Volume 2	Volume 3
A	20	41	81	165
B	20	42	76	142
C	40	83	157	307
D	80	143	294	611

Use of the MIKE 11 structure module ensured the RAFs were modelled at specific sites, with the in-field geometry well represented. This ensured the uncertainty around how the structures were modelled was minimal, as other methods of modelling NBS often rely on separate models or differences in parameterization e.g. Manning's roughness (Nicholson et al., 2020; Ramteke et al., 2020).

For all of the NBS scenarios, the baseline model with the RAFs implemented was run with the final 20 parameter sets. The full spin-up and model time period was used (01/01/2015–17/12/2020) to run the scenarios. However, a comparison between the baseline and the scenarios was made for the period used for calibration and validation (23/02/2018–17/12/2020), to avoid warm-up issues where water stores may not have yet been stabilized. Evaluation of the impact of NBS scenarios was conducted using the mean percentage changes from the baseline (for 20 parameter sets). These were calculated for the Q<sub>10</sub> (high flows), Q<sub>50</sub> (medium flows), Q<sub>95</sub> (low flows), groundwater recharge and groundwater contribution to the stream.

### 3. Results

#### 3.1. Model performance

The baseline model was considered acceptable given that performance was evaluated on the combined ability to represent high and low flows (KGE), as well as overall volumes (VE). Figure 4(c,d) show top performing parameter sets simulated discharge on normal and log scale through the calibration period (KGE max = 0.54; VE max = 0.62; combined objective function (COF) max = 0.5) and validation period (KGE max = 0.65, VE max = 0.60; COF max = 0.6). The improved fit in the validation period for KGE and COF is unusual, but not surprising given that the calibration period included a number of high flow periods and an extreme drought in which flows locally reached the lowest on record (Fennell et al., 2020). This relatively good fit in this extreme calibration period increased confidence that the model could simulate both high and low flows, thus an improved fit in the less extreme validation period.

The model represented the extreme drought period in 2018 quite well with modelled low flows of ~0.52 mm/day (Figure 4(c,d)). Post-drought the simulated stream baseflow increased more rapidly than was observed (October 2018), however the stream response to rainfall was mostly well-reproduced. This simulated early increase in baseflow post-drought was likely due to simulations re-wetting the soils too quickly (Figure 4(d,e)), which is often an issue in hydrological modelling (Ala-aho et al., 2017). Mean baseflow simulations appeared to be overestimated through summer 2020, although the uncertainty band in the model still captured this

variation and hydrograph dynamics were well represented. Peak flows of up to ~20 mm/day were well or slightly over-represented, whereas snow melt generated less of a streamflow response.

The model simulated the dynamics of the observed VWC in grass freely-draining soils quite well, though again the post-drought re-wetting period was less successfully reproduced (Figure 4(e)). Spatial variation between soil types was represented with clear differences between the lower mean VWC for freely-draining and higher mean VWC for poorly-draining soils.

Catchment average recharge rates ranged from < 0.1 mm/day during dry periods to up to ~15 mm/day following precipitation events (Figure 5(a)). Both this range and the average recharge rate across the calibration and validation period (1.3 mm/day) were as expected (Mansour et al., 2018; Ó Dochartaigh et al., 2015). Baseflow reservoir storage and flows were well-captured (Figure 5(c)) given that earlier tracer-based research independently showed similar ranges of between 65-100% groundwater contribution to discharge through the drier period (Fennell et al., 2020). This suggested that the model simulated the storage and flows of different water sources in the catchment reasonably well when evaluating against the observed data and our conceptual understanding of catchment functioning could be justified within the modelling environment (Blöschl, 2017; Fennell et al., 2020).

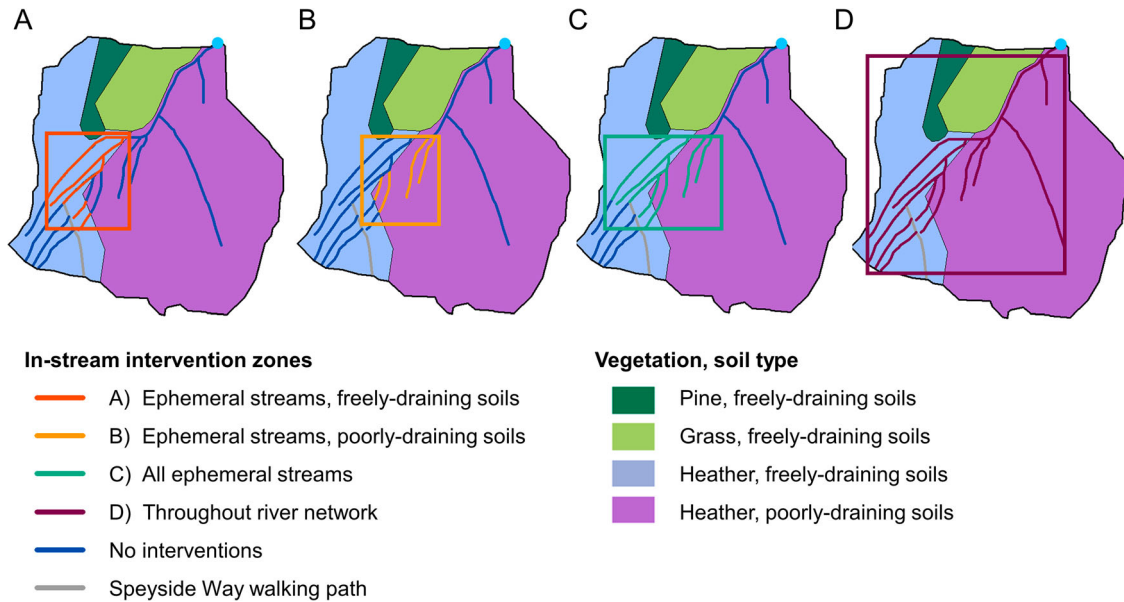
#### 3.2. Impacts of RAFs on recharge and potential for mitigating low and high flows

Time-variable impacts of RAFs are summarized using the currently implemented scenario C1 as an example. The modelled recharge rate averaged across the catchment increased with the implementation of RAFs (Figure 5(b)), and was seasonally variable with a greater increase occurring in late summer (~3–4%) than winter/spring (~0%).

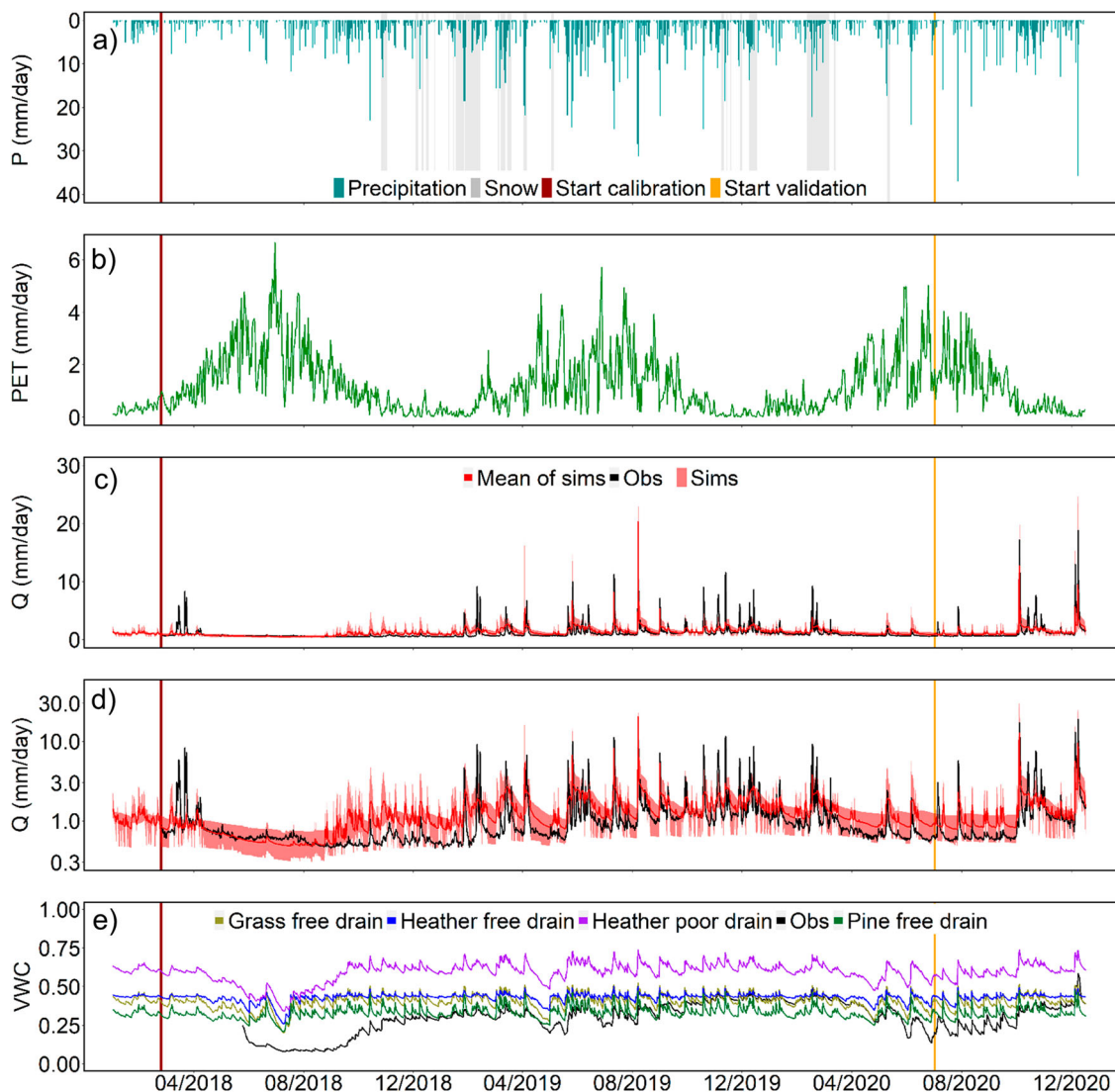
Groundwater contribution to discharge also increased with RAFs, particularly through wetter periods (10–20% above baseline). This was also the case through dry periods (3–5% above baseline) suggesting a small increase in baseflow during drought (Figure 5(d)). This positive impact was not so obvious in some periods; in scenario C1 groundwater contribution to streamflow occasionally dropped below baseline (lowest –0.8%), however this was due to the simultaneous increase in groundwater outflow and discharge through these periods as a result of RAF implementation (Figure 5(d)).

Overall, RAFs reduced mid and high flows (Q<sub>50</sub> and Q<sub>10</sub>), and increased low flows (Q<sub>95</sub>) (Figure 6(a–c)). Across the three year simulation period, total discharge slightly decreased (average ~–2.7%) as a result of increased infiltration to deeper storage zones, and a slower rate of return to river baseflow. Change from baseline was greater for high flow mitigation, where scenario C1 resulted in –5% change in the Q<sub>10</sub>, compared to changes of –1.6% for Q<sub>50</sub> and +1.3% for Q<sub>95</sub>. This must be considered alongside the greater uncertainties associated with modelling of high flows, more likely to be overestimated (Mizukami et al., 2019), however, the direction of changes suggests RAFs could be used to mitigate the impacts of both low as well as high flows.

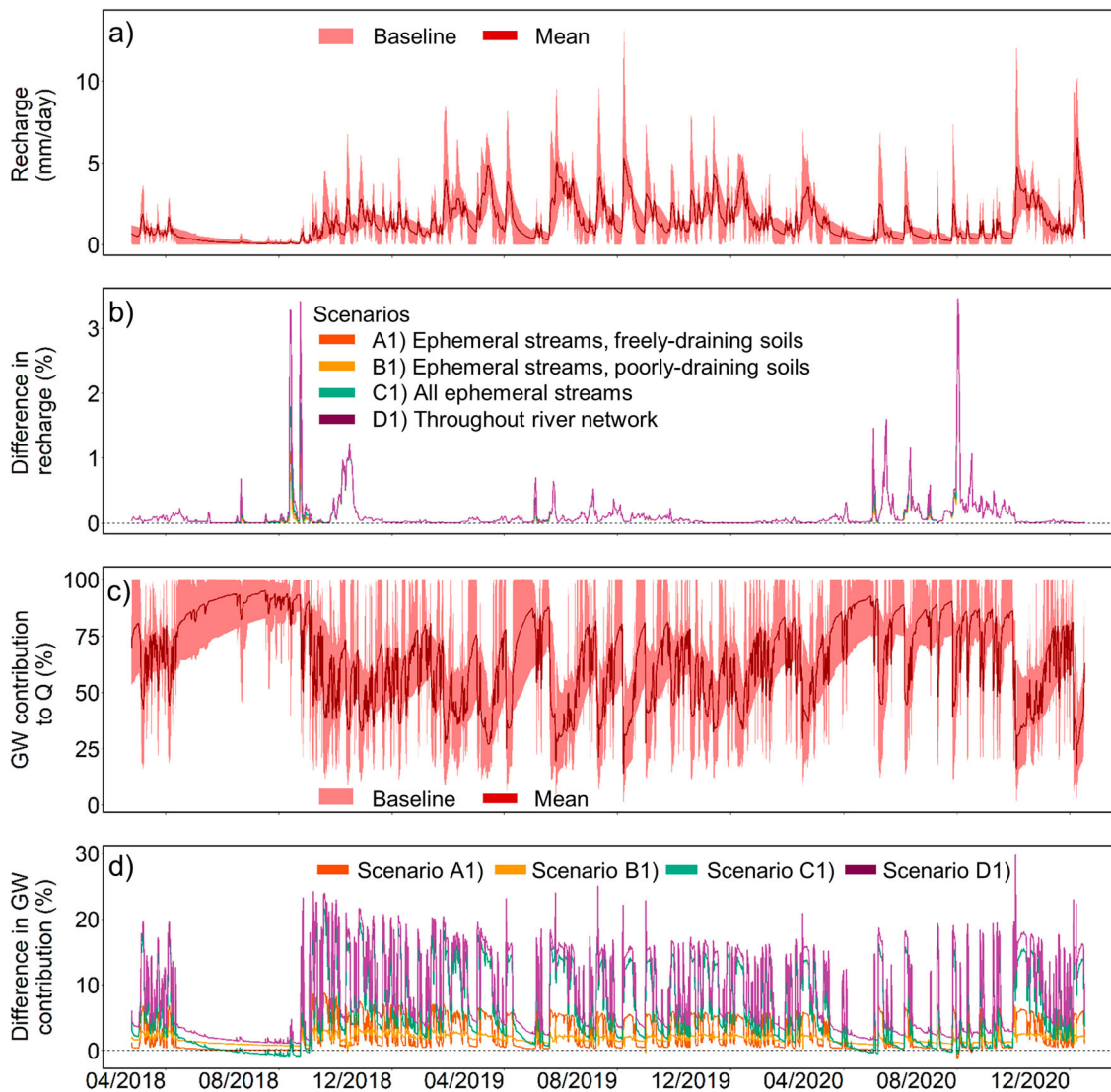




**Figure 3.** Intervention zones (A–D) and associated soil/vegetation type into which RAFs of increasing total combined volumes (volumes 1–3) were placed to form 12 Nature based solution scenarios. Scenarios A1–3 feature RAFs implemented in (A) ephemeral streams, freely-draining soils of volumes 1–3, scenarios B1–3 feature RAFs implemented in (B) ephemeral streams, poorly-draining soils of volumes 1–3, scenarios C1–3 feature RAFs implemented in (C) all ephemeral streams of volumes 1–3 and scenarios D1–3 feature RAFs implemented in (D) throughout river network of volumes 1–3.



**Figure 4.** (a) Observed precipitation and snowfall (b) observed potential evapotranspiration from weather station, used as model input (c) observed discharge and top 20 simulations with mean discharge for Blairfindy stream catchment outlet (d) as (c) in Log scale and (e) mean simulated soil volumetric water content in the four soil/vegetation types.



**Figure 5.** Time series for recharge rate and groundwater contribution to discharge with (a) and (c) baseline results including uncertainty band for 20 top parameter sets, and (b) and (d) mean percentage change from baseline for 4 Nature based solution scenarios A1-D1, (so volume 1 in all 4 intervention zones A-D).

### 3.3. The importance of location and scale on the impacts of RAFs

The location (primarily via contrasting soil properties) of RAFs had a clear impact on their effectiveness (Figure 6). RAFs implemented in freely-draining soils (scenario A1) resulted in greater percentage change than poorly-draining soils (scenario B1) for high and low flows (difference = -2%; +1%, respectively), recharge (+0.001%) and groundwater contribution to discharge (+0.5%) (Figure 6). Freely-draining soils with greater available storage and infiltration rates would allow quicker flow movement to deeper pathways, so RAFs would more likely be empty for the next rainfall event. The main exception was for mid-range flows ( $Q_{50}$ ) where RAFs placed in poorly-draining soils (intervention zone B) resulted in greater percentage decreases, reducing the  $Q_{50}$  by 0.5% more than RAFs in freely-draining soils (intervention zone A) (Figure 6(b)).

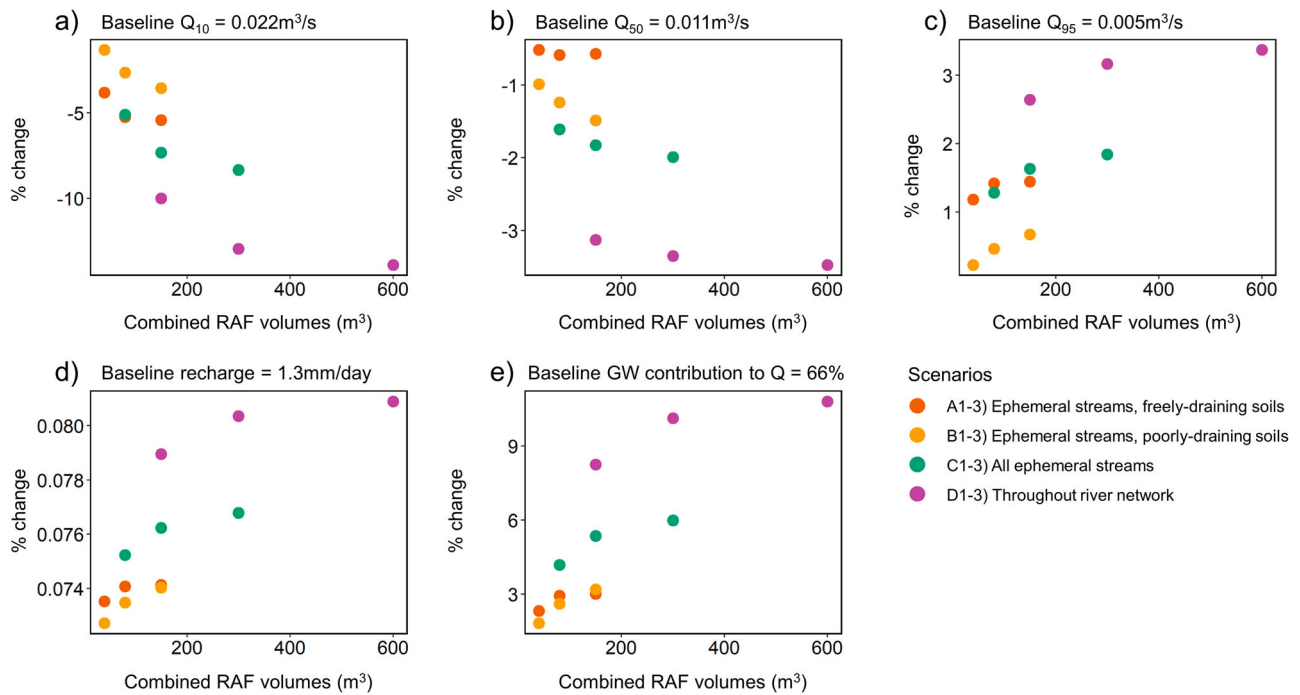
Scale of RAFs was also important, both in terms of volume and total area affected. For currently installed RAFs (scenario C1), doubling RAF volumes (scenario C2) increased impact by ~25% for low flows, ~40% for high flows and ~30% for groundwater contribution to discharge, however doubling the volume again (scenario C3) only

resulted in an additional ~10% change in impact (Figure 6). This trend was also observed in the other three implementation zones.

However, increasing the total area affected by RAFs had a greater impact than an increase in the local volume of RAFs. Spread of the same potential volume across double the number of smaller RAFs over a wider area e.g. Scenario C2 to D1 (both  $150 \text{ m}^3$ ), increased impact on  $Q_{95}$ ,  $Q_{50}$ , recharge and groundwater contribution to discharge two to three times more than doubling the volume in the same location. The one exception was the impact on high flows where, similar to when doubling the RAF volumes, a 40% change was observed (Figure 6(a)).

## 4. Discussion

Whereas previous work has mainly focussed on RAFs for flood management, this study has shown that RAFs have the potential to contribute to water resources management challenges associated with low flow (drought) as well as high flow (flood) conditions. The results also revealed the relative importance of location (orientation and hydrological properties based on soil/vegetation type) and scale, both in



**Figure 6.** Percentage change from baseline values (stated at top of plot; average of 20 parameter sets for model period) for (a)  $Q_{10}$ , (b)  $Q_{50}$ , (c)  $Q_{95}$ , (d) recharge rate and (e) groundwater contribution to discharge, for 12 Nature based solution scenarios A1-3, B1-3, C1-3 and D1-3, given combined RAF volumes (volumes 1–3 in intervention zones A–D).

terms of volumes and area over which the RAFs were spread. Although directly relevant to the context of the distillery at the study site, the potential use of RAFs for low flow management could apply to a wide range of sites in temperate climates which experience periods of water-stress, or indeed those globally more arid. More generally, we've demonstrated the value of our modelling framework informed by data in providing a decision-support tool for efficient design and implementation of RAFs, thereby increasing the potential for wider uptake.

#### 4.1. Simulating spatial and temporal variations in the storage and flow of water sources

Accurate representation of the physiographical properties and the spatial and temporal variations in storage and flow of water sources is key to modelling NBS. Firstly to understand the impacts of these factors on flow and recharge, and secondly for understanding optimal position of NBS, as e.g. RAFs depend on interception of flow pathways for retaining water (Quinn et al., 2013).

Often hydrological models are limited by set-up requirements, data availability and runtime demands (Huang et al., 2019; Huang & Bardossy, 2020). Relatively few are based on catchment process understanding gained through drought, which reveals whether flow pathways in a catchment are ephemeral or perennial in nature (Isokangas et al., 2019). Our baseline model achieved a reasonable balance between detail, flexibility and runtime; results agreed with the conceptual model of catchment functioning based on data collected before, during and after a period of drought (Blöschl, 2017; Fennell et al., 2020). Surface water flow pathways responded quickly to rainfall, and the timing of activation of ephemeral streams and snowfall was consistent with field-based observations (Fennell et al., 2020). High groundwater contributions through the drier

periods (65–100%) were consistent with empirically-based estimates from Blairfindy (Fennell et al., 2020) and other upland catchments in the same region, particularly those with peat soils and groundwater-fed streams (Blumstock et al., 2015).

Although the objective functions were modest, flow dynamics and the water balance were captured, and key processes that would affect the impact of RAFs such as overland flow were represented appropriately and in a fully distributed manner, an improvement on other conceptual approaches (Beven, 2018; Glendenning & Vervoort, 2011). Use of a conceptual approach for the subsurface was appropriate given the availability of data, and provided good representation of baseflow dynamics. This could be developed into a fully-distributed approach, which would benefit from the use of geophysics to characterize the subsurface of soils and drift deposits in upland catchments (Soulsby et al., 2016), to identify zones of storage in a more sophisticated manner than through soil and geology maps.

Some of the model data inputs were less certain, however the parameterization is likely to have more of an impact on the model uncertainty. The relatively high number of calibration parameters, and comparatively low number of Monte Carlo simulations meant the final calibration ranges were quite broad (Table 1). Despite this uncertainty, other modelling studies have successfully used similar approaches (Fabris et al., 2018). Greater confidence in the model internal dynamics was also gained from the model evaluation, which included both discharge-based and soft-calibration through the use of knowledge of catchment function, soil VWC and field observation data. It was thus deemed reasonable to use the model to compare different RAF scenarios, and recognition of the uncertainty in the baseline meant that conclusions were made based on the direction of change rather than absolute values.

#### 4.2. Investigating the impact of RAFs on recharge and potential to mitigate low and high flows

The results suggested RAFs could have a positive, albeit small, impact on the management of low flow periods. RAFs increased overall recharge by between  $\sim 0.07\%$  for currently implemented RAFs (scenario C1; Figure 1(e,f)) to  $\sim 0.08\%$  for max number and volume of RAFs (scenario D3); groundwater contribution to discharge between  $\sim 4\%$  (C1) and  $\sim 10\%$  (D3) and the  $Q_{95} \sim 1\%$  (C1) and  $\sim 3\%$  (D3) from baseline. This direction of change was expected given reported impacts from other studies of RAFs in temperate environments (Kravčík et al., 2012; Pavelic et al., 2012; Somers et al., 2018). On the one hand, small percentage changes to recharge could in fact scale up to larger impacts. Small margins of change in flows can also affect sediment transport and in-stream deposition (Baldan et al., 2021) as well as stream temperatures and volumes of water (Folegot et al., 2018; Piggott, 2017), so RAFs could make an important difference in aquatic habitats. On the other hand, reported projections for climate change suggest more frequent extremes which could have much greater impacts (Visser-Quinn et al., 2021). Further research to integrate this MIKE SHE model with climate change scenarios (as in Thompson, 2012), would likely help to determine how the relative impacts compare (Capell et al., 2013).

RAFs also showed potential to mitigate high flows, decreasing  $Q_{50}$  by  $\sim 2\%$  (C1) to  $\sim 3.5\%$  (D3) and particularly the  $Q_{10}$  by  $\sim 5\%$  (C1) to  $\sim 13\%$  (D3). A similar magnitude of change for high flows has been reported elsewhere, albeit with different volumes of storage at different catchment scales, ranging from 7.5–11% (Metcalfe et al., 2017b; Odoni & Lane, 2010) to 27–30% (Nicholson et al., 2020; Norbury et al., 2021), with an increased number of RAFs increasing impact. Flooding processes – the propagation, synchronization and recession of flood peaks – operate on much shorter timescales than drought-related impacts (Hänssel et al., 2019). This necessitates detailed modelling of overland flow/infiltration processes and small time-steps (van den Bout & Jetten, 2020). To achieve greater detail in our results, the 6-hour time-step would need to be reduced to e.g. 5-minute resolution, as in Nicholson et al. (2020). However, this would then require adjustment of parameters which are highly dependent on time step resolution (Ostrowski et al., 2010). Given the primary focus of the study was on low flows and sub-surface drainage, the 6-hour time-step was deemed appropriate. A shorter time-step might also provide further insights into the drainage rate which affects the storage dynamics of RAFs.

#### 4.3. The importance of location and scale of RAFs

Our results showed that location (determined mainly by soil type) was important, and increasing the soil surface area over which volume was stored was more effective than an increase in volume at the same location for both high and low flow management. For a similar potential maximum storage volume across the RAFs, implementing many small RAFs will likely have more impact than a few larger ones. They are also best placed at locations where more additional sub-surface storage is available. Understanding the spatial variability and properties of soils and geology in a catchment is therefore important for successful placement of RAFs

(Heilweil & Watt, 2011; Reaney, 2022; Standen et al., 2020). These factors affect the rate at which water moves to deeper flow pathways allowing RAFs to drain, or soil or groundwater stores reach saturation and limit RAF drainage.

The results demonstrate the clear benefit from assessing the relative importance of both location and scale together in a modelling framework, which could be transferred to other sites with different soils, geology and climates globally. The use of different scenarios revealed that at this site, the scale in terms of area affected by the RAFs was more important than location, and a tailing-off of impact was observed over a certain volume of storage. This tailing-off of impact was a result of slow drainage leading to ponding so RAFs remained full following rainfall events (Ramchunder et al., 2009; Shanafield & Cook, 2014). Use of a similar modelling framework at other sites to identify optimal local RAF storage volume could, as well as make significant savings on resources and costs, also avoid potential negative impacts. For example, excessive ponding could lead to evaporative losses especially in drier climates, and overall decrease in total streamflow in low flow conditions (Glendenning & Vervoort, 2011; Sharda et al., 2006), or reduce the available storage for multiday storm events, limiting capacity to prevent primary flood peaks when applied to NFM (Nicholson et al., 2020; Wilkinson et al., 2019).

Although RAFs in freely-draining soils had greater impact, increased recharge and managed flows were observed in all scenarios, even when RAFs were placed in poorly-draining soils. This suggests that they may still be beneficial in catchments with soils of limited available storage. Although this may differ with location (vegetation, soils and geology) and climate due to different infiltration and evaporative fluxes, particularly when increasing RAFs surface area (Neumann et al., 2004; Salem et al., 2012; Staccione et al., 2021), this could be tested in a modelling environment. The potential hydrological benefits could then be assessed alongside ecosystem services associated with NBS such as improvements in biodiversity (European Commission, 2020), to help form a full cost-benefit analysis. This in turn, would encourage general uptake of the full approach and implementation of RAFs or similar NBS.

#### 4.4. Future research

More research over longer time scales is required to understand the effects of RAFs on soil properties, the requirements in terms of maintenance, and how these factors change over time (Ramchunder et al., 2009). If the RAFs do not regularly empty, the soils may be constantly saturated, which would prevent infiltration and therefore limit their impact on recharge and potential to mitigate flooding (Hallett et al., 2016). Alternatively, sediment build-up could limit storage capacity of RAFs, or river geomorphology could be affected by alterations in sediment and flow regimes (Baldan et al., 2021). Understanding how RAFs compare with other NBS in this context, with the addition of a cost-benefit analysis would help decision makers determine which solution is most appropriate, and whether financially feasible for their site. Although this study has addressed some of the key issues relating to location and scale, future research could investigate the impact of RAFs over greater scales or in neighbouring catchments (from which the deeper groundwater sources

are likely also fed), and on what scale monitoring would be required (Blöschl et al., 2007).

Long-term empirical data collection post-installation of RAFs in the field would help determine the accuracy of model predictions. A tracer-based, water quality or hydrological analysis of impacts would provide much-needed empirical data to improve the knowledge base on NBS implementation. Comparison to similar studies in other locations in the UK and globally, with different soil types, geology and climates would also help to determine whether similar conclusions can be made on location and scale of RAFs.

## 5. Conclusion

Management of water resources through periods of extremes (floods and droughts) can be challenging, particularly in areas where a balance must be obtained between local communities, the environment and industry. This will likely become more problematic, with climate change projections for the increased frequency of these extreme periods. Investigation into the potential for NBS to mitigate the impacts of floods and droughts, at what scale and where they would be best placed, and whether implementation of RAFs could increase recharge and low flows, could be key to ensuring sustainability of water use e.g. for the distilling industry.

Informed by conceptual understanding of catchment function based on empirical data, a physically-based model was set up to simulate variations in storage and flow of contrasting water sources. Soil moisture dynamics, high groundwater contribution to discharge through low flow periods and hydrograph dynamics were well represented in the model baseline. A range of scenarios tested how implementation of RAFs and their location and scale affected groundwater recharge, contribution to streamflow and low, medium and high flows.

The results suggested that RAFs could help mitigate the impacts of low flows through drought and highlighted the importance of a transferable modelling framework to test scenarios of different locations and scales before installing in a catchment. The direction of change for the impact of RAFs was to reduce mid to high flows, and increase low flows, recharge and groundwater contribution to streamflow. Generally, an increase in storage volume at the same location increased impact to a certain point, after which changes were less marked. Although location of RAFs (soil type) was important, increasing the spread of storage over a wider area was more effective than increasing volumes in the same location (even if on freely-draining soils). This means that many small RAFs spread throughout the river network could be more effective than a few large RAFs.

Future research should aim to collect empirical data on the impacts of implementations, and how their functioning changes through time, in varied locations with different soils and geology. Comparison between different NBS, their potential to mitigate the impacts of climate change and cost-benefit analysis would also help to justify implementation and increase uptake for hydrological and other wider benefits.

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