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Urban base flow with low impact development

Aditi S. Bhaskar,[1]* Dianna M. Hogan[2] and Stacey A. Archfield[3]

[1] *National Science Foundation Postdoctoral Fellow hosted at U.S. Geological Survey, Eastern Geographic Science Center, 12201 Sunrise Valley Drive, MSN 521, Reston, VA, 20192, USA*
[2] *U.S. Geological Survey, Eastern Geographic Science Center, 12201 Sunrise Valley Drive, MSN 521, Reston, VA, 20192, USA*
[3] *U.S. Geological Survey, National Research Program, 12201 Sunrise Valley Drive, MSN 430, Reston, VA, 20192, USA*

Abstract:

A novel form of urbanization, low impact development (LID), aims to engineer systems that replicate natural hydrologic functioning, in part by infiltrating stormwater close to the impervious surfaces that generate it. We sought to statistically evaluate changes in a base flow regime because of urbanization with LID, specifically changes in base flow magnitude, seasonality, and rate of change. We used a case study watershed in Clarksburg, Maryland, in which streamflow was monitored during whole-watershed urbanization from forest and agricultural to suburban residential development using LID. The 1.11-km^2 watershed contains 73 infiltration-focused stormwater facilities, including bioretention facilities, dry wells, and dry swales. We examined annual and monthly flow during and after urbanization (2004–2014) and compared alterations to nearby forested and urban control watersheds. We show that total streamflow and base flow increased in the LID watershed during urbanization as compared with control watersheds. The LID watershed had more gradual storm recessions after urbanization and attenuated seasonality in base flow. These flow regime changes may be because of a reduction in evapotranspiration because of the overall decrease in vegetative cover with urbanization and the increase in point sources of recharge. Precipitation that may once have infiltrated soil, been stored in soil moisture to be eventually transpired in a forested landscape, may now be recharged and become base flow. The transfer of evapotranspiration to base flow is an unintended consequence to the water balance of LID. © 2016 The Authors Hydrological Processes Published by John Wiley & Sons Ltd.

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INTRODUCTION

Soon after World War II, urban stormwater management was designed to evacuate stormwater from roadways as quickly as possible to prevent flooding (NRC, 2009). In the 1970s, centralized detention ponds became standard. These aimed to reduce peak stormflow by delaying stormwater discharge to streams and allow for settling of sediment. This conventional type of stormwater management does not however reduce stormwater volume or treat small storms, which make up the bulk of stormwater quantity and pollution (Booth and Jackson, 1997; Emerson *et al.*, 2005). Because the aim of conventional stormwater management is peak flow reduction, all other alterations of the natural flow regime because of urban development are not mitigated, and resulting degradation of stream quality, geomorphology, and ecology persist (Poff *et al.*, 1997; Meyer *et al.*, 2005; Walsh *et al.*, 2005). The impacts of stormwater on the deterioration of urban streams, aquatic ecosystems, and downstream receiving water bodies have led regulators to demand new types of stormwater management that achieve more than simply urban flood control.

To address these fundamental concerns regarding conventional stormwater management systems, the newest strategy aims to engineer an urban hydrologic system that is 'functionally equivalent' to a natural system (US EPA, 2000). Similar approaches are referred to by many names, such as low impact development (LID), green infrastructure, source-control stormwater management, Environmental Site Design, and distributed stormwater management (Fletcher *et al.*, 2014). We will refer to this practice as 'LID', which is intended to preserve or restore pre-development hydrologic conditions in an urban watershed by retaining and infiltrating stormwater close to its source (Prince George's County, 1999; US EPA, 2000). LID uses a mix of approaches including minimizing impervious surfaces, increasing response time, reducing soil compaction and erosion during urbanization, public education, as well as the use of infrastructure-based stormwater facilities. Stormwater facilities used within LID focus on greater stormwater

*Correspondence to: Aditi S. Bhaskar, Colorado State University, 1372 Campus Delivery, Fort Collins, CO 80523-1372.
E-mail: aditi.bhaskar@colostate.edu

infiltration (e.g. bioretention basins, rain gardens, porous pavement, grassy swales, dry wells, if these do not contain underdrains), retention (e.g. rain barrels), evapotranspiration (e.g. green roofs), and treatment (e.g. filters, separators, and some of the previously mentioned stormwater facilities).

The effectiveness of LID for water quality and flow has been evaluated by a number of researchers on the scale of individual stormwater facilities (Lindsey *et al.*, 1992; Davis, 2008; Wild and Davis, 2009) and neighbourhood or watershed scales (Holman-Dodds *et al.*, 2003; Perez-Pedini *et al.*, 2005; Williams and Wise, 2006; Hood *et al.*, 2007; Dietz and Clausen, 2008; Selbig and Bannerman, 2008; Bedan and Clausen, 2009; Gilroy and McCuen, 2009; Hogan *et al.*, 2014). Studies have also investigated the effect of LID on groundwater recharge, where recharge is that part of infiltrated water not taken up by plants or evaporated but instead reaches the water table (Healy, 2010). These studies have revealed that (1) the effect of stormwater infiltration on recharge depends strongly on pre-development recharge (Ku *et al.*, 1992; Keßler *et al.*, 2012; Stephens *et al.*, 2012; Thomas and Vogel, 2012) and (2) groundwater mounding beneath stormwater facilities is more prevalent with low hydraulic conductivity, centralized spatial arrangement, or positioning at topographic lows (Endreny and Collins, 2009; Carleton, 2010; Machusick *et al.*, 2011). Groundwater mounding may cause unintended damage to infrastructure, such as flooding of underground structures, groundwater leakage into wastewater pipes, vegetation damage, and pollutant mobilization (Göbel *et al.*, 2004).

It is unclear, however, how to translate previous work investigating changes to recharge at the site scale or neighbourhood scale to predict the cumulative effects LID has on watershed-scale recharge (Hamel and Fletcher, 2014). The watershed-scale effects of LID are key because while the implementation of stormwater facilities may be local, a goal of LID is clearly to restore watershed-level hydrologic functioning (Burns *et al.*, 2012; Hamel *et al.*, 2013). Changes to watershed-scale recharge can be most easily observed as changes to base flow, defined here as the groundwater-derived inputs to streamflow, although in urban areas, other inputs to dry-period flow such as wastewater effluent and delayed stormwater pond contributions may be important (Sophocleous, 2002; Liu *et al.*, 2013). Researchers have examined temporal trends in base flow magnitude during urban development (Pluhowski and Spinello, 1978; Brandes *et al.*, 2005; Meyer, 2005; Chang, 2007; Hubbart and Zell, 2013; Townsend-Small *et al.*, 2013; Hogan *et al.*, 2014), while others have used space-for-time substitutions in comparisons of rural and urban areas to infer how base flow magnitude may have changed during urban development (Barringer *et al.*, 1994; Finkenbine *et al.*, 2000; Rose and Peters, 2001; Schwartz and Smith, 2014; Hamel *et al.*, 2015; Hopkins *et al.*, 2015). These studies make it clear that there is no consistent effect of urban development on base flow – both increases and decreases in base flow magnitude with urban development have been observed (O'Driscoll *et al.*, 2010; Price, 2011; Hamel *et al.*, 2013; Bhaskar *et al.*, 2016). Unlike urban stormflow, which is consistently made flashier by directly connected impervious area, there is diversity in urban base flow response (Hopkins *et al.*, 2015) because of the multitude of processes that can affect base flow in addition to imperviousness, such as leakage in and out of piped infrastructure, changes in evapotranspiration, import and export of water and wastewater, and, as is investigated here, evolving modes of stormwater management (Bhaskar *et al.*, 2016).

Almost none of the studies that have investigated trends in base flow magnitude during urban development have done so in watersheds built with LID. Most implementations of LID stormwater facilities are retrofits and piecemeal projects amongst existing conventional stormwater infrastructure, making the effects of LID on base flow at a watershed scale difficult to evaluate (Roy *et al.*, 2008). Furthermore, it has been established that flow is 'the *maestro* that orchestrates pattern and process in rivers' (Walker *et al.*, 1995) and that a naturally variable regime of flow, and not just a minimum low flow, is needed for stream ecosystem health (Poff *et al.*, 1997, 2010). Therefore, to achieve a functionally equivalent hydrologic system using LID, we cannot solely focus on high-flow response nor can we focus just on base flow magnitude (Schwartz and Smith, 2014; Hamel *et al.*, 2015). The variable regime of flow is critical for stream ecosystem health, but we still do not understand how urbanization using novel stormwater management affects ecologically relevant base flow magnitude, seasonality, and rate of change (Rolls *et al.*, 2012). Given the inability of past research to quantify the changes to the base flow regime at the watershed scale resulting from urbanization using novel stormwater management, our objective is to assess changes in base flow magnitude, seasonality, and rate of change in an LID watershed during and after urbanization. We do this in a small, instrumented LID watershed and compare flow changes from 2004 to 2014 to nearby forested and urban control watersheds. We then evaluate the changes observed in base flow regime during and after urbanization in the LID watershed with respect to expected changes from three dominant mechanisms of base flow regime alteration: vegetation removal, introduction of directly connected impervious surfaces, and infiltration through stormwater facilities.

Mechanisms of base flow regime alteration during urbanization with LID

We postulate that there are three mechanisms occurring in short order during urbanization that alter the base flow regime: (1) removal of vegetative cover (agricultural crop fields and forest), (2) paving and introduction of impervious surface cover, and (3) drainage of impervious surface cover to infiltration-focused stormwater facilities. Our general water balance formulation, which will be discussed in the context of our results, is

$$P + Q_{\text{in}} + I = \text{ET} + Q_{\text{out}} + \Delta S, \quad (1)$$

where P is precipitation; Q_{in} is inflow to the watershed via groundwater; I is lawn and garden irrigation using piped water supply; ET is evapotranspiration; Q_{out} is outflow from the watershed via groundwater ($Q_{\text{out,gw}}$) + surface water ($Q_{\text{out,sw}}$), where surface water outflow is made up of a stormflow component (Q_{storm}) + a base flow component (Q_{base}); and ΔS is change in storage (all with depth/time units). Leaky pipes were not included as part of the urban water balance (e.g. Bhaskar and Welty, 2012), as we assumed that a watershed during and directly after urbanization had infrastructure in good condition and minimal leaks. Figure 1 shows a conceptual

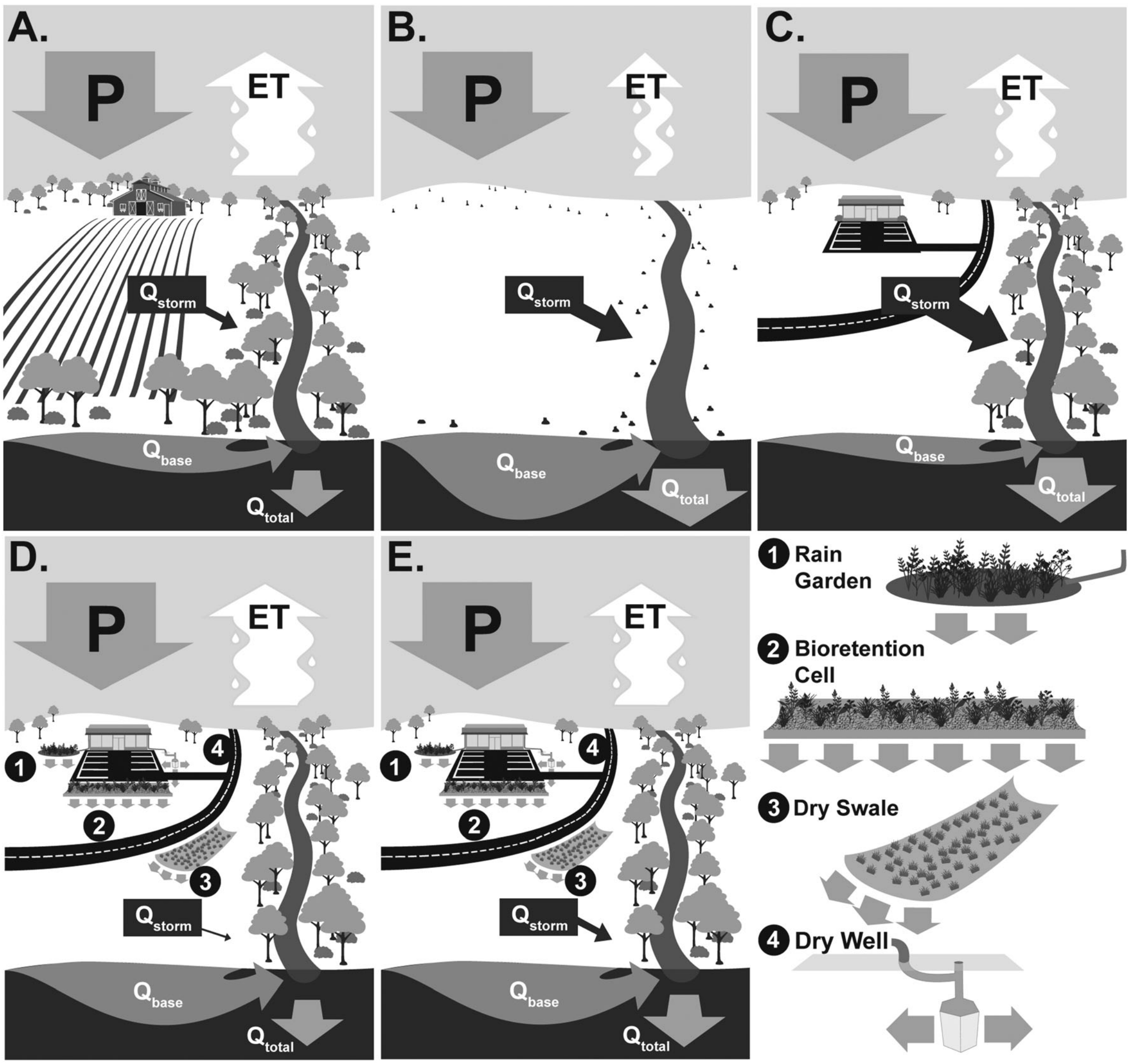

Figure 1. Conceptual diagrams showing expected (not observed) changes from a pre-development hydrologic cycle (A) through different urban processes, where P is precipitation, ET is evapotranspiration, Q_{storm} is stormwater, Q_{base} is base flow, and Q_{total} is total streamflow (stormwater + base flow). Panel (B) shows a hydrologic cycle with vegetation removal, panel (C) shows a hydrologic cycle with directly connected impervious surfaces, panel (A) shows the fluxes of an ideally functioning hydrologic cycle in an urban watershed urbanized using LID (equivalent fluxes as pre-development), and panels (D) and (E) show hydrologic cycles with overcompensating infiltration-focused stormwater facilities (1–4). In panel (D), additional stormwater infiltrated would have been stormwater pre-development, and in panel (E), additional stormwater infiltrated would have been evapotranspiration pre-development (i.e. Figure 1D has smaller Q_{storm} and Q_{total} and larger ET than Figure 1E)

diagram of expected changes to these components of the water balance, starting from pre-development conditions (Figure 1A).

Vegetation removal. Previous studies have found that urban areas generally have reduced evapotranspiration (Dow and DeWalle, 2000; Haase, 2009; Roy *et al.*, 2009; Bhaskar and Welty, 2012; Barron *et al.*, 2013) because of the reduction in transpiration from a less vegetated landscape. If vegetation removal was the sole mechanism of hydrologic alteration during urbanization, we might expect an increase in total streamflow and base flow as evapotranspiration decreases (reduction in ET leading to increase in Q_{out} in Equation 1), a reduction of amplitude in the seasonal cycle of base flow, and a slower base flow recession (Figure 1B shows an un-vegetated hydrologic system).

Introduction of directly connected impervious surfaces. Directly connected impervious surfaces alone would be expected to decrease base flow magnitude by limiting infiltration (Leopold, 1968; Rose and Peters, 2001) and simultaneously increase stormflow magnitude and decrease response time (Figure 1C shows an urban water cycle altered solely by impervious surfaces). This would imply an increase in the stormwater from the watershed (Q_{storm}), which may be compensated for by a decrease in base flow (Q_{base}) (Equation 1).

Infiltration through stormwater facilities. Infiltration-focused stormwater facilities may lead to more recharge from point sources instead of diffuse sources (Appleyard, 1995), which has been suggested to increase annual recharge (Ku *et al.*, 1992; Göbel *et al.*, 2004; Stephens *et al.*, 2012; Barron *et al.*, 2013; Hamel and Fletcher, 2014), although not in all cases (Keßler *et al.*, 2012). Ideally with LID, the goal is for the urbanized water cycle to be equivalent to the pre-development cycle in overall fluxes (i.e. Figure 1A shows both the pre-development and ideal LID hydrologic fluxes). In this case, all excess stormwater generated by impervious surfaces (e.g. extra stormflow in Figure 1C compared with Figure 1A) is re-routed by LID facilities to either evapotranspiration or base flow in the proportions that would have occurred pre-development (e.g. 2/3 to evapotranspiration and 1/3 to base flow for Figure 1C to become equivalent to Figure 1A). This situation is still only a goal however, because urbanization resulting in an unaltered hydrologic cycle has not been demonstrated, although a tailored mix solution for stormwater facilities has been proposed (Askarizadeh *et al.*, 2015).

Where stormwater facilities infiltrate only some of the excess stormwater generated by impervious surfaces, the otherwise expected decrease in base flow volume (due solely to directly connected impervious surfaces) might be partially mitigated. This would result in hydrologic fluxes somewhere between the pre-development watershed (Figure 1A) and the impervious surface-altered watershed (Figure 1C). The opposite situation is also possible (e.g. Appleyard, 1995), where more water is infiltrated through LID facilities than would have infiltrated pre-development, leading to greater recharge and stream base flow, here referred to as overcompensating stormwater recharge. With greater recharge, we would expect greater base flow magnitude and a slower base flow recession. However, the expected changes to other parts of the hydrologic cycle would depend on what fate the recharged stormwater would have had in the pre-development watershed. In terms of Equation 1, a rise in base flow (Q_{base}) may be compensated for by a decrease in stormflow (Q_{storm}), ET, or some other term.

First, we consider the case in which the additional base flow is offset by less stormflow. Where pre-development stormflow is now routed to infiltration facilities, increased base flow comes at the expense of stormflow, not only compared with a watershed with a high proportion of directly connected impervious area (Figure 1C) but even as compared with the pre-development watershed (Figure 1D *vs* Figure 1A). In this case, we would expect that the volume of stormflow would actually decrease after urban development and this decrease to be matched to the increase in base flow (Figure 1D). In other words, the total streamflow would remain the same, but the ratio of stormflow to base flow would decrease.

The second case we consider is where additional base flow is offset by less ET. In a forested landscape, much of the water that infiltrated might have been stored in soil moisture before being taken up by plants or evaporated, with some draining down as diffuse recharge. After urbanization with overcompensating infiltration facilities, a greater fraction of infiltrated stormwater may drain more efficiently to become recharge as opposed to evapotranspiration than would have occurred in the watershed pre-development. In this case, the post-development excess stormwater would be matched with a decrease in ET, and there would be no change in stormflow (Figure 1E). As in the case with vegetation removal (Figure 1B), we would observe an increase in total streamflow and base flow as both Figures 1B and 1E are driven by a reduction in evapotranspiration, although through different processes. Of course, where greater infiltration is observed post-development, there is likely some mix between these two endmembers (Figures 1D and 1E), where increased base flow is offset by some combination of decreases in pre-development stormwater and pre-development ET. We now examine what was observed in the base flow regime while the study watershed urbanized with LID.

Study area

The study focuses on Tributary 104, a 1.11-km^2 watershed northwest of Washington, DC (Figure 2),

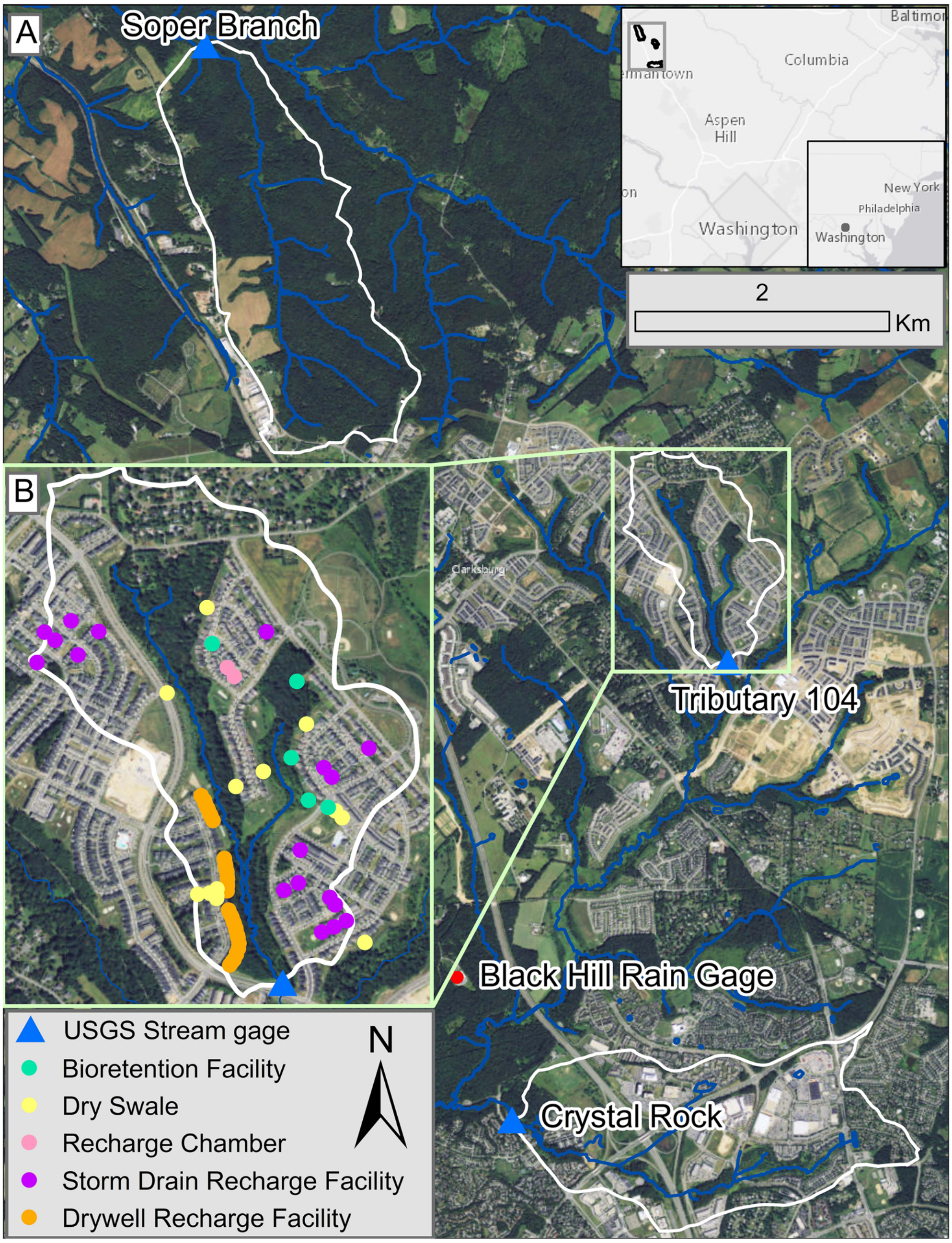

Figure 2. (A) Three study watersheds in Clarksburg, Maryland: Soper Branch (forested control), Tributary 104 (study watershed with LID), and Crystal Rock (urban control). (B) Inset showing locations of 73 infiltration-focused stormwater facilities in Tributary 104 and their types

draining to the US Geological Survey (USGS) stream gage at Little Seneca Creek Tributary near Clarksburg, Maryland (http://waterdata.usgs.gov/nwis/uv?01644371). This stream gage was installed in July 2004, and therefore, we use the period 1 October 2004–30 September 2014 for analysis. Tributary 104 is a recently urbanized watershed that was converted from agricultural and forested land to primarily suburban neighbourhoods between 2002 and 2010 (Figure 3) (Hogan *et al.*, 2014). During urbanization, vegetated land cover declined from 95% in 2002 to 68% in 2012, and impervious surfaces increased to 30% during that same time (Figure 4). We used two nearby control watersheds that had little land cover change over the study period for comparison with changes in Tributary 104 (Figure 4): Soper Branch (at Hyattstown, Maryland; http://waterdata.usgs.gov/usa/nwis/uv?01643395), a 3.03-km^2 watershed that was 85% forested with 3% impervious surface cover in 2012, and Crystal Rock, a 3.50-km^2 suburban watershed draining to the stream gage at Little Seneca Creek

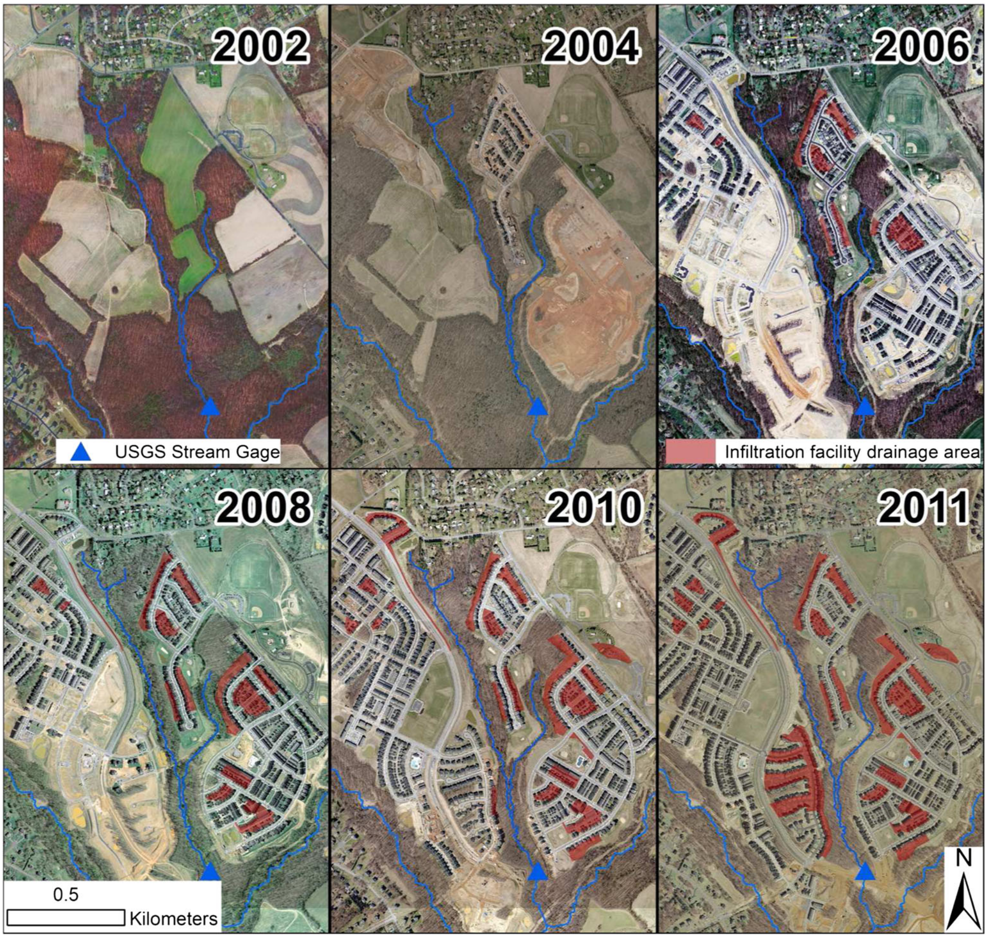

Figure 3. Aerial ortho-images collected by Montgomery County from 2002 to 2011 over Tributary 104 during urbanization. Red shaded areas indicate the areas that have completed residential development and that drain to infiltration-focused stormwater facilities (shown in Figure 2), based on sewershed drainage areas developed by Sparkman (2015). Blue lines and the blue triangle respectively indicate the stream and location of the stream gage

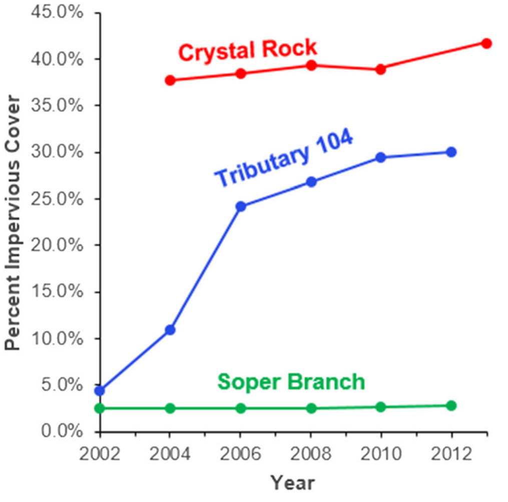

Figure 4. Impervious area in Crystal Rock (urban control), Tributary 104 (LID watershed), and Soper Branch (forested control) over time. Land use/land cover coverages were developed using 2002, 2004, 2006, 2008, 2010, 2012, and 2013 aerial ortho-images (where available) collected by Montgomery County and manually classified and digitized as described in Hogan *et al.* (2014)

Tributary near Germantown, Maryland (http://waterdata.usgs.gov/nwis/uv?01644375). Crystal Rock watershed was 42% impervious in 2013 and was largely urbanized by the 1990s when conventional stormwater management was in use (Rhea *et al.*, 2015). All three watersheds fall within the Piedmont physiographic province and are generally underlain by metamorphic rock (metasiltstone) with granular quartzite of the Marburg Formation (Southworth *et al.*, 2007). The normal annual precipitation is 1178 mm at a nearby National Climatic Data Center site (DAMASCUS 3 SSW, MD US GHCND: USC00182336), and the climate is characterized as humid subtropical with year-round precipitation. Precipitation data from a gage at Black Hill operated by Montgomery County (Figure 2) was used conservatively to exclude time periods when rain occurred for the recession analysis, but the data were not used for annual or monthly totals because it was unclear how representative this gage was of the entire study area.

Tributary 104 falls within the Clarksburg Special Protection Area, a designation used by Montgomery

County, Maryland, to protect high-quality waterways, in part by applying more stringent stormwater management requirements (https://www.montgomerycountymd.gov/DEP/water/special-protection-areas.html). Because of these requirements, Tributary 104 was urbanized using small-scale stormwater facilities distributed throughout the watershed, although the development did not include the reduced impervious cover that is part of LID in other locations. While the study watershed was built such that no impervious area would drain immediately to the stream, 75% of the impervious surface cover does not drain to infiltration stormwater facilities but does undergo water-quality treatment or stormwater detention. Areas draining to water-quality or detention stormwater facilities enter streams with only a relatively short delay and therefore would largely contribute to the stormflow hydrograph rather than base flow. Twenty-five percent of the impervious area in the watershed drains to infiltration-focused stormwater facilities, and as shown in Figure 2, these are located in the upland area of the watershed, not in or adjacent to the stream channels as is standard with conventional stormwater management. There are 73 infiltration-focused stormwater facilities in Tributary 104, composed of 35 dry well recharge facilities, 18 storm drain recharge facilities, 13 dry swales, 5 bioretention facilities, and 2 recharge chambers (Figure 2). Goals for the stormwater facilities in Tributary 104 were to 'protect stream/aquatic life habitat; maintain stream base flow; protect seeps, springs and wetlands; maintain natural on-site stream channels; minimize storm flow run-off increases; identify and protect stream banks prone to erosion and slumping; minimize increases to ambient water temperature; minimize sediment loading; minimize nutrient loading; and control insecticides, pesticides, and toxic substances' (Montgomery County DEP, 2012).

Past research in this study watershed found that sediment and erosion control facilities used during urbanization were not able to prevent adverse hydrologic, geomorphologic, and biotic effects during urbanization (Hogan *et al.*, 2014). Tributary 104 underwent profound topographic change during urbanization with about 80 cm of earth moved for every square meter of watershed between 2004 and 2006 alone (Hogan *et al.*, 2014), which produced abrupt slope changes and disjointed hillslopes (Jones *et al.*, 2014). As one of the last steps in subdivision development, stormwater was routed to stormwater management facilities instead to sediment and erosion control facilities (Figure 3).

METHODS

We used hydrograph separation to examine base flow magnitude and seasonality over time and recession analysis to examine base flow rate of change and compared them to the nearby forested and urban control watersheds. Statistical tests were used to evaluate any observed differences in base flow magnitude. A review of these methods in context is found in Smakhtin (2001).

Hydrograph separation

We used automated hydrograph separation techniques, which are heuristic methods to separate streamflow into 'slow flow' and 'quick flow', representing the slow and fast response components of streamflow without any assumptions about the underlying processes (Schwartz and Smith, 2014). Although we would like to know the component of streamflow that is groundwater-derived (i.e. base flow), information on the process by which water enters the stream is not knowable using streamflow rate alone. Instead, we assume that changes to slow flow are representative of changes to base flow, although we also acknowledge that the magnitude of slow flow as calculated by any one technique does not indicate the magnitude of base flow. Risser *et al.* (2005) found that using multiple slow flow separation techniques bounded estimates of base flow by other methods (e.g. recharge estimates using water balance approaches). Therefore, here multiple flow separation techniques were used (Figure 5). We used a one-parameter digital filter, with a filter parameter of 0.925 (Nathan and McMahon, 1990), executed on the automated web geographic information system-based hydrograph analysis tool (Lim *et al.*, 2005). We also used HYSEP local minimum and HYSEP fixed width methods, executed in the Groundwater Toolbox (Sloto and Crouse, 1996; Barlow *et al.*, 2015). As the one-parameter digital filter gave a mid-range result for slow flow magnitude (Figure 5) and the statistical test results using all three methods were similar, we show results from just this separation method. We carried out analyses of slow flow over 1 October 2004–30 September 2014 in Tributary 104, Crystal Rock, and Soper Branch using R statistical software (R Core Team, 2014). We used a seasonal Mann–Kendall trend test (Hirsch *et al.*, 1982) with significance level of 0.05, implemented in the R package rkt (Marchetto, 2015) for monthly slow flow records. The seasonal Mann–Kendall test is a nonparametric test of trend that takes into account correlation that may exist between months. We separated the period of record into 'during' (1 October 2004–30 September 2010) and 'after' urbanization (1 October 2010–30 September 2014) periods in Tributary 104, based on Montgomery County DEP (2012) and aerial imagery of urbanization (Figure 3). Auto-regression functions were used to examine changes in slow flow seasonality.

Recession analysis

The rate at which streamflow declines over time during dry periods is referred to as the recession rate (Smakhtin, 2001). The recession rate is important in and of itself as an

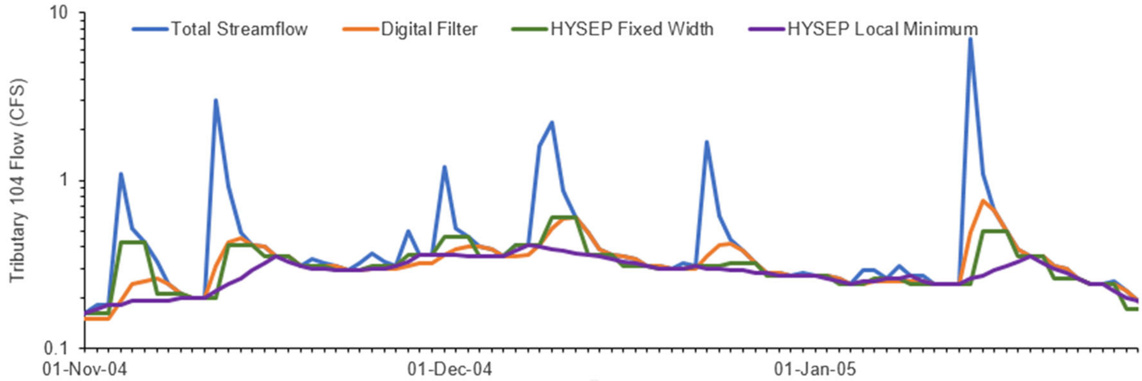

Figure 5. Total streamflow and slow flow separation using multiple methods (digital filter, HYSEP fixed width, HYSEP local minimum) applied to Tributary 104 streamflow record. Only the period from November 2004–January 2005 is shown for ease of comparison

attribute of the flow regime (rate of change) that shapes stream ecological health (Poff *et al.*, 1997; Rolls *et al.*, 2012) and also indicates the relationship between subsurface storage and streamflow, where a slower recession rate generally indicates a larger store of subsurface storage. Brutsaert and Neiber (1977) presented recession rates by examining streamflow decline compared with average streamflow. We used this method to examine base flow recession rates in our study and control watersheds, using binning and fitting through the central tendency of the recession plot (Kirchner, 2009).

This recession method assumes that streamflow recession is primarily related to changes in storage, which would require that precipitation and ET (including riparian ET) are minimal. We assume that we can isolate such time periods by using only rainless nighttimes (Kirchner, 2009). To select rainless nighttimes, we used hourly solar radiation and precipitation data, which were obtained from the Clean Air Status and Trends Network, Beltsville, Maryland site (BEL116; http://epa.gov/castnet/javaweb/site_pages/BEL116.html) and the Black Hill rain gage (Figure 2). We selected times when the average solar radiation from a 3-h window was less than 1 W/m^2 and no rainfall was recorded for the previous 2 or subsequent 6 h (Kirchner, 2009). We also excluded times when either solar radiation data were missing (7% of the hours) or rainfall data were missing (2% of the hours), as well as 45 h when streamflow increased rapidly with no recorded rainfall as we assumed that these were unrecorded rainfall events. We binned the hourly change in streamflow ($-dQ/dt$) *versus* hourly average streamflow (Q) (Kirchner, 2009) and fitted these binned values using a least squares linear fit. As was performed in the base flow separation approach, we divided the streamflow record into two time periods: during and after urbanization.

RESULTS

Figure 6 shows the annual trends in total flow, slow flow, and quick flow in Soper Branch (forested control), Crystal Rock (urban control), and Tributary 104 (LID) over 2004–2014. Using a Mann–Kendall trend test, there were significant ($p < 0.05$) upward trends for total flow, slow flow, and quick flow in Tributary 104, although not in the ratio of slow flow to total flow (Figure 6). The magnitude of these upward trends was lowest for quick flow (Kendall's tau = 0.6) and highest for total flow (Kendall's tau = 0.8) (Figure 6). From 2004 to 2014, there was a 149% rise in annual slow flow magnitude in Tributary 104 (210 to 522 mm), compared with an 86% increase in the forested control and a 32% increase in the urban control. Climatic variation was a confounding factor because Soper Branch (forested control) also showed significant ($p < 0.05$) upward trends in slow (Kendall's tau = 0.6) and total flow (Kendall's tau = 0.7).

To normalize for climatic variation, we examined slow flow in the study watershed (Tributary 104) relative to slow flow in the control watersheds (Figure 7). Monthly slow flow was used here, resulting in a greater number of data points relative to annual data and allowing separation of the period during urbanization (October 2004–September 2010) from the period after urbanization (October 2010–September 2014). Seasonal Mann–Kendall trend tests on the monthly data of these two periods show that there was a significant ($p < 0.05$) increase in monthly slow flow in the LID study watershed (Tributary 104) relative to both the urban and forested control watersheds during urbanization. Tributary 104 monthly slow flow increased 89% relative to Crystal Rock slow flow over 2004–2010 and 84% relative to Soper Branch slow flow, based on the trend lines shown in Figure 7. After urbanization, slow flow in the study watershed was not significantly changing relative to the forested watershed and was increasing relative to the urban control. The same pattern of significant trends was found using the HYSEP local minimum and HYSEP fixed width separation methods. There was no significant trend in monthly slow flow of the urban control relative to the forested control during urbanization (not shown). Conducting seasonal

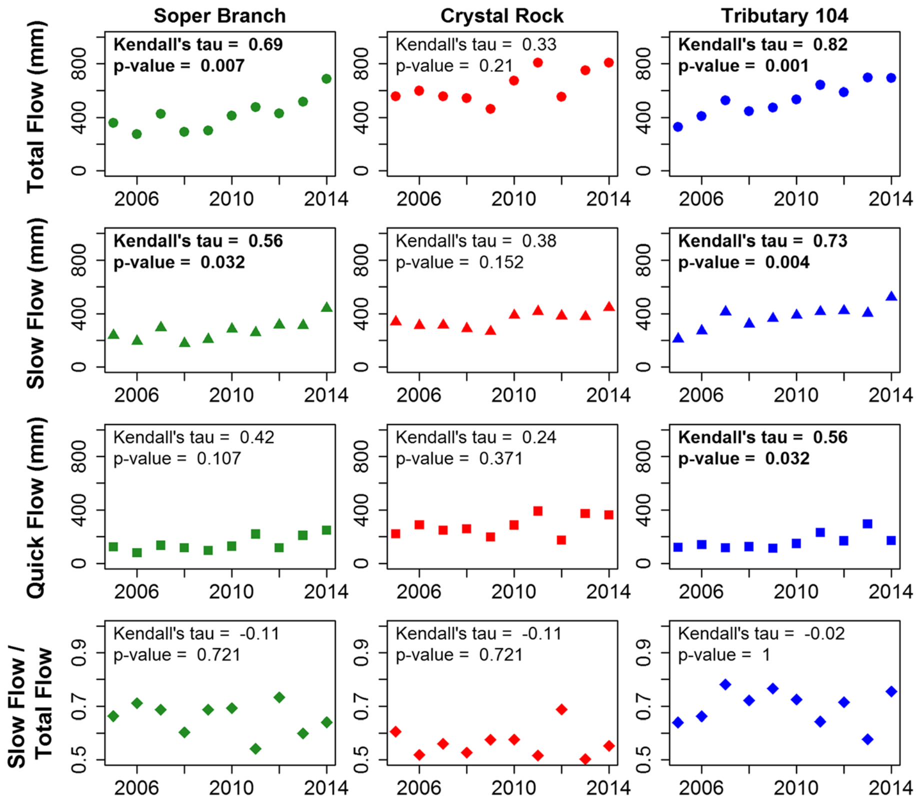

Figure 6. Annual streamflow, slow flow, quick flow, and slow flow as a fraction of total flow (using digital filter separation) from water years 2005–2014 for Soper Branch (forested control), Crystal Rock (urban control), and Tributary 104 (LID watershed). Results from a Mann–Kendall trend test are also shown, where the bolded results indicate values that have a statistically significant (p value <0.05) trend over this time period. Flow is normalized by watershed area to result in values of depth (mm). Water years 2005–2014 spans from 1 October 2004 to 30 September 2014

Mann–Kendall trend tests without separation into two time periods (considering 2004–2014 together) showed that there was a significant rise in slow flow of Tributary 104 relative to Crystal Rock (tau = 0.21 and $p = 0.004$) but not relative to Soper Branch (tau = 0.02 and $p = 0.76$). The fraction of slow flow to total flow did not have a significant trend relative to the control watersheds. Overall, we found that the study LID watershed had a significant rise in slow flow during urbanization and no significant rise after urbanization, relative to the control watersheds.

Trends in total monthly streamflow over time in Tributary 104 relative to the control watersheds (Figure 8) also had a significant positive trend during urbanization. Monthly flow in Tributary 104 increased 58% relative to Crystal Rock and 73% relative to Soper Branch total streamflow based on the trends between 2004 and 2010. After urbanization, there was no significant trend relative to the urban control while relative to the forested control, there was a significant decline. Increasing trends during urbanization were found both for total streamflow (Figure 8) and slow flow (Figure 7), which may not be surprising as over 60% of total flow is made up of slow flow (Figure 6). In contrast to monthly total and slow flow, monthly quick flow does not show a significant trend during either time period relative to the control watersheds (not shown).

We have thus far presented results only on changes in flow magnitude. However, it has been demonstrated that characteristics of the entire flow regime, including magnitude, but also duration, timing, frequency, and variability, are drivers of ecological health (Rolls *et al.*, 2012; Hamel *et al.*, 2015). Therefore, we also examined a measure of slow flow timing, namely the seasonality in slow flow, and later discuss changes in variability, specifically, and rate of change in streamflow. Figure 9 shows the auto-correlation function for the control and study watersheds before and after 1 October 2010. The auto-correlation function shows how slow flow at a given month (e.g. slow flow in October) is correlated to months nearby (e.g. lag of 6 months indicates correlation of slow flow in October with slow flow in April). As we would expect with a strong seasonal base flow cycle in a forested watershed, slow flow in Soper Branch was positively correlated to slow flow 12 months later (because it is the same month) and negatively correlated with slow flow 6 months later (Figure 9). This strong seasonality in slow flow was not evident in the urban control (Crystal Rock), where there was a smaller correlation between slow flow in any given month and those months nearby. In the LID study watershed (Tributary 104), we see that during urbanization (before 1 October 2010), there was strong

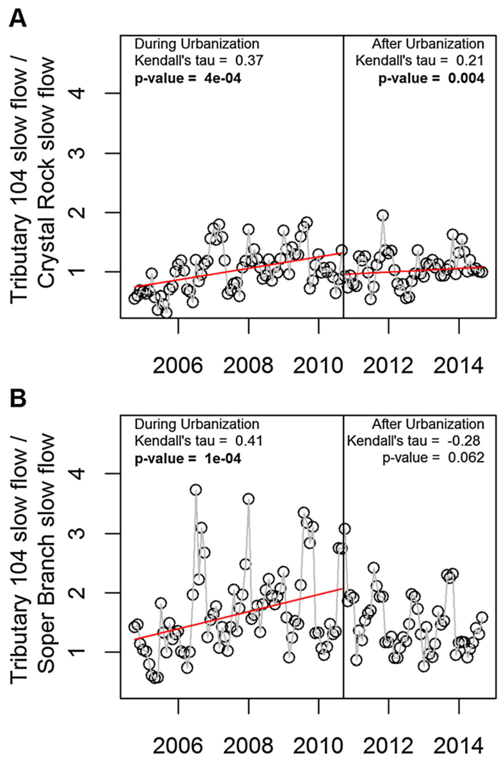

Figure 7. Monthly slow flow (using one-parameter digital filter) in the study watershed normalized by the control watershed, along with results of the seasonal Mann–Kendall trend test during urbanization (1 October 2004–30 September 2010) and after urbanization (1 October 2010–30 September 2014). Where trends are significant (p value <0.05), the p value is bolded and Sen's slopes are also shown. (A) LID watershed (Tributary 104) monthly slow flow divided by urban control (Crystal Rock) monthly slow flow. (B) LID watershed (Tributary 104) monthly slow flow divided by forested control (Soper Branch) monthly slow flow

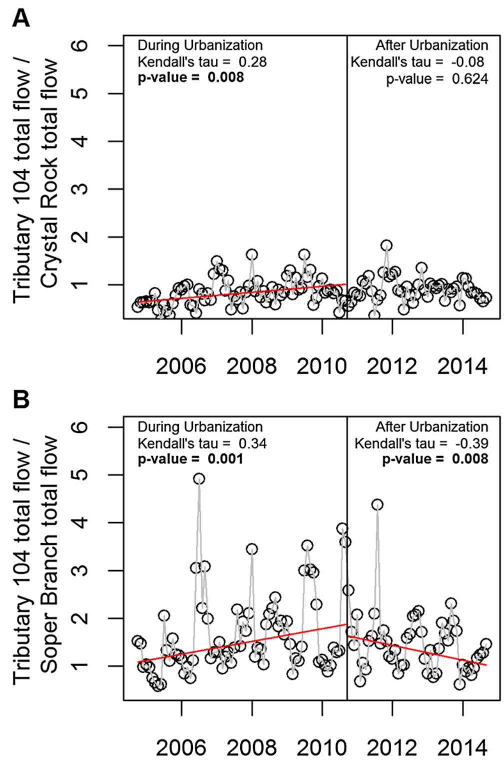

Figure 8. Total streamflow in the LID watershed normalized by control watersheds, along with results of the seasonal Mann–Kendall trend test during urbanization (1 October 2004–30 September 2010) and after urbanization (1 October 2010–30 September 2014). Where trends are significant (p value <0.05), the p value is bolded and Sen's slopes are also shown. (A) LID watershed (Tributary 104) monthly total flow divided by urban control (Crystal Rock) monthly total flow. (B) LID watershed (Tributary 104) monthly total flow divided by forested control (Soper Branch) monthly total flow

seasonality in slow flow and the pattern appeared similar to that in the forested control (Soper Branch). However, after urbanization, there were no monthly lags greater than 3 months that have a positive correlation. Flow seasonality was attenuated post-development and now appears more similar to that of the urban control (Crystal Rock).

The last aspect of the flow regime we examine is rate of change, using recession rates. Recession rates were similar before and after 1 October 2010 in the forested and urban control watersheds, but there was a decrease in the recession rate in the study watershed after urbanization (Figure 10). Testing the equality of the two recession slopes (Larsen and Marx, 2000: 601) using a null hypothesis that the true recession slopes were equal, we find a t-statistic of $2.14 > t_{0.05,65} = 1.997$, allowing us to conclude that the recession slope was significantly different after urbanization in Tributary 104. The slope of the fitted log–log line representing recessions (Figure 10) was smaller after urbanization ($-$dQ/dt [mm/h^2] $= 0.3{*}Q^{2.5}$) than during urbanization ($-$dQ/dt [mm/h^2] $= 3.5{*}Q^{3.5}$) in Tributary 104. The finding that post-development, the hourly decline in streamflow on the falling limb of storm hydrographs was smaller for a given streamflow rate means that in Tributary 104, the recession to pre-stormflow conditions was generally more gradual after urban development.

DISCUSSION

Potential mechanisms of base flow regime alteration

We now interpret our results in light of the three previously discussed mechanisms of base flow regime alteration: (1) removal of vegetation, such that Tributary 104 went from 95% vegetated in 2002 to 68% in 2012 (a decrease of 27%); (2) introduction of impervious surface cover to 30% of watershed area; and (3) drainage of 25% of this impervious surface cover to infiltration-focused stormwater facilities. Our purpose was to observe changes in base flow and total flow magnitude, seasonality, and rate of change during and after urban development as a small watershed was converted from agricultural and forested to suburban land use employing LID stormwater facilities. We found that the study watershed had significant increases in monthly total (Figure 8) and slow

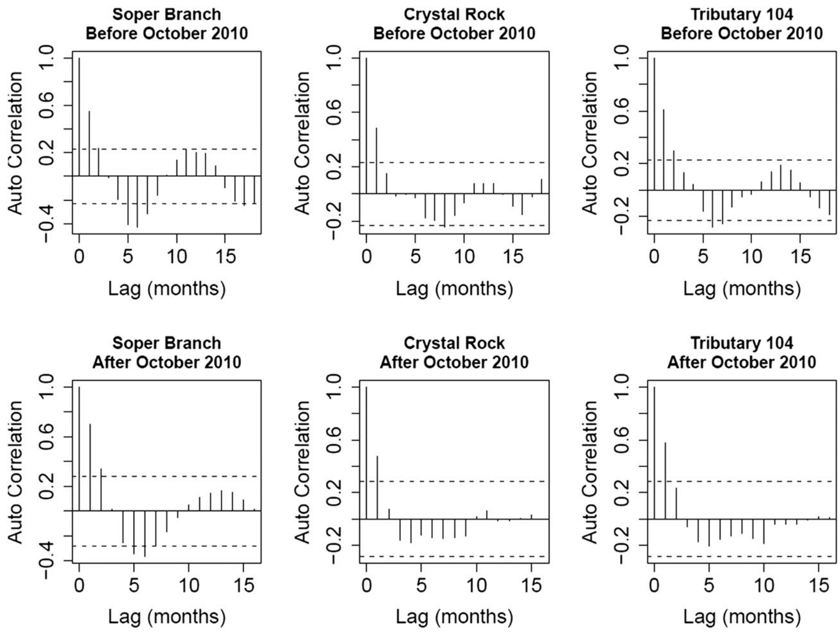

Figure 9. Auto-correlation functions for monthly slow flow (defined by a digital filter) for the forested control (Soper Branch), urban control (Crystal Rock), and LID (Tributary 104) watersheds. The time period has been divided by 1 October 2010, the approximate timing when stormwater infiltration facilities and urban development were completed (refer to Figure 3)

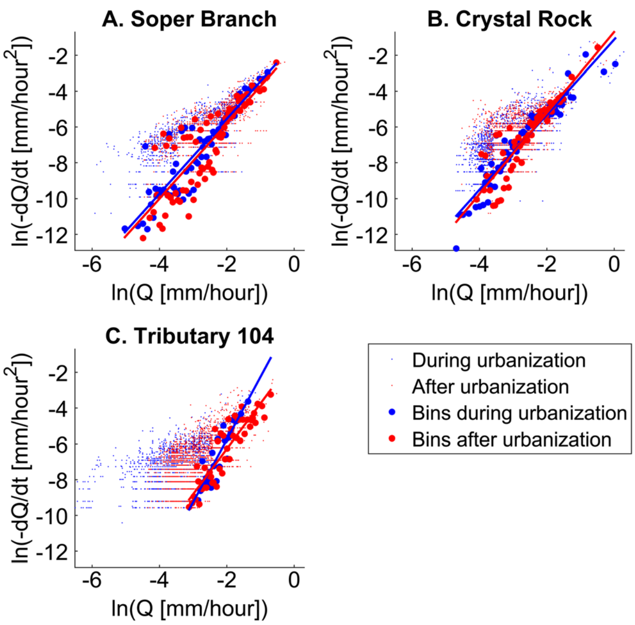

Figure 10. Log–log relation between hourly changes in streamflow recession periods (rainless nighttimes) and streamflow (small dots) for (A) Soper Branch, (B) Crystal Rock, and (C) Tributary 104. Larger dots are binned averages of the small dots. Similar to the procedure laid out in Kirchner (2009), data were binned in linear space but plotted here in log space. The streamflow data were divided into two periods: during urbanization (red; 1 October 2004–30 September 2010) and after urbanization (blue; 1 October 2010–30 September 2014)

flow (Figure 7) (although not quick flow) magnitude relative to control watersheds during urbanization. We also found that seasonality of slow flow was attenuated post-development (Figure 9) and the rate of change on the falling limbs of storm hydrographs (base flow recession) was more gradual post-development in the study watershed (Figure 10). No single proposed mechanism (Figure 1) is able to explain all of these observed changes (Figures 4–10), as we might have expected because multiple mechanisms of base flow alteration occur together during urbanization.

However, we postulate that some mechanisms are more likely to explain the bulk of our results from Tributary 104.

The process of introducing directly connected impervious surfaces alone was expected to reduce base flow magnitude (Figure 1C). As we observed a rise in base flow magnitude, this proposed mechanism is evidently not the dominant process affecting base flow magnitude. The effect of directly connected impervious surfaces might be seen more clearly in stormflow changes however. The impact of impervious surfaces on stormflow appeared smaller than was expected from an urban watershed without LID, as there were no significant trends in monthly quick flow during urbanization relative to the control watersheds. There was a significant increase in annual quick flow from 2004 to 2014, with a rise particularly seen post-development (from 2011 to 2014; Figure 6). Loperfido *et al.* (2014) found that on a storm-event basis, the stormwater benefits of LID were limited to small storms (<27 mm), and stormwater from impervious surfaces was not well controlled for larger events (for example, a higher daily discharge was observed after Tropical Storm Lee in Tributary 104 as compared with Soper Branch and Crystal Rock).

Urbanization with ideally functioning LID (Figure 1A) was not observed, as there were significant trends in multiple components of flow over time. With the process of overcompensating stormwater facilities infiltrating pre-development stormwater (Figure 1D), we had expected to see a rise in base flow magnitude with a mirrored fall in stormflow magnitude compared with pre-development conditions. We observed a rise in base flow, but not a clear decrease in stormflow, and therefore, infiltration of pre-development stormwater is unlikely to be the dominant mechanism of base flow magnitude increase. In terms of alteration to rate of change, we would expect to observe a slower base flow recession with infiltration of pre-development stormflow (Figure 1D), as well as with the other processes that increase base flow magnitude (i.e. those in Figure 1B and Figure 1E). Conventionally, urbanized watersheds were found to have faster base flow recessions in Atlanta, Georgia (Rose and Peters, 2001) and Baltimore, Maryland (Bhaskar and Welty, 2015) compared with less urbanized counterparts, which are the opposite of the slower recession observed after urbanization in Tributary 104. A faster base flow recession was found in a Baltimore watershed even though it was estimated to have a dramatic reduction in ET compared with nearby forested watersheds (Bhaskar and Welty, 2012), and lower ET would be expected to lead to slower recessions (Wittenberg, 2003). Therefore, we postulate that the stormwater facilities, and not a general reduction in ET, were playing a role in slowing base flow recession in Tributary 104. Even detention-focused stormwater facilities might have an effect of slowing stormflow recession on the timescale of hours, and it is not clear here how to separate the effects of different types of stormwater facilities.

The only proposed mechanisms in which base flow magnitude rises along with total flow involve reductions in evapotranspiration, as can be deduced from our water balance formulation of Equation 1. The increase in total flow ($Q_{out,sw}$) in Tributary 104 during urbanization relative to control watersheds with similar precipitation patterns (Figure 8) allows us to assume that Q_{out} is increasing relative to precipitation. With greater infiltration, there also may be increases in other terms on the right hand side of Equation 1, such as greater subsurface storage ($+\Delta S$) or underflow ($Q_{out,gw}$, contributing to a larger overall Q_{out}). Increases in either of these terms, in addition to the rise in $Q_{out,sw}$, would lead to an even larger value on the right hand side of Equation 1, necessitating greater compensation by other water balance components. Groundwater inflow (Q_{in}) is an unlikely compensating term, as there is no clear cause for up-gradient forested and agricultural areas contributing greater groundwater inflow to the study watershed. The two remaining possible compensating terms are decreasing ET or increasing lawn irrigation. In other urban watersheds with similar climatic and geologic settings (Baltimore, Maryland), the reduction in evapotranspiration was estimated to be over 400 mm greater than lawn irrigation (Bhaskar and Welty, 2012), so we assert that the decrease in ET is likely to dwarf lawn irrigation, although both these processes may be occurring.

The proposed mechanisms that would result in decreased evapotranspiration are either overall vegetation removal (Figure 1B) or reduction in ET because of recharge of pre-development ET (Figure 1E). Both of these mechanisms might lead to a decrease in base flow seasonality, as was observed (Figure 9). In addition to an overall lower ET in suburban landscapes, base flow seasonality may be attenuated because of the change in vegetation species, as lawns have greater growth in spring and fall as compared with forests (Peters *et al.*, 2011). Because the changes in fluxes in Figures 1B and 1E are in the same direction as compared with pre-development fluxes (Figure 1A), we cannot clearly distinguish between them. However, the processes at work with vegetation removal (Figure 1B) occur in all urbanizing watersheds, while the overcompensating LID processes (Figures 1D and 1E) are uncommon. Crystal Rock is an urban watershed with conventional stormwater management, where the ratio of base flow to total flow (base flow index) was lower than that of either the forested control or the LID watershed after urbanization (Figure 6). Although streamflow records in Crystal Rock are not available going back to the time of its urbanization, it would not be unreasonable to assume that there was a decrease in the ratio of base flow to total flow in Crystal Rock during urbanization from a more natural state similar to Soper Branch. A decline in base flow index was not observed in Tributary 104 (Figure 6). Therefore, we tentatively suggest that the recharge of pre-development ET through infiltration-focused stormwater management (Figure 1E) is more likely to explain the observed rises in base and total flow than the process of general vegetation removal with urbanization (Figure 1B).

Complicating factors

Legacy of land use change. Alterations to base flow and stream hydrology may be quite different according to previous land use type (Harding *et al.*, 1998). The stream health degradation is likely to be greater where forested land is converted directly to suburban development compared with where agricultural land is converted, because agricultural watersheds already experienced some level of degradation in water quality and hydrology. Because of the diversity in land use legacies, the same base flow increase in response to urban development with LID that was observed here may not occur in even geologically and climatically similar study sites with different land use histories. Furthermore, watersheds are variably vulnerable to changes in base flow because of urban development, based on such factors as location of impervious surfaces relative to recharge areas, pre-development base flow index, and position of the water table (Bhaskar *et al.*, 2016).

Change in watershed area. Across the study time period and area, we consistently used watershed areas defined by topographic boundaries retrieved from the USGS National Water Information System (NWIS). However, in urbanizing watersheds, watershed boundaries can shift over time for multiple reasons. The dramatic topographic alteration of grading and filling has been documented to change the topographic drainage boundary of Tributary 104 (Jones, 2013; Jones *et al.*, 2014). More importantly, however, urban watersheds push us to think beyond our usual definition of topographically defined watershed boundaries and think about the underlying definition of a watershed. If the watershed area is defined as the area over which rain falling will drain to the stream gage, then storm sewersheds need to be considered as well, which may involve storm drains re-routing water across topographic watershed boundaries. The watershed boundaries of Tributary 104 likely changed over our study period because of the topographic change and implementation of storm sewerage during urbanization. Lacking precise information about these changes, we simply compared the watershed area from NWIS based on initial topographic drainage ($1.11\,km^2$) with that derived from storm sewersheds ($1.157\,km^2$) (Sparkman, 2015). The 4% increase in watershed area may have led to greater surface flow to the gage based on a higher drainage area alone, but this effect appears too small to account for the significant increases in total and base flow observed.

Implications for infiltration-focused stormwater infrastructure on a watershed scale

One of the goals of LID in this watershed was to 'maintain stream base flow', and in a strict sense, this goal was achieved. However, a broader goal of LID is to maintain the pre-development water balance and flow regime, which was not observed. In our study watershed, the majority of infiltration-focused stormwater facilities were underground (drywell recharge, storm drain recharge, recharge chambers) rather than vegetated and aboveground (bioretention cells, dry swales). Stormwater facilities that are belowground and un-vegetated limit the potential for evapotranspiration of incoming stormwater and more efficiently recharge stormwater compared with a forested landscape, where more diffuse infiltration may have never made it deeper than the root zone. Recent developments in stormwater quantity management are focused primarily on reducing stormwater volume through infiltration, with little sustained consideration given to collecting stormwater for indoor use or increasing evapotranspiration. The harvesting or evapotranspiration of stormwater in most areas needs to dominate over recharge to maintain pre-development streamflow volumes (Askarizadeh *et al.*, 2015), and these pathways remove flow from the stream altogether rather delaying entry ranging from minutes to years with recharge (Miles and Band, 2015).

SUMMARY

We tracked annual and monthly changes in total streamflow, base flow, and stormflow in a small watershed converted from agricultural and forested cover to 30% impervious surface cover using low impact development stormwater facilities. We compared the changes in streamflow from 2004 to 2014 with changes in nearby control forested and urban watersheds. Our findings can be summarized as follows:

1. Total streamflow increased during urbanization in the LID watershed relative to the control watersheds, by 58% relative to an urban control watershed and 73% relative to a forested control, between 2004 and 2010 (Figure 8).
2. Base flow also increased during urbanization in the LID watershed relative to the control watersheds, by 89% relative to the urban control watershed and 84% relative to the forested control watershed (Figure 7).
3. Seasonality of base flow during urbanization was similar to the forested control watershed and was

attenuated and more similar to the urban control watershed post-development (Figure 9).

4. Base flow recession was more gradual post-development in the LID watershed (Figure 10).
5. A singular cause for these hydrologic alterations cannot be identified using streamflow data alone. However, these observations do not indicate that the infiltration-focused stormwater facilities are infiltrating only excess stormwater generated by new impervious surfaces, because the hydrologic budget would be unchanged post-development if this was the case (Figure 1A). The observations are also not consistent with greater recharge of water that would have been stormflow in a forested watershed (Figure 1D), because this would imply an increase in base flow, which was observed, along with a mirrored decrease in stormflow, which was not observed. Our observations are consistent with urbanization processes that reduce evapotranspiration, such as recharge of water that would have been stored in soil moisture and eventually evaporated or taken up by plants (Figure 1E) or removal of vegetative cover when urbanizing the watershed (Figure 1B).

Our work indicates that in addition to considering the eventual fate of stormwater entering infiltration-based stormwater facilities, we might also consider the pre-development fate of that water. Recharging stormwater might only be beneficial ecologically up to the proportion of that stormwater that would have been recharged pre-development. Not all impervious surface run-off should be recharged if most would have evaporated or been taken up by plants before urbanization, and our ecological objective is to preserve the natural flow regime.

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REFERENCE

Appleyard S. 1995. The impact of urban development on recharge and groundwater quality in a coastal aquifer Near Perth, Western Australia. *Hydrogeology Journal* **3**(2): 65–75. DOI:10.1007/s100400050072.

Askarizadeh A, Rippy MA, Fletcher TD, Feldman D, Peng J, Bowler P, Mehring A, Winfrey B, Vrugt J, AghaKouchak A, *et al.* 2015. From rain tanks to catchments: use of low-impact development to address hydrologic symptoms of the urban stream syndrome. *Environmental Science & Technology*. DOI:10.1021/acs.est.5b01635.

Barlow P, Cunningham WL, Gray M. 2015. U.S. Geological Survey Groundwater Toolbox, A Graphical and Mapping Interface for Analysis of Hydrologic Data (Version 1.0)—User Guide for Estimation of Base Flow, Runoff, and Groundwater Recharge From Streamflow Data. Techniques and Methods.

Barringer TH, Reiser RG, Price CV. 1994. Potential effects of development on flow characteristics of two New Jersey streams. *Journal of the American Water Resources Association* **30**(2): 283–295. DOI:10.1111/j.1752-1688.1994.tb03291.x.

Barron OV, Barr AD, Donn MJ. 2013. Effect of urbanisation on the water balance of a catchment with shallow groundwater. *Journal of Hydrology* **485**: 162–176. DOI:10.1016/j.jhydrol.2012.04.027.

Bedan ES, Clausen JC. 2009. Stormwater runoff quality and quantity from traditional and low impact development watersheds. *Journal of the American Water Resources Association* **45**(4): 998–1008. DOI:10.1111/j.1752-1688.2009.00342.x.

Bhaskar AS, Welty C. 2012. Water balances along an urban-to-rural gradient of Metropolitan Baltimore, 2001–2009. *Environmental & Engineering Geoscience* **18**(1): 37–50. DOI:10.2113/gseegeosci.18.1.37.

Bhaskar AS, Welty C. 2015. Analysis of subsurface storage and streamflow generation in urban watersheds. *Water Resources Research* **51**(3): 1493–1513. DOI:10.1002/2014WR015607.

Bhaskar AS, Beesley L, Burns MJ, Fletcher TD, Hamel P, Oldham CE, Roy AH. 2016. Will it rise or will it fall? Managing the complex effects of urbanization on base flow. *Freshwater Science* **35**(1): 293–310. DOI:10.1086/685084.

Booth DB, Jackson CR. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. *JAWRA Journal of the American Water Resources Association* **33**(5): 1077–1090. DOI:10.1111/j.1752-1688.1997.tb04126.x.

Brandes D, Cavallo GJ, Nilson ML. 2005. Base flow trends in urbanizing watersheds of the Delaware River Basin. *Journal of the American Water Resources Association* **41**(6): 1377–1391.

Brutsaert W, Nieber JL. 1977. Regionalized drought flow hydrographs from a mature glaciated plateau. *Water Resources Research* **13**(3): 637–643. DOI:10.1029/WR013i003p00637.

Burns MJ, Fletcher TD, Walsh CJ, Ladson AR, Hatt BE. 2012. Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning* **105**(3): 230–240. DOI:10.1016/j.landurbplan.2011.12.012.

Carleton GB. 2010. Simulation of groundwater mounding beneath hypothetical stormwater infiltration basins. U.S. Geological Survey Scientific Investigations Report 2010–5102.

Chang H. 2007. Comparative streamflow characteristics in urbanizing basins in the Portland Metropolitan Area, Oregon, USA. *Hydrological Processes* **21**(2): 211–222. DOI:10.1002/hyp.6233.

Davis A. 2008. Field performance of bioretention: hydrology impacts. *Journal of Hydrologic Engineering* **13**(2): 90–95. DOI:10.1061/(ASCE)1084-0699(2008)13:2(90).

Dietz ME, Clausen JC. 2008. Stormwater runoff and export changes with development in a traditional and low impact subdivision. *Journal of Environmental Management* **87**(4): 560–566. DOI:10.1016/j.jenvman.2007.03.026.

Dow CL, DeWalle DR. 2000. Trends in evaporation and Bowen Ratio on urbanizing watersheds in eastern United States. *Water Resources Research* **36**(7): 1835. DOI:10.1029/2000WR900062.

Emerson CH, Welty C, Traver RG. 2005. Watershed-scale evaluation of a system of storm water detention basins. *Journal of Hydrologic Engineering* **10**(3): 237. DOI:10.1061/(ASCE)1084-0699(2005)10:3(237).

Endreny T, Collins V. 2009. Implications of bioretention basin spatial arrangements on stormwater recharge and groundwater mounding.

Ecological Engineering **35**(5): 670–677. DOI:10.1016/j.ecoleng.2008.10.017.

Finkenbine JK, Atwater JW, Mavinic DS. 2000. Stream health after urbanization. *Journal of the American Water Resources Association* **36**(5): 1149–1160. DOI:10.1111/j.1752-1688.2000.tb05717.x.

Fletcher TD, Shuster W, Hunt WF, Ashley R, Butler D, Arthur S, Trowsdale S, Barraud S, Semadeni-Davies A, Bertrand-Krajewski J-L, et al. 2014. SUDS, LID, BMPsWSUD and more—the evolution and application of terminology surrounding urban drainage. *Urban Water Journal* **1–18**. DOI:10.1080/1573062X.2014.916314.

Gilroy KL, McCuen RH. 2009. Spatio-temporal effects of low impact development practices. *Journal of Hydrology* **367**(3–4): 228–236. DOI:10.1016/j.jhydrol.2009.01.008.

Göbel P, Stubbe H, Weinert M, Zimmermann J, Fach S, Dierkes C, Kories H, Messer J, Mertsch V, Geiger WF, et al. 2004. Near-natural stormwater management and its effects on the water budget and groundwater surface in urban areas taking account of the hydrogeological conditions. *Journal of Hydrology* **299**(3–4): 267–283.

Haase D. 2009. Effects of urbanisation on the water balance—a long-term trajectory. *Environmental Impact Assessment Review* **29**(4): 211–219. DOI:10.1016/j.eiar.2009.01.002.

Hamel P, Fletcher TD. 2014. Modelling the impact of stormwater source control infiltration techniques on catchment baseflow. *Hydrological Processes* **28**(24): 5817–5831. DOI:10.1002/hyp.10069.

Hamel P, Daly E, Fletcher TD. 2013. Source-control stormwater management for mitigating the impacts of urbanisation on baseflow: a review. *Journal of Hydrology* **485**: 201–211. DOI:10.1016/j.jhydrol.2013.01.001.

Hamel P, Daly E, Fletcher TD. 2015. Which baseflow metrics should be used in assessing flow regimes of urban streams? *Hydrological Processes*. DOI:10.1002/hyp.10475.

Harding JS, Benfield EF, Bolstad PV, Helfman GS, Jones EBD. 1998. Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Sciences* **95**(25): 14843–14847.

Healy RW. 2010. *Estimating groundwater recharge*. Cambridge University Press: Cambridge, UK.

Hirsch RM, Slack JR, Smith RA. 1982. Techniques of trend analysis for monthly water quality data. *Water Resources Research* **18**(1): 107–121.

Hogan D, Jarnagin ST, Loperfido JV, Van Ness K. 2014. Mitigating the effects of landscape development on streams in urbanizing watersheds. *Journal of the American Water Resources Association* **50**(1): 163–178. DOI:10.1111/jawr.12123.

Holman-Dodds JK, Bradley AA, Potter KW. 2003. Evaluation of hydrologic benefits of infiltration based urban storm water management. *Journal of the American Water Resources Association* **39**(1): 205–215. DOI:10.1111/j.1752-1688.2003.tb01572.x.

Hood MJ, Clausen JC, Warner GS. 2007. Comparison of stormwater lag times for low impact and traditional residential development. *Journal of the American Water Resources Association* **43**(4): 1036–1046. DOI:10.1111/j.1752-1688.2007.00085.x.

Hopkins KG, Morse NB, Bain DJ, Bettez ND, Grimm NB, Morse JL, Palta MM, Shuster WD, Bratt AR, Suchy AK. 2015. Assessment of regional variation in streamflow responses to urbanization and the persistence of physiography. *Environmental Science & Technology* **49**(5): 2724–2732. DOI:10.1021/es505389y.

Hubbart JA, Zell C. 2013. Considering streamflow trend analyses uncertainty in urbanizing watersheds: a baseflow case study in the central United States. *Earth Interactions* **17**(5): 1–28. DOI:10.1175/2012EI000481.1.

Jones DK. 2013. Examining development-induced geomorphic change using multi-temporal LiDAR-derived digital elevation models. University of Maryland, Baltimore County. Available at: http://gradworks.umi.com/15/38/1538460.html [Accessed 22 July 2015]

Jones DK, Baker ME, Miller AJ, Jarnagin ST, Hogan DM. 2014. Tracking geomorphic signatures of watershed suburbanization with multitemporal LiDAR. *Geomorphology* **219**: 42–52. DOI:10.1016/j.geomorph.2014.04.038.

Keßler S, Meyer B, Seeling S, Tressel E, Krein A. 2012. Influence of near-to-nature stormwater management on the local water balance using the example of an urban development area. *Water Environment Research* **84**(5): 441–451. DOI:10.2175/106143012X13347678384729.

Kirchner JW. 2009. Catchments as simple dynamical systems: catchment characterization, rainfall-runoff modeling, and doing hydrology backward. *Water Resources Research* **45** (2): W02429.

Ku HFH, Hagelin NW, Buxton HT. 1992. Effects of urban storm-runoff control on ground-water recharge in Nassau County, New York. *Ground Water* **30**(4): 507–514. DOI:10.1111/j.1745-6584.1992.tb01526.x.

Larsen RJ, Marx ML. 2000. *An introduction to mathematical statistics and its applications*. Prentice Hall: Upper Saddle River, NJ.

Leopold LB. 1968. *Hydrology for urban land planning—a guidebook on the hydrologic effects of urban land use*. Circular 554. U.S. Geological Survey: Washington, D.C.

Lim KJ, Engel BA, Tang Z, Choi J, Kim K-S, Muthukrishnan S, Tripathy D. 2005. Automated Web GIS Based Hydrograph Analysis Tool, WHAT. *Journal of the American Water Resources Association* **41**(6): 1407–1416. DOI:10.1111/j.1752-1688.2005.tb03808.x.

Lindsey G, Roberts L, Page W. 1992. Inspection and maintenance of infiltration facilities. *Journal of Soil and Water Conservation* **47**(6): 481–486.

Liu G, Schwartz FW, Kim Y. 2013. Complex baseflow in urban streams: an example from central Ohio, USA. *Environmental Earth Sciences* **70**(7): 3005–3014. DOI:10.1007/s12665-013-2358-3.

Loperfido JV, Noe GB, Jarnagin ST, Hogan DM. 2014. Effects of distributed and centralized stormwater best management practices and land cover on urban stream hydrology at the catchment scale. *Journal of Hydrology* **519**: 2584–2595. DOI:10.1016/j.jhydrol.2014.07.007.

Machusick M, Welker A, Traver R. 2011. Groundwater mounding at a storm-water infiltration BMP. *Journal of Irrigation and Drainage Engineering* **137**(3): 154–160. DOI:10.1061/(ASCE)IR.1943-4774.0000184.

Marchetto A. 2015. Mann-Kendall Test, Seasonal and Regional Kendall Tests. Available at: https://cran.r-project.org/web/packages/rkt/rkt.pdf.

Meyer JL, Paul MJ, Taulbee WK. 2005. Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society* **24**(3): 602–612. DOI:10.1899/04-021.1.

Meyer SC. 2005. Analysis of base flow trends in urban streams, northeastern Illinois, USA. *Hydrogeology Journal* **13**(5–6): 871–885. DOI:10.1007/s10040-004-0383-8.

Miles B, Band LE. 2015. Green infrastructure stormwater management at the watershed scale: urban variable source area and watershed capacitance. *Hydrological Processes*. DOI:10.1002/hyp.10448.

Montgomery County DEP. 2012. Special Protection Area Program Annual Report 2010. Montgomery County, MD. Available at: https://www.montgomerycountymd.gov/DEP/Resources/Files/downloads/water-reports/spa/2010_spa_report.pdf

Nathan RJ, McMahon TA. 1990. Evaluation of automated techniques for base flow and recession analyses. *Water Resources Research* **26**(7): 1465–1473. DOI:10.1029/WR026i007p01465.

NRC. 2009. *Urban stormwater management in the United States*. The National Academies Press: Washington, D.C.

O'Driscoll M, Clinton S, Jefferson A, Manda A, McMillan S. 2010. Urbanization effects on watershed hydrology and in-stream processes in the southern United States. *Water* **2**(3): 605–648. DOI:10.3390/w2030605.

Perez-Pedini C, Limbrunner J, Vogel R. 2005. Optimal location of infiltration-based best management practices for storm water management. *Journal of Water Resources Planning and Management* **131**(6): 441–448. DOI:10.1061/(ASCE)0733-9496(2005)131:6(441).

Peters EB, Hiller RV, McFadden JP. 2011. Seasonal contributions of vegetation types to suburban evapotranspiration. *Journal of Geophysical Research, Biogeosciences* **116**(G1): G01003 DOI: 10.1029/2010JG001463

Pluhowski EJ, Spinello AG. 1978. Impact of sewerage systems on stream base flow and ground-water recharge on Long Island, New York. U.S. *Geological Survey Journal of Research* **6**(2): 263–271.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime. *BioScience* **47**(11): 769–784. DOI:10.2307/1313099.

Poff NL, Richter BD, Arthington AH, Bunn SE, Naiman RJ, Kendy E, Acreman M, Apse C, Bledsoe BP, Freeman MC, et al. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* **55**(1): 147–170. DOI:10.1111/j.1365-2427.2009.02204.x.

Price K. 2011. Effects of watershed topography, soils, land use, and climate on baseflow hydrology in humid regions: a review. *Progress in Physical Geography* **35**(4): 465–492. DOI:10.1177/0309133311402714.

Prince George's County. 1999. Low-impact development hydrologic analysis Available at: http://water.epa.gov/polwaste/green/upload/lidnatl.pdf.
R Core Team. 2014. *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing: Vienna, Austria. Available at: http://www.R-project.org/.
Rhea L, Jarnagin T, Hogan D, Loperfido JV, Shuster W. 2015. Effects of urbanization and stormwater control measures on streamflows in the vicinity of Clarksburg, Maryland, USA. *Hydrological Processes*. DOI:10.1002/hyp.10505.
Risser DW, Gburek WJ, Folmar GJ. 2005. Comparison of methods for estimating ground-water recharge and base flow at a small watershed underlain by fractured bedrock in the eastern United States. US Department of the Interior, US Geological Survey. Available at: http://pubs.usgs.gov/sir/2005/5038/ [Accessed 26 June 2015]
Rolls RJ, Leigh C, Sheldon F. 2012. Mechanistic effects of low-flow hydrology on riverine ecosystems: ecological principles and consequences of alteration. *Freshwater Science* **31**(4): 1163–1186.
Rose S, Peters NE. 2001. Effects of urbanization on streamflow in the Atlanta area (Georgia, USA): a comparative hydrological approach. *Hydrological Processes* **15**(8): 1441–1457. DOI:10.1002/hyp.218.
Roy AH, Dybas AL, Fritz KM, Lubbers HR. 2009. Urbanization affects the extent and hydrologic permanence of headwater streams in a midwestern US metropolitan area. *Journal of the North American Benthological Society* **28**(4): 911–928. DOI:10.1899/08-178.1.
Roy AH, Wenger SJ, Fletcher TD, Walsh CJ, Ladson AR, Shuster WD, Thurston HW, Brown RR. 2008. Impediments and solutions to sustainable, watershed-scale urban stormwater management: lessons from Australia and the United States. *Environmental Management* **42**(2): 344–359. DOI:10.1007/s00267-008-9119-1.
Schwartz SS, Smith B. 2014. Slowflow fingerprints of urban hydrology. *Journal of Hydrology* **515**: 116–128. DOI:10.1016/j.jhydrol.2014.04.019.
Selbig WR, Bannerman RT. 2008. A comparison of runoff quantity and quality from two small basins undergoing implementation of conventional and Low-Impact-Development (LID) strategies: Cross Plains, Wisconsin, Water Years 1999–2005. U.S. Geological Survey Scientific Investigations Report 2008–5008.
Sloto RA, Crouse MY. 1996. *HYSEP, a computer program for streamflow hydrograph separation and analysis*. US Department of the Interior, US Geological Survey. Available at: http://pubs.usgs.gov/wri/1996/4040/report.pdf [Accessed 10 August 2015].
Smakhtin VU. 2001. Low flow hydrology: a review. *Journal of Hydrology* **240**(3–4): 147–186. DOI:10.1016/S0022-1694(00)00340-1.
Sophocleous M. 2002. Interactions between groundwater and surface water: the state of the science. *Hydrogeology Journal* **10**(1): 52–67. DOI:10.1007/s10040-001-0170-8.
Southworth S, Brezinski DK, Burton WC, Froelich AJ, Reddy JE, Denenny D, Daniels DL. 2007. Geologic Map of the Frederick 30′ X 60′ Quadrangle, Maryland, Virginia, and West Virginia Available at: http://pubs.usgs.gov/sim/2889/ [Accessed 26 May 2015]
Sparkman S. 2015. Effects of spatial distribution of stormwater management infrastructure on pollutant removal efficiencies at the watershed scale: a geographic information systems approach. M.S. Thesis George Mason University, Fairfax, Virginia. Available at: http://mars.gmu.edu/handle/1920/9761 [Accessed 11 September 2015].
Stephens DB, Miller M, Moore SJ, Umstot T, Salvato DJ. 2012. Decentralized groundwater recharge systems using roofwater and stormwater runoff. *Journal of the American Water Resources Association* **48**(1): 134–144. DOI:10.1111/j.1752-1688.2011.00600.x.
Thomas BF, Vogel RM. 2012. Impact of storm water recharge practices on boston groundwater elevations. *Journal of Hydrologic Engineering* **17**(8): 923–932. DOI:10.1061/(ASCE)HE.1943-5584.0000534.
Townsend-Small A, Pataki DE, Liu H, Li Z, Wu Q, Thomas B. 2013. Increasing summer river discharge in southern California USA linked to urbanization. *Geophysical Research Letters*. DOI:10.1002/grl.50921.
US EPA. 2000. Low Impact Development (LID) Literature Review and Fact Sheets. US EPA Document # EPA-841-B-00-005. Available at: http://water.epa.gov/polwaste/green/lidlit.cfm [Accessed 7 July 2015]
Walker KF, Sheldon F, Puckridge JT. 1995. A perspective on dryland river ecosystems. *Regulated Rivers-Research and Management* **11**(1): 85–104.
Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP II. 2005. The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society* **24**(3): 706–723.
Wild T, Davis A. 2009. Simulation of the performance of a storm-water BMP. *Journal of Environmental Engineering* **135**(12): 1257–1267. DOI:10.1061/(ASCE)EE.1943-7870.0000106.
Williams ES, Wise WR. 2006. Hydrologic impacts of alternative approaches to storm water management and land development. *Journal of the American Water Resources Association* **42**(2): 443–455. DOI:10.1111/j.1752-1688.2006.tb03849.x.
Wittenberg H. 2003. Effects of season and man-made changes on baseflow and flow recession: case studies. *Hydrological Processes* **17**(11): 2113–2123. DOI:10.1002/hyp.1324.

