Will it rise or will it fall? Managing the complex effects of urbanization on base flow

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Abstract: Sustaining natural levels of base flow is critical to maintaining ecological function as stream catchments are urbanized. Stream base flow responds variably to urbanization. Base flow or water tables rise in some locations, fall in others, or remain constant. This variable response is the result of the array of natural (e.g., physiographic setting and climate) and anthropogenic (e.g., urban development and infrastructure) factors that influence hydrology. Perhaps because of this complexity, few simple tools exist to assist managers to predict baseflow change in their local urban area. We address this management need by presenting a decision-support tool that can be used to predict the likelihood and direction of baseflow change based on the natural vulnerability of the landscape and aspects of urban development. When the tool indicates a likely increase or decrease, managers can use it for guidance toward strategies that can reduce or increase groundwater recharge, respectively. An equivocal result from application of the tool suggests the need for a detailed water balance. The tool is embedded in an adaptive-management framework that encourages managers to define their ecological objectives, assess the vulnerability of their ecological objectives to changes in water-table height, and monitor baseflow responses to urbanization. We tested our framework with 2 different case studies: Perth, Western Australia, Australia and Baltimore, Maryland, USA. Together, these studies show how predevelopment water-table height, climate, and geology together with aspects of urban infrastructure (e.g., stormwater practices, leaky pipes) interacted such that urbanization led to rising (Perth) and falling (Baltimore) base flow. Greater consideration of subsurface components of the water cycle will help to protect and restore the ecology of urban fresh waters.
Key words: urban development, decision framework, groundwater, urban stream management, water sensitive urban design, low impact development

Stream base flow is derived from delayed water sources, such as groundwater (water below the water table), unsaturated areas, and riparian zones (Smakhtin 2001) and is closely related to water-table height (where the fluid pressure is equal to atmospheric pressure, which is generally close to the depth below which all pore spaces are saturated). Base flow has critical functions in stream ecosystems. These functions include moderation of water temperature and water quality (Klein 1979, Menció and Mas-Pla 2010, Price 2011), nutrient and C processing (Hancock 2002, Groffman et al. 2003, Gift et al. 2010), habitat for aquatic species (Poff et al. 1997, Finkenbine et al. 2000) and support for riparian vegetation (Kondolf and Curry 1986, Nilsson et al. 2003, Naiman et al. 2008).

The effects of urbanization on groundwater recharge and base flow are variable among physiographic and climatic regions and types of urban development (Meyer 2005, Walsh et al. 2005, Hopkins et al. 2015). Many studies suggest that groundwater recharge and stream base flow decrease in urban areas because impervious surface cover

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prevents infiltration and increases surface runoff relative to nonurban lands (Leopold 1968, Ferguson and Suckling 1990, Rose and Peters 2001, Konrad et al. 2005, Hardison et al. 2009). However, other investigators have reported rising water tables and stream base flow (Harris and Rantz 1964, Hollis 1977, Jakovljev et al. 2002, Stephens et al. 2012, Townsend-Small et al. 2013, Barron et al. 2013c). Authors of recent reviews (O'Driscoll et al. 2010, Price 2011, Hamel et al. 2013) identified many factors that affect groundwater recharge. Natural factors include those that affect precipitation (e.g., climate), runoff and infiltration (e.g., geology, topography, and soil type), and evapotranspiration (e.g., climate and vegetation). Anthropogenic factors include urban density (e.g., impervious area), stormwater management, water pumping, irrigation, vegetation alteration, and the condition of water infrastructure (e.g., leaky water-supply and sewage pipes). These factors, which vary widely across watersheds, result in diversity and complexity of water-table and baseflow responses to urbanization.

Despite recognition that baseflow responses to urbanization are varied, surprisingly little guidance exists to assist managers in predicting the severity and direction of change of either stream base flow or water tables in a given area. Understanding how water tables and stream base flow are altered by urbanization is a management imperative because a falling water table creates a different set of human and ecological risks and requires different management approaches than a rising water table (Table 1). For example, a water table that rises to the land surface (or just below it) can cause damage to houses and underground transportation structures (e.g., subways; Kim et al. 2001). Conversely, a falling water table and base flow can amplify water tem-

Table 1. Ecological alterations/risks to humans associated with changing water tables/base flow.

Falling water table/base flow	Rising water table/base flow
Ecological alteration/risk	*Ecological alteration/risk*
• Increased extreme water temperatures (Klein 1979, Price 2011)	• Reduction in extreme water temperatures (Klein 1979)
• Increased likelihood of channel drying (Brunke and Gonser 1997, Roy et al. 2009)	• Increasing flow permanence and damping of seasonal fluctuations in water depth (Davis and Froend 1999)
• Reduced water depth for fish survival and recruitment (Finkenbine et al. 2000)	• Increase in nutrient loads (Stanford and Ward 1993)
• Reduced water quality because of increased contaminant concentrations (Menció and Mas-Pla 2010)	• Increasing salinity of surface soil and water (Cramer and Hobbs 2002)
• Falling O_2 levels associated with reduced flow velocity (Finkenbine et al. 2000)	• Reduction in species that rely on riffle habitat for feeding or spawning (Poff et al. 1997)
• Altered in-stream species assemblage structure (Shields et al. 1994)	• Altered in-stream species assemblage structure (Bunn and Arthington 2002, Riley et al. 2005, Reich et al. 2010)
• Reduced nutrient processing in the riparian zone (Groffman et al. 2003, Gift et al. 2010)	• Increased invasion by competitive non-native species (Poff et al. 1997, Bunn and Arthington 2002)
• Reduced in-stream nutrient processing associated with reduced groundwater upwelling (Hancock 2002)	• Altered in-stream and riparian vegetation (Bunn and Arthington 2002)
• Terrestrialization of the riparian vegetation community (Naiman et al. 2008)	
• Reduced health of deep-rooted vegetation (groundwater-dependent) across the catchment (Barron et al. 2013b)	
Human risk	*Human risk*
• Reduced water quality because of increased contaminant concentrations (Menció and Mas-Pla 2010)	• Flooding of houses (Barron et al. 2013a)
• Reduced access of existing bores to ground water	• Flooding of underground infrastructure (subway train lines) (Kim et al. 2001)
• Reduced volume of water for household use and irrigation (only in urban areas where ground water contributes to water use)	• Increasing contamination of ground- and stream water by septic systems
	• Increased leakage of ground water into wastewater systems leading to wastewater treatment plants treating ground water

perature extremes and increase the likelihood of stream drying, both of which cause stress to stream biota (Menció and Mas-Pla 2010, Rolls et al. 2012).

Most studies describing the effect of urbanization on water-table height or stream base flow have been based on direct measurements through time or across space (e.g., a space-for-time substitution along an urban-to-nonurban gradient) (Beighley and Moglen 2002, Brandes et al. 2005, Burns et al. 2005, Meyer 2005, Barron et al. 2013c). These pragmatic approaches are useful but require water-table or stream data. Such data may not be available, particularly for attempts to forecast the consequences of future development. Ideally, managers need hydrological models that link changes in the urban environment to changes in groundwater recharge (Barron et al. 2013a, b) and, ultimately, baseflow conditions. Such models can be subject to substantial uncertainty, particularly in data-poor situations. In addition, building and executing these models can require considerable human and time resources. A decision-support tool that can be used to assess the likelihood of water-table or baseflow change, predict the direction of the change, and summarize the associated benefits and risks could help managers estimate potential effects of urbanization on base flow in their precinct (or urban area of interest, which may range from neighborhood to metropolitan scale) and catchment. Such a tool would enable them to evaluate the need to undertake a detailed water-balance analysis.

This paper fills the management gap by proposing a decision-support framework. We first describe the natural and anthropogenic factors that affect water-table height and base flow and present examples from the literature showing when each factor has been identified as critical to driving water-table responses. We then present the decision framework, which we designed from an ecological perspective, and outline sequential steps to be taken. The steps include: 1) identifying the ecological objectives for the urban precinct and broader catchment, 2) assessing whether changes in water-table height or base flow will affect objectives at both spatial scales, 3) completing a checklist to assess the likely direction of the change at relevant spatial scales and the need to undertake a detailed water-balance analysis, and 4) identifying and gathering the data required for a water-balance model. We provide guidance on the management solutions applied within an adaptive management framework that will be most effective in combatting rising or falling water tables. Last, we use case studies from Perth (Australia) and Baltimore (Maryland, USA) to illustrate the applicability of our decision-support framework.

DIVERSE MECHANISMS AFFECTING WATER TABLES AND BASE FLOW

The height of the water table is affected by factors that contribute water to ground water (e.g., rainfall, irrigation, or leakage from piped infrastructure), remove water from ground water (e.g., evapotranspiration, pumping, or leakage into piped infrastructure), or alter how water infiltrates into the soil (e.g., creation of impervious areas and soil compaction). The factors that determine water-table height are shaped by natural physiographic features and by characteristics of urban infrastructure and development (Fig. 1). Below, we describe how features of urban infrastructure and development can drive changes in water-table height or base

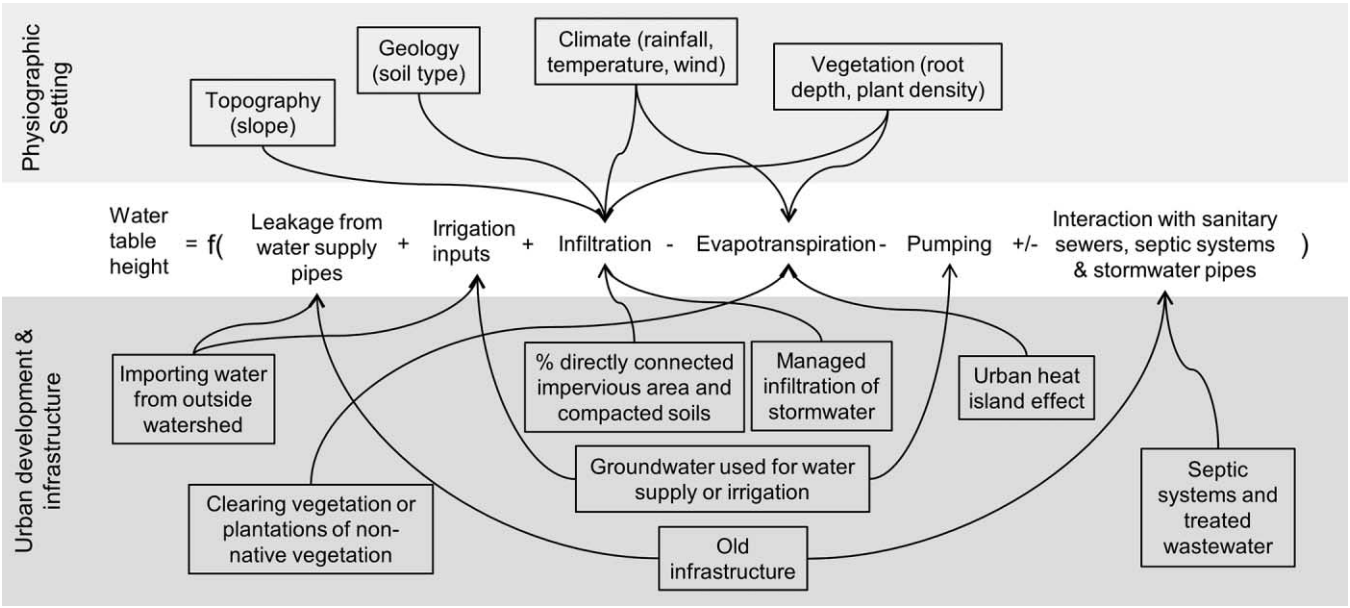

Figure 1. Water-table height is a function of multiple factors that affect the water balance of an area. These factors are, in turn, shaped by physiographic features of the environment and characteristics of urban development and infrastructure.

flow and present studies in diverse settings that attributed these changes to causal mechanisms.

Impervious areas and stormwater management

Increasing impervious surface cover can be correlated with decreasing base flow and water tables (Leopold 1968, Brun and Band 2000, Finkenbine et al. 2000, Rose and Peters 2001, Kauffman et al. 2009). The effect of impervious cover on base flow will be greatest when these surfaces drain directly to the stream via stormwater drainage, thereby limiting opportunities for infiltration (Göbel et al. 2004, Roy and Shuster 2009; Fig. 1). Disconnected impervious areas allow potential infiltration and recharge throughout catchments. However, impervious surfaces are not the only land cover in urban areas that have reduced infiltration. What is normally called the pervious portion of the urban landscape (e.g., lawns and other vegetated areas) may be severely compacted and underlain by urban fill resulting in low infiltration capacity (Booth and Jackson 1997, Price et al. 2010, Ossola et al. 2015). Natural soil permeability also influences the baseflow response to increased impervious cover; e.g., recharge is reduced to a greater extent where naturally permeable soils are paved over because of the greater reduction in infiltration capacity compared to areas with naturally low-permeability soils (Konrad and Booth 2005, Haase 2009).

In some urban areas, stormwater runoff is managed locally with storage and infiltration techniques, such as detention ponds, and these can alter the height of the water table and the source and timing of base flow (Hamel et al. 2013, Liu et al. 2013, Hogan et al. 2014). Approaches that mimic natural infiltration, such as swales, rain gardens, and irrigation from water tanks, are more likely to attenuate the impact of falling water tables driven by conventional stormwater management rather than cause an increase in the water table (Hamel et al. 2013). However, the piping of roof runoff to underground sumps or the transport of runoff to unlined basins may result in unnaturally high rates of recharge and cause the water table to rise (Ku et al. 1992, Appleyard 1995, Stephens et al. 2012, Barron et al. 2013a, c). These focused infiltration strategies are typically implemented in regions with highly permeable soils (Ku et al. 1992, Barron et al. 2013b).

Evapotranspiration

Evapotranspiration (ET) describes water losses associated with evaporation from the land surface, the surface of water bodies (reservoirs, wetlands), and transpiration from vegetation. ET may decline in urban areas as native vegetation is replaced with impervious surfaces (Roy et al. 2009, Barron et al. 2013a), which may increase urban base flow (Fig. 1). However, in areas with low levels of development, ET may remain constant or increase if low-water-using native vegetation is replaced with high-water-using nonnative vegetation, such as *Eucalyptus* or *Pinus* (Di Tomaso 1998, Yesertener 2005), or if the heat-island effect increases advection (Oke 1979). Outdoor irrigation of lawns and gardens can also increase ET (Grimmond and Oke 1986, Stephenson 1994, Claessens et al. 2006), and water-table rises if over-irrigation does not run off (Berg et al. 1996), but these localized effects have not been readily observed in stream base flow.

The extent to which reduced ET will drive a change in water-table height depends on the extent to which groundwater recharge is maintained during urbanization. If semi-natural recharge rates are maintained, then reduced ET will result in an increase in water-table height and base flow (Ku et al. 1992, Barron et al. 2013a, c). However, if recharge is dramatically reduced, then changes in ET will have little effect on water-table height. The importance of ET as a driver of changes in water-table height also will reflect its importance in the natural water balance. Cities built in areas that naturally have very high ET losses will be most susceptible; this includes, those: 1) built in flat landscapes with permeable soils (sand or gravel), because water is more likely to infiltrate (Meyer 2005); 2) with historically dense and deep-rooted vegetation (e.g., forest as opposed grassland), because plants can access a greater volume of soil moisture (Zhang et al. 2001, Yesertener 2005); and 3) in regions with hot dry climates, because they create a strong atmospheric demand for water (Zhang et al. 2001).

Potable water supply and wastewater discharge

Withdrawing ground water for human use via pumping can cause water tables in urban areas to fall (Roach et al. 2008, Sekhar et al. 2013), and the cessation of pumping can cause water tables to rebound (Fig. 1). For example, water tables in Barcelona, Spain, have risen as land use has shifted away from industrial uses (Vázquez-Suñé et al. 2005). Some urban areas require groundwater drainage or pumping to protect underground infrastructure—e.g., 200,000 m^3/d is pumped from subway tunnels in Seoul, South Korea (Kim et al. 2001).

In watersheds with primarily septic systems, wastewater disposal can result in increased groundwater recharge (e.g., Burns et al. 2005; Fig. 1). This increase in water is often observed in developing countries, where drinking water infrastructure typically precedes sewage infrastructure, resulting in a reliance on septic systems and, therefore, a net increase in water in urban areas (Meyer 2005). The construction of stormwater and sanitary sewer networks can lead to dramatic decreases in base flow, as on Long Island, New York (Pluhowski and Spinello 1978, Simmons and Reynolds 1982). The effect of type of wastewater system (e.g., combined or separate sanitary and storm drain systems) may play a large role in the effect of the drainage system on base flow. For example, in an area with a combined sanitary and sewer system, impervious areas may not drain

to the local stream and may instead contribute only to discharge downstream of the wastewater treatment plant (e.g., Hopkins et al. 2014). Wastewater treatment plants (WWTPs) that discharge into streams are a continuous source of water to downstream ecosystems that results in higher median and minimum discharges (White and Greer 2006, Wang and Cai 2010). Treated wastewater effluents can mask otherwise reduced base flow from urbanization, resulting in no net change through time on a watershed scale (Brandes et al. 2005) or dramatic increases in base flow (Townsend-Small et al. 2013). Streams undergoing both withdrawals (ground water or surface water) and direct inputs (e.g., treated wastewater discharges) were referred to as 'churned' by Weiskel et al. (2007). The net effect on base flow may be negligible at a watershed scale, but churned streams can have locally reduced or increased base flow, depending on the locations of withdrawals and inputs (Brandes et al. 2005).

Infrastructure leakage

In urban areas, leakage from pressurized water-supply infrastructure can be a major source of groundwater recharge (Harris and Rantz 1964, Jakovljev et al. 2002, Lerner 2002, Garcia-Fresca 2006; Fig. 1). For example, in his review, Lerner (2002) reported estimates of annual leakage ranging from 15 to 30% of the drinking-water supply, although cases of higher (up to 50%) and lower leakage exist depending on the age and quality of the infrastructure (Garcia-Fresca and Sharp 2005). Typically, water is imported to urban areas from distant reservoirs (Foster 1990), resulting in a net gain of base flow, but in cases where potable water is drawn from ground water, the reverse may be true.

In areas with aging infrastructure, leakage into and out of sanitary, storm, and combined sewers may greatly alter water tables and base flow (Fig. 1). Whether sewer leakage through cracks results in a net loss of ground water (via conveyance to WWTPs) or a net gain of ground water (via infiltration) depends on the locations of sewers relative to the water table. Sewers below the water table probably will gain ground water (Wittenberg 2003), resulting in reduced recharge and lower base flow in the headwaters (Schwartz and Smith 2014), whereas leakage from sewers above the water table can serve as groundwater recharge (Lerner 1986, 2002). Leakage can also vary temporally. De Benedittis and Bertrand-Krajewski (2005) found that infiltration rates in sewers varied on an hourly basis based on groundwater pumping and the amount of water in the pipes. Wittenberg (2003) found that groundwater infiltration into sewers was highest during months of high precipitation. Many authors have attributed increases and decreases in base flow to leaking infrastructure (e.g., Meyer 2005, Townsend-Small et al. 2013), but the extent to which leakage contributes to altered base flow is largely unknown.

Channel morphology

Stream base flow is affected by the height of the water table and by the depth and incision of the stream channel (Fig. 1). In general, the response of base flow to channel incision will reflect the influence of regional ground water on the stream. If water recharged over a large area flows through the subsurface and into the stream, then channel incision will cause base flow to increase (Schilling 2002). If only ground water in the vicinity of the stream influences base flow, then channel incision will drain the local water table and cause base flow to fall (Gölz 1994, Groffman et al. 2002, Micheli and Kirchner 2002). Streambed clogging by sediments from urban development also can decrease base flow (Hancock 2002, Sophocleous et al. 2002).

DECISION-SUPPORT FRAMEWORK

The array of mechanisms and possible baseflow responses to urbanization make it difficult for managers to know whether base flow is likely to be higher or lower post-urbanization and how to respond to hydrologic alteration in local streams. Moreover, environmental managers and urban designers frequently have incomplete knowledge about the natural or current water dynamics and risks posed by past or future urbanization. To assist managers, we have developed a decision-support framework (Fig. 2). Within the framework, we encourage managers to define their ecological objectives and then ask a series of ecologically relevant questions to ascertain whether alteration to water-table height will create ecological risks. Where risks are likely, we direct managers to an assessment checklist that uses characteristics of the physiographic setting and urban development/infrastructure to predict whether stream base flow will rise or fall (Fig. 3). In instances where the directional change is unclear, we suggest that managers undertake a detailed water-balance analysis. Last, we encourage managers to monitor baseflow changes and offer a suite of water-management options to counter the perturbation and protect ecological objectives (Table 3).

The questions posed within the framework should be considered at local and regional scales because urban alteration of the water balance can create effects at both scales. For example, the framework may reveal few or no threats to ecological objectives in the local urban precinct because the water table is naturally deep, but urbanization may contribute to baseflow alterations downstream in the catchment. We now explore the key steps underpinning the decision-support framework.

Defining ecological objectives

Objectives for stream and water management may include both ecological (e.g., water quality, native species) and human objectives (e.g., recreation, provision of drinking water, protection of infrastructure), but we focus here on ecological objectives. Smith et al. (2016) provided a de-

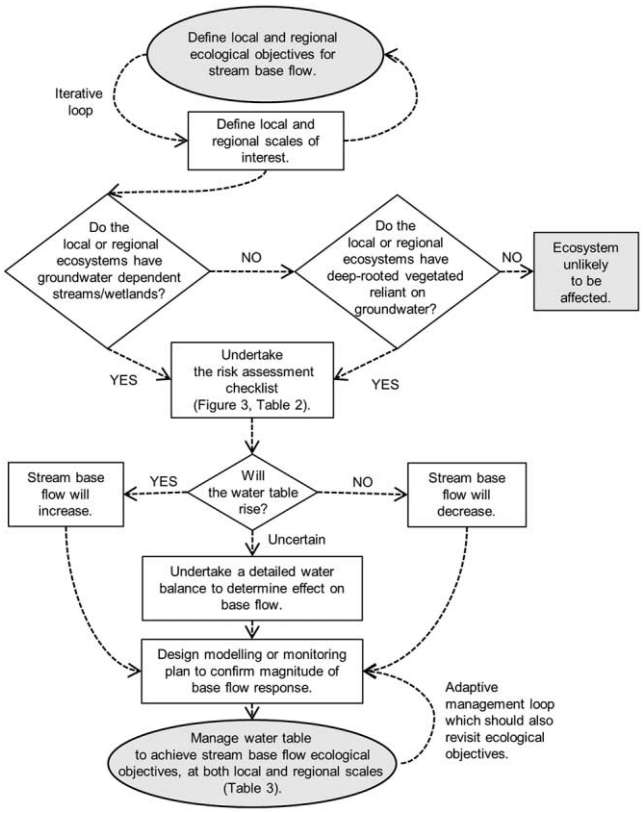

Figure 2. Decision-support framework to assist managers to assess the risk of baseflow perturbation during urbanization. Ovals indicate start and finish of the flow chart, diamonds indicate decisions or questions, and rectangles indicate actions.

tailed discussion of societal objectives and how they may fit within a broader ecosystem framework.

Changes in base flow threaten water quality, ecosystem functions, and native biotic assemblages (Table 1) and, therefore, merit attention. The importance of the natural flow regime to the ecological integrity of streams is so well recognized (Poff et al. 1997, 2010) that it is widely accepted that alterations to the magnitude, duration, timing, frequency, and variability should be managed in urban settings to restore ecosystems (Burns et al. 2012, Walsh et al. 2012). Determining which aspects of base flow are most critical and the extent to which these should reflect pre-urban conditions will be specific to the system under study. Improving the health of an urban stream is the overall ecological objective, but some examples of site-specific ecological objectives may be to promote salmonid spawning or to eradicate an invasive species. The ecological objective for base flow in the 1st case may be to increase the baseflow magnitude and in the 2nd case to alter baseflow timing such that no-flow periods occur in summer. Identifying the appropriate goal or endpoint for restoration or protection can be difficult for urban or peri-urban streams (Smith et al. 2016) because previous landuse change (e.g., agriculture) already has caused degradation, making a return to natural conditions difficult. In these cases, a return to pre-urban development conditions (rather than true 'natural' conditions) may be more appropriate.

Predicting the likelihood of change

The factors driving baseflow responses to landuse change are numerous, complex, and interactive in their effects, but a systematic checklist (Fig. 3) can be used to guide predictions of likely change as a function of both natural and urban characteristics. The checklist helps identify types of information required to make predictions of baseflow alteration. The checklist is separated into 2 parts that address the catchment's natural vulnerability to baseflow change and characteristics of urban development and infrastructure that may lead to baseflow change.

Part 1: Landscape vulnerability to change The vulnerability of a catchment to baseflow change depends on natural factors that determine the extent to which base flow contributes to natural stream flow and the extent and location of the urban footprint. Streams with naturally high base flow are vulnerable to decreases. Streams with little base flow may be vulnerable to increases where management of stormwater or other urban water components may create an 'artificial' baseflow stream or to decreases where loss of year-round flow would lead to large ecological consequences. A catchment's vulnerability is best assessed by the use of hydrologic data wherever they are available. Assessment can be done directly by using relevant hydrologic metrics (e.g., baseflow index, or recession rates; see Price et al. 2011, Hamel et al. 2013) to estimate the fraction of natural flow that is base flow or indirectly by using depth to water table. A depth range of 2 to 4 m to the water table can be used as a representative threshold for vulnerable systems because it describes conditions under which ground water is likely to play a large role in stream flow and will interact with urban infrastructure (water and sewer pipes) and natural vegetation via root-derived water use (Schenk and Jackson 2002).

In the absence of hydrologic data, vulnerability to change could be inferred on the basis of physiographic features of the landscape (slope, soil type, geology, vegetation, and climate). No physiographic classification system has emerged yet (Hrachowitz et al. 2013), but a conceptual approach drawing on relevant local expertise can be used. Care should be taken when considering individual physiographic factors because the factors can interact in complicated ways. Considering the influence of each factor in isolation can be misleading. For example, catchment steepness may lead to naturally low base flow, but a very wet climate or very permeable soils may counteract this effect. The relative importance of physiographic factors also may vary among catchments in the same region (Hamel et al. 2013) and over

time. Jencso and McGlynn (2011) illustrated this variability in a study of a dozen catchments in Montana (USA) in which slope, geology, and vegetation all affected baseflow characteristics but their relative importance varied with season.

The scale and location of the urban development in the catchment (particularly in relation to the location of recharge areas) also will influence the likelihood of changes to base flow (Price 2011, Hamel et al. 2013). If the urban footprint is very small, changes to groundwater recharge may be insignificant unless urbanization occurs in high-recharge areas (Alberti 2005), although changes to peak flow hydrology may still occur (Walsh et al. 2012). However, defining a threshold urban footprint above which measurable effects occur is challenging because the effect of urban development depends on its location in the catchment and characteristics of urban development, including potential changes to subsurface drainage pathways brought about by the creation of infrastructure conduits (Hamel et al. 2013). Walsh et al. (2012) showed that a catchment with only 7% effective impervious area (EIA) still had substantially lower base flow than nearby undisturbed catchments, but generalizing such thresholds is precluded by complexities, such as changes to pervious area infiltration capacity and hydrologic behavior (Ossola et al. 2015).

Part 2: Urban development and infrastructure Characteristics of urban development also clearly affect how urbanization shapes base flow (Fig. 3). We offer predicted baseflow responses given the impervious area and stormwater management, alterations to the level of ET, the type of potable water supply and wastewater discharge, infrastructure leakage, and changes to the stream channel through incision and streambed clogging. The combined direction and likelihood of predicted change of the aforementioned factors will influence the overall effect of urban development on base flow.

Synthesizing natural vulnerability and urban development to inform the likelihood of baseflow change. Because many factors are likely to interact, the overall outcome is unlikely to be a simple addition of the likely change indicators (– or +) in Fig. 3. However, in the absence of knowledge of prediction about such interactions, the user may wish to begin with a simple additive approach, while identifying a plan to improve understanding about likely interactions within the catchment. The challenge is to predict whether urbanization exerts dominant control on subsurface flow or climate, topography, and geology will overwhelm any influence of landuse change. This question cannot be answered simply. One pragmatic approach is to consult the regional literature or literature pertaining to a hydrologically similar region. In regions with active research, models may have been developed to indicate if landuse change is a dominant control over groundwater recharge and base flow. For example, Singh et al. (2014) developed a conceptual model relating relief and climate with land use for catchments in the USA that suggested that landuse change, including urbanization, has a strong effect on subsurface flows in the lowlands of humid regions. Similar regional models have been developed for the UK (Chiverton et al. 2014) and eastern Australia (Zhang et al. 2014).

If the regional models mentioned above are not available, managers can use the decision matrix in Table 2 as a guide. The table combines the outputs of both parts of the checklist presented in Fig. 3 to describe the likelihood of change to base flow and the implications of uncertainty on the prediction. The table is based on the premise that change to base flow will be greatest in vulnerable landscapes with a strong urban development risk. Where changes to base flow are less clear, the matrix indicates that managers should consider monitoring base flow or undertaking a water-balance analysis to provide clarity.

Establishing a water balance. If the assessment checklist and associated decision matrix suggest that the effect of urbanization on base flow is unclear, a detailed water-balance analysis is needed. A water-balance analysis requires a quantitative (data-driven) estimation of the mechanisms outlined previously and in the bottom half of Fig. 1. The challenges are to estimate these urban processes quantitatively by discovering, obtaining, and synthesizing the necessary data over the relevant spatial and temporal scales. Data may be available but may be unpublished or unavailable in publicly served electronic format (i.e., 'dark data'; Heidorn 2008). Although difficult to obtain, dark data can be useful. For example, municipal data can be used to estimate leakage from water-supply pipes by subtracting the sum of metered potable water from the sum of treated or withdrawn water, and leakage into wastewater pipes can be estimated by subtracting flow entering WWTPs from metered potable water (Semádeni-Davies and Bengtsson 1999).

The importance of various urban mechanisms may be assessed by comparing estimates to annual predevelopment (or nearby undeveloped) recharge rates, precipitation, or stream base flow. To make such comparisons, investigators first must decide what spatial scale is relevant to the ecological objectives. Healy (2010) discusses application of groundwater budgets at multiple scales and the potential discrepancy between surface and groundwater flow boundaries. If the scale is large, i.e., a large urban watershed, then an assumption of steady-state groundwater storage conditions often is made, and changes to surface inflows and outflows in that watershed (compared to precipitation and stream base flow) are the focus of the water-balance investigation. In contrast, a water-balance analysis examining an urban unconfined aquifer would specifically include changes in

	Factor	Example	Explanation and Information	Predicted vulnerability (H, M or L)
(1) Landscape vulnerability to change	Analysis of hydrological data			
	Hydrologic metrics	High base flow contribution (i.e. high Base Flow Index) Low base flow contribution (i.e. low Base Flow Index)	Hydrologic metrics provide the most direct means of assessing the vulnerability of a catchment to water table and base flow changes. These analyses typically require monitoring data and knowledge of the local hydrology. Where groundwater recharge is large or where there is a strong base flow contribution to total flow, the effects of vegetation change, impervious area creation, and interactions with urban infrastructure are likely to be stronger. Catchments with naturally low groundwater recharge and base flow contribution, e.g. with intermittent streams, may be less affected by such change, although may be more prone to baseflow increase.	
		Depth to water table < 4m Depth to water table > 4m	We propose 2-4 m as a representative depth to water table for urban infrastructure and natural vegetation rooting depth (Schenk and Jackson 2002). The highest vulnerability to change will thus be catchments with shallow depth to water table (< 4 m) and high base flow index, with the least vulnerable being catchments with deep groundwater and low base flow index streams.	
	OR			OR
	Analysis of catchment			
	Climate Soils/geology Topography Scale & location		Catchment characteristics such as the aridity index (climate), percentage of high porosity bedrock (soils/geology), mean slope, and elevation (topography), are typical indices that can be used to characterize the runoff response (e.g., Yadav et al. 2007, Singh et al. 2014). As a general rule: Climate: A more arid climate is likely to already have low base flows (except where depth to water table is shallow) and thus less likely to see a decrease in base flow due to creation of impervious areas. Conversely, such areas might be more vulnerable to increases due to other issues (e.g., leaking infrastructure). In less arid climates where existing base flows are high, there is a higher chance of post-urbanization decreases. Soils/geology: Where more permeable soils are present, the pre-development hydrology is likely to contain more base flow (O'Loughlin 1981) and be less flashy than in catchments with very low permeability. However, the influence of soils/geological properties will be highly dependent on topography. For example, in very flat catchments, significant infiltration will occur even where there are quite impermeable soils, through for example local recharge areas (O'Loughlin 1981). Topography: All things being equal, a steeper catchment will have a greater proportion of runoff (and thus lower groundwater/base flow contributions), but this will depend (as above) on soil properties and on climate (e.g. rainfall intensity and temporal distribution within a year). Scale & location of the urban footprint: The size and location of the urban footprint will affect the likelihood of change. For example, if the urban footprint makes up only a very small area of the catchment, and is located in areas that do not significantly contribute to recharge, any change is unlikely. Conversely, where the impervious area makes up a significant part of the catchment or is located in high recharge areas, significant changes will most likely occur. There is unlikely to be a simple threshold of urbanization above which effects are measurable. However, local knowledge may be available to derive a safe threshold (for example, in SE Australia, Walsh et al. (2012) demonstrated that significant baseflow impacts for a catchment with a connected impervious area of only 7% of the catchment). In addition to connected impervious areas driving hydrologic changes, modification of urban soils properties can as well, due to compaction and removal of organic matter (Ossola et al. 2015). **IMPORTANT: As all these factors interact, they should be considered together, with the final result considering not only the likely result for each separate category, but the potential interactions. For example, while a very steep catchment will likely have lower natural base flow, a very wet climate or very permeable soils will counteract this.**	

Factor	Description	Main driver	Predicted change	Score (+,++, 0, -, --)	Explanation and Information
(2) Urban development & infrastructure					
Impervious areas and stormwater management		'Classic' (evacuation)	- - likely		Effect of stormwater controls can be estimated using a simple rainfall-runoff or stormwater model such as SWMM (US EPA 1994) or MUSIC (eWater CRC 2012). Increased contributions to groundwater and base flow would require greater infiltration than would have occurred in the natural state, which can be predicted from a water balance analysis (e.g., Zhang et al. 1999).
		Detention	- likely		
		Widespread, distributed infiltration	+ to ++		
		Limited, localized Infiltration	- possible		
Evapotrans-piration (ET)	Existing vegetation dense with high ET demand	Loss of vegetation or reduced ET demand	++ likely		Existing vegetation affects the pre-urbanization subsurface flow and hence the influence of a land use change (Price 2011). For example, if a grassland catchment is urbanized and this involves the planting of substantial area of trees, this may increase over all ET. This prediction could be made by calculation the pre- and post-development areas of different vegetation types and using estimates of ET for each type (a starting point for such calculations is the global data provided by Zhang et al. 1999).
		No change	no change		
		Increased vegetation cover or ET demand	- or - - likely	OR	
	OR				
	Existing vegetation sparse or has low ET demand	Further loss of vegetation or reduced ET demand	+ possible		
		No change	no change		
		Increased vegetation cover or ET demand	- or - - likely		
Potable water supply and wastewater discharge		Imported from external catchments and discharged locally	+ possible, particularly if widely used for irrigation		Import and export could be by gravity-fed or by pumping. Water import/export can be determined from a water balance study, which will provide insight as to the volume of over-irrigation, relative to the annual baseflow volume, for example.
		Locally-derived water exported to external catchments	- likely		
		Locally-derived water infiltrated e.g. for irrigation	+ possible		
Infrastructure leakage		Leaks from water or wastewater networks to groundwater	+ likely		Leaks may be determined directly using water balance data or use of tracers to identify infrastructure-derived water (Bottrell et al. 2008, Lerner 2002, Wolf et al. 2012) or time-series modelling (Kracht and Gujer 2005).
		Leaks to sewer network	- likely		
		Few leaks	no change		
Stream channel morphology		Channel incision but regional groundwater input < stream export	- possible		Channel surveys (if existing) or geomorphic studies (based on proposed stormwater control measure strategy) can be used to predict likely geomorphic state of streams post-urbanization.
		Channel incision but regional groundwater input > stream export	+ possible		
		Bed colmation	- possible		Bed surveys (if existing) or geomorphic studies (based on proposed stormwater control measure strategy) to predict likely geomorphic state post-urbanization.
Potential impact of urban development = summary of scores above					

Figure 3. Assessment checklist for likely direction of change in water table and base flow. The checklist is made up of 2 categories: 1) *Natural landscape vulnerability* and 2) *Urban development and infrastructure*. In assessing landscape vulnerability (as High, Medium or Low), the user can use existing flow data (if available) or consider the combined effect of the catchment's characteristics (climate, topography, and soils/geology). The likelihood of change in base flow (increase [+ or ++], decrease [– or – –], or no change [0]) can then be predicted by considering the factors relating to the existing land use and urban infrastructure (e.g., vegetation change, nature of stormwater management). Predicted change that is very strong change should be indicated with 2 symbols (++ or – –). Where insufficient information is available to make an assessment of an individual item, the item should be left blank. The final result, obtained by summarizing the scores taking into account likely interactions, can be used to provide confidence of the overall direction of change to base flow. In the absence of knowledge about interactions, the user may wish simply to add the – and + scores, but this procedure should be done only with the explicit recognition that interactions are ignored and the uncertainty remains very high. In that case, further investigation to understand interactions better should be given priority as part of any proposed management response. The overall result should be considered together with the assessed vulnerability to give a qualitative assessment of the overall likelihood of change using the decision matrix provided in Table 2. The checklist can be carried out at the local (precinct) or catchment scale, but when it is used at the local scale, consideration should still be given to the potential effects at the larger (catchment) scale downstream. Guidance on the information requirements to inform the assessment is outlined in the last column. For more details see Hamel et al. (2013).

Table 2. Suggested decision matrix for interpretation of results of assessment checklist.

Landscape vulnerability	Predicted strength of directional change in base flow caused by urban development		
	Small (0)	Moderate (–, +)	Large (≤– –, ≥++)
Low	• Change extremely unlikely • Uncertainty has little effect on prediction • Revisit risk assessment when planning landuse change	• Change unlikely • Uncertainty has medium effect on prediction • Monitor base flow	• Change possible • Uncertainty may have strong effect on prediction • Undertake water balance
Medium	• Change unlikely • Uncertainty has medium effect on prediction • Monitor base flow	• Change possible • Uncertainty may have high impact on prediction • Undertake water-balance analysis	• Change likely • Uncertainty may have medium impact on prediction • Implement management options and monitor base flow
High	• Change possible • Uncertainty may have strong effect on prediction • Undertake water balance analysis	• Change likely • Uncertainty has low effect on prediction • Implement management options and monitor base flow	• Change extremely likely • Uncertainty may have medium effect on prediction • Implement management options and monitor base flow

unconfined groundwater storage and the effect of urban features on groundwater discharge and recharge and stream base flow. Water balances in urban areas frequently cross administrative and regulatory boundaries and the watershed boundaries that define the surface drainage area. The mismatch between the spatial scales required to establish a relevant urban water balance and the more frequently used municipal boundaries emphasizes the need to define a new spatial scale of interest that we call the 'urban water system'.

The water-balance approach described above is often difficult to use because direct measurements or proxies for the different water fluxes are rare and data availability is poor. A variety of techniques can be used to infer the water balance through modeling (Schirmer et al. 2013). These techniques include classical hydrologic approaches to water-flow analysis and methods developed for urban hydrogeology based on contaminant loads and concentration across space and time. For example, regional analyses can be used to understand the likely effects of urbanization on groundwater recharge (Bhaskar and Welty 2012, Hamel et al. 2015). In such an approach, the substitution of space for time allows analysts to infer the magnitude of the change in recharge caused by urban development (Stephenson 1994). A review of these techniques is outside the scope of our paper, but interested readers are referred to work by Schirmer et al. (2013) and Vázquez-Suñé et al. (2005).

Despite the considerable challenges in establishing an urban water balance, collaborative development (even conceptually) encourages multiple stakeholders to agree on dominant water sources and sinks in their urban system. This type of assessment may never have occurred before. More importantly, establishing an urban water balance will highlight gaps in both data and understanding. If we eschew a water balance, gaps in monitoring data are likely to continue for decades and capacity will not be built where it is needed most.

MANAGING THE WATER TABLE AND BASEFLOW RESPONSE TO URBANIZATION

Guidance on appropriate strategies to attenuate changes in base flow associated with urbanization is provided in Table 3. Where base flow is expected to increase, managers can focus on reducing recharge by harvesting stormwater or promoting ET (Walsh et al. 2016). Water may be trapped and stored in lined bioretention systems, but care is needed to ensure that controlled outlets do not increase base flows above natural conditions (Mitchell et al. 2007, DeBusk and Wynn 2011, Hamel et al. 2013). Harvested stormwater could be used to irrigate vegetation, but only when evaporative demand exceeds infiltration (Mitchell et al. 2003). Groundwater abstraction also could be used to irrigate vegetation where ET is reduced and water levels have risen with urban development (Barron et al. 2013a), but the effect of abstraction on dry-period base flow may need to be assessed. Catchment-wide tree planting to promote ET will be particularly effective in regions with naturally deep-rooted vegetation and where evaporative demand is high. Repairing leaking water-supply and wastewater pipes also may be important.

Where base flow is expected to decrease, management approaches can be focused on a combination of stormwater infiltration and harvesting to reduce the alteration to baseflow magnitude (Walsh et al. 2016). Infiltration can be designed to make up any predicted deficit between the natural baseflow level and the predicted postdevelopment

Table 3. Management strategies to attenuate increasing or decreasing stream base flow.

Management to attenuate increasing base flow	Management to attenuate decreasing base flow
• Stormwater harvesting for domestic use to reduce recharge • Lined bioretention with controlled outlet or distributed 'trickle tanks' to recreate natural baseflow patterns • Catchment-wide tree planting to promote evapotranspiration • Summer irrigation with stormwater up to, but not above, evapotranspiration demand • Summer irrigation using pumped ground water for urban areas that receive most of their rainfall during winter to lower water table and create a buffer for winter rises • Treatment of local ground water for drinking to reduce the volume of water imported into the catchment • Repair leaks from water-supply or wastewater infrastructure	• Infiltration of stormwater throughout the catchment, particularly in recharge areas • Repair leaks from wastewater and storm drain infrastructure if significant leakage of ground water into piped infrastructure • Reduce groundwater pumping, particularly during low-flow seasons when bores are close to the stream

level. In the absence of local data, simple guidance can be obtained from the 'Zhang curve' (Zhang et al. 2001), which provides broad annual streamflow coefficients for given rainfall zones and vegetation types (thereby giving a useful target water balance based on the likely state before urbanization). Infiltration generally is best applied as close to the source as possible for both hydrologic and urban amenity-related reasons (Mikkelsen et al. 1996, Chocat et al. 2001, Freni et al. 2010), but the overall volume of infiltration can be maximized by targeting areas that are naturally conducive to high rates of infiltration (Lerner 2002, Shuster et al. 2007). Reducing groundwater pumping also may be important in many catchments. Ultimately, an integrated water-cycle-management approach has the best opportunity to address baseflow concerns in heterogeneous urban landscapes.

Given that the responses of water tables and base flow to urbanization can be quite uncertain, the strategies adopted must be implemented within an adaptive-management framework. The measures proposed in Table 3 can be used to guide the initial efforts to mitigate urbanization effects, but their implementation has to be considered in the context of a longer-term effort to improve understanding of trends in base flow. Designing a long-term monitoring plan is a key step to reducing uncertainties of urban effects. Epting et al. (2008) described an adaptive management framework for ground water in urban areas. The framework hinges on the establishment of numerical groundwater models and creation of a network of groundwater and streamflow observation systems. Such a framework would allow detection of changes over time, but more importantly, would allow an assessment of whether the stated goals (e.g., in terms of desired flow regime) are being achieved. Trends over time can be compared to those predicted by the numerical model to improve understanding of the system. Given ecological targets, the observation network should consider including ecological monitoring (Westgate et al. 2013).

APPLICATION OF OUR FRAMEWORK ILLUSTRATED BY CASE STUDIES

The usefulness of our framework hinges on its ability to predict the directional change in water-table height and base flow correctly for most cases worldwide. Validation of our framework could be undertaken retrospectively based on studies that have documented changes in water-table height or base flow with urbanization, but the framework also can be applied to future development. Below we present 2 case studies to illustrate application of our framework, one from Perth, Western Australia, Australia, and the other from Baltimore, Maryland, USA. These case studies have opposite conclusions. In Perth, management strategies based on ET and stormwater harvesting would mitigate rising base flow, whereas in Baltimore, management strategies based on remediating leaky infrastructure and infiltration of stormwater might mitigate falling base flow.

Perth, Western Australia, Australia

The Perth Metropolitan Area, situated on the Swan Coastal Plain, has a Mediterranean climate with ~80% of the annual rainfall (~720 mm) falling during the Austral

winter between May and September. Evaporation is low during the wet winter months and very high during the dry summer months. The Swan Coastal Plain consists of a series of deep sand deposits with interbedded clay layers that support the unconfined aquifer (McArthur and Bettanay 1960). The landscape is relatively flat, and the drainage network consists predominantly of wetlands and streams. With the existing drainage network—a combination of agricultural, urban, and natural drains—the annual average depth to the water table varies considerably across the Plain from >10 m in highlands (sand dune crests) to <2 m in low-lying areas (sand dune valleys). Such variability commonly occurs over short distances (600 m). In areas with high water tables, creeks and drains receive substantial water from shallow ground water and have high base flow. At an annual scale, water balances indicate that groundwater recharge from rainfall/irrigation makes up 81 to 94% of the water fluxes, and up to 40% of that becomes discharge in storm drains or creeks (Barron et al. 2013a).

Increased demand for urban development in areas with high water tables has required a nuanced understanding of development guidelines that typically recommend, where appropriate, capture and infiltration of surface runoff within the development precinct instead of the classical evacuation. Implementation of infiltration strategies (e.g., soak wells, rain gardens, compensation basins) in high water table areas has caused flooding from groundwater inundation in the Perth region. The variability in water-table height across a catchment means that water management practices may need to vary between adjacent development precincts and possibly within a precinct. The current ecological objectives of urban development include restoration of stream habitat and reduction of nutrient export. However, these objectives are compromised by intrusion of nutrient-rich ground water into streams, drains, and wetlands and by associated rises in base flow, which are turning intermittent systems into perennial streams (Davis and Froend 1999).

We used the decision-support tool and the risk assessment checklist to explore appropriate management responses to 3 urbanization scenarios: Joondalup, Armadale 1, and Armadale 2. Scenarios Joondalup and Armadale 1 represent real situations, whereas Armadale 2 is hypothetical. This latter scenario highlights how the tool could be used to predict the outcome of different development practices in new precincts, i.e., the use of classic evacuation style stormwater management in Armadale. The physiographic settings are broadly the same for all scenarios (Mediterranean climate, highly permeable soils, flat topography, and possible loss of vegetation under urbanization), except that the depth to water table varies markedly between the localities, as does the vulnerability of the landscape to change to base flow. The ecological objective for all scenarios at both precinct and catchment levels is to restore water-table height so that streams, wetlands, and other groundwater-dependent ecosystems (including terrestrial vegetation) have more natural wetting and drying regimes.

In the Joondalup scenario, depth to the water table is >10 m. The tool (Appendix S1) suggests that stream base flow is unlikely to be altered by urbanization (vulnerability low), but that urban development will result in a substantial rise in the water table (++), making the effect of urbanization unclear. The health of deep-rooted vegetation may be affected by a change in water-table height. Table 2 directs us to undertake a water-balance analysis.

In the Armadale 1 scenario, the urban locale contains groundwater-dependent wetlands and streams, and the depth to the water table is <2 m. The tool suggests that stream base flow is highly vulnerable to urbanization. Widespread infiltration of stormwater management at the lot and precinct scales suggests with high confidence that base flow will increase (++). Table 2 suggests implementation of management strategies to reduce recharge.

In the Armadale 2 scenario, the shallow water table is likely to make stream base flow (and wetland hydroperiod) affected by urbanization. However, with classic evacuation of stormwater, base flow may change little (0). The high uncertainty around our prediction (low vulnerability but confidence in directional change) indicates that a water-balance analysis should be undertaken. A detailed description of the workings for the Perth scenarios is provided in Appendix S1.

Baltimore, Maryland, USA

Baltimore is situated at the transition between the Piedmont physiographic province and the Atlantic Coastal Plain and surface-water drainages discharge to the Chesapeake Bay. The water table in Baltimore has been affected by many urban stressors. In 1945, pumping removed about 34 million gallons of ground water/d (1.5 m^3/s) from confined aquifers underlying the industrial area around the harbor. This withdrawal created cones of depression in ground water >46 m around Sparrows Point (Bennett and Meyer 1952). Groundwater storage recovered later in the century after cessation of industrial pumping. Groffman et al. (2002) found that water tables were lower in urban than in less-developed riparian zones and attributed this pattern to incision of the urban stream channel. Schwartz and Smith (2014) found that watersheds in Baltimore did not have consistent baseflow responses and, instead, were affected by various processes, such as connected impervious surfaces, stormwater ponds, interbasin transfers, and leaking infrastructure (both recharge and drainage).

When the decision framework was applied to Baltimore, the ecological consequences of changing base flow were examined. During all but the driest conditions, streams in the Baltimore region are fed by shallow ground water, meaning

that water-table changes will have ecological consequences, such as the isolation of stream flow from riparian vegetation and potential reduction in denitrification (Groffman et al. 2002). Therefore, the overall ecological goal is to restore baseflow conditions to the predevelopment flow regime. The assessment checklist (Fig. 3) was applied to Baltimore (see Appendix S2 for details). Baltimore has a humid climate (1060 mm annual rainfall) with groundwater-supported streamflow, relatively flat topography (Hopkins et al. 2015), and depth to water table ≤2 m in stream riparian areas (Striz and Mayer 2008). These factors indicate that landscape vulnerability is high. In terms of urban development, dense forested vegetation was lost when the area was developed (which, in isolation, would suggest a rise in water table); most areas have either classic evacuation or detention in terms of stormwater controls (water-table fall very likely); water is imported from external catchments and, in some cases, used for irrigation (potential water-table rise); infiltration of ground water to the sanitary sewer network is high (water-table fall very likely); and channel incision has been observed in some Baltimore streams with some regional and some riparian inputs of ground water to the stream (resulting baseflow change is hard to predict) (Striz and Mayer 2008). The overall prediction is that urban development has led to a water-table drop in Baltimore (–), but that the effects are expected to be strongly spatially variable because some existing factors would lead to water-table rises in some locations (e.g., under water-main leaks).

Based on the combination of landscape vulnerability and urban development, change in base flow is likely. A water-balance study comparing urban and rural watersheds in the Baltimore metropolitan region was undertaken to evaluate the effect of urban development on the water budget (Bhaskar and Welty 2012). This water-balance study was focused on watersheds rather than on an aquifer as the spatial unit of analysis and, therefore, did not explicitly consider changes in water-table height. The analysis revealed that groundwater leakage into wastewater pipes is the primary factor driving the urban alteration to the water balance (Fig. 4A, B) and has a larger effect on subsurface storage than impervious surfaces (Bhaskar et al. 2015). Baltimore has separate wastewater and storm systems, but the wastewater infrastructure is aged and leaky and, therefore, receives groundwater inflows. Groundwater discharge into wastewater pipes means that the sanitary sewer infrastructure serves as a drain and controls water-table levels in some parts of the city. This situation suggests that at a watershed scale, stream base flow will fall with greater urban development, as has been observed over Baltimore area watersheds with a range of urban intensities (Groffman et al. 2002, Hopkins et al. 2015). The management strategies (Table 3) that might mitigate this decrease in base flow are those that focus on remediating leaky infrastructure and infiltration and ET of stormwater throughout the catchment.

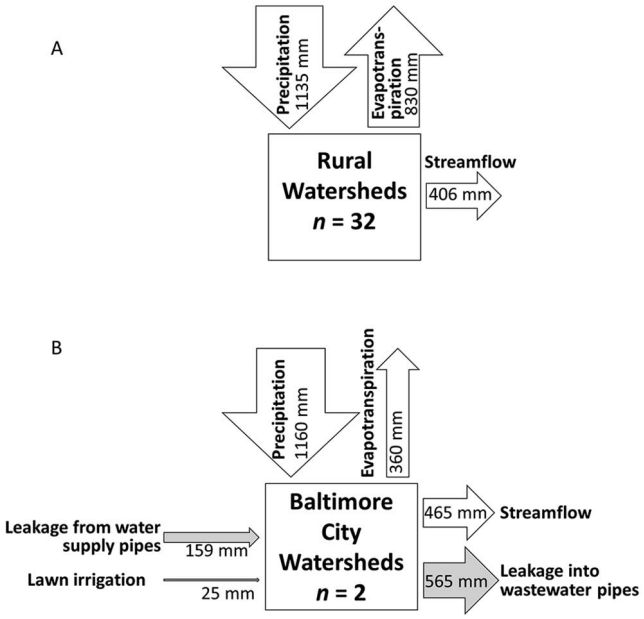

Figure 4. Comparison of the annual water-balance components between 32 rural (impervious area <5% and without reservoirs) (A) and 2 urban (Gwynns Run and Moores Run) (B) watersheds in Baltimore City, Maryland, USA. The width of the arrows corresponds to the magnitude of the flows (mm/y). Unshaded arrows correspond to natural water-balance components, whereas shaded arrows indicate urban water-balance components. Components were estimated separately (not by subtraction), so inflows do not necessarily equal outflows. Streamflow captures total annual flows at US Geological Survey gages, including stormwater. Figure modified from Bhaskar and Welty (2012).

DISCUSSION

Urbanization affects stream base flow in diverse and complex ways to create different ecological risks. Our framework helps predict the direction and likelihood of baseflow change and suggests the most appropriate management strategies to mitigate these changes. Preliminary assessment of our framework, via case studies, demonstrates potential utility of the tool, but it must be tested more widely to assess its broad-scale applicability. One important attribute of our framework is that it elucidates the array of factors that influence urban water tables and stream base flow. As a result, the tool should aid managers in gathering the types of information needed to make the best appraisal of the effect of urbanization on stream base flow.

Our framework, particularly our assessment tool, has several limitations that warrant discussion. First, the tool does not weight the importance of each urban factor. Weighting can be undertaken only if data are available about the extent to which urban factors are increasing or decreasing recharge. Such data are unlikely to be available in many locations. We encourage managers to apply their

own weighting where possible. Second, the tool does not capture interactive or synergistic relationships among factors. This stems partly from a desire to keep the framework simple, but also reflects the lack of general, transferable quantitative relationships available (but see Jencso and McGlynn 2011). Future broad-scale studies assessing how landscape vulnerability interacts to affect recharge would be useful and would be likely to lead to improvements in the tool. Managers faced with equivocal results from the tool should consider constructing a water-balance model. Last, we have assumed that urban-induced changes to the water table equate to changes in stream base flow, but this may not always be the case. For instance, in-stream water withdrawals or additions (i.e., wastewater inputs) may override the consequences of local groundwater changes.

The potential importance of numerous natural (e.g., physiographic) and anthropogenic (e.g., urban development and infrastructure) factors on stream base flow suggests it is unwise for managers to extrapolate the likely impacts of urbanization from one region to another, particularly regions with dissimilar physiographic settings and development practices. This issue is highlighted by our case studies and is in keeping with the growing awareness of the heterogeneity inherent with the urban stream syndrome (Booth et al. 2016). The importance of physiographic factors (topography, soil type, climate) is underscored in the Perth case study, where the flat, sandy landscape and shallow water table result in rising base flow and increased groundwater-derived nutrients from urbanization. However, a similar outcome may not occur in steeper, less permeable landscapes. Natural landscape differences can be used to identify physiographic aspects that can confer resilience to urbanization. For example, natural water-storage capacity of a catchment may be able to buffer urban streams from surface-water hydrological stress (Utz et al. 2016) and watershed capacitance may indicate probable fates of infiltrated stormwater (Miles and Band 2015).

The importance of urban development and infrastructure is highlighted by the Baltimore case study, which revealed that underground infrastructure, particularly groundwater leakage into sanitary sewers, can exceed effects of impervious surface cover on base flow. Future studies examining the effect of urbanization on stream base flow will benefit from greater consideration of below-ground urban infrastructure, particularly in older cities.

Our case studies also highlight the potential for small spatial-scale variation (caused by infrastructure, vegetation, or water-use differences) to create meaningful differences in the baseflow response to urbanization. Where fine-scale differences are important, scientists and managers should be aware that space-for-time studies that attempt to assess the effect of urbanization on base flow through time may be confounded. Small spatial-scale differences associated with physiography could be used by managers to prioritize the location of future urban densification.

From an ecological perspective, restoring or preserving stream base flow will be successful only if it improves or maintains ecological health and function. More studies are needed to assess the extent to which management practices can protect or restore predevelopment base flow and generate improvements in stream and riparian ecosystem health and functioning. Ultimately, tailored solutions that help to restore a region's natural flow regime, including base flow, will bring us closer to healthier urban stream communities—both ecological and human.

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