Optimisation of Urban-Rural Nature-Based Solutions for Integrated Catchment Water Management

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Highlights

- Runoff attenuation features and floodplains are prioritised nature-based solutions for optimal water availability
- Optimal water quality and flood management have trade-offs with water availability caused by large-scale regenerative farming
- Extensive urban green space expansion is preferred by optimal phosphorus management but induces significant economic costs

Abstract

Urban-rural nature-based solutions (NBS) have co-benefits for water availability, water quality, and flood management. Searching for optimal integrated urban-rural NBS planning to maximise these co-benefits is important for catchment scale water management. This study develops an integrated urban-rural NBS planning optimisation framework. In this framework, the CatchWat-SD model is developed to simulate a multi-catchment integrated water cycle in the Norfolk region, UK. Three rural (runoff attenuation features, regenerative farming, floodplain) and two urban (urban green space, constructed wastewater wetlands) NBS interventions are integrated into the model at a range of implementation scales. A many-objective optimization problem with seven water management objectives to account for flow, quality and cost indicators is formulated, and the NSGAII algorithm is adopted to search for optimal NBS portfolios. Results show that rural NBS have more significant impacts across the catchment, which increase with the scale of implementation. Integrated urban-rural NBS planning can improve water availability, water quality, and flood management simultaneously, though trade-offs exist between different objectives. Runoff attenuation features and floodplains provide the greatest benefits for water availability. While regenerative farming is most effective for water quality and flood management, though it decreases water availability by up to 15% because it retains more water in the soil. Phosphorus levls are best reduced by expansion of urban green space to decrease loading on combined sewer systems, though this trades off against water availability, flood, nitrogen and suspended solids. The proposed framework enables spatial prioritisation of NBS, which may ultimately guide multi-stakeholder decision-making.

Keywords: nature-based solutions, integrated planning, integrated water management, systems analysis, many-objective optimisation, co-benefits and trade-offs

1. Introduction

Sustainable infrastructure development challenges are present in resource use (e.g., water scarcity (Mancosu et al., 2015)), natural hazards mitigation (e.g., extreme climate events (Allard, 2021)) and environmental management (e.g., ecological degradation (Nyumba et al., 2021)). 'Hybrid' solutions that explicitly integrate blue, grey, and green infrastructure have been recently emerging (Depietri and McPhearson, 2017). Green infrastructure (GI), under a philosophy of 'working with nature' (Calliari et al., 2019), includes several concepts, such as low impact developments (LIDs) and sustainable urban drainage systems (SUDs) for urban regeneration, and best management practices (BMPs) in both urban and rural contexts (Fletcher et al., 2015; Matsler et al., 2021; Ramírez-Agudelo et al., 2020). In this work, we adopt the term Nature-Based Solutions (NBS), defined as "solutions that are inspired by, supported by or copied from nature" and can "simultaneously provide environmental, social and economic benefits and help build resilience" (Bauduceau et al., 2015). NBS can provide multiple benefits to water resources management (Ramírez-Agudelo et al., 2020; Sonneveld et al., 2018), including enhancing water provision (Keesstra et al., 2018; Water, 2018), water purification (Jessup et al., 2021; Tanner et al., 2005) and flood peaks mitigation (Majidi et al., 2019; Vojinovic et al., 2021).

NBS generate water resources management benefits by intervening at multiple points across the water cycle. Some NBS are primarily targeted towards the rural water cycle; for example, wetlands (Lane et al., 2018) and floodplains (Burt et al., 2002) may increase recharge to groundwater and maintain baseflows during low-flows. These interventions also remove nutrients through active biochemical processes such as denitrification (Jones et al., 2015; Lane et al., 2018; Roley et al., 2012) and sedimentation of suspended solids (Braskerud, 2002; Kløve, 2000; Tockner et al., 1999). They can attenuate surface runoff generated during heavy rainfall and reduce flood risks (Acreman et al., 2003; Wright et al., 2008; Wu et al., 2020). Other NBS target the urban water cycle. SUDs increase infiltration into soil and percolation down to groundwater on the urban surface (Hamouz and Muthanna, 2019; Zölch et al., 2017) and remove pollutants from stormwater runoff (Drake et al., 2014; Lim et al., 2015). Constructed wastewater treatment wetlands filter and remove nutrients and solids to purify urban effluent before it is discharged into rivers (Hickey et al., 2018; Zhang, D. Q. et al., 2014).

Considering NBS from an integrated viewpoint that covers both urban and rural systems can potentially reveal larger co-benefits when incorporated into a whole catchment water management strategy. Such integrated NBS implementation has recently been advocated in the 'urban-rural partnerships' considering the catchment-scale dependencies (Banzhaf et al., 2022). However, NBS implementation has been previously studied in isolation using models that can simulate either urban or rural water cycles, but not both (Cheng et al., 2009; Chiang et al., 2014; Mao et al., 2017). Hence, evaluating NBS co-benefits at a catchment scale and how they interact with existing infrastructure systems requires an integrated modelling approach that simulates both urban and rural water systems.

NBS planning is difficult because a given catchment or region will have a variety of options, locations and scales available in a range of configurations. In addition, urban-rural water cycle interactions may result in non-linear system responses and thus unpredictable performance metrics (Liu et al., 2021). The result is a large and complex decision space that cannot be exhaustively searched (Deb, 2011; Gunantara, 2018). Thus, identifying portfolios of NBS options in a region typically requires a search driven simulation-optimisation approach that combines hydrological models and multi-objective evolutionary algorithms (MOEA) (Artita et al., 2013; Mao et al., 2017; Maringanti et al., 2011; Qi et al., 2020; Zhang, K. and Chui, 2018). MOEA approaches require objectives (performance indicators) to optimise, most commonly economic costs. NBS optimisation studies also tend to focus on water quality (e.g., nutrients and sediment loadings (Chaubey and Maringanti, 2009; Veith et al., 2004)) or flood management objectives (e.g., flood risk (Duan et al., 2016)), or a combination of both (e.g., stormwater pollution loadings and flow reduction (Gao et al., 2015; Xu et al., 2017)). Although there have been studies that model NBS benefits for water availability (Kumar et al., 2021), it has not yet been included as an objective to optimise for NBS planning. The choice of objectives should reflect the preferences of decision makers (Kasprzyk et al., 2013), but should also facilitate the wider goal of water systems models, which is to better understand the interactions and dominant processes present in the integrated water cycle and how these manifest as objectives (Loucks, 1992). For example, to understand the systems-level impacts on water quality, both nitrogen and phosphorus should be included as objectives, as they have different environmental impacts and their main loadings are from different sources (cropland and urban wastewater effluent, respectively) in the UK (Liu et al., 2021).

Including the wide range of water management objectives required for urban-rural NBS evaluation designates the problem as a, so called, “many-objective” optimisation problem (Di Matteo et al., 2017; Maringanti et al., 2011; Seyedashraf et al., 2021). Such problems normally have more than four objectives and the solutions are accompanied by co-benefits and trade-offs among the objectives. For example, Seyedashraf et al. (2021) searched optimal planning of the SUDs in an urban catchment for six objectives and found co-benefits for improving total suspended solids, flood volume, and average runoff peaks. Todman et al. (2019) searched future optimal agricultural landscape planning and found trade-offs between agricultural yield and N_2O emissions, and between N_2O emissions and soil organic carbon (SOC). Pareto fronts are obtained to illustrate the trade-offs between objectives, which most commonly exist between costs and other objectives (Di Matteo et al., 2019; Maringanti et al., 2011; Todman et al., 2019). Finally, it is important to understand how different NBS interventions may result in trade-offs between water management objectives, which have not been fully revealed in previous studies. A holistic analysis of NBS performance at a catchment scale in the context of water quantity and quality in rural and urban systems is still missing.

This study proposes an integrated urban-rural NBS planning optimisation framework. In this framework, the CatchWat model from Liu et al. (2021) has been redesigned as a semi-distributed tool (CatchWat-SD) to simulate integrated urban-rural water cycles of the Norfolk region, UK. Five types of NBS are conceptualised and parameterised in the CatchWat-SD modelling framework so that their performance can be compared. Their benefits for water availability, water quality, and flood management are evaluated at the catchment scale for a range of implementation sizes. A many-objective optimisation problem is then formulated, and solutions are obtained using NSGAII genetic algorithm. Finally, NBS co-benefits and trade-offs in the context of water availability, water quality, and flood management are investigated to understand the implications of multi-objective analysis on integrated urban-rural NBS planning at a catchment scale.

2. Study area

To better demonstrate the importance of integrated urban-rural NBS planning, we choose the Wensum and Yare catchments in Norfolk, UK, as a case study, an area with both active urban and rural water cycles (Fig. 1). The study area is 1,324 km^2. The land cover is predominantly rural, with 63% categorised as arable and 19% as grassland in 2015 (Rowland et al., 2020). 9% is populated urban or suburban areas, primarily surrounding the city of Norwich.

We delineate the study region into 32 sub-catchments (SC) whose boundaries are based on Water Framework Directive (WFD) River Water Bodies Cycle 1 (Environment Agency, 2021b). The average annual rainfall varies from 711 mm (SC10) to 664 mm (SC31) (Marsh and Hannaford, 2008). Hydrogeological conditions are defined by highly permeable chalk in the north (SC5, 10, 14) and east (SC4, 6, 9, 28, 29, 30, 32), driving significant baseflow (Dils et al., 2009), and less permeable loamy soil in the rest area (Cranfield Soil and Agrifood Institute, n.d.). The gauged mean flow for River Wensum is 4.1 m^3/s (SC9) and for River Yare is 1.6 m^3/s (SC31) (UK Centre for Ecology and Hydrology, 2020). The predominant WFD classifications on surface water ecological status are 'poor' to 'moderate' in the study area (Environment Agency, 2022), thus we consider it an area that would benefit greatly from NBS interventions.

For water resource reliability, most of the study area is evaluated to have at least 50% of the time when consumptive abstraction is available (Environment Agency, 2017). 32% of licensed irrigation for agriculture in Broadland rivers comes from surface water and 68% from groundwater (Knox et al., 2017). The largest licensed river water abstraction for domestic use is > 40 Ml/day in SC9 on River Wensum, which is for Norwich water supply (Soley et al., 2012). 11 wastewater effluent discharge points are identified, with the largest discharge point located in SC32 downstream Norwich city (European Commission, 2022).

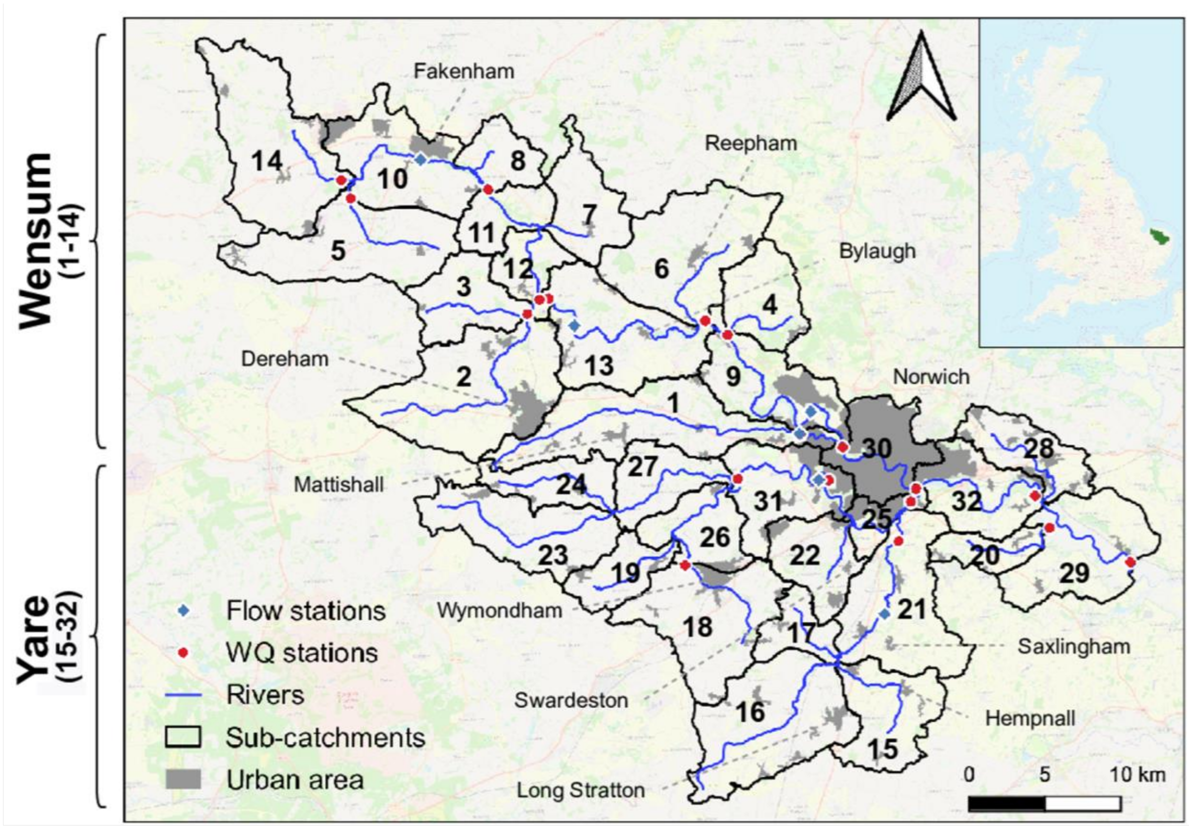


Fig. 1 Wensum and Yare catchments with the locations of river flow and water quality monitoring stations

3. Methodology

3.1 An integrated urban-rural NBS planning optimisation framework

This study develops an integrated urban-rural NBS planning optimisation framework (Fig. 2). It starts by initialising a population of NBS solutions. In each solution scenario, the optimised design for the proposed NBS will be generated for all sub-catchments. These will be input into the CatchWat-SD model for simulation. Then, objective values are calculated based on the simulation results, and the constraints are evaluated for each solution. The NSGAII genetic algorithm is adopted to obtain the next generation of solutions via non-dominant sorting, crossover, and mutation (Maringanti et al., 2011; Yang, Guoxiang and Best, 2015). The new generation of solutions will be input into the CatchWat-SD model for simulation, and the process is repeated until the maximum number of generations is achieved or no further improvements to objective functions can be made. The details of each procedure are illustrated in the following sections.

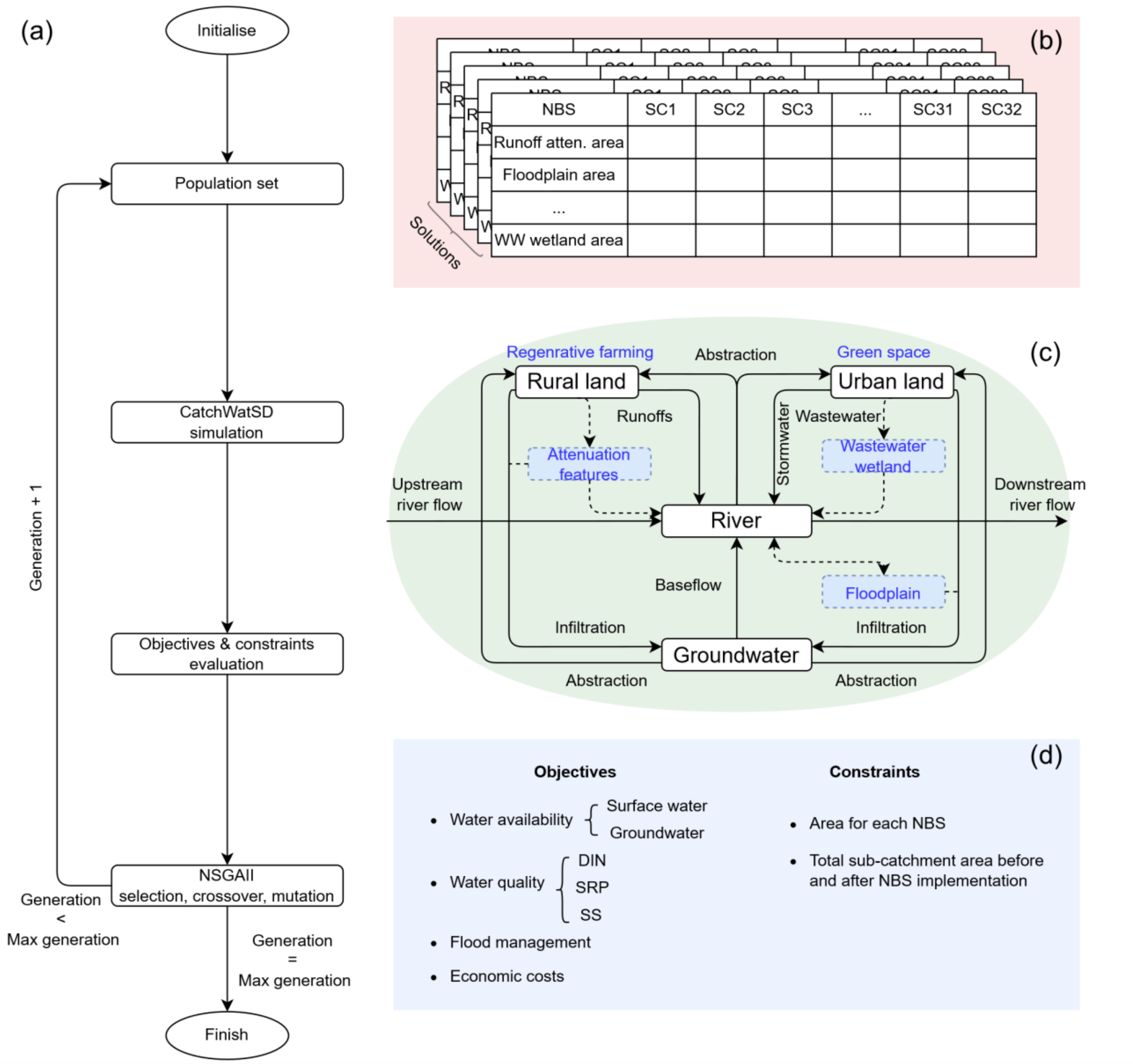


Fig.2 An integrated urban-rural NBS planning optimisation framework (a), with details on solutions (b), CatchWat-SD simulation (c), objectives and constraints (d). The blue text and boxes in (c) indicate different NBS options

3.2 CatchWat-SD modelling framework

CatchWat is an integrated model that was developed to simulate surface water dominated urban-rural systems lumped at a catchment scale (Liu et al., 2021). It simulates runoff generation and routing, water use (abstraction and effluent discharge), and in-river biochemical processes. Its modelling structure enables simulating urban-rural NBS' effects on the integrated water cycle. In this study, CatchWat is expanded to represent sub-catchment processes in a semi-distributed configuration for multi-catchment systems (Fig. 1). To represent the behaviour of different NBS options, CatchWat-SD simulates two additional physical processes: (i) groundwater storage and baseflow representations; and (ii) a river storage module that receives flows from upstream sub-catchments along with the local runoff as inflows. Detailed explanations of new CatchWat-SD modules are presented in Supplementary material S1.

The information that is used to set up and validate the CatchWat-SD model is summarised in Table 1; see Liu et al. (2021) for a full description of the different data sources.

Table 1 Data input for CatchWat-SD model set-up and validation

Data use	Variables	Source	Temporal resolution	Spatial resolution
Model set-up	Hydroclimatic	HadUK (Hollis et al., 2019)	1990-2018 daily/monthly	12x12 km
	Land use & vegetation	Crop Map of England (CROME) (Rural Payments Agency, 2020)	2019	4156 m2
	Crop calendar and crop parameters	International Production Assessment Division (IPAD) and FAO-56 (Allen et al., 1998)	-	-
	Population	Local Authority District Population for England and Wales - Census 2011 (Pope, 2017)	2011	-
	Water use	Rural: Agricultural demand forecast (Knox et al., 2017)	n.d.	-
		Urban: EA abstraction compliance (Environment agency, 2022)	2019	
	Atmospheric deposition	Concentration Based Estimated	1986-2012	5x5 km

		Deposition (CBED) dataset (Levy et al., 2020)		
	Fertilisers	UK Centre for Environment and Hydrology (CEH) Fertilisers Map	2010-2015 average	1x1 km
	Manure	ADAS manure report (Nicholson, F. A. et al., 2008)	n.d.	-
	Urban treated effluent concentration	UK EA WIMS database (Environment Agency, 2021a)	2000-2018 less frequent than monthly	11 urban wastewater discharge points
Model validation	River flow	National River Flow Archive (NRFA) (UK Centre for Ecology and Hydrology, 2020)	2000-2018 daily	6 stations
	River DIN concentration	EA WIMS database (Environment Agency, 2021)	2000-2018 monthly	18 stations
	River SRP concentration			
	River SS concentration			

The simulation period starts on 1990/01/01 and ends on 2018/12/31. We validate the CatchWat-SD model for the case study region using simulated flows and water quality against historic data. The metrics for evaluating model validation performance in this study are Nash-Sutcliffe Efficiency (NSE) and percentage of bias (PB) as common indicators used to evaluate the performance of both river flow and water quality simulations (Moriasi et al., 2007). Given the large number of parameters (> 50 per sub-catchment) in the integrated model, a formal calibration may obtain results with high performance metrics but based on 'wrong reasons' (Dobson et al., 2021). In this study, some parameters (e.g., crop coefficients, percentage of irrigated water from surface and groundwater) are selected by the best publicly available evidence (Section 2), while the others (e.g., runoff coefficients, runoff routing time) are adjusted according to expert knowledge. This might not obtain the best performance metrics against observations but can provide insights into systems reactions to parameter values, which ultimately better serves this study's purpose in understanding systems mechanisms.

3.3 CatchWat-SD representations of urban-rural NBS

In this study, we implement five types of urban-rural NBS, selected to cover interventions across the entire water cycle (Fig. 2). They are summarised in Fig. 3 and described in detail below, with equations given in Supplement material S1.

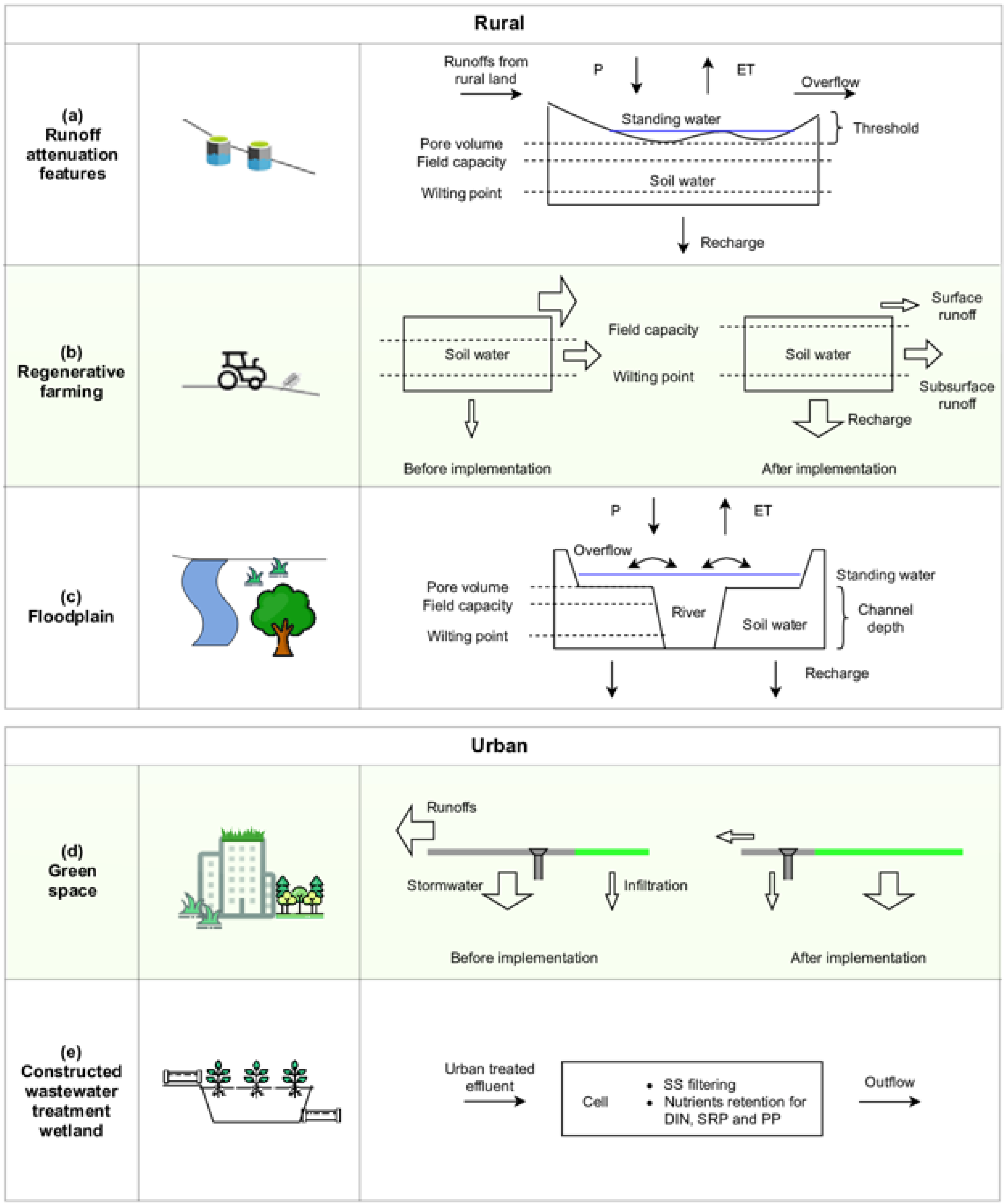


Fig. 3 CatchWat-SD model conceptualisation of five urban-rural NBS (green shade shows that NBS is simulated via parameters change, while blank background shows that NBS is simulated via developing new modules)

Runoff attenuation features are soft-engineered NBS that increase catchment runoff storage capacity during rainfall events and have various forms such as on-site ponds/wetlands and dams/barriers (Nicholson, A. R. et al., 2012; Nicholson, Alexander R. et al., 2020). We generalise runoff attenuation features as farm wetlands that are a conceptual soil water tank in the model (Fig. 3 (a)). Soil hydrological parameters (wilting point, field capacity, and total porosity) and processes (precipitation, evapotranspiration, recharge) are included in simulations. Runoff from rural land will interact with soil water in wetlands, and excess soil water that is above total porosity becomes standing water on the land surface. Water quality processes in the standing water include denitrification, suspended solids sedimentation, and macrophyte nutrient uptake. The depth of runoff attenuation features is set as a threshold, above which the standing water will flow downstream to rivers. Detailed equations on outflows and water quality processes in standing water are adopted from the HYPE model (HYPE Model Documentation, 2021) and are illustrated in Supplementary material S1.

Regenerative farming aims to improve soil health and agricultural productivity in general (Sherwood and Uphoff, 2000). Hydrologically, regenerative farming includes soil management measures that loose compacted soil structure, such as non-reversing tillage and cover crops (Jan et al., 2020). Soil pore volume is enlarged, which enables soils to hold more water. This is conceptualised by increasing field capacity up to 0.4 based on the comparison between compacted and non-compacted soil characteristics (Fig. 3(b)) (Houšková, 2016). These measures are also reported to enhance soil infiltration and groundwater recharge, both of which are hindered by compacted soil structure (SARE, 2012). Cover crops, as one regenerative farming technique, were shown to increase infiltration compared to conventional tillage by 34.8% on average (Basche and DeLonge, 2019). In the model, the original percolation coefficient increases by this percentage on the arable land with regenerative farming, limited by a maximum of 0.85. The surface and subsurface runoff coefficients decrease proportionately. The two sets of parameter values (with and without regenerative farming) are weighted based on their area to obtain the parameters for the whole rural land within a sub-catchment.

Floodplains have close interactions with rivers and wetland-related functions in flood peaks attenuation (Acreman et al., 2003; Wright et al., 2008), baseflow maintenance (Burt et al., 2002), nutrients removal (Doll et al., 2020; Jones et al., 2015; Roley et al., 2012) and sediment retention (Kløve, 2000; Tockner et al., 1999). This is due to its similar characteristics (e.g., high soil moisture content (Yin et al., 2019)) and processes (e.g., standing water storage (Cole and Brooks, 2000)) to wetland (Bradley and Gilvear, 2000; Doll et al., 2020), which makes them widely termed 'floodplain wetland' (Grapes et al., 2006). Hence, this study designs a floodplain module based on the farm wetland module. The major difference is that it now only interacts with river water (Fig. 3(c)): when the river water depth is above the standing water level, water will flow from the river to the floodplain and is added as standing water, and vice versa.

Implementing SUDs in cities, including green roofs, rain gardens and swales, expands urban green space. This is simulated by increasing pervious area and decreasing impervious area at an urban-lumped scale (Fig. 3(d)). Less urban surface runoff and in-pipe stormwater will be generated than before (Riechel et al., 2020), and fewer pollution loadings from the urban area will be directly discharged into rivers (Yang, Wenyu et al., 2021). Infiltration into soil water and groundwater is increased through the green space (Bai et al., 2018; Gillefalk et al., 2021).

Constructed wastewater treatment wetlands have the main function of nutrient retention and pollutants removal for urban treated effluent from wastewater treatment plants (Land et al., 2016). Though there have been some large-scale (> 1 km^2) modified natural wetlands used to treat wastewater effluent, we are simulating those small-scale (< 1 km^2) wetlands that are specifically designed and engineered for only receiving wastewater effluent (Kadlec, 2016). In various designs, this type of constructed wetlands normally has porous material beds that create a submerged environment for nutrients transformation, plant uptake, and solids filtering (Hickey et al., 2018). It usually has a low hydraulic retention time (< 4 days) (Tonderski et al., 2005), and thus its impacts on flows passed through can be neglected (HYPE Model Documentation, 2021). We, therefore, conceptualise it as a tank, with its inflow and outflow having the same quantity (Fig. 3(e)). Nutrient retention and solids filtering are simulated in the tank, the equations of which are adopted from HYPE (HYPE Model Documentation, 2021).

To test the performance of water management, each NBS is implemented at small to large sizes (See Section 3.4) across the whole sub-catchments in the model simulations, respectively. The NBS performance is evaluated as the relative changes in water availability, water quality, and flood management from the baseline scenario. These results are presented in Section 4.2.

$$Performance_{size,objective} = \frac{Objective_{size} - Objective_{baseline}}{Objective_{baseline}} \times 100\% \quad (1)$$

3.4 Optimisation formulation

A many-objective optimisation problem is then formulated. Seven objectives are integrated to represent different aspects of water availability, water quality, flood management, and economic cost.

For water availability, both surface water and groundwater availability are maximised. Surface water availability is calculated as the ratio of mean daily river flow ($\bar{q}$) in low-flow period (May-Oct) over daily maximum surface water licensed abstraction (LA_{sw}). Groundwater availability is calculated as the ratio of mean groundwater storage ($\bar{S}$) in low-flow period over daily maximum groundwater licensed abstraction (LA_{gw}). The objectives are calculated as the average ratio in 2014-2018 ($nyear = 5$) and averaged for all sub-catchments ($nsub = 32$).

$$Obj1: Maximise\ \frac{1}{nsub}\sum_{i}^{nsub}\frac{1}{nyear}\sum_{j}^{nyear}\frac{\bar{q}}{LA_{sw}} \quad (2)$$

$$Obj2: Maximise\ \frac{1}{nsub}\sum_{i}^{nsub}\frac{1}{nyear}\sum_{j}^{nyear}\frac{\bar{S}}{LA_{gw}} \quad (3)$$

For water quality, the annual mean river concentration of DIN ($\overline{c_{DIN}}$), SRP ($\overline{c_{SRP}}$) and SS ($\overline{c_{SS}}$) are minimised. Using this indicator complies with the current surface water quality regulation standard (DEFRA, 2014).

$$Obj3-5: Minimise\ \frac{1}{nsub}\sum_{i}^{nsub}\frac{1}{nyear}\sum_{j}^{nyear}\overline{c_k}, k = DIN, SRP, SS \quad (4\text{-}6)$$

For flood behaviour, the median of the annual maxima of river flows (QMED) during the 5-year simulation period is calculated (Kjeldsen, 2015). It is then averaged across all sub-catchments, which is to be minimised.

$$Obj6: Minimise\ \frac{1}{nsub}\sum_{i}^{nsub} median(\max(q)) \quad (7)$$

Economic costs are evaluated as the sum of capital and management costs and are minimised. Capital costs are only accounted for once, while management costs are calculated for the whole of 2014-2018. They are both calculated as the unit cost (UC_c for capital and UC_m for management) multiplied with the areas of NBS, which are then summed for all NBS together.

$$Obj7: Minimise\ \sum_{j}^{nNBS}\sum_{i}^{nsub}(UC_c \times Area_j + UC_m \times Area_j \times nyears) \quad (8)$$

The evaluation of both unit costs (Table 2) is based on existing project examples illustrated in UK NBS design guidelines (Keating, Keeble et al., 2015; Keating, Pettit et al., 2015).

Table 2 Cost evaluation and constraints on NBS implementation sizes

	Economic costs				Constraints	
NBS	*Unit capital cost (GBP/km2)*	*Reference values*	*Unit management cost (GBP/km2 /year)*	*Reference values*	*Min*	*Max*
Runoff attenuation features	5e5	Around £1/m^3 for ponds and wetlands. This study simulates shallow wetlands with an average depth of 0.5 m (Babbar-Sebens et al., 2013), which gives £0.5/m^2	1e5	£0.1-£2/m^3 of pond volume for retention (wet) ponds	0	WWNP runoff attenuation features 1% AEP
Regenerative farming	0	No documented capital cost for soil management	12300	£123/ha/year for soil management based on an arable farm system	0%	100% maximum arable land area
Floodplain	3e5	£3000/ha for floodplain woodland on average	7500	£75/ha/year	0	WWNP floodplain woodland potential
Urban green space	12.5e6	£10-£15/m^2 swale area	1e5	£0.1/m^2 for swale surface area	25% urban area	82% urban area
Constructed wastewater treatment wetland	27.5e6	£25-£30/m^3 treated volume for urban constructed wetland, our design of constructed wastewater wetlands has an average depth of 1 m (Solano et al., 2004; Zhang, L. et al., 2010)	1e5	£0.1/m^2 for wetland surface area	0	33 ha

The decision variables are the areas of each five NBS (i.e., five decision variables per catchment), whose constraints are shown in Table 2. The constraints for runoff attenuation features and floodplain are obtained from the Working With Natural Processes (WWNP) dataset (Environment Agency, 2020). This dataset includes evaluated potential area for runoff attenuation features under 1% annual exceedance probability and floodplain woodland, respectively. The dataset is aggregated for each sub-catchment as the maximum potential area for the implementation of these two NBS. The average maximum constraints across all sub-catchments are 0.18 and 1.40 km^2 for runoff attenuation features and floodplain woodland, respectively. The maximum area for implementing regenerative farming techniques is set as the whole arable land area within each sub-catchment, whose average value is 28 km^2. The minimum percentage of urban green space is 25%, which is the baseline evaluation in Norwich city (Office for National Statistics, 2020). The maximum percentage of urban green space is 82%, which is the evaluated green space accessibility potential for global cities (Huang et al., 2021). Kadlec (2016) reviewed 87 small constructed wastewater treatment wetlands and found the maximum area up to 33 ha, which is adopted as a constraint in the study.

The simulation period is between 2014-2018 for the optimization problem. The population size of solutions for the NSGAII algorithm is 200, with the number of generations set as 10000.

4. Results

4.1 Model evaluation

We present simulated river flow and water quality indicators against available observed data at all the monitored stations and summarise the performance metrics (e.g., Nash-Sutcliffe efficiency (NSE)) in full in Supplementary material S2. The simulated river flow generally has a good match with the observed data series at all stations (Fig. 4(a)), always above 0.7, and with a percent bias (PB) within ±20%.

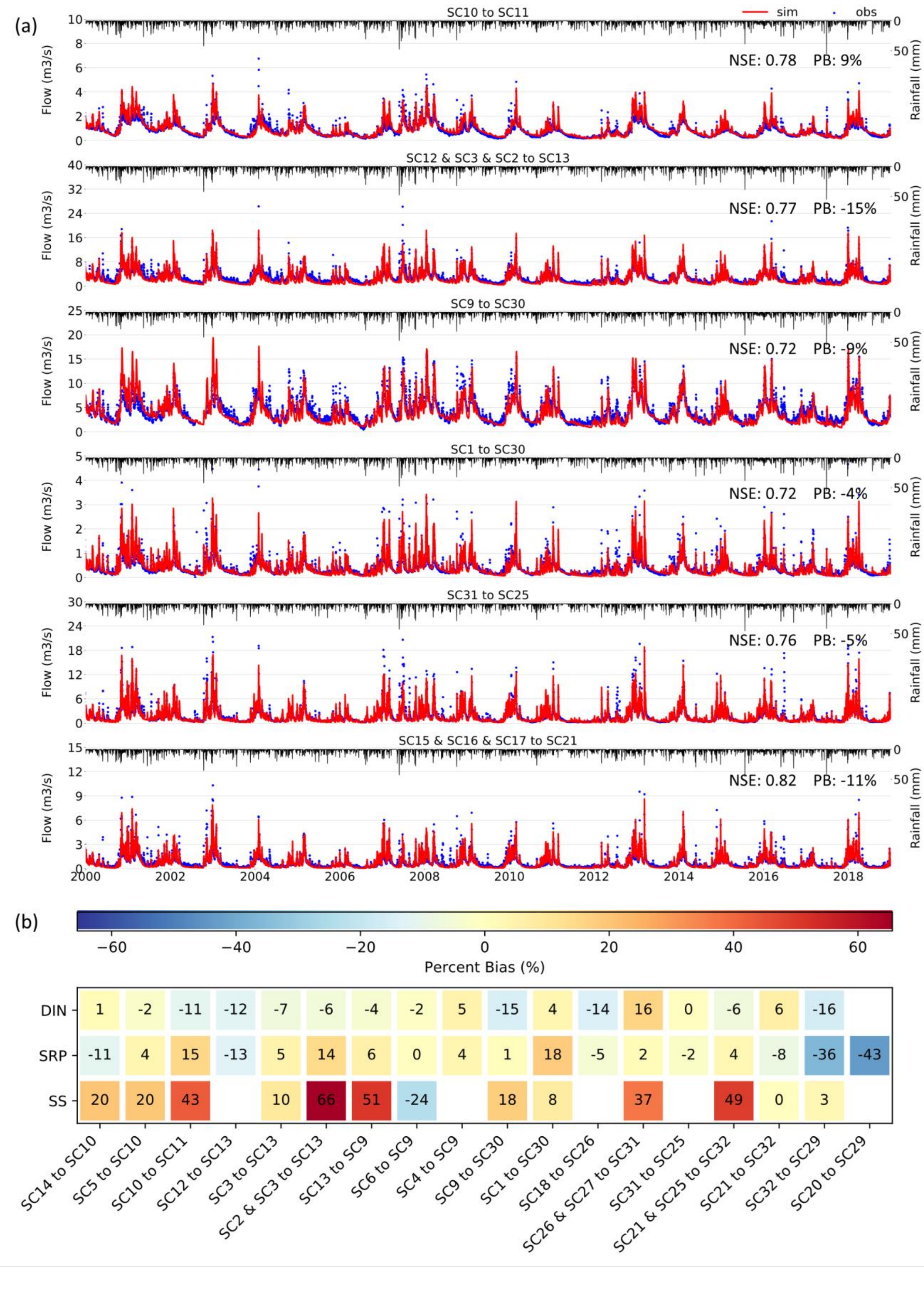

(a)
SC10 to SC11
sim
obs
NSE: 0.78 PB: 9%
SC12 & SC3 & SC2 to SC13
NSE: 0.77 PB: -15%
SC9 to SC30
NSE: 0.72 PB: -9%
SC1 to SC30
NSE: 0.72 PB: -4%
SC31 to SC25
NSE: 0.76 PB: -5%
SC15 & SC16 & SC17 to SC21
NSE: 0.82 PB: -11%
Flow (m3/s)
Rainfall (mm)
2000 2002 2004 2006 2008 2010 2012 2014 2016 2018
(b)
−60 −40 −20 0 20 40 60
Percent Bias (%)
DIN 1 -2 -11 -12 -7 -6 -4 -2 5 -15 4 -14 16 0 -6 6 -16
SRP -11 4 15 -13 5 14 6 0 4 1 18 -5 2 -2 4 -8 -36 -43
SS 20 20 43 10 66 51 -24 18 8 37 49 0 3
SC14 to SC10
SC5 to SC10
SC10 to SC11
SC12 to SC13
SC3 to SC13
SC2 & SC3 to SC13
SC13 to SC9
SC6 to SC9
SC4 to SC9
SC9 to SC30
SC1 to SC30
SC18 to SC26
SC26 & SC27 to SC31
SC31 to SC25
SC21 & SC25 to SC32
SC21 to SC32
SC32 to SC29
SC20 to SC29

Fig. 4 Simulated river flows against observed data at six validation stations (a) and a heatmap for percent bias between simulated results and observed data for river water quality at 18 validation stations (b) (DIN = dissolved inorganic nitrogen, SRP = soluble reactive phosphorus, SS = suspended solids, blank = no available sampled data series for validation).

In Fig. 4(b), we specifically present the results of percent bias (PB) for water quality. We show percent bias because the water quality objectives are based on the mean change in concentration between the scenarios before and after NBS implementations (Section 4.2). Simulated dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) results have a generally small percent bias (less than ±20%). SRP in SC20 (-43%) and SC32 (-36%) is significantly underestimated, especially during dry seasons when urban effluents induce SRP peaks (Fig. S3 in Supplementary material S2). Though the magnitude in the upstream Wensum (SC14 and SC5) matches with the sampled SRP data (PB < ±20%), capturing the temporal trend in the data could be improved. The slight rising trend during summer seen in Fig. S3 might be induced by unaccounted wastewater discharge (Cooper et al., 2022) or more complex phosphorus processes in the river (e.g., sediment exchange) (Demars and Harper, 2005; Roberts and Cooper, 2018). In contrast, suspended solids (SS) simulations capture temporal peaks well (see Fig. S4 in Supplementary material S2), but are generally overestimated (> 30%), particularly during wet seasons when rural soil erosion drives significant fluctuations in river SS concentration. Overall, the performance of the model is within the comparable range with existing catchment scale water quality studies (Hankin et al., 2019).

4.2 NBS performance on water availability, water quality and flood management

The performance of the five urban-rural NBS for water availability, water quality, and flood management benefits, which are evaluated as the percentage change from the baseline scenario (Eq. 1), varies with NBS implementation sizes (Table 3). Within the maximum implementation size (Table 2), the larger an NBS is implemented, the more significant performance it has. However, the performance is different between urban and rural NBS.

Table. 3 NBS performance on water availability, water quality and flood management averaged across all sub-catchments, with economic costs evaluation

NBS	**Implementation sizes**	**Water availability**		**Water quality**			**Flood management**	**Economic cost (Million GBP)**
		Surface water	Ground water	DIN	SRP	SS	QMED	
Runoff attenuation features	0.15 km^2	8.1%	52.1%	-3.4%	-10.9%	-27.5%	-12.4%	2.88
	0.3 km^2	15.2%	71.5%	-4.9%	-13.5%	-31.8%	-22.1%	5.76
	0.45 km^2	19.1%	77.2%	-6.2%	-14.7%	-33.6%	-27.5%	8.64

Regenerative farming	25%	-4.2%	-3.5%	-3.0%	-0.9%	-7.6%	-10.3%	14
	50%	-6.7%	-6.8%	-6.1%	-1.6%	-14.0%	-21.4%	27.5
	75%	-8.7%	-9.9%	-9.0%	-2.4%	-19.3%	-29.1%	41.3
Floodplain	1.5 km^2	20.1%	89.7%	-4.0%	-13.0%	-13.3%	-49.6%	16
	3 km^2	31.5%	116.6%	-6.7%	-17.8%	-17.2%	-58.7%	32.4
	4.5 km^2	38.0%	121.1%	-8.9%	-20.9%	-19.4%	-63.0%	48.6
Urban green space	45%	-1.1%	0.1%	-1.1%	-2.5%	0.0%	-0.3%	113.4
	65%	-2.3%	0.1%	-2.3%	-5.3%	-0.1%	-0.6%	226.7
	85%	-3.6%	0.2%	-3.7%	-8.6%	-0.2%	-0.9%	340.1
Constructed wastewater treatment wetland	0.1 ha	0.0%	0.0%	-0.2%	-0.8%	-0.2%	0.0%	0.028
	1 ha	0.0%	0.0%	-1.5%	-4.2%	-1.0%	0.0%	0.28
	10 ha	0.0%	0.0%	-5.6%	-9.8%	-2.3%	0.0%	2.8

Runoff attenuation features and floodplains are more effective measures for increasing water availability, providing an additional 19.1% and 38.0% of surface water and 77.2% and 121.1% of groundwater resources, respectively. Both NBS can divert surface runoff into groundwater storage during the wet period, which will increase the baseflow into rivers during the dry period. These processes attenuate high river flows, which has been observed in previous studies (Acreman et al., 2003; Blanchette et al., 2019; Hensel and Miller, 1991).

In contrast, regenerative farming covering 75% of cropland across the region slightly decreases the water availability overall, by almost 10% for both surface and groundwater. Though the NBS can generally increase the groundwater recharge rate when the soil moisture content reaches the field capacity, it also increases field capacity so that more water will be stored in the soil and less water for recharge will be generated. The results show that the latter effect is more significant across the catchment. As a result, the groundwater storage and baseflow to rivers are decreased in dry periods.

Due to the small percentage of area (9%) across the study region, urban green space expansion generally has limited effects on water availability. It slightly increases groundwater availability (< 0.2%), as it allows more rainfall to infiltrate into the soil and recharge groundwater. However, it slightly decreases surface water availability (by up to 3.6%), mainly due to reduced stormwater and urban surface runoff generation. This is notable in summers when both flows account for a higher percentage of water in rivers. Finally, given that small constructed wastewater wetlands are assumed to not significantly affect natural hydrological processes in this study, their effects on water availability are minor.

In the context of water quality management, rural NBS can reduce river DIN, SRP, and SS concentration to a larger extent than urban NBS, by 9%, 20.9%, and 33.6%, respectively. Runoff attenuation features and regenerative farming intervene in soil processes. Both NBS decrease surface runoff generation, which reduces the quantity of runoff pollutants reaching rivers. This is particularly effective for SS (33.6% and 19.4%, respectively), as heavy rainfalls and the resulting runoff cause significant soil erosion on rural land (Peng and Wang, 2012). In addition, runoff attenuation features include processes for soil adsorption and plant uptake of SRP, which more significantly (by 14.7%) reduces the SRP than regenerative farming. Floodplains directly interact with pollutants in rivers by diverting river water into riparian soil and groundwater. Soil nutrient processes and solids filtering processes in the floodplain are triggered, which results in a good performance in removing river pollutants (8.9% for DIN and around 20% for SRP and SS).

Urban NBS are generally effective in reducing river SRP, by up to 8.6%. Urban green space expansion generates less stormwater and reduces total wastewater amount in a combined sewer system, which consequently reduces discharged SRP loadings. In contrast, constructed wastewater wetland directly treats wastewater effluent via biochemical processes, including adsorption and plant uptake (Kadlec, 2016). Both NBS decrease SRP loadings discharged into rivers and hence improve river SRP. Urban NBS are less effective for managing DIN and particularly SS reduction across the study region. This is because the main sources of DIN and SS are fertiliser application and soil erosion, both of which impact the system through the rural water cycle.

At a catchment scale, rural measures are more effective in flood mitigation than urban measures, with up to 63% QMED reduction at the catchment outlet. As surface runoff generated on rural land is the main cause of river flood peaks, three rural measures all provide more water storage and divert the stored water into groundwater that takes a longer time to route into rivers. These processes significantly reduce the flow peaks across all the sub-catchments. Urban green space expansion can also decrease high flow peaks by generating less flashy urban runoff and in-pipe stormwater. However, such effects are minor because urban runoff and in-pipe stormwater account for very little proportion of the river flow during the wet period.

With much higher capital costs than the other NBS (Table 2), urban green space expansion is the most expensive NBS, costing more than 100 million GBP to implement region-wide. Floodplains and regenerative farming can cost up to 50 million GBP if each of them is implemented to its maximum extent. Given the limited maximum area for runoff attenuation features and the proposed size of constructed wastewater wetlands, their costs are the lowest (< 10 million GBP).

4.3 NBS implementation co-benefits and trade-offs

The objective values of the final population (200 solutions) from the MOEA and the baseline are shown in Fig. 5. The solutions are coloured to show the variation in economic costs. Results show that achieving the maximum surface and groundwater availability requires lower investment (blue scenarios), while achieving the optimal DIN, SS, and flood management objectives will be more costly (green and yellow scenarios). Achieving maximum SRP reduction is the most expensive solution (red scenarios).

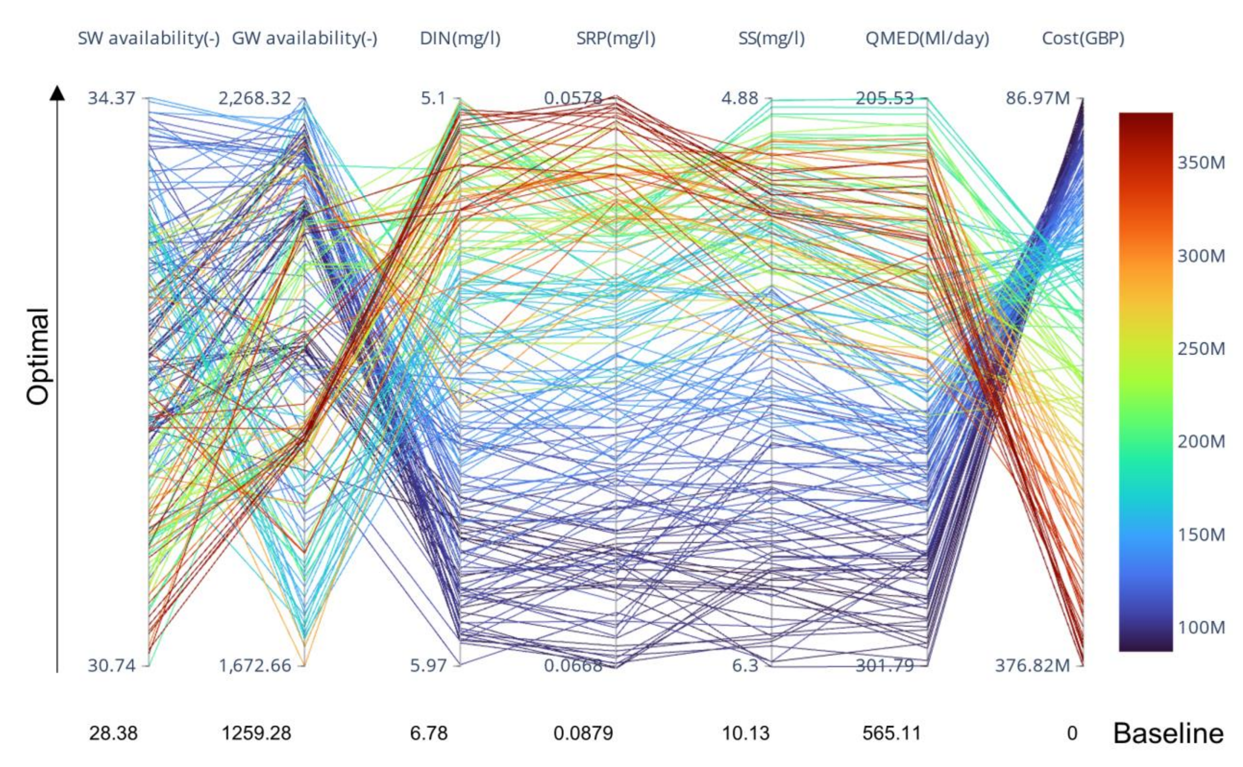


Fig. 5 Parallel coordinates plot for integrated objectives of the final population (200 solutions) from the MOEA (coloured based on costs)

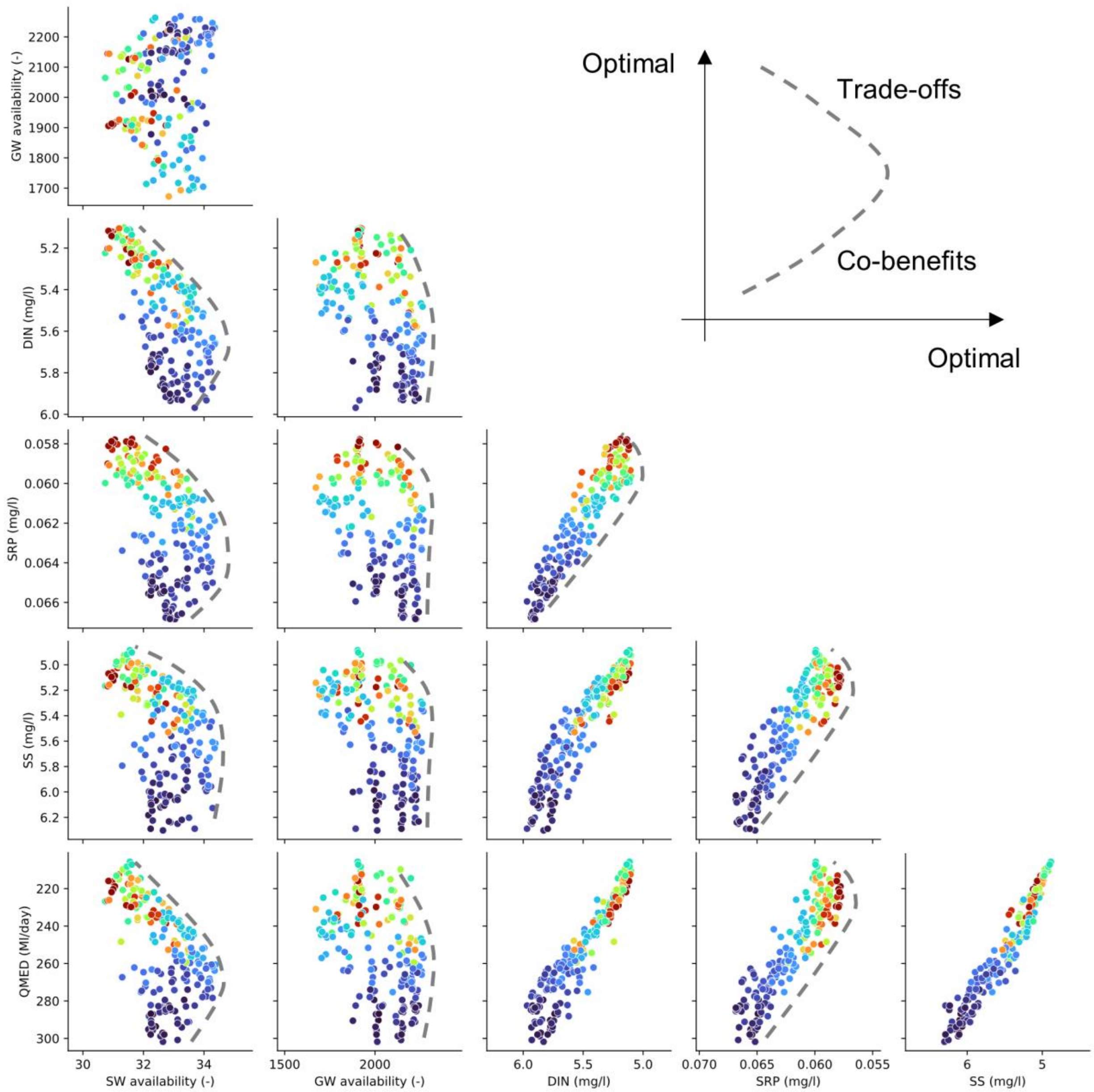


Fig. 6 Pair-wise plots between water management objectives of the final population (200 solutions) from the MOEA, illustrating co-benefits and trade-offs. Scatters are coloured based on the costs.

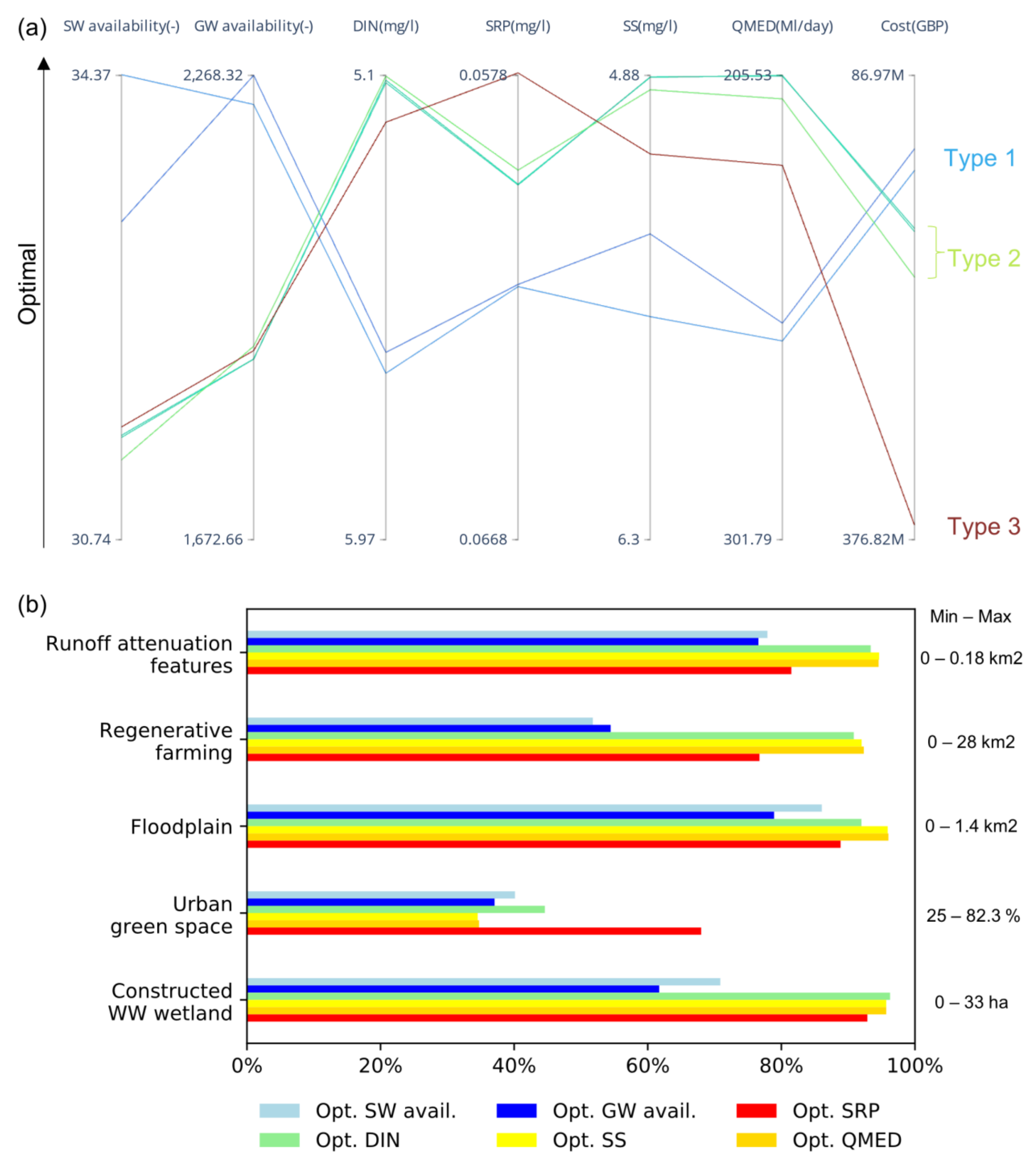
While achieving multiple objectives comes at a range of investment costs, it should be noted that even scenarios with the least optimised values for water management objectives provide improvement, compared to the baseline scenario. This shows that the integrated implementation of five selected urban-rural NBS is able to improve water availability, water quality, and flood management simultaneously, which demonstrates strong co-benefits for water management. However, no single solution can bring all the water management objectives to their optimal value simultaneously, indicating that trade-offs exist.

To better illustrate the co-benefits and trade-offs, objective values are plotted between water availability, water quality, and flood management objectives in Fig. 6, respectively. A clear positive correlation exists between DIN, SS, and flood objectives, indicating that these objectives can be co-improved to their optimal values simultaneously by implementing integrated urban-rural NBS. Similarly, SRP and DIN, SS, and flood objectives can be improved together as well (a general positive correlation). However, there is a change in trend where SRP reaches its optimal value, with DIN, SS, and flood objectives values starting to deteriorate, which indicates trade-offs between SRP and the other three objectives.

The most significant trade-off pattern is between water availability and water quality and flood management objectives. Water availability and the other objectives can be co-improved initially when implementing integrated urban-rural NBS, especially for increasing surface water availability. However, there is a clear tipping point, after which the other objectives are still improved but both surface and groundwater availability are decreased. This indicates that improving water quality and flood objectives comes at a cost of failing to maximise water availability, though the scenarios still increase water availability compared to the baseline.

4.4 Optimal NBS implementation for integrated objectives

To maximise NBS implementation potential, it is crucial to understand catchment system mechanisms that explain co-benefits and trade-off phenomena. We select six solution scenarios that achieve optimal values for each water management objective to explore the mechanisms (Fig. 7(a)). The NBS implementation sizes for each scenario are averaged across the whole catchment and standardised based on introduced implementation constraints (Fig. 7(b)).



434

435 Fig. 7 Six solutions that achieve individual water management objectives to their optimal are
436 clustered into three types (a); the NBS implementation sizes averaged across all sub-catchments in
437 these six solutions (b), which are standardised in 0 – 100% based on each NBS constraints
438 (illustrated on the right side).

439 We categorise selected solution scenarios into three types based on the optimal objectives they
440 achieve, the similarity in the NBS implementation sizes, and the corresponded economic costs. The

solutions that achieve the optimal surface and groundwater availability are classified into Type 1. Both scenarios implement around 80% maximum implementation sizes (MIS) of runoff attenuation and floodplain, 50% MIS of regenerative farming and constructed wastewater wetland, and 40% MIS of urban green space (Fig. 7(b)). This demonstrates the runoff attenuation features and floodplains are prioritised NBS for improving water availability due to their better performance (Table 3). Both solutions require an investment of about 150 million GBP (Fig. 8).

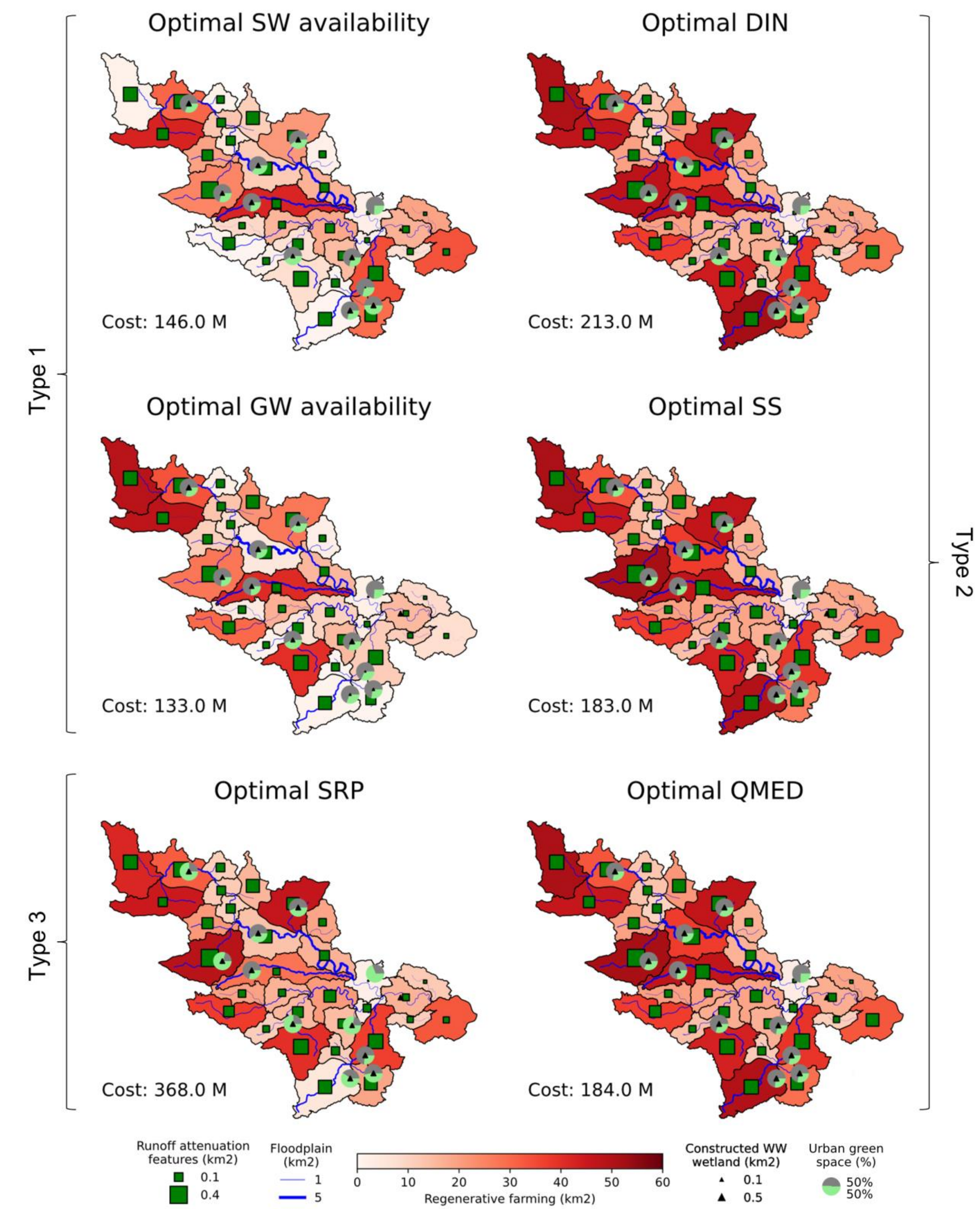


Fig. 8 Spatial configurations of NBS implementation in the six solutions that achieve individual optimal objectives

The DIN, SS, and flood can be improved to their optimal simultaneously, whose optimal scenarios are classified into Type 2. Compared to Type 1, runoff attenuation and floodplain NBS implementation slightly increases to more than 90% MIS, respectively. Regenerative farming almost doubles compared to Type 1 solution to around 90% MIS. This highlights the effects of regenerative farming in generating less surface runoff, which will reduce flood peaks and carried DIN and SS loadings. However, larger implementation of regenerative farming reduces water availability (Table 3) by increasing soil water storage, creating the trade-offs between water availability and other objectives (Fig. 6). Although there is also a significant increase in constructed wastewater wetland implementation in Type 2 solutions by approximately 30%, this NBS helps to improve DIN rather than SS or flood based on its performance in Table 3. Urban green space maintains a similar design to Type 1 solutions. The expansion of the implementation sizes in this type increases the total costs to 180 – 210 million GBP.

Finally, SRP optimal scenario is classified into Type 3, as its NBS implementation shows a different pattern. It significantly increases the sizes of urban green space uptake to more than 65% MIS, with the other NBS designs comparable to Type 1 and Type 2 solutions. This highlights the effects of urban green space in reducing stormwater generation and consequently reducing wastewater SRP loadings in combined sewage systems. The high capital cost of such significant urban green space expansion increases the total investment to about 370 million GBP, which is the most expensive NBS implementation among all six solutions. It should be noted that Type 3 does not require as much regenerative farming as Type 2, with approximately 75% MIS. This is largely due to the ineffective SRP reduction by regenerative farming (Table 3). As a result, DIN, SS, and flood objectives cannot reach their optimal values as in Type 2 solutions. This explains the trade-offs between SRP and the other three objectives shown in Fig. 6.

4.5 Spatial configurations of optimal NBS implementation

In the optimal scenarios, the spatial configurations of NBS implementation exhibit a heterogeneous pattern (Fig. 8), where catchments are prioritised locations for implementing certain types of NBS. Some of these patterns are driven by physical mechanisms. For optimal groundwater availability, regenerative farming is priortised in the upstream Wensum catchment (SC5 and 14), which increases groundwater recharge. For optimal surface water availability, regenerative farming is not a preferred option to be implemented in the upstream Yare catchment (SC16-19 and SC23-24), due to its impact on decreasing groundwater recharge. MIS also affects such a pattern. For example, regenerative farming is implemented at a general minimum size in SC30, as it is a very urbanised sub-catchment with the least rural land area. Finally, observed patterns might be driven by randomness in solution generation by MOEA. In the optimal surface water availability scenario, SC5 has a much larger regenerative farming implementation than SC14, even though both sub-catchments have similar physical conditions and MIS. A similar pattern also exists in regenerative farming implementation in SC18 and SC19 in the optimal groundwater availability scenario.

5. Discussion

The first contribution of the study is to develop an integrated modelling approach to simulate urban-rural NBS implementation. The proposed CatchWat-SD model successfully simulates urban-rural NBS, which enables the comparison of their performance at a catchment scale. The scale of analysis is found to significantly affect evaluating urban-rural NBS performance in catchment water management. In this analysis, rural NBS' performance is more significant than urban interventions when analysed at the catchment outlet. This is because the river flows in most sub-catchments (SC) are mostly from the rural water system given the dominant rural land use in Norfolk. Nevertheless, urban NBS can still generate significant local benefits, especially in highly impervious areas (e.g., SC30 that contains the city of Norwich, Fig. 1). Results show that expanding urban green space from 25% (baseline) to 45% in Norwich can increase the groundwater availability in SC30 by 153%. This demonstrates the need for integrated urban-rural NBS implementation, whose performance can be tested using developed modelling framework.

Although NBS performance is generally difficult to validate, CatchWat-SD results are comparable with previous studies. For example, Yang et al. (2010) simulated that wetland restoration can reduce 23.4% peak discharge, which is in the range of 12.4%-27.5% for runoff attenuation features in this study; Roley et al. (2012) showed a less than 10% nitrate loading reduction caused by a constructed floodplain in an agricultural catchment, which is aligned with the 8.9% maximum DIN reduction obtained in this study. However, because it is formulated as part of a conceptual modelling framework, the main value in CatchWat-SD is to contextualise NBS within the wider integrated urban-rural water cycles and thus to explore the plausible catchment scale changes that may result from their implementation. This makes it a complementary tool for simulating and comparing the performance of different integrated urban-rural NBS planning at a catchment scale. We highlight that CatchWat-SD is not a substitute for a detailed hydraulic representation of NBS nor for monitoring and empirical evidence, which are needed to verify that an NBS of a given size/location behaves in a given way.

The second contribution of this study is demonstrating that integrated NBS planning has co-benefits for water availability, water quality, and flood management for catchment water management. This is supported by evidence presented in Fig. 5 that all 200 optimal solutions (where a solution is a portfolio of NBS options) achieve improvements in each objective compared to the baseline scenario (i.e., no NBS options). However, there is no solution that dominates individual objectives; instead there are a range of trade-offs between multiple water management objectives.

The most significant trade-offs exist between water availability and the other indicators. For example, regenerative farming can improve water quality and flood management but decreases water availability by storing more water in the soil. However, whether this can be interpreted as decreasing the water availability as a whole might depend on how water availability is defined. More soil water means more 'green' water availability for plants, which may generate ecological benefits such as increased biodiversity (Bykova et al., 2019). Only considering its effects on 'blue' water for human use cannot reveal the water availability in a wider context that includes ecological benefits. In addition, results show that trade-offs exist between different pollutants, which are driven by different pollution sources (e.g., N and SS are from the rural water cycle, while P is from the urban water cycle (Liu et al., 2021)). Both findings highlight the need for incorporating indicators in a wider systems context as objectives in the optimisation.

The third contribution of this study is to provide evidence that could be used to support decisions around NBS implementation. For example, results show that green space in all cities should be expanded for optimal phosphorus management to reduce the significant loading from urban effluent. However, given the high capital costs involved (Table 2), such large-scale implementation is likely to be extremely expensive (>300 million GBP for a 5-year construction and management period) and might not be cost-efficient. Identifying prioritised locations for NBS implementation is needed in decision-making, especially when bounded by a limited budget. Our results also provide potential prioritised locations based on physical mechanisms that are explained in Section 4.5. However, socio-economic factors should be considered in decision-making as well. For example, among all the cities included in the study (Fig. 1), Norwich might be the prioritised location for urban green space expansion, considering the number of people that would benefit from the implementation. Further detailed analysis that integrates local citizen perspectives with multi-stakeholder engagement (Peters and Landström, 2021) is needed for determining prioritised NBS locations.

6. Conclusions

This study developed an integrated NBS planning optimisation framework and applied it in the Norfolk region in the UK. In the framework, CatchWat-SD is developed to enable simulating five selected urban-rural NBS, including runoff attenuation features, regenerative farming, floodplain, urban green space, and constructed wastewater wetland, and their effects on the integrated water cycle. Their performance in water availability, water quality, and flood management, as integrated water management objectives, is evaluated at the catchment scale, with the associated economic costs. A many-objective optimization problem is formulated to find optimal NBS planning.

This study finds that integrated urban-rural NBS planning has co-benefits for water availability, water quality, and flood management. Floodplain and runoff attenuation features are prioritised NBS for implementation. Achieving optimal water quality and flood management will result in trade-offs with water availability. Such trade-offs are caused by a large-scale implementation of regenerative farming across multiple sub-catchments, which overall decreases water availability by increasing soil water storage. For water quality objectives, maximising SRP reduction requires expansion of urban green space to decrease the wastewater loading in combined sewer systems, which is accompanied by significant economic costs. These systems insights are useful information for multi-stakeholder decision-making in integrated urban-rural NBS planning at a catchment scale. Robust NBS design and planning require detailed modelling tools and additional social-environmental information.

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