Assessing the impact of spatial allocation of bioretention cells on shallow groundwater – an integrated surface-subsurface catchment-scale analysis with SWMM-MODFLOW

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Abstract: Well-designed and implemented green infrastructure (GI) can help to recover the natural hydrologic regimes of urban areas. Large-scale GI planning requires understanding the impact of GI spatial allocation on surface-subsurface hydrologic dynamics. This study develops a coupled surface-subsurface hydrological model (SWMM-MODFLOW) that simulates fine-temporal-scale two-way interactions between GI and groundwater at a catchment scale. The model was calibrated and validated using monitoring data from an urban catchment within Kitsap County, WA, US. Based on the validated model, a series of hypothetical simulations were then performed to evaluate how the spatial allocation of GI, in particular bioretention cells, influences and correlates with surface runoff and groundwater table dynamics. The spatial allocation was represented by the implementation ratio (i.e., area), the aggregation level (i.e., density), and the location of bioretention cells. The dynamics were quantified by peak and volume reductions of surface runoff, as well as groundwater table rise and the standard deviation of groundwater levels. The implementation ratio of bioretention cells was found to be the main spatial feature that governed both surface runoff and groundwater table dynamics. With the implementation of more bioretention cells, greater amounts of runoff could be controlled and the groundwater table rose, but the spatial uniformity of regional groundwater levels (i.e., the standard deviation of groundwater levels) was not significantly affected. Bioretention cells should therefore be allocated in a distributed pattern when groundwater table depth is relatively uniform. Allocating bioretention cells in upstream areas can generally raise the groundwater levels downstream, but their exact locations must still be determined based on the geophysical conditions and spatial variations within the catchment. Bioretention cells with greater surface runoff control efficiencies lead to higher groundwater table rises, which highlights the importance of considering tradeoffs between surface runoff control and groundwater protection in GI planning.

Keywords: low impact development; bioretention cell; stormwater management; integrated modeling; groundwater modeling; urban planning

1. Introduction

Excessive urbanization has significantly deteriorated natural hydrological, ecological, and biological regimes. There is now a general consensus that more sustainable and environmentally friendly development approaches are needed (Song, 2005). Green infrastructure (GI) has been proposed as an approach for this conceptual revolution (Brown et al., 2009). However, the definition of GI can vary. For fields concerned with hydrology and stormwater management, GI is analogous to concepts such as low impact development (LID), sustainable urban drainage systems (SUDS), and water sensitive urban design (WSUD), which represent a group of semi-natural spatially distributed stormwater management practices (Potter, 2006; Young et al., 2014; Fletcher et al., 2015). Compared with traditional drainage systems, these possess more diverse functionalities, which include collecting, storing, and infiltrating rainfall runoff, and recovering natural hydrological cycles (Chui et al., 2016). Representative practices include bioretention cells, porous pavements, green roofs, etc. Alternatively, for fields such as landscape design and urban planning, GI can include forests or other green spaces that provide other environmental benefits such as urban heat island mitigation and biodiversity improvement (EC, 2013; Zhang and Chui, 2019). GI as referred to in this study follows the first definition, focusing on hydrology and stormwater management.

Among the benefits that GI can provide, such as reduction of peak runoff and control of non-point source pollution, groundwater recharge is one benefit that attracts relatively little attention (Jefferson et al., 2017; Sohn et al., 2019). One reason for this may be that recharging groundwater using GI comes with many challenges, which are particularly prominent in shallow groundwater areas. For example, a groundwater mound can form when the groundwater recharge rate exceeds the dissipation rate. This may slow down or inhibit surface infiltration, and increases the risk of groundwater contamination due to a shorter traveling distance and more carried pollutants (Fischer et al., 2003; Datry et al., 2004; Göbel et al., 2004; Endreny and Collins, 2009; Machusick et al., 2011; Stewart et al., 2017; Zhang and Chui, 2017; Zhang and Chui, 2018a). However, recharging groundwater using GI can increase the baseflow, recover the hydrological cycle, and help maintain urban water supplies (Newcomer et al., 2014; Bhaskar et al., 2016, 2018; Bradshaw and Luthy, 2018). It should therefore be promoted in the appropriate conditions, such as in locations with suitable subsurface soil properties and a relatively deep groundwater table (Trinh and Chui, 2013; Chui and Trinh, 2016).

For the reasons aforementioned, the objectives and constraints of groundwater recharge should be thoroughly considered in GI planning. However, maximizing the control of surface runoff often remains the dominant objective in GI implementation. As reviewed by Zhang and Chui (2018b), many studies considered the peak and volume control of surface runoff (Perez-Pedini et al., 2005; Damodaram and Zechman, 2013; Sebti et al., 2016; Giacomoni and Joseph, 2017; Lim and Welty, 2017; Voter and Loheide, 2018). Other studies considered pollution mitigation of surface runoff (Maringanti et al., 2009; Rodriguez et al., 2011; Chiang et al., 2014; Chen et al., 2015, 2016), and some examined the two aspects together (Lee et al., 2012; Liu et al., 2016a, 2016b; Mao et al., 2017; Xu et al., 2018). For the relationship between GI and groundwater, some studies assessed the response of shallow groundwater to GI (Endreny and Collins, 2009; Trinh and Chui, 2013, Chui and Trinh, 2016; Zheng et al., 2018), while others proposed recommendations about suitable distances between GI and the groundwater table (Locatelli et al., 2015; Zhang and Chui, 2017; Muñoz-Carpena et al., 2018; Lauvernet and Muñoz-Carpena, 2018). However, the impact of GI spatial allocations on shallow groundwater table dynamics remains to be evaluated.

The spatial allocation of GI is hypothesized to affect the groundwater table dynamics in a number of aspects. Based on the study of Zhang and Chui (2018b), the spatial allocation of GI can be mainly represented by the implementation ratio (i.e., area), aggregation level (i.e., density), and location. First, the implementation ratio of GI is a major factor because it determines the amount of rainfall that can be infiltrated. With a higher implementation ratio, more water can be recharged, and the groundwater table should rise higher. Second, the aggregation level of GI is also influential, because more-aggregated GI practices can cause local water infiltration and result in groundwater mounds as reported by Endreny and Collins (2009). Third, the location of GI also matters, as the land use and geologic conditions (e.g., the hydraulic properties of in-situ soil, groundwater table depth) can be very different at different locations. Specifically, in areas of higher imperviousness, more permeable soils, and shallower groundwater tables, GI can affect the groundwater table dynamics more dramatically, due to greater surface runoff and higher infiltration and recharge rates. The concept of variable source area explains the impacts of these factors (Miles and Band, 2015; Lim, 2016). Additionally, the spatial allocation of GI can be determined, based only on land use and geologic factors, by using spatial analysis tools without hydrological analysis (Martin-Mikle et al., 2015; Johnson and Sample, 2017).

Although there are many different numerical models that can simulate the hydrological processes of GI, they all have limitations in simulating GI in shallow groundwater environments. As reviewed by Zhang et al. (2018), variably saturated porous media software, e.g. COMSOL Multiphysics, VS2D, Hydrus 1D/2D/3D, have been used in some cases (He and Davis, 2010; Stewart et al., 2017; Zhang and Chui, 2017). However, they generally cannot handle, or are not suitable for, catchment-scale studies because they simplify or cannot simulate rainfall-runoff generation and surface runoff routing. Alternatively, some surface-subsurface hydrological models (e.g., MODHMS, MIKE-SHE, and VELMA) can better simulate rainfall-runoff processes and are more widely used at the catchment scale (Barron et al., 2013; Trinh and Chui, 2013; Locatelli et al., 2017; Hoghooghi et al., 2018). However, they mostly operate at relatively coarse temporal and spatial resolutions, which are beyond the normal scales of individual GI practices. Thus, in all current models certain time- and space-sensitive hydrological processes important to GI are overly simplified. Many of these tools are also commercial or non-open source software, which makes them harder to improve or integrate with other tools for data analysis and optimization. Importantly, neither type of model can simulate urban hydraulics, such as storm sewer systems, which limits their usage in urban areas where GI may have the greatest impact.

As an urban hydrologic-hydraulic model, SWMM has been widely adopted to simulate GI, including to assess the hydrological and water quality treatment performance of GI (Qin et al., 2013; Palla and Gnecco, 2015; Chui et al., 2016; Jayasooriya et al., 2016; Avellaneda et al., 2017; Kong et al., 2017). It is also used to evaluate the optimal designs and allocations of GI (Elliott et al., 2009; Lucas and Sample, 2015; Giacomoni and Joseph, 2017; Macro et al., 2018; Yang and Chui, 2018a, 2018b; Zischg et al., 2018). However, SWMM is not as capable in simulating the subsurface hydrological performance of GI. First, it highly simplifies the simulation of unsaturated and saturated flows by assuming a linearized soil water retention curve. Second, it neglects the impact of groundwater on the hydrological processes of GI (e.g., exfiltration, percolation, underdrain flow) (Lee et al., 2018; Zhang et al., 2018). To partially overcome these deficiencies, Zhang et al. (2018) improved SWMM by creating an interface to incorporate groundwater levels into the simulation of GI, and showed that the modified SWMM is appropriate for simulating the performance of GI in shallow groundwater environments. However, it cannot simulate groundwater dynamics and requires the direct input of groundwater levels, which greatly hinders its application.

This study integrates the modified SWMM, named SWMM-LID-GW, with MODFLOW, which is a finite-difference groundwater flow model developed by U.S. Geological Survey, to develop a loosely coupled surface-subsurface hydrological model, named SWMM-MODFLOW. It is loosely coupled because the two models were integrated through external file input and output without internal function calls. The coupling approach utilized in this study is similar to that of Zhang et al. (2018), however, the groundwater dynamics are simulated instead of being input, and two-way interactions between GI and groundwater are realized. The model was calibrated and validated using monitoring data from an urban catchment at Silverdale, WA, in the US. Then, using a bioretention cell (BC) that allows exfiltration as a representative GI, a series of hypothetical scenarios of different spatial allocation patterns of BCs was simulated, which covered different implementation ratios, aggregation levels, and locations of BCs within the same catchment. The influence of the spatial allocation of BCs on surface runoff and groundwater table dynamics was also evaluated. Finally, the correlations between surface runoff and groundwater table dynamics were examined. This study focused on groundwater, rather than surface runoff, dynamics because they are less studied and understood.

2. Methodology

2.1 Modeling framework of SWMM-MODFLOW

A two-way coupled surface-subsurface hydrological model, SWMM-MODFLOW, was developed and utilized. It is a loosely coupled model linking SWMM and MODFLOW, and its structure is shown in Fig. 1. Retaining the main structures of SWMM and MODFLOW, the coupling was performed through file input and output without internal function calls between the source codes of the two models. More specifically, the surface infiltration rate in non-GI-pervious areas, or the exfiltration rate at the bottom of the GI, was sent from SWMM to MODFLOW, while groundwater table depth was sent from MODFLOW to SWMM (Fig. 1). Based on the groundwater table depth obtained from MODFLOW, the hydrological processes of GI including underdrain flow, exfiltration rate, and surface runoff, were calculated using the equations of SWMM-LID-GW developed by Zhang et al. (2018) (Eq. 1-2). Furthermore, MODFLOW can also simulate groundwater table dynamics after receiving the infiltration and exfiltration rates from SWMM. The governing equations of underdrain flow of GI, exfiltration rate of GI, and groundwater flow in MODFLOW are shown by Eq. 1, 2 and 3 below respectively:

$$f_{drain} = \begin{cases} A \times (h_{WS} - h_{offset})^B & (if\ h_{WS} \geq h_{GW}) \\ A \times (h_{GW} - h_{offset})^B & (if\ h_{WS} < h_{GW}) \end{cases} \quad \text{(Eq. 1)}$$

where f_{drain} is the rate of underdrain flow; A and B are the two coefficients of underdrain, which depend on the size and the density of holes on the underdrain, respectively; h_{offset} is the distance between the underdrain and the bottom of the GI; h_{WS} is the depth of water storage within the GI; and h_{GW} is the distance between the groundwater table and the bottom of the GI.

$$f_{exfil} = \frac{\theta_s - \theta}{\theta_s - \theta_i} \times K_s \quad \text{(Eq. 2)}$$

where f_{exfil} is the exfiltration rate; K_s is the saturated hydraulic conductivity of the in-situ soils; θ_s and θ_i are the saturated and initial moisture contents, respectively, of the in-situ soils; and θ is the current moisture content of the in-situ soil near the bottom of the GI, which depends on the groundwater table depth. The exfiltration rate varies with the groundwater table depth and the soil moisture of in-situ soils. This rate becomes equal to K_s when the soil moisture of in-situ soils is equal to θ_i, and the rate reduces to zero when the groundwater table rises to, or above, the bottom of the GI. It should be noted that the initial moisture content of the in-situ soils (θ_i) was updated for every time step, and was based on the groundwater table depth of nearby GI. Some of the variables mentioned above are illustrated in Fig. 2.

$$\frac{\partial}{\partial x}(K_x \frac{\partial h}{\partial x}) + \frac{\partial}{\partial y}(K_y \frac{\partial h}{\partial y}) + \frac{\partial}{\partial z}(K_z \frac{\partial h}{\partial z}) = S_s \frac{\partial h}{\partial t} \quad \text{(Eq. 3)}$$

where K_x, K_y, and K_z represent the hydraulic conductivity in the x, y, and z directions; S_s represents the specific storage; and h represents the average groundwater head of grids beneath the GI, which was used to calculate the soil moisture at the bottom of the GI (i.e., θ in Eq. 2) through the Van Genuchten equation.

Overall, the coupled model can characterize the variations of hydrological processes of GI, with respect to groundwater table depth. When the groundwater table fluctuates, the underdrain flow and exfiltration rates can vary, as characterized by the equations above. As a result, the percolation rate from soil layer to storage layer, the surface infiltration rate, and the rate of surface runoff may also vary according to the water balance inside the different layers of GI. For example, when the groundwater table is shallower than the bottom of both the GI and the underdrain pipe, the exfiltration stops and the underdrain flow is governed by the groundwater table instead of the percolation rate. Thus, the maximum percolation and surface infiltration rates should be equal to the rate of underdrain flow. More details about these features can be found in Zhang *et al*. (2018).

The data exchange was governed by a MATLAB controller. The detailed mechanisms of the MATLAB controller and the data exchange are also shown in Fig. 1. The controller realized temporal synchronization between the two models by using the "hot-start" functions, which allowed the two models to pause and re-start from a previous time step when necessary. More specifically, one simulation was segmented into multiple time steps. After the completion of each time step, one model was paused with the results of this time step stored into external hot-start files, and then sent to the controller for processing. The other model was then activated by the controller after receiving the processed results (Fig. 1). The hot-start function in SWMM was already improved by Zhang et al. (2018) to support the storage and extraction of GI simulation results including soil moisture of the soil zone, water depth at the surface, and water depth inside the storage layer, into the external hot-start files (Fig. 1). However, there is no similar built-in function in the current version of MODFLOW. Thus, a hot-start function was developed in MODFLOW by transferring and modifying the source code from GSFLOW, another surface-subsurface hydrological model coupled by the Precipitation-Runoff Modeling System (PRMS) and MODFLOW (Regan et al., 2015). With similar functionalities to those of SWMM, the hot-start function of MODFLOW can store and extract groundwater heads and percolation rates in different layers of grids at the end and start of each time step. Details about the hot-start functions in SWMM and MODFLOW can be found in Zhang et al. (2018) and Regan et al. (2015) respectively.

The controller performed the spatial mapping between two models by processing the results from each model into organized and transferable formats. This is necessary because the spatial representation approaches of the two models are completely different. SWMM is a sub-catchment-based spatially lumped model, but MODFLOW is a grid-based finite-difference model, as shown in Fig. 3. The controller extracted the infiltration rate of pervious areas, or the exfiltration rate of GI, from the output files of SWMM and discretized them into matrixes, which were then updated into the input files of MODFLOW (Fig. 1 and 3). Simultaneously, the controller extracted matrixes of groundwater heads from the output files of MODFLOW, modified these into groundwater table depths, lumped them into sub-catchment-based values, and then updated these values into the input files of SWMM (Fig. 1 and 3). Although the locations of GI within sub-catchments cannot be specified in SWMM, they can be accurately located in MODFLOW (yellow rectangles in Fig. 3) with three indicators: the index of sub-catchment; the index of GI within each sub-catchment; and the coordinates of grids of GI. This mapping process is schematically illustrated in Fig. 3.

2.2 Study area and data

An urban catchment in Silverdale, Kitsap County, WA, US was selected as the study area (Fig. 4). The study area is 197 ha in size with approximately 80% urbanization, and is located on the Kitsap Peninsula (Fig. 4b) lying at the northern tip of the Dyes Inlet, which is connected to the Puget Sound. The area is of equable oceanic climate with generally mild temperatures, and moderate to heavy precipitation (Sceva, 1957). It has warm dry summers and relatively mild winters. The precipitation averages 1103.6 mm/year with 161 precipitation days, and the reference evapotranspiration (ET) averaged 741.7 mm/year during 1991-2018. The detailed properties of the sub-catchments can be found in Table 1.

The catchment is located at a Pleistocene depositional unit in Kitsap County. The major soil types of the area include, but are not limited to, bedrock, advanced outwash, gravels, lacustrine, peat, and till, based on local geologic surveys (Sceva, 1957) and the SSURGO soil database (NRCS and USDA, 2017). The northern and western parts of the catchment are mountainous with an average slope of 10–14%, so the highest locations were assigned as the domain boundaries in these two directions, which were assumed as no-flow boundaries in the groundwater model (the light gray boundary in Fig. 4c). The southwestern and eastern sides of the catchment lie in the Strawberry Creek (the blue boundary in Fig. 4c) and the Clear Creek (the green boundary in Fig. 4c), respectively. The southern boundary is connected to the Dyes Inlet (the yellow boundary in Fig. 4c). In addition, discrete groundwater level data within the catchment (black triangles in Fig. 4c) were retrieved from the Environmental Information Management System (EIM) of Washington State, and were used to estimate the initial groundwater levels of the catchment through extrapolation. The groundwater levels observed are within 30 m below the land surface, and were highest during the late spring months and lowest in the late fall and early winter months (Sceva, 1957).

One year of monitoring was performed by Kitsap County at an urban catchment near the Central Kitsap County Campus (CKCC), which is located at the central part of the catchment (Fig. 4c and 3d). The CKCC site is 2.63 ha in size, within which nine BCs and 10 parcels of porous pavements are implemented. The BCs are 35–146.8 m^2 in size, which allows a surface ponding depth of 100 mm, and consist of soil and storage layers of 400 mm and 380 mm in thickness, respectively. The soil layers were filled by an amended soil mix and the storage layers were filled by washed aggregated (AASHTO No. 57). The porous pavements are 238–

1710 m^2 in size, consisting of pavement and storage layers of 50 mm and 750 mm in thickness, respectively. The pavement layer is made up of Eco-Priora concrete pavers and AASHTO No. 8 aggregate in the openings. The storage layer is made up of an open-graded base and a subbase, which are filled by AASHTO No. 57 and No. 2 aggregates, respectively. These GI practices are connected to the storm sewer system via 150-mm underdrains. The hydrologic properties of the different layers of both types of GI practices are shown in Table 2, and their detailed designs can be found in Herrera (2013) and Zhang *et al*. (2018). Five datasets are available for 1 year from October 1, 2011 to September 30, 2012. These include surface runoff from the rooftop of the Haselwood Family YMCA building (0.46 ha) ($ROOF\_SR$), surface runoff from one impervious area of 0.068 ha (IP_SR), underdrain flow of one parcel of porous pavement (0.17 ha) (PP_UD), the pipe flow at the sewer outlet of the catchment (2.63 ha) ($OUTLET$), and the groundwater table depth at one location within the site (GW_DEPTH). All of the datasets were continuous, and were at a temporal resolution of 5 min. The locations of the monitoring stations are shown in Fig. 4d, and more details about the monitoring approaches and devices can be found in Zhang *et al*. (2018).

2.3 Model settings

2.3.1 SWMM

Given that SWMM is a spatially lumped model, the catchment was separated into 14 sub-catchments, or hydrologic response units, in SWMM (Fig. 4c). The hydrologic response units were delineated based on the topography and soil type using ArcGIS. Due to the relatively consistent geophysical and hydrological characteristics within each unit, each sub-catchment was considered as homogeneous in its hydrologic responses. The area and the imperviousness of the sub-catchments ranged from 7.67 ha to 24.23 ha, and from 47.49% to 100%, respectively (Table 1).

The Dynamic Wave model was used for flow routing in the storm sewers, and the Green-Ampt equation was used as the infiltration model, which allowed the consideration of surface ponding. This model may slightly overestimate the hydraulic conductivity (Triadis and Broadbridge, 2012), but it obtained similar results as that calculated by the Richards equation for GI practices (Dussaillant *et al*., 2004). GI practices were placed within sub-catchments in a lumped manner instead of being independently represented. The detailed hydrologic properties of GI practices can be found in Table 2. After implementing GI practices into sub-catchments, the properties of the sub-catchments were modified, including the imperviousness, width, and percentage of impervious area treated by GI. The details of these modifications are elaborated in the following sections as necessary.

2.3.2 MODFLOW

The grid size in MODFLOW was set as 23 m × 23 m, considering the trade-offs between sub-catchment size and the normal size of GI, and between computation accuracy and cost. Based on the geologic conditions, the area was segmented into three subsurface layers, the thicknesses of which were 0–30 m, 0–15 m, and 0–15 m, respectively. The upper two layers were set as convertible (i.e., can switch between unconfined and confined) and the bottom layer was set as confined. The properties of the layers and the flow between layers and grids were simulated using the Layer Property Flow (LPF) package, which allowed the simulation of dewatered conditions.

The surface infiltration (retrieved from SWMM in each time step) was simulated as a specified flux into the subsurface layers using the Recharge (RCH) package. The ET was simulated using the EVT package, and the monthly averaged ET rate in the study area was used with an ET root depth of 0.5 m. The River (RIV) package was used to simulate the river boundaries (i.e., Clear Creek and Strawberry Creek), which were represented as head-dependent flux. The river stages of these two rivers, retrieved from the Kitsap Public Utility District, were simply assigned as the heads of these two boundary conditions, assuming the rivers are connected to the unconfined aquifer underneath. This assumption is reasonable because the base flow of these two creeks is primarily from groundwater discharge during summer (Sebren, M.B., 2017). In addition, the General-Head Boundary (GHB) package was used to simulate the boundary of the sea (i.e., Dyes Inlet), and the sea level, retrieved from the NOAA Tides and Currents database, was assigned as the head of the boundary condition.

In addition, it should be noted that the coupled SWMM-MODFLOW ran hourly, but SWMM and MOFWLO ran at a time step of 5 min. The Preconditioned Conjugate-Gradient (PCG) package was used to solve the finite difference equations in MODFLOW, with a maximum number of iterations of 150, a relaxation parameter of 0.97, and a maximum absolute change in head of 0.01 m.

2.4 Model calibration and validation

Although some types of information about the catchment, like the thickness of the aquifer and the soil type distribution, are available, the exact modeling parameter values may still be unknown, because some parameters are not directly observable. These include the drainage coefficient of underdrain, Manning's n of overland flow and conduit, and the depression storages of impervious and pervious areas. Other parameters may vary significantly in their ranges, such as the hydraulic conductivity, specific yield, and specific storage of soils. Thus, the parameters were calibrated first.

The model was calibrated and validated using the monitoring datasets at the CKCC site, including $ROOF\_SR$, IP_SR, PP_UD, $OUTLET$, and GW_DEPTH as mentioned. Data from the first 5 months (from October 1, 2011 to February 29, 2012) were used for calibration, while those for the last 7 months (from

March 1 to September 30, 2012) were used for validation. Although both SWMM and MODFLOW were run at 5-min time steps, they were coupled hourly to save computations, so SWMM and MODFLOW each were run for 12 time steps before each round of data exchange. Updating groundwater table depths or recharge rates in SWMM and MODFLOW every hour was considered fine-grained enough to capture the surface runoff and groundwater table dynamics during the 1-year simulation. Particularly, the general groundwater level in the catchment does not fluctuate at a very fine scale according to historical monitoring (Herrera, 2013). It should be noted that the monitoring data used for model calibration was from the central catchment of the area (i.e., the CKCC site, red rectangular area in Fig. 4c). However, the model was built to cover the whole catchment and reach the catchment boundaries.

The calibration approach was similar to that of Zhang *et al*. (2018). More specifically, a non-dominated sorted genetic algorithm (NSGA-II) originally developed by Seshadri (2009) in MATLAB was utilized after integration with SWMM-MODFLOW. The parallel computing package of MATLAB was utilized, using four cores of the CPU to save computation time. The algorithm first initialized the parameters for calibration mentioned above, then invoked the execution of SWMM-MODFLOW. The Nash-Sutcliffe Efficiency (NSE) values of the datasets (i.e., $ROOF\_SR$, IP_SR, PP_UD, $OUTLET$, and GW_DEPTH) were then computed as the performance indicators. Based on the objective of maximizing the performance indicators (i.e., the NSE values), the algorithm generated new populations (i.e., new sets of parameters) through the processes of parent selection, crossover (crossover probability = 0.9), and mutation (mutation probability = 0.1) out of the population generated (number of populations = 24). The parameters were improved after iterations of generations. The calibration was considered completed when the assigned total number of generations (10) was reached. This number of generations was sufficient because the NSE values of the datasets were found to reach their near optimums after 8 generations. The final calibrated parameters were manually selected from the last generation of populations by striking a tradeoff among the NSE values of the datasets.

The SWMM parameters to be calibrated included underdrain coefficient, underdrain exponent, the offset height of underdrain, saturated hydraulic conductivity, the width of sub-catchment, Manning's n for overland flow, the depression storage of impervious area, and the roughness of the conduit. Additionally, hydraulic conductivity and specific yield were calibrated in MODFLOW.

2.5 The hypothetical case studies

A series of hypothetical scenarios were formulated to represent different spatial allocation patterns of BCs. One-year continuous simulations were then performed using the validated model, but with different spatial allocation patterns. The simulation duration was considered sufficient to capture the groundwater table dynamics because the response time of shallow groundwater was shorter, and the model was warmed up sufficiently. The results obtained were also considered representative given that the 1-year period covers a range of rainfall events and groundwater levels.

The initial conditions of the above simulations were the results from warm-up simulations in which there were no BCs. Using the 10-year rainfall from 2001 to 2011 in the study area as the input, the warm-up simulations were run repeatedly until they reached a dynamic equilibrium where the difference of groundwater levels, at 10 selected grids from different parts of the catchment between two consecutive simulations, was within 0.5%.

2.5.1 Rules of spatial allocation of bioretention cells

The rules of allocating the BCs within the catchment are illustrated in Fig. 5. The BCs were allocated as a cluster of practices within each sub-catchment. The size of each BC was the same, and equal to the grid size of MODFLOW (23 m × 23 m). First, the number of BCs within each sub-catchment was determined, which represented the implementation ratio of the BCs. Then the central location of the BC cluster was determined (BC_4 in Fig. 5), which represented the approximate location of BCs within each sub-catchment. After that, the remaining BCs simply surrounded the central BC circle-by-circle, following a rectangular-shaped pattern until reaching the total number of BCs. A certain gap (i.e., 0–69 m in this study) was kept between each BC, the magnitude of which determined the aggregation level of the BC cluster. The physically unavailable locations, i.e., pervious areas and locations out of the sub-catchment, were skipped during the process (indicated by yellow cells with red crosses in Fig. 5). The spatial allocation assumed a uniform allocation pattern within each sub-catchment. This simplification is considered acceptable because this study focused on generating generic understanding instead of making detailed planning decisions. In addition, the surface flow routing between BCs was neglected, which may affect the runoff control performance of the BCs. This simplification was considered negligible because this study focused on groundwater table dynamics, and limited information about the site, such as topography, land use, and sewer systems, also hinders the consideration of flow routing between BCs.

Four dimensionless indicators (i.e., $RATIO$, GAP, LOC_c, and LOC_{sc}) were used to represent the different aspects of spatial allocation of BCs using equations shown in Fig. 5. These four indicators were selected because they represent the main factors that need to be considered in GI planning as proposed by Zhang and Chui (2018b):

- $RATIO$ represents the implementation ratio of BCs, which is the ratio of the total area of BCs (N_GI_n) to the total area of available locations for BCs (A_n) (Eq. 4).

- GAP represents the aggregation level of BCs, which is calculated by the ratio of average gap size between BCs (L_{GAP}) and the average size of BCs (L_{BC}) (Eq. 5).

- LOC_c represents the relative location of BCs within the catchment, which ranges from 0 to 1. A value closer to 0 or 1 means BCs are mainly allocated in upstream or downstream areas respectively. After labeling the sub-catchments from 1 to N approximately from upstream to downstream, the relative location index of each BC was calculated by dividing the index of the sub-catchment it belonged to (LOC_n) by N. Then LOC_c was obtained by taking the average of the relative location indices of all BCs (Eq. 6).

- LOC_{sc} represents the relative location of BCs within the sub-catchments, which ranges from 0 to 1. Similar to LOC_c, a value closer to 0 or 1 means that BCs are mainly allocated near the upper or lower ends of the sub-catchments, respectively. For a specific sub-catchment, the available locations for BCs were first labeled from 1 to A_n approximately from upstream to downstream, and the relative location index of each BC was calculated by dividing the index of the grid it belonged to ($LOC\_BC_{m,n}$) by A_n. Then the LOC_{sc} of this sub-catchment was obtained by taking the average of the relative location indices of all BCs within this sub-catchment (Eq. 7).

$$RATIO = \frac{N\_BC_n}{A_n} \qquad \text{(Eq. 4)}$$

$$GAP = \frac{L_{GAP}}{L_{BC}} \qquad \text{(Eq. 5)}$$

$$LOC_c = \frac{1}{\sum_{n=1}^{N} N\_BC_n} \sum_{n=1}^{N} \frac{N\_BC_n \times LOC_n}{N} \qquad \text{(Eq. 6)}$$

$$LOC_{sc} = \frac{1}{\sum_{n=1}^{N} N\_BC_n} \sum_{n=1}^{N} \sum_{m=1}^{N\_BC_n} \frac{LOC\_BC_{m,n}}{A_n} \qquad \text{(Eq. 7)}$$

where N represents the total number of sub-catchments (14 in this case); N_BC_n represents the total area of BCs in sub-catchment n; A_n represents the total area of available areas for BCs; L_{BC} and L_{GAP} represent the size of BCs and the gap in between BCs, respectively; LOC_n represents the index of sub-catchment n; and $LOC\_BC_{m,n}$ represents the index of the grid for the m^{th} BC in sub-catchment n. The exact ranges of these four indicators are elaborated in the next section.

The properties of the BCs, such as their thickness and the hydraulic conductivity of media soils, followed those of the BCs at the CKCC site mentioned above. More specifically, each BC constituted a 300 mm soil layer and a 380 mm storage layer. Notably, however, no underdrain pipe was used. For each scenario, some sub-catchment parameters needed to be modified in SWMM. First, the width of the sub-catchments, which is one main parameter used to calculate overland flow in SWMM, was adjusted by multiplying the original value by the ratio of $\frac{A_n - N\_BC_n}{A_n}$, because a proportion of the impervious area was replaced by BCs. This approach is recommended in SWMM manuals and adopted by some SWMM users (Rossman, 2015). In addition, the percentage of the impervious area treated by BCs was also adjusted, allowing BCs to receive surface runoff from impervious areas that are, at most, 20 times larger. This is within the recommended range of most design standards, which range from 5 to 20 times the area of the BC (Dhalla and Zimmer, 2010; Roseen and Stone, 2013; Woods Ballard *et al.*, 2015).

2.5.2 Modeling scenarios and outputs

A total of 144 scenario-based simulations were performed to evaluate the impact of the spatial allocation of BCs on surface runoff and groundwater table dynamics. The scenarios covered four different implementation ratios of BCs ($RATIO$ of 0.625%, 1.25%, 2.5%, and 5%); four different aggregation levels (GAP of 0, 1, 2, and 3); three different locations within the whole catchment (LOC_c of approximately 0.5, 0.3, and 0.75 when BCs were distributed throughout all of the sub-catchments, only in upstream, or only in downstream sub-catchments, respectively); and three different locations within sub-catchments (LOC_{sc} of approximately 0.1, 0.45, and 0.9 when BCs were allocated near the upper end, middle section, or lower end of sub-catchments, respectively). The ranges of parameters for the hypothetical scenarios are illustrated in Table 3 below.

The scenario without BCs was also simulated, and was treated as the base case. The indicators representing the surface runoff and groundwater table dynamics were then calculated by comparing the results of the base case with those of the hypothetical cases. More specifically, the peak reduction (PR) and volume reduction (VR) of surface runoff throughout the year in different sub-catchments, and for the whole catchment, were extracted to represent the surface runoff dynamics. The peak (GR_P) and temporally averaged (GR_M) groundwater table rises in different sub-catchments and for the whole catchment, as well as the standard deviation of groundwater level in the catchment (GL_{STD}) throughout the year, were extracted to represent the groundwater table dynamics. The three parameters together can comprehensively represent the local and regional changes of groundwater levels, as well as the uniformity of groundwater levels. Note that higher runoff control efficiency (i.e., higher PR and VR) and more spatially uniform groundwater levels (i.e., lower GR_P, GR_M, and GL_{STD}) are generally preferred, because a more uniform groundwater level results from higher recharge in areas with a deeper groundwater table and lower recharge in areas of a shallower groundwater table. The outcome is beneficial because the two objectives of enhancing groundwater recharge in deeper areas and minimizing groundwater mounding in shallow groundwater areas can be realized simultaneously.

3. Results and discussion

3.1. Model calibration and validation

Fig. 6 shows the time series of different datasets (i.e., rainfall, PP_UD, IP_SR, $ROOF\_SR$, $OUTLET$, and GW_DEPTH) during both calibration and validation periods. The light gray and dark gray sections in Fig. 6a-6f correspond to the calibration and validation periods, respectively. One event on December 23, 2011 is shown specifically in Fig. 6g-6l. This event is considered representative given its medium rainfall intensity (12 mm/h), runoff amount (23 mm), and groundwater table fluctuation (0.3 m) during the period, which can be observed in Fig. 6. In addition, the final calibrated parameters in both SWMM and MODFLOW are shown in Table 4 and are all within physically reasonable ranges.

The pipe flow at the outlet of the catchment ($OUTLET$) and surface runoff of the building roof ($ROOF\_SR$) showed very good fits with the monitoring data, with NSE of 0.80 and 0.62 during calibration, and 0.64 and 0.51 during validation, respectively (Fig. 6d and 6e). The underdrain flow (PP_UD) and surface runoff (IP_SR) were slightly underestimated at rainfall peaks and rises of the groundwater table (Fig. 6h and 6i), but were still of reasonable goodness of fit. The NSE of PP_UD and IP_SR were 0.44 and 0.42 during calibration, and 0.50 and 0.41 during validation, respectively. Particularly, the goodness of fit of PP_UD was improved during the selected event with an NSE of 0.72 (Fig. 6h). Generally, the goodness of fit values of these datasets were acceptable, and comparable to those obtained by Zhang *et al*. (2018) using SWMM-LID-GW with groundwater monitoring data as direct input.

Comparatively, the goodness of fit for GW_DEPTH was not as satisfactory, with NSE values of -0.26 and 0.33 during calibration and validation periods, respectively (Fig. 6f). However, SWMM-MODFLOW captured the general fluctuations of the groundwater table during the overall period and the selected event

(Fig. 6f and 6l), and the goodness of fit during the selected event was reasonable (NSE of 0.61) (Fig. 6l). The good fits of PP_UD and $OUTLET$ also confirmed the accuracy of groundwater simulation to some extent, because they are both highly related to groundwater table depth, as shown in Zhang *et al*. (2018). The discrepancy could be reduced if there were additional monitoring data for model calibration, and if a better understanding of the spatial variations of subsurface geophysical conditions (e.g., hydraulic conductivity and specific yield) in the study area could be obtained.

3.2. Surface runoff dynamics

Fig. 7 shows the surface runoff response, represented by peak reduction (PR) and volume reduction (VR) of surface runoff, at different sub-catchments for different implementation ratios of BCs. It should be noted that the values shown in the graph (Fig. 7) are the averaged results of different scenarios. Fig. 7 explicitly shows the results for different values of $RATIO$, but the $RATIO$ values in the graph are the averaged results for different values of GAP, LOC_C, and LOC_{SC}. This also applies to other similar graphs (i.e., Fig. 8-13).

As expected, PR and VR were greater when there were more BCs. More specifically, when the implementation ratio of BCs increased from 0.625% to 1.25%, 2.50%, and 5.00%, respectively, PR for the whole area (shown as dashed lines in Fig. 7b) increased from $1.9 \pm 1.3\%$ to $8.6 \pm 2.7\%$, $22.3 \pm 2.8\%$, and $43.6 \pm 4.8\%$, and VR for the whole area (shown as dashed lines in Fig. 7d) increased from $4.2 \pm 1.7\%$ to $15.1 \pm 2.7\%$, $35.2 \pm 3.1\%$, and $54.5 \pm 13.5\%$. Considering that this is a highly impermeable catchment (approximately 80% urbanized), the impact of BCs may not be as significant for other more permeable catchments.

Furthermore, it was found that the impact of the implementation ratio was different in different sub-catchments. Using the scenario of an implementation ratio of 5.00% as an example, the PR values of sub-catchments S7, S8, and S9 were $56.7 \pm 41.2\%$, $53.9 \pm 40.6\%$, and $52.2 \pm 39.9\%$, which were significantly higher than the PR values of sub-catchments S1, S4, S10, and S14, which were approximately $0.0 \pm 0.0\%$, $30.4 \pm 26.2\%$, $43.3 \pm 37.0\%$, and $32.9 \pm 30.7\%$, respectively (Fig. 7b). This was also true for the VR values (Fig. 7d). These sub-catchment differences were likely caused by differences of imperviousness, slope, and the permeability of in-situ soils. Compared with the imperviousness values of sub-catchments S7, S8, and S9 (57%, 50.7%, and 66.2%, respectively), the imperviousness values of sub-catchments S4, S10, and S14 were higher (79.1%, 95.3%, and 78.7%, respectively), and the soils of sub-catchments S4, S10, and S14, which are near the two rivers and the sea, are of lower permeability than the other sub-catchments, which resulted in lower runoff control efficiency in these areas. Although sub-catchment S1 was lower in imperviousness (i.e., 47.5%), its slope (i.e., 14%) was significantly greater than that of other sub-catchments, which was not beneficial for runoff control. This was consistent with the mechanisms of GI and runoff generation found in some other studies (Shuster *et al.*, 2005; Dietz and Clausen, 2008; Ahiablame and Shakya, 2016).

3.3. Groundwater table dynamics

Figs. 8-13 compare the response of the groundwater table with BCs of different spatial allocations. More specifically, Figs. 8, 11, 12, and 13 show the spatial variation of peak (GR_P) and temporally averaged (GR_M) groundwater table rises within the catchment. Fig. 9 shows the spatial variation of mean groundwater table depth throughout the year, and Fig. 10 illustrates the standard deviation of groundwater levels in the catchment (GL_{STD}) throughout the year.

3.3.1. Impact of the implementation ratio of bioretention cells

A small number of BCs can significantly change the groundwater dynamics. For example, when averaged over the whole catchment, if 0.625%, 1.25%, 2.50%, and 5.00% of the impervious area were replaced by BCs, the peak groundwater rise (GR_P) would be approximately 0.13 $\pm$ 0.05 m, 0.34 $\pm$ 0.08 m, 0.82 $\pm$ 0.11 m, and 1.31 $\pm$ 0.38 m, respectively (see dashed lines in Fig. 8b and Fig. 8c). At some specific locations, the groundwater table rise can be a few meters or higher (> 5 m) during some specific time (Fig. 8d). This is a significant change and can be problematic, considering that the thickness of the unsaturated zone was only from 0 to 5 m in approximately half of the catchment, and 5 out of 14 sub-catchments had groundwater tables very close to the ground (Fig. 9b). In addition, it was found that BCs not only affect the local groundwater table, but also influence regional groundwater levels. When 5% of the catchment was replaced by BCs, the groundwater table of the catchment was as much as 1 m closer to the ground compared with an implementation ratio of 0.625% (Fig. 9c). This can also be seen from the significantly large areas of groundwater table rise (darker green regions) shown in Fig. 8a and 8d. A similar observation was obtained by Bhaskar *et al*. (2018). This illustrates the importance of considering the groundwater table condition in the spatial planning of GI.

Fig. 8d and 8h show the exceedance probability curves of GR_P and GR_M within the catchment, illustrating the proportion of the catchment area with different levels of GR_P and GR_M. Similarly, Fig. 11d, 11h, 12d, 12h, 13d, and 13h illustrate the same information. They provide another perspective on the groundwater table dynamics as a result of BCs. With more BCs implemented, the proportion of catchment areas with lower groundwater rises (e.g., < 1.0 m for peak rise and < 0.4 m for temporally averaged rise) was lower, while the proportion of catchment areas with higher groundwater rises (e.g., > 1.0 m for peak rise and > 0.4 m for temporally averaged rise) was higher (Fig. 8d and 8h). For example, when only 0.625% of the impervious area was replaced by BCs, approximately 100% of the catchment had a GR_P of less than 1.0 m and a GR_M less than 0.2 m. However, when 5.00% of the catchment was replaced by BCs, only 48% and 39% of the catchment had GR_P and GR_M values less than 1.0 m and 0.2 m, respectively. Overall, areas comprising 33%, 14%, 4%, and 2% of the catchment had GR_P values of 1–2 m, 2–3 m, 3–4 m, and 4–5 m, respectively (Fig. 8d and 8h).

Similar to surface runoff, the groundwater table response also varied among different sub-catchments. Comparatively, sub-catchments around the center of the catchment (e.g., S3 and S9) showed higher groundwater table rises than those closer to the catchment boundary (e.g., S1 and S14) (Fig. 8b and 8f). A similar observation can be obtained from Fig. 11-13. This variation can be explained by noting that the upstream areas were of steeper topography and groundwater hydraulic gradients, while the downstream areas were of gentler topography and groundwater hydraulic gradients. As a result, the groundwater tended to gather in the central sub-catchments (e.g., S3 and S9), rather than sub-catchments nearer the boundary. This is quite a common phenomenon in sloped areas. Focusing on a catchment of similar terrain (i.e., steeper or gentler in upstream or downstream directions, respectively) in China, Cai *et al*. (2015) also found that the groundwater level was higher in the medium section of the catchment. However, the same phenomenon may not occur in catchments of different topographies. For example, a more uniform groundwater table rise is expected in relatively flat catchments.

Notably, increasing the implementation level of BCs could slightly decrease the average uniformity of the groundwater table due to the greater maximum and minimum GL_{STD} values (Fig. 10a), although the differences were not that significant. During approximately 50% of the time, the uniformity of the groundwater table condition among different implementation ratios was very similar (Fig. 10a). Thus, implementing more BCs with an appropriate allocation strategy may not be unfavorable to the regional groundwater dynamics.

3.3.2. Impact of the aggregation level of bioretention cells

Fig. 11 compares the response of the groundwater table to different aggregation levels of BCs. The spatial variation in groundwater table rise was different for different BC aggregation levels, particularly for GR_P (Fig. 11a). When BCs were more aggregated, with a smaller GAP, the groundwater table rises were also more concentrated (Fig. 11a). As a result, BCs allocated in a more aggregated pattern formed slightly higher GR_P for the overall catchment (the dashed lines in Fig. 11b and Fig. 11c). GR_P of the catchment was 0.70 ± 0.55 m, 0.66 ± 0.51 m, 0.65 ± 0.50 m, and 0.64 ± 0.48 m when GAP was 0, 1, 2, and 3, respectively. This was expected because more densely aggregated BCs with a relatively short distance them between may form an overlapped groundwater mound (Endreny and Collins, 2009). When the BCs were more aggregated, the proportion of lower GR_P areas was smaller, while that of higher GR_P areas was greater (Fig. 11d). For the same reason, the GL_{STD} of the catchment was slightly higher when the BCs were allocated in a more aggregated pattern (Fig. 10b).

However, GR_M for the catchment was slightly greater when the BCs were allocated in a more distributed pattern (Fig. 11f). When the BCs were more distributed, the proportion of lower GR_M areas was smaller, while the proportion of higher GR_M areas was greater (Fig. 11h), which was different from the pattern in Fig. 11d. This is possibly because more distributed BCs can affect a greater proportion of the total area. This can be seen from the spatial variation of GR_M of the catchment, in which the shaded areas are larger when GAP is greater (Fig. 11e). As a result, the overall groundwater table rise of the catchment was higher, although with a relatively low local groundwater table rise at specific locations. However, the impact of the aggregation level on the regional groundwater table dynamics was relatively minimal, compared with that of the implementation level. This can be seen from the minimal difference in groundwater table depth shown in Fig. 9d.

Thus, to achieve a more spatially uniform groundwater level, which is a generally desired condition, BCs with more distributed patterns are preferred when the groundwater table depth is relatively uniform. When BCs need to be allocated in a more aggregated pattern, due to site constraints, it would be better if they could be allocated in places with a deeper groundwater table.

3.3.3. Impact of the location of bioretention cells

Figs. 12 and 13 compare the responses of the groundwater table to BCs allocated at different locations within the catchment and within sub-catchments, respectively. When the BCs were spatially distributed, or in upstream areas of the catchment, GR_P and GR_M were greater than when the BCs were in downstream areas. When the BCs were spatially distributed, only in upstream areas, or only in downstream areas, the corresponding GR_P for the whole catchment was 0.72 ± 0.66 m, 0.70 ± 0.47 m, or 0.54 ± 0.33 m, respectively (dashed lines in Fig. 12b and Fig. 12c), and the GR_M for the whole catchment was 0.45 m, 0.46 m, or 0.31 m, respectively (dashed lines in Fig. 12f and Fig. 12g). In addition, the proportion of areas of lower groundwater table rise (both GR_P and GR_M) was smaller, and the proportion of areas of higher groundwater table rise was greater, when the BCs were spatially distributed or only in upstream areas (Fig. 12d and 12h).

These phenomena have two possible explanations. First, as shown in Fig. 9, the groundwater table in upstream areas (i.e., S1, S3, S4, S5, and S6) was closer to the ground surface, so the groundwater table could respond and rise more quickly and more significantly. Second, the regional groundwater hydraulic gradient may play a role. Generally, the recharge from BCs in upstream areas can affect the groundwater table conditions in downstream areas more quickly and to a larger spatial extent, while the recharge in downstream areas normally extends to the surrounding areas without obvious directionality. This can be seen clearly in Fig. 11b and 11f. When the BCs were allocated in upstream sub-catchments (S1-S7), obvious groundwater table rises were observed in some downstream sub-catchments (S8-S10). In contrast, when the BCs were allocated in downstream sub-catchments (i.e., S8-S14), the groundwater table also rose at some of the upstream sub-catchments (i.e., S2, S3, and S7) but the rise was relatively minimal. However, the magnitude of this effect may differ for different regional groundwater hydraulic gradients and hydrologic connectivities (Jones *et al*., 2019). For example, the effect may not be as significant in areas with relatively flat topographies.

A similar phenomenon was observed through comparing the GR_M values for BCs allocated at different locations within sub-catchments (Fig. 13). Compared with BCs at the lower end of the sub-catchments (0.31 $\pm$ 0.22 m), GR_M for the overall catchment was greater when the BCs were at the upper end or in the middle section of the sub-catchments (0.44 $\pm$ 0.32 m and 0.46 $\pm$ 0.36 m, respectively) (Fig. 13f and 13g), because the proportion of higher GR_M areas was greater (Fig. 13h). In addition, the groundwater levels within the catchment were less uniform (represented by larger GL_{STD} values) when the BCs were allocated in upstream areas and near the upper end of sub-catchments, and vice versa (Fig. 10c and 10d).

Thus, when the groundwater table is relatively deep, BCs are generally better allocated in upstream areas to result in greater regional groundwater recharge and groundwater table rise. When the groundwater table is relatively shallow, it is generally better to allocate BCs in the downstream areas to minimize groundwater table rise and its potential effects on the performance of BCs. However, the optimal allocation may vary case-by-case, as other geophysical conditions like soil distribution should also be considered. Furthermore, it was found that both GR_P and GR_M of upstream sub-catchments (S1-S7) were greater when the BCs were allocated at the upper end of the sub-catchments. These values for downstream sub-catchments (S9-S14) were greater when the BCs were allocated at the lower end of the sub-catchments (Fig. 13b and 13f). For example, when the BCs were located at the upper end, middle section, and lower end, GR_P of S2 decreased from approximately 1.01 $\pm$ 0.86 m to 0.68 $\pm$ 0.61 m and 0.51 $\pm$ 0.61 m, respectively (Fig. 13b and 13c), and GR_M of S2 decreased from 0.67 $\pm$ 0.62 m to 0.47 $\pm$ 0.43 m and 0.30 $\pm$ 0.38 m, respectively (Fig. 13f and 13g). In comparison, GR_P of S9 increased from 0.72 $\pm$ 0.65 m to 1.03 $\pm$ 0.93 m and 1.38 $\pm$ 1.23 m, respectively (Fig. 13b and 13c), and GR_M of S9 increased from 0.43 $\pm$ 0.39 m to 0.54 $\pm$ 0.42 m and 0.61 $\pm$ 0.46 m, respectively (Fig. 13f and 13g). This was because the extent of groundwater rise was greater near BCs (Machusick *et al*., 2011; Thomas and Vogel, 2011; Nemirovsky *et al*., 2014). More specifically, when the BCs (in all sub-catchments) were allocated at the upper end of each sub-catchment, those within the downstream sub-catchments were closer to the upstream sub-catchments. Therefore, the groundwater recharge by BCs in downstream sub-catchments more easily affected upstream sub-catchments, resulting in higher groundwater table rises in upstream areas. Conversely, when the BCs were allocated at the lower end of each sub-catchment, those within the upstream sub-catchments were closer to the downstream sub-catchments. Then the groundwater recharge by BCs in upstream sub-catchments more easily affected downstream sub-catchments, resulting in higher groundwater table rise in downstream areas.

3.4. Relationships between surface runoff and groundwater table dynamics

Fig. 14 illustrates the inter-correlations between the responses of surface runoff and groundwater table levels to different implementation ratios of BCs. Each dot in the graph represents the data (PR, VR, GR_P,

GR_M, or GL_{STD}) for one sub-catchment.

One observation can be obtained from the bar plots along the diagonal of Fig. 14. More specifically, when there were more BCs ($RATIO$ of 5.00%), the occurrences of higher PR, VR, GR_P, GR_M, and GL_{STD} values were greater, which was consistent with observations in pervious sections. As expected, PR and VR correlated with each other closely, with an R^2 of 0.95. For groundwater table rises, only GR_P and GR_M were closely correlated (R^2 of 0.91), while GL_{STD} was less correlated to the other two indicators (R^2 of 0.59 and 0.42 for GR_P and GR_M respectively). In addition, the indicators of surface runoff (i.e., PR and VR) were also correlated with the indicators of groundwater table rises (i.e., GR_P, GR_M, and GL_{STD}) at a relatively lower, but still significant, level (Fig. 14). More specifically, PR correlated with GR_P, GR_M, and GL_{STD} with R^2 values of 0.85, 0.82, and 0.45, respectively, and VR correlated with GR_P, GR_M, and GL_{STD} with R^2 values of 0.86, 0.84, and 0.45, respectively. The observed correlations were not surprising, because the reduction of surface runoff and the increase of groundwater recharge were simultaneous outcomes of enhanced infiltration and recharge by GI.

These observations together reflect the importance of considering the tradeoffs between surface runoff control and groundwater protection in GI planning. A more ideal GI strategy should reduce surface runoff, but also maintain a relatively minimal influence on groundwater dynamics.

4. Concluding remarks

A coupled surface-subsurface hydrological model, SWMM-MODFLOW, was developed to evaluate the surface runoff and groundwater table dynamics of green infrastructure of different spatial allocations at catchment scale. The model was calibrated and validated using the monitoring data at an urban catchment at Kitsap County, WA, US.

Using bioretention cells as the representative green infrastructure, a series of hypothetical simulations was performed. The influence of spatial allocations of bioretention cells, represented by the implementation ratio, aggregation level, and location, on the surface runoff and groundwater table dynamics, was quantified. The primary findings are summarized as follows.

- The implementation ratio of the bioretention cells was the main spatial feature that governed both surface runoff and groundwater table dynamics. With higher implementation ratios, the peak reduction and volume reduction of surface runoff were greater, and the peak and temporally averaged groundwater table rises were higher. However, implementing more bioretention cells may not affect the uniformity of regional groundwater levels if an appropriate allocation strategy is selected.
- Bioretention cells with more distributed allocation patterns resulted in slightly lower peak groundwater table rises, higher temporally averaged groundwater table rises, and a lower standard deviation of groundwater levels in the catchment. Thus, if a more uniform groundwater level is desired, bioretention cells should be allocated in a more distributed way when the original groundwater table depth is relatively uniform. Conversely, when bioretention cells need to be allocated in a more aggregated pattern (e.g., due to site constraints), it would be better if they could be allocated in places with a deeper groundwater table.
- Allocating bioretention cells in upstream areas can raise the groundwater levels downstream due to the regional hydraulic gradient. Thus, when the groundwater table is relatively deep, bioretention cells should generally be allocated in upstream areas to produce a greater regional groundwater recharge and

groundwater table rise. In cases when the groundwater table is relatively shallow, it is generally better to implement bioretention cells in the downstream areas to minimize groundwater table rise and the potential influence on the performance of bioretention cells. In addition, the geophysical conditions and spatial variations within the catchment should be considered when allocating bioretention cells.

- Bioretention cells of greater surface runoff control efficiencies led to higher groundwater table rises. Thus, it is of great importance to consider the tradeoff between surface runoff control and groundwater protection in the planning of green infrastructure.

This study carries certain limitations. First, the coupled model considered the impact of shallow groundwater on some hydrological processes such as exfiltration, underdrain flow, and surface runoff, but other processes were neglected. For example, the impact of shallow groundwater on evapotranspiration of GI was not considered, which could be influential in some conditions such as areas of shallow groundwater or arid climate. Second, the rules for allocating bioretention cells spatially in the hypothetical simulations were simplified. The detailed land uses (e.g., buildings, roads) and physical constraints (e.g., underground infrastructures) were not considered due to the unavailability of relevant information, so a relatively uniform allocation pattern was assumed. As a result, the main insights obtained in this study may have limited contribution at the scales of single GI practices, but they can be beneficial to the higher-level planning of GI. The simplified rule of spatial allocation of GI practices therefore should not affect the insights that work at regional scales. In fact, considering the specific physical and/or legal constraints of the study area might even have affected the transferability of the insights gained, because the constraints can be very different in different areas. Third, the simulations in this study only considered the boundary conditions and hydrogeological conditions of one catchment due to data availability. However, the results from this study can serve as a general reference for others, and the developed model and study methodology can be applied to other catchments to obtain more specific and accurate findings. Future studies should examine more catchment characteristics, such as through the use of more hypothetical catchments, and more spatial allocation rules for various GI practices. They should also explore the optimal spatial allocation of green infrastructure for the restoration of surface-subsurface hydrology.

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