Lessons learned from 20 y of monitoring suburban development with distributed stormwater management in Clarksburg, Maryland, USA

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Abstract: Urban development is a well-known stressor for stream ecosystems, presenting a challenge to managers tasked with mitigating its effects. For the past 20 y, streamflow, water quality, geomorphology, and benthic communities were monitored in 5 watersheds in Montgomery County, Maryland, USA. This study presents a synthesis of multiple studies of monitoring efforts in the study area and new analysis of more recent monitoring data to document the primary lessons learned from monitoring. The monitored watersheds include a forested control, an urban control with centralized stormwater management, and 3 suburban treatment watersheds featuring low-impact development and a high density of infiltration-focused stormwater facilities distributed across the watershed. Treatment watersheds were monitored before development, during construction, and after development. Monitoring was initiated to inform adaptive management of stormwater and impervious cover limits within the study area, with a focus on the impacts of distributed stormwater management. Results from our synthesis indicate that distributed stormwater management is advantageous compared with centralized stormwater management in numerous ways. Hydrologic benefits were greater with distributed stormwater infrastructure, demonstrating the ability to mitigate runoff volumes and peak flows and, for small storms, replicate predevelopment conditions. Baseflow temporarily increased during the construction phase in the treatment watersheds. Water-quality benefits were mixed, with declines in baseflow nitrate concentrations but limited changes to nitrate export and increases in specific conductance after development. Substantial topographic changes occurred during construction in the treatment watersheds, including changes within the riparian zone, despite riparian buffer protections. Ecological monitoring indicated that even though index of biotic integrity scores rebounded in some cases, sensitive benthic macroinvertebrate families did not fully recover in the treatment watersheds. Lessons learned from this synthesis highlight the importance of tracking multiple indicators of stream health and considering past land use and that more stormwater facilities distributed across the watershed is beneficial but cannot mitigate the effects of all urban stressors on aquatic ecosystems.

Key words: urban, suburban, impervious surfaces, stormwater, green infrastructure, low-impact development, best management practices, stream health, flow regime, water quality, geomorphology, benthic macroinvertebrates

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Urban and suburban development continue to expand globally. In the United States of America (USA) between 1990 and 2010, populations grew and impervious surface cover expanded in almost every major metropolitan area, with the greatest growth along major transportation corridors between Atlanta, Georgia, and Washington, DC (Falcone et al. 2018). Urban and suburban development leads to consistent declines in the physical, ecological, and biological health of streams, collectively termed urban stream syndrome (Meyer et al. 2005, Walsh et al. 2005, Cunningham et al. 2009). For example, ~53% of streams in the Piedmont located in the southeastern USA are expected to experience >25% loss in sensitive invertebrate taxa by the y 2030 because of suburban sprawl and infill development along highway corridors (Van Metre et al. 2019). For headwater streams in the Piedmont within the Chesapeake Bay watershed, impervious cover is the most important predictor of poor benthic macroinvertebrate condition (Maloney et al. 2018).

Mitigating the impacts of development on stream health is primarily accomplished by minimizing impervious surfaces and installing stormwater facilities. Stormwater facilities, such as green stormwater infrastructure that use vegetation and soils to capture, infiltrate, transpire, and filter stormwater, are increasingly used to address stormwater challenges (McPhillips and Matsler 2018). Other engineered stormwater facilities, such as water-quality filters, sand filters, infiltration trenches, and porous pavement, are also being implemented to reduce runoff and improve downstream water quality. These types of facilities, when implemented densely within a neighborhood, are sometimes referred to as low-impact development, water-sensitive urban design, or sustainable urban drainage systems (Fletcher et al. 2015).

There are 2 general arrangements of stormwater facilities: 1) centralized infrastructure, with 1 large facility to treat stormwater from a large drainage area; and 2) distributed infrastructure, with smaller facilities treating stormwater close to the source. Distributed stormwater facilities can be stand-alone practices or placed in treatment trains, with a series of facilities feeding into each other. The types and placement of stormwater facilities influence the effectiveness of stormwater treatment, which has been demonstrated in many modeling studies (Bell et al. 2020), but there are few corresponding empirical studies on a watershed scale (Jefferson et al. 2017, Li et al. 2017). The few empirical studies that do assess distributed stormwater management have found reduced runoff volumes, water-quality pollutants, or both compared with traditional development with limited or centralized stormwater management (Dietz and Clausen 2008, Page et al. 2015, Wilson et al. 2015, Jarden et al. 2016, Woznicki et al. 2018). These studies assessed impacts of distributed stormwater management at the scale of a single street or neighborhood block (<0.1 km²), but the cumulative effects of distributed stormwater management at larger watershed scales (>1 km²) remain limited, especially when including ecological responses in receiving streams. Demonstrated effects on a watershed scale for hydrologic regulation, water quality, and stream health are necessary to inform more widespread use of distributed stormwater facilities.

Monitoring efforts over the past 20 y in Clarksburg, Maryland, USA, provide a unique opportunity to assess how ecological function changes during construction and after watershed-scale (>1 km²) implementation of distributed stormwater facilities. More stringent development regulations were established in a portion of this area in the late 1990s. The Montgomery County Planning Board and the County Department of Environmental Protection began to use watershed-scale distributed stormwater facilities in Clarksburg to monitor ecological impacts and inform future development requirements (MCDPP 1994). Stormwater regulations required new development to use environmental site design to the greatest extent practicable in an effort to replicate the hydrology of “woods in good condition” (MDE 2000 chapter 5, page 17). Environmental site design, which uses principles similar to those of low-impact development, includes minimizing impervious surfaces, limiting large-scale grading during construction, adding riparian buffer protection, and installing infiltration-focused stormwater facilities (e.g., sand filters, infiltration trenches, recharge chambers) distributed throughout the watershed.

The aim of this study was to synthesize the lessons learned from existing studies and long-term monitoring data on the hydrologic, ecological, geomorphic, and water-quality impacts of suburban development with distributed stormwater facilities arranged in treatment trains in Clarksburg. This study provides a summary of past research in the study area, supplements those studies with additional monitoring data when possible, and interprets the findings in the context of distributed stormwater management impacts on ecological function. We addressed the following questions for this synthesis: how does the use of distributed stormwater facilities on a watershed scale in Clarksburg affect 1) peak flows and runoff volumes; 2) baseflow; 3) water quality, including pollutants and ionic concentrations; 4) topography and drainage patterns; and 5) the benthic macroinvertebrate assemblage.

**METHODS**

Our synthesis examined all published studies that focused on 1 or more of 5 watersheds located in Clarksburg, Montgomery County, Maryland, USA (Fig. 1A, B). We identified studies through a literature search on Google Scholar and through the authors’ knowledge of studies in the area. Results from these studies were used to develop lessons learned for this synthesis. We also supplemented previous studies with more recent datasets on streamflow, water quality (specific conductance [SC]), topographic change, and benthic macroinvertebrate counts to expand on and refine lessons learned.
Study area

The study area includes 5 watersheds located in Clarksburg, Montgomery County, Maryland, USA (Fig. 1A, B). Portions of Clarksburg were designated as a Special Protection Area in 1994, requiring additional levels of stormwater management for new development in areas with high-quality water resources and where land-use change would threaten these resources (MCDEP 2001). The region is experiencing substantial new suburban development due to its proximity to large metropolitan areas of Washington, DC, and Baltimore, Maryland (Fig. 1A–D).

The study design includes 3 treatment watersheds within the Special Protection Area that transitioned from predominantly row-crop agricultural to suburban development with a high density of distributed stormwater facilities (Fig. 1A–D). South of the treatment watersheds is an urban control watershed which provides a comparison with 1980s development practices with predominantly centralized detention-based stormwater facilities, typically detention ponds. North of the treatment watersheds is a forested reference watershed, which is primarily a county park and the least disturbed of the monitored watersheds. The study area, including the
forested control, has a legacy of land clearing for row-crop agriculture dating back to the early 1900s. Monitored watersheds range in size from 0.9 to 3.4 km². The climate is humid continental with hot summers and cold winters, with mean annual precipitation (2006–2020) of 1235 mm, and mean annual temperature of 12.4°C (National Oceanic and Atmospheric Administration, https://www.ncei.noaa.gov/access/us-climate-normals/#dataset = normals-monthly&timeframe = 15). The study area is in the crystalline Piedmont Plateau province with rolling uplands underlain by siltstone and quartzite and with a number of incised streams (Reger and Cleaves 2008).

The study was designed to examine and monitor changes in watershed function as the 3 treatment watersheds underwent suburban development with a high density of stormwater facilities and compared the treatment watersheds with the urban control and forested control (Fig. 1A–D). We monitored conditions in the treatment watersheds during 3 phases: 1) predevelopment, 2) construction, and 3) developed. The construction phase of development in all 3 treatment watersheds required the use of sediment and erosion control facilities that included sediment basins with forebays, series of multiple basins, super silt fences, and 125% of the normal storage volumes for sediment traps and forebays (MDE 2011). Sediment and erosion control basins were converted to stormwater facilities (sand filters and dry ponds) when construction was completed. All 3 treatment watersheds were developed with stormwater facilities that treat 100% of the impervious area within the watershed. Stormwater facilities are primarily arranged in treatment trains with facilities in series that terminate into a sand filter, which drains to a dry pond that discharges to a stream. Most stormwater facilities in the treatment watersheds were designed to mitigate peak flows for a 2.54-cm (1-in) precipitation event or a 66-mm, 1-y, 24-h event (2.6 in) for dry detention basins (MDE 2000). Stormwater facilities in Treatment 1 primarily consist of recharge chambers and infiltration trenches, and development is primarily single-family detached homes. Treatment 2 has twice the stormwater facility density of Treatment 1 because facilities are smaller, consisting mostly of tree-box filters, infiltration trenches, and underground detention structures, and development is a mix of single-family detached housing and attached townhouses (Table 1). Treatment 3 was undergoing development at the time of this synthesis. Development in Treatment 3 will include a mix of detached single-family homes and attached townhouses, with distributed stormwater facilities throughout the watershed.

### Monitoring framework

Monitoring efforts within the study area occurred as part of the Clarksburg Integrated Study Partnership, which included scientists from the Montgomery County Department of Environmental Protection (MCDEP), the United States Geological Survey (USGS), the United States Environmental Protection Agency, and the University of Maryland. Monitoring goals were to evaluate changes to the health of streams in the study area by assessing hydrologic change, common water-quality parameters (e.g., nutrients, SC), topographic change through light detection and ranging (LiDAR), and benthic macroinvertebrate assemblage composition.

Streamflow, water quality, and benthic macroinvertebrate monitoring have been ongoing in the study area since the early 2000s. The USGS began monitoring streamflow at 5 locations in the study area in 2004 to assess hydrologic changes in the treatment watersheds compared with the 2 control watersheds (Fig. 1A–D). Stream discharge was collected at 5- or 15-min intervals (USGS, https://doi.org/10.5066/F7P55KJN). In this synthesis, we used streamflow data to supplement the analysis by Bhaskar et al. (2016b) with an

| Site name          | USGS stream gage number, name, location | Watershed area (km²) | AL 1959 (%) | IC 2002 (%) | IC 2017 (%) | SWFD 2019 (facility/km²) | Housing style                    |
|--------------------|----------------------------------------|----------------------|-------------|-------------|-------------|-------------------------|---------------------------------|
| Forested control   | 01643395, Soper Branch near Hyattstown, MD | 3.4                  | 45          | 2           | 2           | 2                       | Not applicable                   |
| Urban control      | 01644375, Little Seneca Creek Trib near Germantown, MD | 3.1                  | 82          | 38          | 40          | 47                      | Commercial and single family detached |
| Treatment 1        | 01644371, Little Seneca Creek Trib near Clarksburg, MD | 1.2                  | 74          | 4           | 33          | 105                     | Single family detached          |
| Treatment 2        | 01644372, Little Seneca Creek Trib near Brink, MD | 0.9                  | 76          | 2           | 45          | 274                     | Single family detached and townhomes |
| Treatment 3        | 01644380, Cabin Branch near Boyds, MD     | 2.1                  | 61          | 2           | 11          | –                       | Single family detached and townhomes |
additional 6 y of data and 2 additional sites (Treatment 2 and Treatment 3) to see if baseflow patterns persisted across treatment watersheds during the construction and developed phases. We estimated annual baseflow yield with mean daily discharge data spanning 1 January 2005 through 31 December 2020 by separating mean daily streamflow into baseflow and quickflow components. For this analysis, we used the EcoHydRology package (version 0.4.12.1; Fuka et al. 2018) in R (version 4.0.2; R Project for Statistical Computing, Vienna, Austria) with a filter parameter set to 0.925 with 3 passes. We tested trends in annual baseflow yield with a non-parametric Mann–Kendall trend and the trend package (version 1.1.4; Pohlert 2020) in R. We also compared the ratio of annual site baseflow yield with the urban control and the forested control to assess the magnitude of baseflow departures relative to the control sites.

Water-quality monitoring of nitrate was conducted by USGS, University of Maryland, and MCDEP, with surface water samples collected monthly from October 2004 through September 2006 and storm and baseflow samples collected during the summers of 2010, 2011, and 2012. Groundwater in Treatment 1 was sampled annually or biannually from 2003 to 2004 and 2011 to 2016 by MCDEP. Soils in selected stormwater facilities in Treatment 1 were collected by USGS in 2015 for microbial community assessment. MCDEP began monitoring annual benthic communities near the USGS stream gages in the study watersheds in the early 1990s and continue monitoring today. County benthic monitoring followed the Maryland Biological Stream Survey protocol, which includes sampling best available habitat with a D-frame net over a 75-m reach in the spring, typically March or April (MDNR 2007). A benthic index of biotic integrity (IBI) score was calculated from 8 metrics describing biological function (MCDEP 1997). At the time of benthic sampling, a water-quality sonde was used to measure instream pH, temperature, SC, and dissolved oxygen. Previous studies have not examined these datasets. Our synthesis assessed trends in SC, benthic IBI scores, and the relative abundance of Chironomidae using Mann–Kendall tests and Sen’s slope tests with the trend package in R.

In this synthesis, we used remotely sensed datasets to assess topographic changes in the study area by supplementing topographic data in the study area with LiDAR data collected in 2013 and 2018, and LiDAR flights with data processed into digital elevation models for 2002, 2008, 2013, and 2018. Elevation change, interpreted as sediment erosion/excavation or deposition/filling, was measured with the Geomorphic Change Detection Tool (version 7, https://gcd.riverscapes.xyz/; Wheaton et al. 2010). Change across the entire watershed and within a 45–m riparian buffer was measured between 4 timeframes: 2002 to 2008, 2008 to 2013, 2013 to 2018, and 2002 to 2018 in Treatment 1, Treatment 2, and the forested control (Mettes and Jones 2021, Metes et al. 2021). Treatment 3 was excluded because construction was still ongoing as of 2018. The digital elevation models were aligned together at a resolution of 0.9 m, and the vertical error of each dataset was used to calculate a propagated error (Table S1; Wheaton et al. 2010).

RESULTS AND DISCUSSION

To date, 13 studies have been published that specifically focus on assessing changes to watershed function in the Clarksburg study area (Table 2). Eight of these studies assessed hydrologic changes in the study area, with 3 water-quality studies and 2 topographic change studies. Two approaches were commonly used to assess change: 1) watershed comparisons and 2) tracking changes over time within one watershed. Lessons learned from these studies are summarized in the following sections, which correspond with our 5 overall research questions about peak flows and runoff patterns, baseflow, water quality, topography and drainage patterns, and the benthic macroinvertebrate assemblage, to provide an understanding of the effects of development with distributed stormwater on stream function (Table 2).

Distributed stormwater management can reduce runoff yields and peak flows compared with centralized stormwater management

The streamflow regime (e.g., rapid increases in flow or prolonged periods of low flow) influences the mosaic of conditions that disturb aquatic habitats and species (Resh et al. 1988, Poff et al. 1997, Poff and Zimmerman 2010), and flow alteration is a major stressor for macroinvertebrate communities in urban streams across the Piedmont province (Waite et al. 2019). Results from studies in Clarksburg suggest that distributed stormwater management in developed watersheds can mitigate changes to streamflow and, in some cases, replicate reference conditions.

The effects of distributed stormwater management on rainfall–runoff response were observed at 2 distinct spatial scales in Clarksburg. At the street scale (<0.05 km²), runoff was monitored from May through December 2012 along Treatment 1 streets: 1) a street with disconnected downspouts and vegetated swales along the roadways, hereafter referred to as the Green Street; and 2) a street with traditional curb and gutter drainage, hereafter referred to as the Grey Street. Runoff yields from the Green Street were generally lower than runoff yields from the Grey Street (Fig. 2A; Woznicki et al. 2018). Precipitation depths <6 mm produced no detectible runoff (~55% of all events) in the Green Street, and most events between 6 and 20 mm produced less runoff than the Grey Street. However, as precipitation depth increased, the Green Street began producing similar runoff yields as the Grey Street, demonstrating an inability to mitigate runoff from larger precipitation events (Woznicki et al. 2018).
Table 2. Summary of lessons learned from monitoring in the Clarksburg, Maryland, USA, study area.

| Stream function                                | Lesson learned from Clarksburg                                                                 | Citations                                      | Ecological relevance                                      |
|------------------------------------------------|-----------------------------------------------------------------------------------------------|------------------------------------------------|----------------------------------------------------------|
| Hydrology – runoff and peak flows              | Distributed stormwater management can reduce runoff yields and peak flows compared with centralized stormwater management | Loper et al. 2014, Rhea et al. 2015, Woznicki et al. 2018, Giese et al. 2019, Hopkins et al. 2020 | Lessen the impact of disturbance on instream taxa       |
| Hydrology – baseflow                           | Baseflow temporarily increased during the construction phase of suburban development            | Hogan et al. 2014, Bhaskar et al. 2016b, 2018   | Temporarily buffer instream habitats                     |
| Water quality – nitrate                        | Nitrate declined but remains elevated because of past agricultural land use                     | Hopkins et al. 2017b, Sparkman et al. 2017, Hall et al. 2021 | Contributes to excessive algal growth and low dissolved oxygen |
| Water quality – specific conductance           | Rising specific conductance trends in all 3 treatment watersheds are likely driven by increased impervious cover | Williams et al. 2021a                          | Increase chloride in groundwater feeding summer baseflow during a time with lower flow, thereby concentrating chloride |
| Geomorphology – elevation change across the watershed and riparian zone | Most topographic change occurred during the construction phase, with substantial excavation and fill across the entire watershed and deposition within the riparian area | Jarnagin 2010, Jones et al. 2014                 | Loss of intermittent and ephemeral stream channels and substantial sediment mobilization within the riparian area during construction |
| Ecology – benthic macroinvertebrates            | Stormwater facilities may provide some protection to streams from biological impairment        | Williams et al. 2021a                          | Sensitive family recovery may be inhibited by stressors not adequately managed by stormwater facilities |

* Indicates a data release.

A similar pattern was observed at the watershed scale during a study that analyzed streamflow patterns from 1 October 2004 through 30 September 2018 (1–3 km²; Fig. 2B; Hopkins et al. 2020). Runoff yield from the forested control and Treatments 1 and 2 was similar for events with <10 mm of precipitation (negligible runoff from ~66% of events). However, as precipitation depth increased, runoff yields from the treatment watersheds began to converge with the urban control watershed (Loper et al. 2014, Rhea et al. 2015, Hopkins et al. 2020). Runoff yields in the Treatment 2 watershed began to mimic the urban control watershed at smaller precipitation depths than Treatment 1, indicating that streamflow response may depend on total impervious area, density of distributed stormwater facilities, or other landscape characteristics that distinguish Treatment 1 and Treatment 2 (Loper et al. 2014, Hopkins et al. 2020). The urban control had higher runoff yields than the treatment watersheds and the forested control for precipitation events <20 mm (Hopkins et al. 2020). Mitigating small and moderate precipitation events with distributed stormwater management may reduce the frequency of flow perturbations that affect aquatic species in urban streams, allowing more time for benthic recovery after flow disturbances. Buffering flow perturbations is particularly important because modeling has indicated that autumn and winter runoff may increase under future climate scenarios in the study area (Giese et al. 2019).

Peak discharges, normalized by drainage area, were also compared at the street scale and watershed scale (Fig. 2C, D). At the street scale, peak runoff from the Green Street was less than the Grey Street for all events except the most infrequent, extreme events, indicating that street-side swales provided enhanced flow attenuation compared with curb and gutter (Woznicki et al. 2018). At the watershed scale, most peak runoff events from the developed treatment watersheds fell between the peaks from the forested and urban control watersheds (Hopkins et al. 2020). However, at exceedance probabilities of ~0.1 and smaller, peak runoff magnitudes in both the Treatment 1 and Treatment 2 watersheds were similar to peak runoff magnitudes from the urban control watershed (Fig. 2D). Despite the inability of the distributed stormwater management to reduce peaks during extreme events, the total number of disturbance events was reduced because the peaks from most smaller storms were mitigated.

Hydrograph lag metrics (e.g., time to peak) were calculated at the street scale and watershed scale and show that distributed stormwater facilities can delay the timing of peak flows (Rhea et al. 2015, Woznicki et al. 2018, Hopkins et al. 2020). High extreme-event peak discharges from the treatment watersheds may suggest that when runoff volume exceeds the storage capacity of the distributed stormwater facilities, peak discharges are lagged and tributary flows synchronize, causing higher peaks at the watershed.
scale (Petrucci et al. 2013). The frequency and magnitude of rising and falling stages are important instream characteristics for benthic community health (Carlisle et al. 2017, Maloney et al. 2021). More frequent high-flow events can promote invertebrate drift by entraining bed sediment habitat and can lead to depletion of sensitive taxa (Gibbins et al. 2010). Mitigating peak discharges with distributed stormwater facilities may lessen the impact of disturbance on instream taxa by reducing the severity of flow events, thereby resulting in fewer disturbance events with lower velocities to entrain bed sediment and disrupt habitat. The question remains whether increased peaks for extreme events offset the ecological benefits of reducing disturbance frequencies for smaller events.

**Baseflow temporarily increased during the construction phase of suburban development**

Sufficient baseflow is critical to healthy aquatic communities in lotic environments (Poff and Allan 1995, Konrad and Booth 2005). Urban development can result in a rise or decline in baseflow depending on how groundwater interacts with local climate, geology, and water infrastructure (Bhaskar et al. 2016a). The installation of a high density of infiltration-focused stormwater facilities, in combination with the conversion of vegetation to impervious surfaces, could lead to increased baseflow in streams after suburban development. Initial streamflow monitoring in Treatment 1 indicated that during the construction phase, baseflow increased and seasonality in baseflow was attenuated compared with the centralized urban control (Hogan et al. 2014, Bhaskar et al. 2016b). Increased baseflow in Treatment 1 also affected patterns in groundwater depths, with the most gradual recession rate in the well closest to the stream and a faster recession rate in the well farthest from the stream (Bhaskar et al. 2018). The additional analysis of streamflow in this synthesis indicated that the increasing baseflow trend in Treatment 1 found by Bhaskar et al. (2016b) ended in 2015 (Mann–Kendall $\tau = 0.53, p = 0.03$), ~5 y after the construction phase ended (Fig. 3, dashed line). Treatment 1 baseflow yield was greater than that of the urban control in all but 4 y and of the forested control in all but 1 y, although this difference from the forested control decreased over the monitoring period (Fig. 4A, B). Potential reasons for these temporal baseflow patterns in Treatment 1 include the reduced infiltration capacity of stormwater facilities over time (e.g., because of clogging), and greater urban evapotranspiration over time as vegetation is established.

Our updated analysis of baseflow found that annual baseflow increased from 2005 to 2020 in Treatment 2 and Treatment 3, but there is not enough data to determine if these baseflow increases were maintained after development (Fig. 3). Baseflow in both treatments were less than the urban control in the predevelopment phase (Fig. 4A). Treatment 2 baseflow increased but remained lower than
the urban control during all but the last year of the construction phase, whereas Treatment 3 baseflow became more similar to the urban control during construction (Fig. 4A). In all 3 treatment watersheds, baseflow increased relative to the forested control during the construction phase (Fig. 4B). Baseflow levels appear to decline back to levels similar to the forested control 10 y post development in Treatment 1. These common trends in baseflow across all 3 treatments over time and development periods indicate that distributed stormwater management in Clarksburg can increase baseflow during the construction phase and post development, but increases may be temporary. Elevated baseflow during construction may temporarily buffer instream habitats from intermittency and enhance baseflow stability.

**Nitrate declined but remains elevated because of past agricultural land use**

Nutrient delivery to urban streams is influenced by the type and density of urban development and stormwater management facilities on the landscape. Patterns in baseflow nitrate were only assessed in Treatment 1 (spanning the construction and developed phases) and the control watersheds from 2004 through 2015. No nitrate data were collected in Treatments 2 or 3. Baseflow N (measured as NOx) concentrations are elevated (>1 mg/L) within the
Clarksburg study area, including the forested watershed (Hopkins et al. 2017b), and these elevated levels are associated with a history of agriculture in the region. Baseflow N concentrations in Treatment 1 surface water and groundwater declined from ~6 mg/L during construction to ~3 to 4 mg/L in the developed phase (Fig. 5). Although concentrations declined, overall N export remained roughly the same because of elevated baseflow in Treatment 1 (Hopkins et al. 2017b). N concentrations in Treatment 1 were higher than the urban control and forested control throughout the entire monitoring period (Fig. 5). Modeling to compare pollutant removal efficiencies in Treatment 1 and the urban control contrast these empirical results by indicating that distributed stormwater facilities in Treatment 1 could remove 78 kg more N, 3 kg more P, and 1592 kg more suspended sediment/unit area (km$^2$) annually than the centralized stormwater system in the urban control (Sparkman et al. 2017). The infiltration promoted by distributed stormwater management may decrease baseflow N concentrations in both surface water and groundwater. Modeling results indicated that Treatment 1 would have greater pollutant removal efficiency than the urban control; however, baseflow water-quality monitoring indicated that nitrate would continue to be a stressor and may contribute to excessive algal growth and low dissolved oxygen.

In addition to physical removal of pollutants through improved infiltration, the diversity of stormwater facility types used in Clarksburg for distributed stormwater management may provide enhanced opportunities for denitrification. A bacterial assemblage of >1000 genera, including several known denitrifiers, were identified in the soils of bioretention facilities, dry ponds, and surface sand filters in Treatment 1 (Hall et al. 2021). The presence of denitrification genes, the assumption of bacterial viability (using extracted RNA), and suitable conditions for denitrification suggest that denitrification can occur in the stormwater facilities sampled, including surface sand filters, which are thought to have low denitrification potential because filters are composed of predominantly inorganic material (Hall et al. 2021). Further research is needed to better understand N cycling in stormwater facilities (Gold et al. 2019) and how stormwater facilities function when implemented in tandem (i.e., stormwater from 1 facility drains into another facility), such as the treatment trains in this study area.

**Rising SC trends in all 3 treatment watersheds were likely driven by increased impervious cover**

Freshwater salinization is a growing concern in temperate and northern stream ecosystems where urban and suburban development result in an increase in ion concentrations (Kaushal et al. 2018). Sources of elevated ion concentrations in urban settings include deicer applications during winter weather events, weathering of building materials, and non-point source pollution (e.g., application of fertilizer). Elevated ion concentrations, often expressed as SC, can disrupt osmotic regulation of benthic macroinvertebrates and have been identified as a major ecological stressor in urbanized landscapes (Roy et al. 2003). Research has shown that SC increases with impervious cover (Moore et al. 2020), but less is known about the effects of distributed stormwater management on ionic concentrations in streams.

In this synthesis we conducted an additional analysis to examine temporal patterns in SC across the 5 study watersheds in Clarksburg to determine whether suburban development in the treatment watersheds increased SC. SC was measured annually during spring benthic macroinvertebrate sampling. Reference SC levels of Piedmont streams do not typically exceed 100 μS/cm (Olson and Cormier 2019), but median SC levels were higher than reference levels in all studied watersheds. Median SC for the forested control was slightly higher than regional reference conditions (137 μS/cm; Table 3), whereas median SC in the urban control was >7× higher than Piedmont reference conditions (722 μS/cm; Table 3). Treatment 1 had the 2nd-highest median SC (321 μS/cm), likely because of this watershed being developed first (Fig. 1A–D).

To understand patterns in changing SC as a function of development and stormwater facility implementation, we computed Kendall τ trend tests for each of the 5 watersheds for years in which there were data available for all sites (2004–2020; Table 3). SC levels in the forested watershed increased slightly over time (Table 3; Sen slope estimate = 4.3 μS cm$^{-1}$ y$^{-1}$ increase). Although annual SC varied widely in the urban control watershed, an upward trend was not detected (Fig. 6). SC increased over time in all 3 treatment watersheds as the watersheds underwent development (Table 3, Fig. 6; Kendall τ 0.46–0.83, p < 0.01), with Treatment 1 experiencing the greatest rate of increase on average (32 μS cm$^{-1}$ y$^{-1}$ increase).
Because distributed stormwater management alters hydrologic routing in the watershed, the implementation of stormwater management could also affect ion storage and transport, thereby altering stream SC patterns. Rather than flush ions quickly through winter runoff pulses, infiltration stormwater facilities could act as a store for ions during the winter months, slowly releasing ions throughout the spring. The temporal frequency of the SC data (annual measurements) precluded formally testing this hypothesis in Clarksburg. However, other recent studies have examined the effects of different types of stormwater management on SC and ion patterns in streams and suggest that stormwater facility type and design, as well as the surrounding groundwater dynamics, can greatly affect ion retention patterns. For example, 2 stormwater management ponds in Ontario, Canada, retained vs released SC differently based on their interactions with groundwater (Lam et al. 2020). A larger study in Baltimore, Maryland, found that streams with more stormwater management ponds had higher SC and ion concentrations in baseflow than urban sites with fewer or no ponds, suggesting that ponds may concentrate ions during the winter (Snodgrass et al. 2017).

In contrast, infiltration-based stormwater management, such as the facilities implemented in Clarksburg, may route ion-rich stormwater into the subsurface and reduce surface ion loads (Burgis et al. 2020). A modeling study in the Appalachian Mountains of North Carolina found maximum stream ion concentrations were reduced by low-impact development implementation via increased dilution with groundwater, although mean annual chloride concentrations were not affected by low-impact development (Gu et al. 2019). Distributed stormwater management has the potential to retain ions, but ion exchange in infiltration media may increase metal loading from stormwater.

Table 3. Mann–Kendall test and Sen’s slope test (with lower and upper slope confidence intervals [CI]) for trends in specific conductance (SC; μS/cm) in spring samples at each site in the Clarksburg, Maryland, USA, study area, with a consistent time period (2004–2020) used for comparison.

| Site             | n  | Median SC (μS/cm) | Kendall tau (τ) | Sen slope estimate | p-value | Slope estimate lower CI | Slope estimate upper CI |
|------------------|----|------------------|-----------------|--------------------|---------|-------------------------|------------------------|
| Forested control | 15 | 137              | 0.42            | 4.3                | 0.03    | 0.8                     | 6.0                    |
| Urban control    | 15 | 722              | 0.30            | 19.4               | 0.14    | -9.4                    | 47.4                   |
| Treatment 1      | 16 | 321              | 0.83            | 32.0               | <0.001  | 23.5                    | 40.6                   |
| Treatment 2      | 16 | 182              | 0.70            | 20.1               | <0.001  | 10.1                    | 27.5                   |
| Treatment 3      | 17 | 178              | 0.46            | 5.8                | 0.01    | 1.0                     | 9.0                    |

Figure 6. Specific conductance measured annually in the spring (typically March or April) during benthic macroinvertebrate sampling at the study watersheds, Clarksburg, Maryland, USA. Loess smoothing line shown for reference.
facilities (Mullins et al. 2020). Enhanced salt management is essential to minimizing the impacts and tradeoffs associated with road salt. Retaining substantial road salt in a stormwater management pond would protect downstream biota from pulses directly following snowmelt events, but the retention of road salt would also lead to an increase in chloride in groundwater, which would feed summer baseflow with more concentrated chloride and may ultimately be more detrimental to biotic health. More research into these tradeoffs is needed to fully understand the implications of ion retention and release in different types of distributed stormwater management facilities and the resulting effects on seasonal fluxes of SC in streams.

Most topographic change occurred during the construction phase, with substantial excavation and fill across the entire watershed and deposition within the riparian area

Urban development has direct impacts on landscape topology through land clearing and redistribution of surface drainage patterns (Csima 2010) and indirect impacts through landscape adjustments resulting from altered rates and quantities of sediment and runoff draining to adjacent waterways (Wolman 1967). Stream morphology can be sensitive to surrounding landscape changes and can further influence benthic and vegetative characteristics of riparian habitat quality.

Several geomorphic evaluations were conducted in Clarksburg, including a watershed-scale topographic change analysis of the construction phase (Jones et al. 2014). Topographic changes associated with land clearing and grading for development can dramatically alter natural drainage patterns, resulting in the burial of ephemeral stream channels and changes in the distribution of slopes across a watershed. Special Protection Area regulations in Clarksburg aimed to minimize land grading during the construction phase. Jones et al. (2014) evaluated watershed-scale topographic changes in Treatments 1 and 2 and the forested control using LiDAR collected between 2002 and 2008, spanning construction in Treatment 1 and some minor land grading in the headwaters of Treatment 2. They observed more elevation change outside the riparian zone in Treatment 1 and in the disturbed portion of Treatment 2 relative to the forested control. Slope in Treatment 1 transitioned from relatively smooth profiles to abrupt slope changes, expressed as upland flattening and increased valley entrenched, across the landscape associated with land grading (Jones et al. 2014).

The updated analysis for this synthesis indicated that over the entire timeframe (2002–2018), Treatments 1 and 2 experienced $-61 \pm 5$ and $-57 \pm 5$ cm of total excavation and $52 \pm 3$ and $71 \pm 5$ cm of total fill across the watershed, respectively (Table 4). The amount of change in the forested watershed was substantially less, with $-3 \pm 2$ cm of total excavation and $<1$ cm of total filling across the watershed. There was large-scale fill and excavation during construction in Treatments 1 and 2, including valley burial and substantial cutting on ridgetops (Fig. 7A, B). The timing of large-scale topographic change occurred during the land-clearing

| Site                  | Type of change | 2002–2008 | 2008–2013 | 2013–2018 | 2002–2018 |
|-----------------------|----------------|-----------|-----------|-----------|-----------|
| Watershed-wide change in elevation (cm) | | | | | |
| Forested              | Excavation     | $-1 \pm 0$ | $-1 \pm 1$ | $-2 \pm 1$ | $-3 \pm 2$ |
|                       | Fill           | $2 \pm 1$ | $1 \pm 0$ | $0 \pm 0$ | $1 \pm 0$ |
| Treatment 1           | Excavation     | $-56 \pm 5$ | $-4 \pm 1$ | $-4 \pm 2$ | $-61 \pm 5$ |
|                       | Fill           | $54 \pm 5$ | $5 \pm 1$ | $0 \pm 0$ | $52 \pm 3$ |
| Treatment 2           | Excavation     | $-21 \pm 3$ | $-41 \pm 5$ | $-12 \pm 2$ | $-57 \pm 5$ |
|                       | Fill           | $42 \pm 3$ | $39 \pm 6$ | $7 \pm 2$ | $71 \pm 5$ |
| Riparian buffer change in elevation (cm) | | | | | |
| Forested              | Excavation     | $-1 \pm 0$ | $-1 \pm 1$ | $-2 \pm 1$ | $-3 \pm 2$ |
|                       | Fill           | $2 \pm 2$ | $1 \pm 0$ | $0 \pm 0$ | $0 \pm 0$ |
| Treatment 1           | Excavation     | $-4 \pm 1$ | $-2 \pm 1$ | $-4 \pm 2$ | $-10 \pm 4$ |
|                       | Fill           | $26 \pm 3$ | $2 \pm 1$ | $0 \pm 0$ | $23 \pm 2$ |
| Treatment 2           | Excavation     | $-1 \pm 1$ | $-3 \pm 1$ | $2 \pm 1$ | $-5 \pm 2$ |
|                       | Fill           | $2 \pm 1$ | $16 \pm 2$ | $1 \pm 0$ | $18 \pm 2$ |
phase of construction in Treatment 1 (2002–2008) and between 2002 to 2013 in Treatment 2, which underwent a longer time period of land disturbance during construction and more overall excavation and filling than Treatment 1 (Fig. 7C).

Substantial deposition was also observed within the 45-m riparian buffer during the land-grading phase of construction in treatments 1 and 2, despite Special Protection Area protections (Fig. 7D). During the 2002 to 2008 timeframe, when construction began in Treatment 1, there was $10\times$ more elevation change in Treatment 1 than the forested control. Treatment 1 had less change in the later construction phase, but the amount of overall change was still twice the forested watershed change from 2008 to 2013 (Table 4). Substantial excavation and fill occurred within the Treatment 2 riparian buffer during the construction phase (2008–2013; $16 \pm 2$ cm) as well, but it had $\sim \frac{1}{2}$ the amount of deposition as the Treatment 1 riparian buffer.

Figure 7. Elevation changes in Treatment 1 (A) and Treatment 2 (B) between 2002 and 2018 in the Clarksburg, Maryland, USA, study area. Darker grays show areas that were excavated, and lighter grays show areas filled in. Barplots show overall elevation change normalized to area between 3 timeframes for the entire watershed (C) and within a 45-m buffer of the stream corridor (D).
Changes detected with LiDAR indicated that most of the topographic change occurred during the construction phase of development within the treatment watersheds when large-scale land grading resulted in substantial excavation and fill along the ridges and substantial deposition within the riparian buffer. Topographic change likely resulted in redistribution of surface drainage patterns, the burial of intermittent and ephemeral stream channels, and substantial sediment mobilization within the riparian area, which likely negatively affected instream habitat quality.

Stormwater facilities may provide streams some protection from biological impairment

One of the primary motivations for implementing distributed stormwater management is to protect the ecological integrity of aquatic ecosystems from the negative impacts of development. Clarksburg provides an opportunity to observe how benthic assemblages in the treatment streams responded to development and stormwater management. Annual measures of benthic assemblages in the forested watershed revealed consistently high IBI scores throughout the monitoring period, with a mean of 35 (good). By contrast, IBIs averaged much lower in the urban control watershed, with a mean of 17 (fair). The IBI in Treatment 1 was, on average, similar to the forested IBI predevelopment but declined during construction (Fig. 8A; Kendall $\tau = -0.43$, Sen slope $-0.4$, $p = 0.006$) and, during the developed phase, remained, on average, slightly lower than the forested control and predevelopment levels. Treatment 2 followed a similar pattern, though the decline in IBI during construction was larger (Kendall $\tau = -0.67$, Sen slope $-1$, $p < 0.001$), and IBI remained low during the developed phase. Finally, IBI in Treatment 3 generally remained in the good category during construction until the last 2 y of the record, when IBI declined into the fair category (Fig. 8A). IBI scores in Treatments 1 and 2 declined, with a faster rate of declining IBI scores in Treatment 2 (Table 5).

Changes in IBIs were partially driven by shifts in the dominant families at each site (Fig. 8B). The forested watershed has consistently low dominance, with multiple sensitive families, including Leptophlebiidae, Ephemeroptera, and Nemouridae. The largely tolerant family Chironomidae increased in relative abundance in the forested stream during the monitoring period (Table 5). By contrast, the benthic macroinvertebrate assemblage in the urban control watershed was dominated by Chironomidae, which did not change in relative abundance during the monitoring period.
Stream assemblages in all 3 treatment watersheds follow a similar pattern to the urban control by becoming dominated by Chironomidae in the developed phase, with a substantial increase in Chironomidae relative abundance during the construction phases (Table 5, Fig. 8B). Critically, even though IBI rebounded in some cases, sensitive families did not recover in the treatment watersheds.

Many factors shape stream benthic assemblages over space and time, including instream habitat suitability (Roy et al. 2003); flow regimes (Buchanan et al. 2013); water quality, such as nutrients (Matthaei et al. 2010) or salinity levels (Wallace and Biastoch 2016); riparian conditions (Ono et al. 2020); food resources, temperature, and climate (Piggott et al. 2012); landscape connectivity (Brown et al. 2018); and the occurrence and prevalence of toxic contaminants, such as pesticides (Nowell et al. 2018). Almost all of these factors can be altered by urban and suburban development (Paul and Meyer 2001, Moore and Palmer 2005, Waite et al. 2019), and many of these factors may need to be considered when designing stormwater facilities to provide protection against changes to benthic stream assemblages.

There is strong evidence that stormwater facilities in the Clarksburg watersheds mitigated some hydrologic disturbance from development, especially in the Treatment 1 watershed. However, the stormwater facility designs used in the region may not substantially address other stressors, like water quality. Although these stormwater facilities can trap particles or remove oil and grease, they are not often designed to treat soluble water-quality constituents, such as chloride from deicers. The low impervious cover (and presumably lower pollutant loads) in Treatment 1 (33% impervious cover) may be why IBI scores did not decline as much as they did in Treatment 2 (45% impervious cover). Minimizing impervious cover and implementing distributed stormwater management may be an effective way to better protect aquatic assemblages.

Table 5. Mann–Kendall test and Sen’s slope test for trends in benthic index of biotic integrity (IBI) scores and relative abundance of Chironomidae in each watershed during the monitoring period. See Fig. 8 for IBI scores and relative abundances over time.

| Site          | Benthic IBI scores | Relative abundance of Chironomidae |
|---------------|--------------------|-----------------------------------|
|               | n  | Mean (SD) | Kendall tau | p-value | n  | Mean (SD) | Kendall tau | p-value |
| Forested      | 24 | 35 (2.8)  | 0.07        | 0.68    | 23 | 16% (11.8%) | 0.56        | <0.001   |
| Treatment 1   | 23 | 26 (5.5)  | -0.43       | 0.006   | 23 | 45% (23.7%) | 0.37        | 0.02     |
| Treatment 2   | 20 | 25 (7.0)  | -0.67       | 0.001   | 20 | 49% (20.5%) | 0.54        | 0.007    |
| Treatment 3   | 20 | 30 (5.6)  | -0.11       | 0.55    | 20 | 37% (21.1%) | 0.44        | 0.007    |
| Urban control | 16 | 17 (2.8)  | 0.26        | 0.20    | 16 | 52% (16.1%) | -0.05       | 0.82     |

Finally, the ways in which stream biological conditions are assessed may not be ideal for fully understanding the effects of stormwater management on stream assemblages. Traditional IBIs often rely on the presence and abundance of Ephemeroptera, Plecoptera, and Trichoptera families, which are often considered sensitive to water quality. Therefore, IBIs may not be optimized to detect biological responses to the hydrological changes expected from stormwater management. Bioassessments that incorporate traits or even ecosystem function could improve the characterization of biological responses to changes in land use and management implementation because taxonomic information alone may not be sufficient to understand causal relationships between biota and stressors (Menezes et al. 2010). Trait-based biological assessments, which leverage information on the life history, physiological, and reproductive characteristics of individual families, may be sensitive to specific stressors, like flow regime alteration or temperature (Poff et al. 2010). Benthic macroinvertebrates in the study area were only identified to family and, therefore, there are insufficient data to conduct a trait-based assessment. Measures of functional diversity or redundancy, rather than taxonomic diversity, is another promising approach for ecosystem bioassessment programs (Bruno et al. 2016).

Broader implications

Lessons learned from the Clarksburg study area demonstrate how development with distributed stormwater management can affect watersheds and stream ecosystems in a myriad of ways (Table 2). Development with distributed stormwater management resulted in mitigation of peak flows...
and runoff volumes, especially for smaller events (<20 mm). Distributed stormwater facilities were not designed to mitigate extreme precipitation events; therefore, it is not surprising that the precipitation events that were mitigated had rainfall amounts less than the design standard of 25.4 mm (1 in). Observed deposition in riparian areas, loss of sensitive biota, and elevated ion concentrations point to the need for additional research into new stormwater management technologies that target these ecosystem stressors, especially during the construction phase of development.

Results from the Clarksburg study area provided clear evidence of the benefits of distributed stormwater management for protecting stream ecosystems. The benefits were most evident in the Treatment 1 watershed, where runoff yields, peak discharge, and runoff thresholds were similar to the forested control. These hydrologic protections may be the reason for only moderate declines in IBIs after development in Treatment 1. By contrast, streamflow was more altered after development in Treatment 2, and IBI scores declined more severely despite having a much higher density of stormwater facilities. Effective impervious cover was essentially 0 in Treatments 1 and 2 (i.e., 100% of impervious surfaces drain to a stormwater facility), but Treatment 2 had 12% more impervious cover than Treatment 1. These contrasting examples show that the amount of impervious cover, not just stormwater facility density, likely plays a critical role in overall impacts of development on stream function. Other landscape factors that disturb the riparian area, including excavation and fill during construction, sediment deposition in the channel, and loss of canopy cover, could also contribute to the success or failure of benthic community recovery.

This case study also demonstrates the limitations of distributed stormwater management because stormwater facilities are often not designed to address water-quality stressors common in urban landscapes (e.g., conductivity, pesticides, nitrate). Distributed stormwater management coupled with pollutant source reduction plans could mitigate these additional stressors. There may be unintended consequences of distributed stormwater management as well, in which altered hydrology exacerbates a water-quality problem. For example, nitrate concentrations declined in Treatment 1, but overall export remained roughly the same because of elevated baseflow. More research is needed to understand how the combination of different management strategies could affect multiple elements of stream ecosystems (e.g., flow regime, habitat, water quality, temperature).

Partnerships were critical to sustaining the monitoring program in Clarksburg over the last 20 y. An informal partnership, the Clarksburg Integrated Monitoring Partnership, began around 2007 to work towards assessing the impacts of distributed stormwater management on stream health in Clarksburg. Much of the effort grew from a small amount of seed funding that demonstrated the value of data collection. Co-location of agency scientists in the early years of monitoring allowed for enhanced coordination and collaboration. Success may also be attributed to a culture of openly sharing datasets between federal scientists, county staff, and university researchers. Open data sharing allowed members of the partnership to combine geospatial and field datasets to tell a more coherent story of land-cover change impacts on a wide array of stream ecosystem functions.

Results from the Clarksburg Integrated Monitoring Partnership’s studies informed the placement of impervious cover limits and stormwater management approaches for new development in Clarksburg. The County’s monitoring effort indicated that using treatment trains was more effective than single facilities at reducing pollutant loads. The County’s effort also suggested that stormwater facilities alone could not minimize the impacts of development on stream health in the sensitive headwater streams in the Clarksburg Special Protection Area. Other factors, such as limiting the amount of impervious cover, land grading, soil compaction, and culvert and road placement, are critical for maintaining stream health during development. The lessons learned from monitoring efforts in Clarksburg can help developers, watershed planners, and local water resources agencies better assess how large-scale green stormwater infrastructure might influence stream health and function in their region.

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Data accessibility: All datasets are available through the following data releases: Peak flow and runoff data (Hopkins et al. 2019, USEPA 2020), water chemistry data (Hopkins et al. 2017a, Williams et al. 2021), soil microbial communities and soil chemistry data (NCBI 2020), geomorphic data (Mettes and Jones 2021, Mettes et al. 2021, Williams et al. 2021), and land-cover datasets (Williams et al. 2018, Woznicki and Hopkins 2019).

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