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Variation in physicochemical responses to urbanization in streams between two Mid-Atlantic physiographic regions

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Abstract. Urban development substantially alters the physicochemistry of streams, resulting in biodiversity and ecosystem function loss. However, interregional comparisons of physicochemical impact in urban streams suggest that geoclimatic heterogeneity may influence the extent of degradation. In the Mid-Atlantic United States, the adjacent Coastal Plain and Piedmont physiographic provinces possess distinctly different hydrogeomorphic properties that may influence how stream ecosystems respond to urbanization. Recent bioassessments have demonstrated that biotic sensitivity to urbanization is relatively acute in the Piedmont, suggesting that physicochemical change as a consequence of urbanization may be greater in that province. We compared hydrologic, chemical, and thermal characteristics of Mid-Atlantic Coastal Plain and Piedmont first- through fifth-order streams along gradients of impervious surface cover (ISC) at multiple spatial scales. Linear models were applied to test if conditions in rural streams and the degree of impact from ISC varied between provinces. Mean and maximum summer temperatures in Piedmont streams increased more per unit of ISC than in the Coastal Plain. Contrary to expectations, however, variables that quantified high-flow event frequency, magnitude and duration, exhibited significantly greater impact along the ISC gradient in the Coastal Plain. Most chemical changes associated with increasing ISC were similar in the two provinces, although the interregional chemical composition of rural streams differed substantially for most parameters. Our findings demonstrate consistent interregional heterogeneity in stream ecosystem responses to urbanization. Landscape-scale management decisions with stream ecosystem conservation, mitigation, or restoration as a goal must therefore carefully consider the geoclimatic context in order to maximize effectiveness.

Key words: high-flow events; impervious surface cover; regional differences; stream physicochemistry; urbanization; urban streams.

Introduction

Urban encroachment ranks among the most pervasive drivers of stream ecosystem degradation. During precipitation events, impervious surfaces associated with urbanization route water directly to stream channels that would otherwise infiltrate the catchment. Moderate to extreme changes in the thermal, chemical, geomorphic, and hydrologic regimes of streams ensue, resulting in extensive loss of biodiversity (Paul and Meyer 2001, Walsh et al. 2005). Although the spatial extent of such degradation has traditionally been localized in the vicinity of urban centers, the expansion of exurban growth has exacerbated the scale of risk to aquatic ecosystems. For instance, approximately 40% of all fifth-order streams within the conterminous United States will be somewhat impacted by urban development by 2030 (Theobald et al. 2009).

A wealth of mostly case study and local-scale efforts has led to a consensus on the general nature of physicochemical change that occurs in streams following watershed urbanization. Repeatedly detected trends in impacted sites among multiple locales have been observed and are now considered central tenets of urban stream ecology. Examples include heightened high-flow event frequency and magnitude (Konrad et al. 2005, Roy et al. 2005, Schoonover et al. 2006); elevated temperatures (Wang and Kanehl 2003, Moerke and Lamberti 2006, Nelson and Palmer 2007); increased concentrations and loads of nutrients, metals, dissolved organic carbon, and suspended solids (Lenat and Crawford 1994, Groffman et al. 2004, Schiff and Benoit 2007); and geomorphic adjustment such as channel widening and/or incision (Arnold et al. 1982, Pizzuto et al. 2000, Grable and Harden 2006). Each of the aforementioned studies and the majority of related research were typically limited to a single watershed, paired watersheds, or group of sites within a relatively homogeneous geoclimatic region (Brown et al. 2009)

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Comparing impacts in two or more geoclimatic settings may reveal important discrepancies in the degree of change caused by urbanization. Physiographic provinces and/or ecoregions (Omernik 1987) have long been recognized as meaningful spatial divisions that demarcate disparities in stream form and function (Hughes et al. 1986, Johnson 2000). Such natural variability may convey inherent differences in vulnerability to landscape stressors, yet only a handful of efforts have explicitly explored the role of physiographic or regional variation on physicochemical change in urban streams. The most comprehensive among such studies was conducted by the National Water-Quality Assessment Program (NAWQA) of the United States Geological Survey (Sprague and Nowell 2008, Brown et al. 2009, Cuffney et al. 2010). Findings from the NAWQA assessment demonstrated substantial heterogeneity in biological, chemical, geomorphic and hydrologic responses to urbanization among nine metropolitan regions located in different ecoregions. Specifically, streams in most urban areas surveyed by NAWQA exhibited similarity in the nature of ecosystem response to urbanization, but the severity of impact differed substantially among ecoregions. In partial contrast, Poff et al. (2006*a*) highlighted differences in both the nature and magnitude of response between urbanization and multiple hydrologic metrics among four regions of the United States.

Despite the insight gained from such efforts and explicit calls to examine how physiography may regulate stream ecosystem responses to urbanization (Walsh et al. 2005), related efforts remain scarce. Yet determining how geoclimatic variation mediates the effects of landscape stressors could prove useful for the management of impacted systems: if streams in one region are relatively more vulnerable to change for a given environmental parameter, conservation or restoration efforts could target that parameter over others. For instance, many urban stream remediation and mitigation actions meant to improve water quality emphasize high-flow event water management. In arid streams, however, the majority of urban-derived pollutants may be exported during dry conditions with low discharge relative to floods, rendering management actions that target only flood flows ineffective at decreasing total pollutant loads (Stein and Ackerman 2007).

The Coastal Plain–Piedmont physiographic boundary of the eastern United States likely represents a meaningful transition affecting stream ecosystem responses to land use. Both physiographic provinces possess distinct hydrogeomorphic properties: soils are thicker and infiltration rates are greater in the Coastal Plain while topographic relief is typically more substantial in the Piedmont (Markewich et al. 1990, Swain et al. 2004). Different physicochemical responses to land use change have been observed as a result of such hydrogeomorphic disparities. For instance, agricultural land use causes more nitrogen export in Piedmont streams relative to the Coastal Plain (Jordan et al. 1997*a*, *b*, Liu et al. 2000). Although a very large proportion of research on urban stream ecology has been conducted in these regions (Schueler et al. 2009), no direct quantitative comparisons of physicochemical change along urban gradients have been conducted. Recent bioassessment efforts have revealed that the loss of biotic integrity along increasing gradients of urbanization is heightened in the Piedmont for both fish (Morgan and Cushman 2005, Utz et al. 2010) and macroinvertebrate (Goetz and Fiske 2008, Utz et al. 2009) assemblages, however. Piedmont macroinvertebrate communities have also been shown to be more sensitive to urbanization relative to assemblages in ecoregions other than the Coastal Plain (Cuffney et al. 2010). Such consistently observed disparities in biological responses suggest that at least some physicochemical change induced by urbanization may be more severe in the Piedmont physiographic province.

Given the discrete hydrogeomorphic differences between the Coastal Plain and Piedmont, we sought to determine if physicochemical responses to urbanization were distinct between these two adjacent ecoregions. The previously observed disparity in biodiversity loss between provinces was used to develop our hypothesis: because biotic degradation due to urbanization is relatively more severe in the Piedmont, we predicted greater physicochemical change would be observed along gradients of urban development in the Piedmont. We focused on hydrologic, chemical, and thermal properties of streams commonly affected by urbanization, many of which are considered agents of biological degradation. For each quantified physicochemical parameter, we tested for province-specific differences in rural streams and in the degree of change along a gradient of impervious surface cover (ISC). Analysis of covariance (ANCOVA) was used to construct linear models along ISC gradients; a province term (which compared intercepts) tested for provincial differences in undeveloped streams and an interaction term (which compared regression slopes) determined if the degree of change along the ISC gradient varied between provinces.

Study Area

The eastern Piedmont and Coastal Plain physiographic provinces encompass approximately 197 000 and 423 000 km^2 of land, respectively, in the eastern United States (Fig. 1). Both provinces are characterized by distinct geologic, topographic, and hydrogeomorphic attributes. Gneiss–schist and shale–sandstone crystalline rock formations underlay a 1–2 m layer of soil in the Piedmont province (Markewich et al. 1990, Swain et al. 2004). The topographic relief of the Piedmont is best described as undulating ridges and valleys that typically range from 15 to 100 m deep; elevation above sea level ranges from 60 to >500 m (Thornbury 1965). Piedmont streams are of steep to moderate gradient and bed

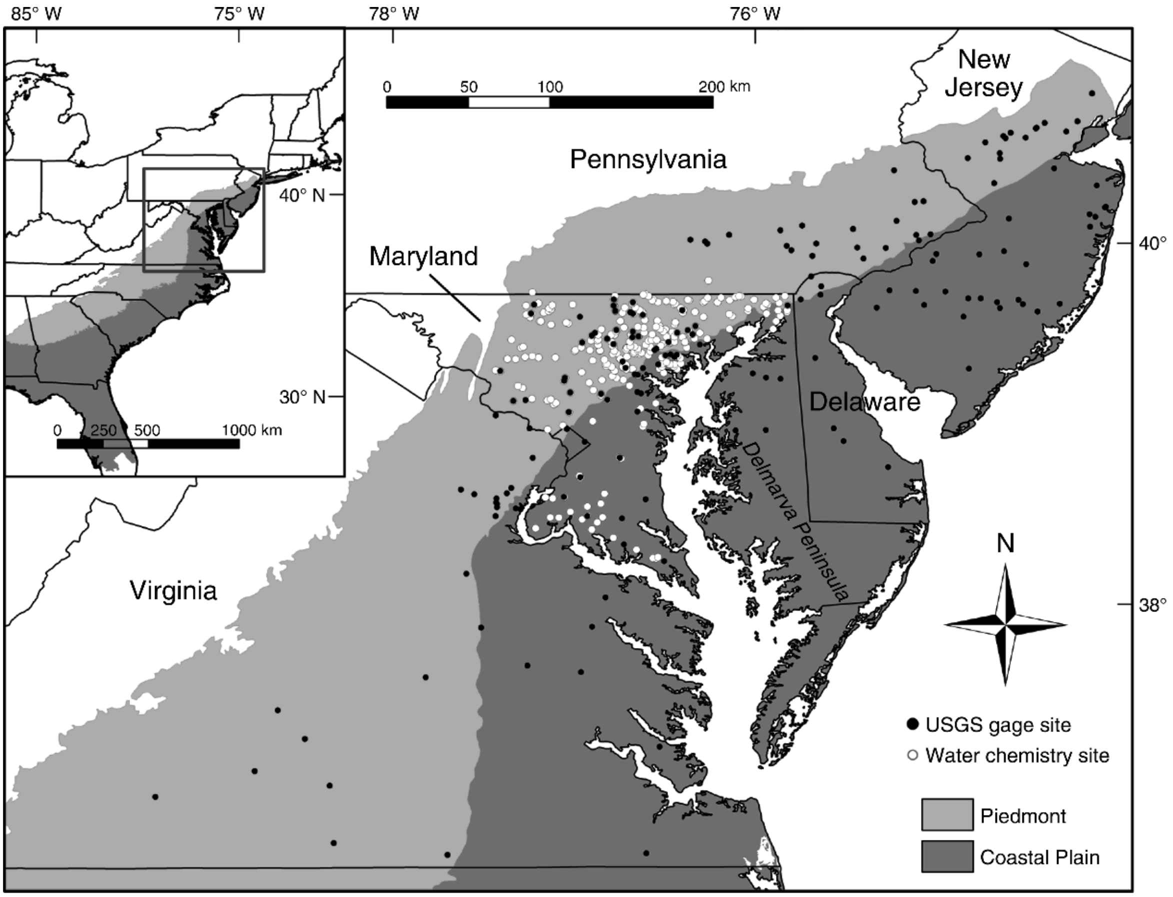

FIG. 1. Map of the Coastal Plain and Piedmont physiographic provinces shown at the scales of the eastern United States and Mid-Atlantic region. Gage and water chemistry (Maryland Department of the Environment [MDE] and Baltimore County [BC] sites) sites are shown; Maryland Biological Stream Survey (MBSS) sites were excluded due to the large sample size.

sediments consist of boulders, cobble, gravel, and occasional bedrock outcrops. In the Coastal Plain, crystalline basement rock is buried by a wedge of unconsolidated, mostly siliciclastic sediments in depths ranging from <10 m near the Piedmont border (i.e., the "Fall Line") to >3000 m along the coast of North Carolina (Ator et al. 2005). Uppermost Coastal Plain elevations are 80 to 100 m and topography varies from steeply incised to nearly flat. Streams in the Coastal Plain are of moderate to low gradient and bed sediments are a heterogeneous site-specific mix of cobble, sand, clay, and silt.

Data analyses were limited to reaches within the five state Mid-Atlantic region, an area that includes the northernmost extent of both provinces. Hydrologic data were derived from watersheds throughout the region, while chemical and temperature data were assessed in reaches exclusively within the state of Maryland. Although the physiographic divide of interest extends further south than the area we considered, we restricted analyses to the Mid-Atlantic region for which we were able to acquire data. Watershed delineation in the outer reaches of the Delmarva Peninsula is particularly difficult due to low topographic relief and a prevalence of agricultural drainage ditches (Baker et al. 2006). We therefore excluded Delmarva watersheds from chemical and temperature analyses and included only northern Delmarva watersheds in hydrologic analyses.

METHODS

Hydrologic data

Data used to assess hydrologic regimes were derived from United States Geological Survey (USGS) continuous stream discharge records (*available online*).[4] Collected data were limited to the years 1996–2006, inclusive, to coincide with the year that our selected land use coverage represented (2001). The majority of data (86%) represented discharge readings collected at 15-minute intervals; time increments for the remainder of data were no less than 5 minutes and no greater than 60 minutes. If the available record for a given site–year was <90% complete, data from that year were excluded from calculations. Further, if the time increment between successive readings

[4] ⟨http://waterdata.usgs.gov/nwis/rt⟩

at a given site was adjusted (for instance, from 30 minutes to 15 minutes) by the USGS gaging program, data from the transitional year were omitted.

Six hydrologic metrics were calculated to quantify flow regime change with a primary focus on characterization of high or low-flow event frequency, magnitude, or duration. These metrics have demonstrated previous success in detecting flow regime alteration in an urban setting (Konrad et al. 2005, Roy et al. 2005, Poff et al. 2006*a*). Because the amount of data varied across sites, variables were averaged across complete years of record so that gages with more available records were not overrepresented in analyses. High-flow event frequency was quantified by counting the instances in which discharge exceeded three times the monthly median discharge as recommended for quantifying such events in Mid-Atlantic perennial streams (Olden and Poff 2003). We initially considered the metric $T_{Q\mathrm{mean}}$ (Konrad et al. 2005), the fraction of time in which discharge exceeded the monthly mean, to assess high-flow event duration. Since $T_{Q\mathrm{mean}}$ was highly correlated with high-flow event frequency (Pearson correlation coefficient = 0.84, $P < 0.0001$), the mean duration of high-flow events (time elapsed while above three times the monthly median) was used instead. An acute low-flow event was defined as discharge falling below 25% of the annual median as this metric has been shown to detect low-flow hydrologic change in urban Piedmont streams (Roy et al. 2005). Both the frequency of occurrence and yearly maximum duration (i.e., duration of potential drought conditions) of acute low-flow events were calculated and used as metrics. The maximum and minimum observed mean daily discharges were divided by the watershed area (km^2) to provide normalized high- and low-flow magnitude metrics (Olden and Poff 2003, Poff et al. 2006*a*).

Chemical data

Stream water chemistry data were obtained from three sources. Sampling protocols and assessed parameters varied among data sets so each was assessed separately. The largest data set was provided by two rounds (1995–1997, 2000–2004) of the Maryland Biological Stream Survey (MBSS; Klauda et al. 1998), a statewide stream monitoring program that includes assessment of water chemistry. Water quality sampling in the MBSS protocol includes one collection during spring (1 March–30 April) when flows are low enough to allow macroinvertebrate sampling (i.e., approximately baseflow). Although most MBSS reaches were sampled once, about 20% were sampled 2–10 times over the course of both rounds. The total maximum daily load program of the Maryland Department of the Environment (MDE) provided the second largest stream chemistry data set (*available online*).[5] In the MDE program, water samples were collected ≥10 times at each site throughout the year between 1997 and 2006 regardless of flow or weather conditions. The smallest data set was provided by the Baltimore County Watershed Management and Monitoring program (2008; BC). This data set included samples collected during baseflow conditions 6–10 times annually in the spring, summer, and fall of 2003–2006.

All samples from each program were subjected to laboratory chemical analysis, with protocols following those outlined by the United States Environmental Protection Agency (United States Environmental Protection Agency 1983, 2004). The variables analyzed included: conductivity, concentrations of sulfate (SO_4), dissolved organic carbon (DOC), total nitrogen (TN), and total phosphorous (TP) in the MBSS data set; conductivity, total suspended solid (TSS), TN, DOC, and chlorophyll *a* concentrations in the MDE data set; and hardness, total solid (TS), SO_4, TN, and TP concentrations in the BC data set.

Temperature data

Summer water temperature data were derived from the MBSS round-two data set. Temperature loggers were programmed to read every 20 minutes starting on 1 June and ending near the onset of fall; loggers were collected between 25 August at the earliest and 18 September at the latest. Records were visually assessed in a graphing program to determine if dewatering had likely occurred; data from loggers that appeared to temporarily go dry as well as those collected dry were excluded from analyses.

Four temperature variables were calculated: (1) mean temperature, (2) maximum temperature, (3) the number of days in which a temperature surge (presumably associated with a high-flow event) occurred, and (4) the mean observed temperature surge duration. A temperature surge was defined as an increase of ≥1.3°C between readings and was assumed to persist until temperatures had reached ≤1.3°C of the pre-surge temperature. The 1.3°C per 15-minute increment criteria was successfully applied previously by Nelson and Palmer (2007) to detect thermal regime shifts in urban Piedmont streams. The number of days in which a surge was recorded was standardized to account for variable record lengths by multiplying the value by the total record length in days and dividing by 93 (i.e., the record length in days between 1 June and 30 August). We restricted our analyses to increasing temperature spikes as such events can induce substantial physiological stress on aquatic organisms (Caissie 2006).

Geographic data and procedures

Watersheds corresponding to each stream sampling location were delineated using GIS software and impervious cover was quantified upstream of each gage location and sampling point. A sink-corrected 30-m resolution national digital elevation data set was used to

[5] ⟨http://www.mde.state.md.us/Programs/WaterPrograms/TMDL/index.asp#current⟩

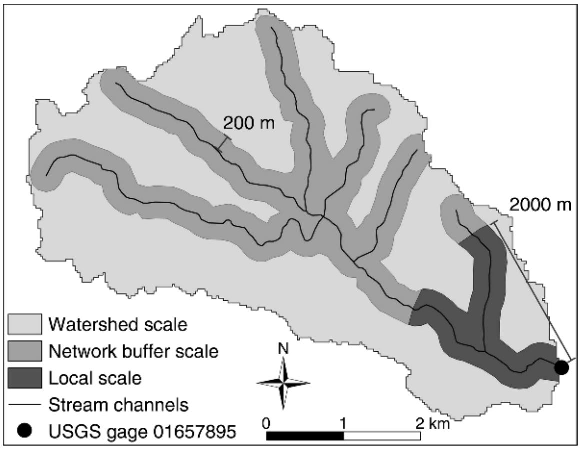

FIG. 2. Map of the Powells Creek (Virginia) watershed illustrating the three spatial scales in which percent impervious surface cover (ISC) estimates were quantified.

determine watershed boundaries above each point coordinate (data set *available online*).[6] The impervious surface cover layer was provided by the National Land Cover Database (NLCD), a raster data set representative of the year 2001 (*available online*).[7] Each 30×30 m pixel in the NLCD data set includes an estimate of percent impervious cover between 0 and 100; these values were summarized to calculate total impervious cover at the scales described in the next paragraph. We also calculated percent agricultural cover (sum of pasture and row crop classes) from the land use layer of the NLCD data set. Percent physiographic composition of each watershed was calculated from the Level III Ecoregion layer provided by United States Environmental Protection Agency (United States Environmental Protection Agency 2010).

Percent ISC at the watershed, network buffer, and local scales (Fig. 2) relative to the gage or sample point was calculated for use as predictor variables. The watershed scale was defined as the percent ISC of the entire basin upstream from the gage or sampling location. For the network buffer scale, we calculated percent ISC in a buffer zone that was 200 m perpendicular to both sides of the entire stream channel network upstream of the sampling point. We assumed that a 200-m riparian area ensured the inclusion of all near-stream ISC that may have been directly connected to site channels. At the local scale, percent ISC was represented as the land in the network buffer zone within a 2000-m radius upstream of the gage or sample point. Stream channels were derived from the flowline component of the National Hydrography Data Set Plus, which includes all perennial stream channels mapped at a 1:100 000 scale (data set *available online*).[8] Percent agricultural cover at the watershed and network buffer scales was also calculated for each site.

Statistical analyses

Sampling sites within each data set met a number of criteria established a priori before being included in the statistical analyses. Only fifth-order ($\leq$282 km^2; Knighton 1998) or smaller watersheds were selected. Watersheds with a major impoundment on the main stem of the stream network were excluded. Further, each watershed was $\geq$95% within either the Coastal Plain or Piedmont (the majority were 100% within either province) in order to reduce variation that may have been introduced by using watersheds that span both provinces. Table 1 shows the final number of watersheds in each data set delineated by physiographic province. Considerable disparities in watershed area existed among some of the data sets, though catchment size differences between ecoregions within data sets were much less profound (Fig. 3).

The statistical test for each quantified parameter was an analysis of covariance (ANCOVA) model with province (Coastal Plain or Piedmont), ISC, and an interaction term as predictor variables. While multivariate approaches such as principal components analysis may allow for pattern recognition across groups, the data sets used were not suitable for aggregation into a larger whole due to differences in the attributes measured, environmental parameters available for analysis, and locations of sites assessed among data sets. The use of ANCOVA models allowed us to use the same inferential approach to test for differences in each data set. Furthermore, our intent was to identify the ecoregion-specific variation in impact induced by a specific independent variable (ISC) rather than model the entire suite of environmental attributes that affected each dependent variable. Within each ANCOVA analysis, the most appropriate scale of ISC (watershed, buffer, or local) was chosen by running the ANCOVA models with each scale of ISC and selecting the one with the lowest Aikake's information criterion (AIC) score (Burnham and Anderson 2002). Normality of residuals was visually assessed with normal distribution probability plots and data were $\log_{10}$-transformed to approximate normality when necessary. Variance homogeneity with respect to the physiographic province term was tested using Levene's test (Levene 1960). If variances were highly ($P < 0.01$) heterogeneous, an ANCOVA with variance heterogeneity incorporated into the model was used. The assumption of variance homogeneity was also explored with respect to the continuous variable by plotting the residuals against the predicted value separately for each province and visually assessing if variances increased substantially along the gradient (Zar 1999).

Select chemical data were omitted or filtered to reduce redundancy and ensure that ISC impact was relatively isolated from potential confounding effects. Within all

6 ⟨http://ned.usgs.gov/⟩
7 ⟨http://www.mrlc.gov/⟩
8 ⟨http://www.horizon-systems.com/NHDPlus/⟩

TABLE 1. Sample size of each data set.

Data set and subset	Coastal Plain	Piedmont
Hydrologic	63	107
Chemical		
Maryland Biological Stream Survey (MBSS)	374 (320)	631 (138)
Maryland Department of the Environment (MDE)	44 (43)	149 (37)
Baltimore County (BC)	15 (10)	91 (43)
Temperature	187	164

Note: Values in parentheses indicate the number of samples remaining after highly agricultural sites (≥40% coverage at the watershed scale) were excluded for the analysis of total nitrogen, total phosphorus, dissolved organic carbon, and chlorophyll *a*.

chemical data sets, if two or more of the response variables were highly correlated (Pearson correlation > 0.75), only the variable with the most complete data record was assessed. In instances where sites were sampled more than once, dependent values were averaged. A number of chemical parameters (TN, TP, DOC, and chlorophyll *a* concentrations) are strongly influenced by agricultural land use but are also known to increase along a forested to urban land use gradient (King et al. 2005). We visually assessed preliminary ISC–nitrogen and –phosphorus regression models and concluded that omitting sites with ≥40% watershed agriculture removed the majority of streams with agriculturally-derived high nutrient concentrations. Therefore, the sites included in analyses varied somewhat among a subset of chemical parameters. The exclusion of watersheds heavily impacted by agriculture allowed the comparison of statistical metrics (i.e., coefficients, P, and r^2 values) among parameters and data sets for models that primarily quantified the effects of ISC and physiography. While lower levels of agriculture may induce some impact on these variables, very few watersheds in Maryland lack agricultural cover altogether and setting lower criteria would have omitted a majority of our sites.

Temperature data were also filtered, transformed, and/or blocked to minimize variability not associated with ISC impact. Data were derived from five years (2000–2004) and were assumed to be affected by interannual climatic fluctuations. We therefore blocked all temperature analyses by year. Further, stream temperature attributes vary by watershed size (Vannote and Sweeney 1980, Nelson and Palmer 2007). To account for this factor we included a watershed size (km^2) model term in each ANCOVA model as a predictor variable that was log-transformed to linearize the relationships with dependent metrics. Finally, sites with ≥40% agricultural cover in the network buffer zone were excluded from all temperature analyses to remove potential confounding effects derived from a lack of shading in agricultural riparian zones.

In most data sets, certain sites were located some distance upstream of others (hereafter referred to as nested) and we were concerned that such nonindependence among sample units could have affected the statistical analyses. We explored this possibility by using a Monte Carlo approach of running the ANCOVA models on randomly selected non-nested subsets of data for each high-flow event variable. The hydrologic data set included 57 nested sites in 19 groups and 113 sites that were non-nested. Two permutation procedures with 1000 iterations each were run: one with data subsets consisting of one randomly selected site from each of the 19 nested groups along with the 113 original non-nested sites and the second with 132 randomly selected nested and non-nested sites (to see if any disparity in results could be attributed to a reduction in sample size). We chose 1000 iterations in order to obtain a sufficiently large number of comparisons to minimize random effects in analysis results and to approximate the true statistical result. The percentage of P values below $\alpha = 0.05$ and mean P value for each term (province, impervious surface, and interaction) from both permutation procedures were calculated to determine if nestedness and/or a reduction in sample size affected statistical outcome.

RESULTS

Hydrologic change

Province-specific differences in hydrochemical responses along impervious surface gradients were observed for all variables that characterized high-flow conditions. ANCOVA models for high-flow event frequency (df = 3, 166, $F = 98.8$, $P < 0.0001$; $r^2 = 0.64$), $\log_{10}$-transformed duration (df = 3, 166, $F = 17.5$,

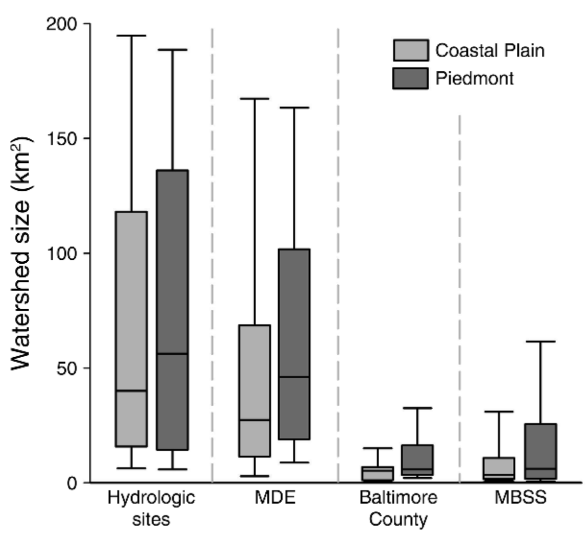

FIG. 3. Distribution of watershed sizes among data sets. The box lines represent the lower, median, and upper quartile, while the whiskers illustrate the range (i.e., minimum and maximum values).

TABLE 2. Hydrologic variable ANCOVA model details.

Variable	Scale of ISC	% ISC		Province		Interaction	
		F	P	F	P	F	P
Number of high-flow events (yr^{-1})	B	257.1	<0.0001	38.5	<0.0001	9.8	0.0021
$\log_{10}$(high-flow event duration)†	B	23.0	<0.0001	29.1	<0.0001	4.1	0.0452
Maximum daily flow/watershed size ($m^3 \cdot s^{-1} \cdot km^{-2}$)	W	73.9	<0.0001	16.8	<0.0001	5.0	0.0275
Maximum duration of low-flow events (min)	B	6.0	0.0582	3.2	0.0741	3.8	0.0533

Notes: Only models that were found to be statistically significant are shown. The scale of impervious surface cover (ISC) that proved to be the best fit in each statistical model is also shown (B, 200-m riparian buffer; W, watershed). All df = 1, 166.
† Measured in minutes.

$P < 0.0001$; $r^2 = 0.24$), and maximum daily flow (df = 3, 166, $F = 29.1$, $P < 0.0001$; $r^2 = 0.35$) were statistically significant; the continuous, province, and interaction terms were significant in each model (Table 2, Fig. 4). Percent ISC at the network buffer scale proved the best model for high-flow event frequency and duration, while percent ISC at the watershed scale provided the best model of daily maximum flow. High-flow events in rural Piedmont streams were more frequent and of shorter duration, while annual maximum flows were larger relative to those observed in rural Coastal Plain streams (Table 3). Increased ISC affected high flows in Coastal Plain streams to a greater degree, however; regression models appear to converge at high levels of ISC (Fig. 4).

Low-flow event attributes did not appear to vary between provinces or change substantially along impervious surface gradients. The $\log_{10}$-transformed frequency (df = 3, 166, $F = 1.12$, $P = 0.3422$; network buffer scale) of low-flow events and minimum annual daily flow (df = 3, 166, $F = 0.3$, $P = 0.8184$; watershed scale) ANCOVA models were not statistically significant. Analysis of $\log_{10}$-transformed low-flow event duration was significant in the full ANCOVA model (df = 3, 166, $F = 3.5$, $P = 0.0176$, $r^2 = 0.06$; network buffer scale), however, no individual term was independently significant (Table 2).

Chemical change

Chemical composition varied between physiographic provinces, and multiple variables were correlated with ISC. The degree of change along ISC gradients varied between provinces for only a few parameters, however. In the MBSS data set, overall ANCOVA models were significant for conductivity (df = 3, 1002, $F = 255.6$, $P < 0.0001$, $r^2 = 0.44$; network buffer scale), SO_4 (df = 3, 1002, $F = 121.2$, $P < 0.0001$, r^2=0.27; watershed scale), $\log_{10}$-transformed DOC (df = 3, 454, $F = 44.1$, $P < 0.0001$, $r^2 = 0.23$; watershed scale), and $\log_{10}$-transformed TP (df = 3, 454, $F = 11.0$, $P < 0.0001$, $r^2 = 0.10$; watershed scale). The province and ISC (watershed scale) terms were also significant in a mixed model for TN (Table 4). While the ISC and province terms were significant for most parameters, only the $\log_{10}$-transformed TP model included a significant interaction term. Rural Piedmont streams were more conductive and had higher concentrations of TN, while rural Coastal Plain

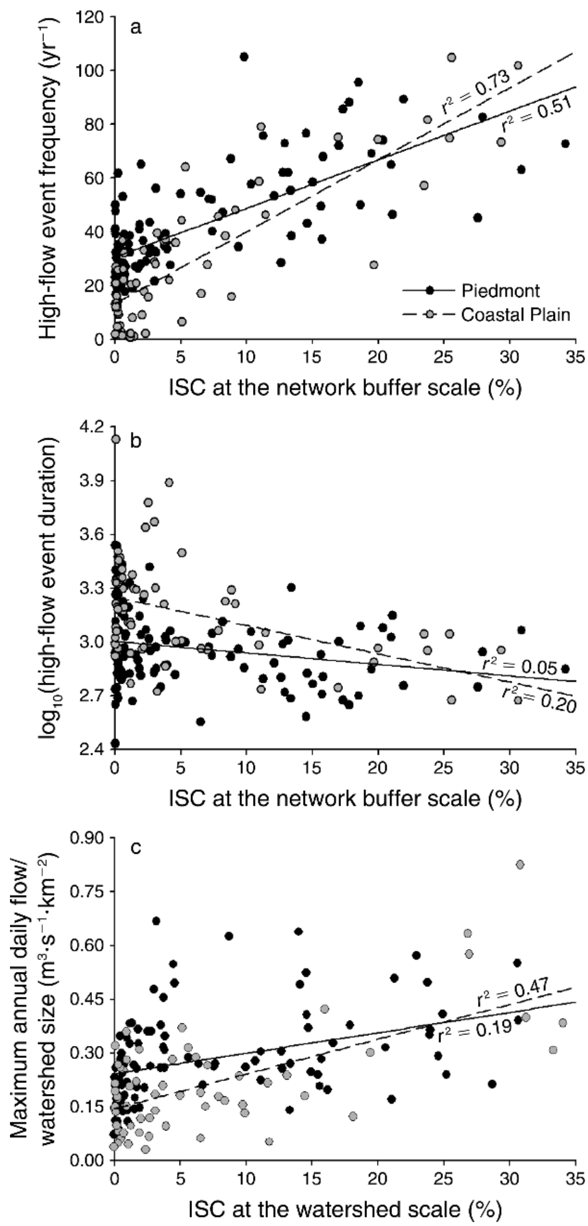

FIG. 4. Shifts in high-flow event (a) frequency, (b) duration, and (c) magnitude along gradients of impervious surface cover delineated by physiographic province. For panel (b), units for the high-flow event duration were originally in minutes.

TABLE 3. Comparisons of intercepts and slopes (±95% confidence intervals) for significantly different parameters.

Data set and parameter	Intercept		Slope	
	Coastal Plain	Piedmont	Coastal Plain	Piedmont
Hydrologic				
Frequency of high-flow events (no./yr)	13.3 ± 4.3	**30.6 ± 3.5**	**2.7 ± 0.4**	1.8 ± 0.3
$\log_{10}$(high-flow event duration)†	**3.3 ± 0.1**	3.0 ± 0.1	**−0.0158 ± 0.0081**	−0.0064 ± 0.0053
Maximum daily flow ($m^3 \cdot s^{-1} \cdot km^{-2}$)	0.14 ± 0.04	**0.24 ± 0.03**	**0.0097 ± 0.0027**	0.0057 ± 0.0024
Chemical (MBSS)				
Conductivity (μS/cm)	100.5 ± 16.5	**183.8 ± 11.3**		
SO_4 (mg/L)	**14.3 ± 1.1**	9.6 ± 0.6		
$\log_{10}$(DOC)‡	**0.51 ± 0.03**	0.28 ± 0.05		
TN (mg/L)	0.65 ± 0.09	**1.75 ± 0.20**		
$\log_{10}$(TP)‡	**−1.6 ± 0.1**	−1.8 ± 0.1	*−0.0013 ± 0.0039*	**0.0079 ± 0.0053**
Chemical (BC)				
Hardness (mg/L)			4.0 ± 2.1	**8.2 ± 1.3**
TS (mg/L)			7.5 ± 4.8	**13.0 ± 1.6**
$\log_{10}$(TP)‡	**0.035 ± 0.005**	0.016 ± 0.004	*−0.00014 ± 0.00034*	**0.00032 ± 0.00029**
Chemical (MDE)				
Conductivity (μS/cm)	119.0 ± 41.9	**202.7 ± 22.3**		
TN (mg/L)	0.84 ± 0.15	**1.73 ± 0.25**		
$\log_{10}$(DOC)‡	**0.77 ± 0.04**	0.34 ± 0.07	*−0.0026 ± 0.0038*	**0.0083 ± 0.0056**
$\log_{10}$(chlorophyll *a*)§	**0.64 ± 0.15**	0.36 ± 0.13	−0.016 ± 0.015	**0.012 ± 0.009**
Temperature				
Mean (°C)	**20.2 ± 0.8**	17.8 ± 0.7	0.032 ± 0.023	**0.076 ± 0.027**
Maximum (°C)	**26.3 ± 3.3**	22.9 ± 1.5	0.11 ± 0.05	**0.17 ± 0.05**
$\log_{10}$(surge duration)¶	**2.3 ± 0.2**	2.1 ± 0.2	0.022 ± 0.008	**0.033 ± 0.009**

Notes: Values with a greater actual (for intercepts) or absolute (for slopes) magnitude are shown in boldface type; slopes not significantly different from zero are shown in italic type. Abbreviations are DOC, dissolved organic carbon; TN, total nitrogen; TP, total phosphorus; TS, total sulfur. Empty cells represent terms that were not statistically significant in the ANCOVA model.

† Measured in minutes.

‡ Measured in mg/L.

§ Measured as μg/L.

¶ Measured in hours.

TABLE 4. Chemical ANCOVA model details.

Data set and variable	Scale of ISC	df	ISC (%)		Province		Interaction	
			F	*P*	*F*	*P*	*F*	*P*
MBSS								
Conductivity (μS/cm)	B	1, 1002	705.1	<0.0001	68.5	<0.0001	0.4	0.5052
SO_4 (mg/L)	W	1, 1002	200.7	<0.0001	61.9	<0.0001	0.5	0.4622
$\log_{10}$(DOC)†	W	1, 454	6.1	0.0136	45.2	<0.0001	0.8	0.3798
TN (mg/L)	W	1, 454	15.3	0.0001	93.8	<0.0001	0.0	0.9406
$\log_{10}$(TP)†	W	1, 454	3	0.0868	27.6	<0.0001	5.7	0.0180
BC								
Hardness (mg/L)	B	1, 102	51.6	<0.0001	0.1	0.8643	6.2	0.0146
TS (mg/L)	B	1, 102	83.8	<0.0001	0.1	0.7431	6.1	0.0154
$\log_{10}$(TP)†	W	1, 49	6.6	0.0136	14.9	0.0004	13.6	0.0006
SO_4 (mg/L)	W	1, 102	11.3	0.0017	1.2	0.2799	0.1	0.7870
MDE								
Conductivity (μS/cm)	W	1, 189	149.0	<0.0001	12.1	0.0006	0.3	0.6181
TSS (mg/L)	B	1, 189	5.0	0.0283	2.2	0.1442	0.1	0.7686
TN (mg/L)	B	1, 76	14.9	0.0003	42.4	<0.0001	2.2	0.1466
$\log_{10}$(DOC)†	B	1, 76	2.8	0.0968	110.7	<0.0001	10.5	0.0018
$\log_{10}$(chlorophyll *a*)‡	B	1, 76	0.2	0.6993	8.7	0.0043	9.3	0.0032

Notes: Only models found to be statistically significant are shown. The scale of ISC that proved the best model fit is also provided (B, 200-m riparian buffer; W, watershed).

† Measured as mg/L.

‡ Measured as μg/L.

TABLE 5. Details of temperature variable ANCOVA models.

Variable	ISC (%) F	ISC (%) P	Province F	Province P	Interaction F	Interaction P
Mean (°C)	35.2	<0.0001	134.7	<0.0001	7.2	0.0079
Maximum (°C)	51.0	<0.0001	60.8	<0.0001	6.5	0.0113
$\log_{10}$(surge duration)†	70.2	<0.0001	5.2	0.0237	5.5	0.0196
Number of days with a surge per summer	84.8	<0.0001	1.0	0.3199	3.8	0.0529

Note: In each case, ISC in the 200-m riparian buffer zone proved to be the best scale fit among models.
† Measured in hours.

streams exhibited higher concentrations of SO_4, DOC, and TP (Table 3). Impervious surfaces appeared to cause TP to increase in the Piedmont, but not in the Coastal Plain (the slope in the Coastal Plain was not significantly different from zero).

Results from the BC data set included some interactive effects between province and ISC gradients (Tables 3 and 4). ANCOVA models for hardness (df = 3, 102, $F = 59.4$, $P < 0.0001$, $r^2 = 0.67$; network buffer scale), TS (df = 3, 102, $F = 94.4$, $P < 0.0001$, $r^2 = 0.76$; network buffer scale), $\log_{10}$-transformed TP (df = 3, 49, $F = 5.1$, $P = 0.0041$, $r^2 = 0.25$; watershed scale) and SO_4 (df = 3, 102, $F = 13.8$, $P < 0.0001$, $r^2 = 0.50$; watershed scale) were statistically significant, while the model for TN was not (df = 3, 49, $F = 0.9$, $P = 0.4350$; watershed scale). Hardness and TS concentrations were not significantly different between provinces in rural streams yet both increased more along ISC gradients in the Piedmont. The TP model concurred with findings from MBSS streams; TP increased along an ISC gradient in the Piedmont, but not in the Coastal Plain. Only the ISC model term was statistically significant in the SO_4 model.

Trends observed in MDE models largely reflected those of the MBSS data set (Tables 3 and 4). Conductivity (df = 3, 189, $F = 50.5$, $P < 0.0001$, $r^2 = 0.48$; watershed scale), TSS (df = 3, 189, $F = 3.3$, $P = 0.0248$, $r^2 = 0.13$; network buffer scale), and TN (df = 3, 76, $F = 35.7$, $P < 0.0001$, $r^2 = 0.62$; network buffer scale) were statistically significant as well as some terms in the $\log_{10}$-transformed DOC (network buffer scale) and $\log_{10}$-transformed chlorophyll *a* (network buffer scale) mixed models. TN concentrations and conductivity differed between provinces in rural streams, and both increased along ISC gradients but neither model included a significant interaction term. In contrast, $\log_{10}$-tranformed DOC and $\log_{10}$-transformed chlorophyll *a* concentrations were significantly higher in Coastal Plain rural streams and only increased along an impervious surface cover gradient in the Piedmont.

Temperature change

Most temperature attributes were affected by ISC to a greater degree in Piedmont streams. For each temperature variable considered, the network buffer was the selected scale of ISC and at least one mixed model term was found to be significant (Table 5). Mean and

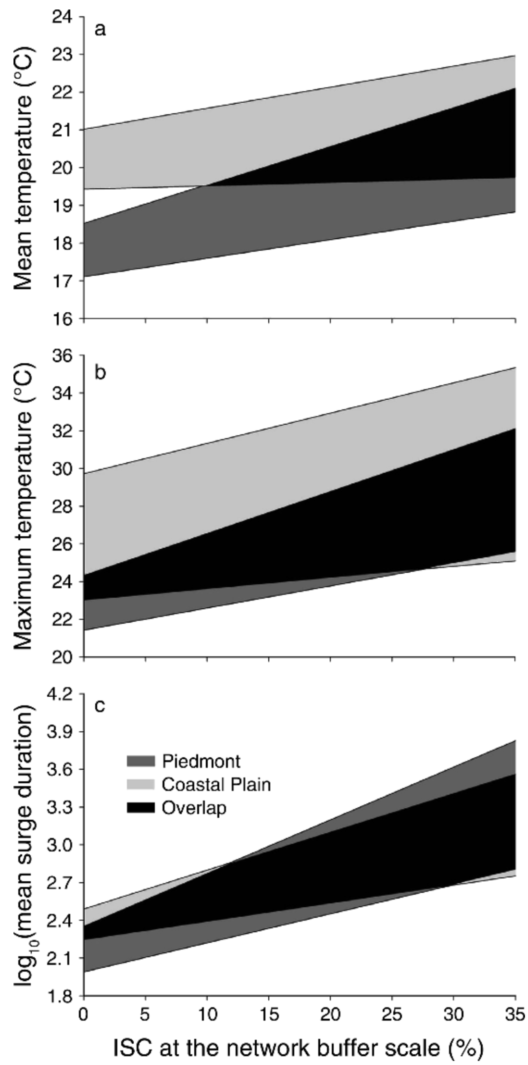

FIG. 5. Relationships between impervious surface cover and (a) mean, (b) maximum, and (c) surge duration of summer water temperature delineated by physiographic province. Because data were affected by natural fluctuations among years, model parameters are illustrated rather than raw data. Shaded areas represent 95% confidence intervals of regression parameters. For panel (c), units for surge duration were originally in minutes.

Table 6. Permutation procedure details for high-flow event ANCOVA models.

Variable and parameter	Nested removal		Random removal	
	Mean P	Iterations with $P < 0.05$ (%)	Mean P	Iterations with $P < 0.05$ (%)
Frequency of high-flow events (no./yr)				
ISC (buffer)	<0.0001	100	<0.0001	100
Province	<0.0001	100	<0.0001	100
Interaction	0.0018	100	0.0109	95.1
log(mean duration of high-flow events)†				
ISC (buffer)	<0.0001	100	0.0001	100
Province	<0.0001	100	<0.0001	100
Interaction	0.1081	2.8	0.1037	31.5
Maximum daily flow/watershed ($m^3 \cdot s^{-1} \cdot km^2$)				
ISC (buffer)	<0.0001	100	<0.0001	100
Province	0.0005	100	0.0007	100
Interaction	0.3415	0	0.0881	52.4

Notes: "Nested removal" refers to the iterative process where only nested watersheds were targeted for removal; the final random subset represented only independent sites (132 out of 170). "Random removal" represents iterations of the models where 132 sites were randomly chosen. Both processes represent 1000 iterations.

† Measured in minutes.

maximum temperatures were higher in rural Coastal Plain streams, yet each increased to a greater degree along the ISC gradient in Piedmont streams (Fig. 5, Table 3). Similarly, the mean duration of temperature surges increased along an ISC gradient but significantly more so in Piedmont streams. The only parameter that did not differ between provinces or exhibit variable relationships with ISC between provinces was the number of days in which a surge occurred.

Effect of nested watersheds

Permutation of ANCOVA models for high-flow event variables suggested that the presence of nested watersheds did not substantially affect statistical outcomes (Table 6). For high-flow event frequency, the P value for each model term was below 0.05 in the nested removal permutation procedure for 100% of the iterations, which matched conclusions using the full data set. Although the ISC and province terms were statistically significant in 100% of the iterations for high-flow event duration and maximum daily flow, the interaction term was not found to be statistically significant for the majority of iterations. Randomly reducing the sample size produced a similar effect, however.

Discussion

Our results demonstrate that the magnitude of urbanization-induced physicochemical change in streams may be strongly influenced by physiography. As biological assessments (Morgan and Cushman 2005, Goetz and Fiske 2008, Utz et al. 2009, 2010) have suggested may be the case, thermal properties of streams are more affected by watershed imperviousness in the Piedmont than in the Coastal Plain. Contrary to what we expected, however, hydrologic attributes associated with high-flow events are apparently affected by watershed imperviousness to a greater degree in Coastal Plain systems. In further contrast, few chemical parameters showed province-specific change along gradients of impervious surface cover. Therefore, it appears that physicochemical degradation caused by urbanization may vary between geoclimatic settings, but not necessarily in a uniform and predictable manner among environmental attributes.

The disparate patterns of change along ISC gradients caused interregional homogenization of many physicochemical properties. For example, temperatures in naturally cooler Piedmont streams were more strongly elevated by impervious surfaces relative to those in the Coastal Plain. As a result, temperature regimes became increasingly similar between regions as urbanization increased. Comparable patterns of interregional congruence with increasing ISC were also observed for hydrologic attributes associated with high-flow events. Urban development appears to cause a loss of thermal and hydrologic regime diversity at the interregional spatial scale. Such trends make intuitive sense: as urbanization increases, impervious surfaces prevent water from interacting with the surficial and shallow geologic attributes that render physiographic regions distinct. The loss of physicochemical diversity parallels, and could potentially contribute to, the homogenization of biotic assemblages at the landscape scale also observed in urban streams (Roy et al. 2005, Scott 2006).

The dissimilar hydrogeologic properties of the two provinces suggest a possible mechanistic explanation for the observed differences in hydrologic response to ISC. First, a comparison of published results from long-term intensive watershed studies conducted in the Piedmont (Dougherty et al. 2007) and Coastal Plain (Correll et al. 1999) provinces suggest that the observed differences in hydrologic responsiveness at zero imperviousness (Fig. 4) should not be attributed to differences in hydroclimatology. Dougherty et al. (2007) reported long-term (1979–2002) annual precipitation and runoff for four gaged headwater basins in the Occoquan River watershed (Piedmont) were 983 mm and 353 mm, respectively, while Correll et al. (1999) reported similar values (1080 mm and 332 mm, respectively) for six sub-watersheds in the nearby Rhode River watershed (Coastal Plain) for the 25-year period from 1972 to 1996. The modest differences (<10%) in hydroclimatological conditions between these systems appear inconsistent with the magnitude of the hydrologic differences shown in Table 3. Rather, the shallow basement rocks, relatively steep gradients, and the presence of soils with relatively low infiltration capacity in the Piedmont (Markewich et al. 1990, Swain et al. 2004) likely facilitate frequent, high magnitude, short duration high-flow events associated with moderate precipitation as is evident in our results. In contrast, the extensive depth of unconsolidated sediments, relatively low topographic relief, and relatively pervious soils of the Coastal Plain (Markewich et al. 1990, Ator et al. 2005) may attenuate to some degree the small, recurrent floods that characterize the average response of Piedmont streams. Therefore, replacing natural or agricultural land with impervious surfaces in the Coastal Plain induces greater hydrologic impact relative to the Piedmont.

Furthermore, our findings generally concur with previous comparative hydrologic assessments. Regional flood frequency equations developed for Maryland streams by Dillow (1996) predict higher flood magnitudes for Piedmont streams relative to similarly sized Coastal Plain counterparts. For comparable 26 km^2 (10 square miles) watersheds with 100% forest cover, the peak discharge with a two-year return period for a Piedmont stream is about 16 m^3/s (560 cubic feet per second), compared to 5.8 m^3/s (210 cubic feet per second) for a stream in the Western Coastal Plain. Such findings may not be surprising, given the fact that our results and the empirical models of Dillow (1996) are based on field data from some of the same USGS gaging stations in the two provinces. A portion of our results and those of Dillow (1996) appear to somewhat contrast findings reported by Jordan et al. (1997*b*), who concluded that the proportion of the annual budget exported as baseflow was higher in the Piedmont compared to the Coastal Plain. Yet this study included a substantial number of central and southern Eastern Shore watersheds, a region that was excluded from our analyses. Further, our finding that Coastal Plain high-flow events are longer in duration agrees with the conclusions of Jordan et al. (1997*b*). Regardless, our results confirm the importance of considering the regional hydrogeologic context when assessing hydrologic shifts resulting from land use change (Poff et al. 2006*a*, *b*, Chang 2007).

Differences in temperature regime response to ISC gradients between provinces may be explained by the hydrogeologic, geomorphic, and climatic attributes that characterize each region. One means by which temperatures are elevated in urban streams is the delivery of precipitation-generated runoff that has moved over impervious surfaces warmed by solar radiation (Herb et al. 2008). Piedmont streams are naturally cooler, possibly due to spring seepage from basement rocks that exist only in the Piedmont and/or by the slightly lower air temperatures (0.5–1.0°C difference by monthly average) relative to the Coastal Plain (data *available online*).[9] Therefore, floodwater runoff must reach a higher temperature to produce a detectable thermal effect in naturally warm Coastal Plain streams, which offers a mechanistic explanation as to why Piedmont streams are relatively more thermally impacted by urbanization. A second means of temperature elevation in urban streams is the removal of canopy cover that enhances the exposure of sediments in the wetted channel to solar radiation (LeBlanc et al. 1997, Krause et al. 2004). Coastal Plain streams tend to be deeper and water is often colored by dissolved organic material (blackwater streams; Mallin et al. 2004), both properties that may buffer solar radiation-driven sediment conduction (Caissie 2006, Houser 2006, Webb et al. 2008). The above mechanistic explanations of ecoregion-specific thermal responses to urbanization are speculative and require further research. Regardless, our findings pertaining to thermal properties further highlight the need to consider regional context when assessing impact in urban streams.

Contrary to hydrologic and thermal regimes, most chemical properties changed to a similar degree along impervious surface gradients between regions. The majority of chemical concentrations in watersheds lacking ISC differed significantly between the two provinces as previously observed (Kaufmann et al. 1991, Zipper et al. 2002, Stoddard et al. 2006). However, only total phosphorus was found to be consistently divergent in response to impervious surface gradients (with concentrations affected only in Piedmont streams) among data sets. A similar lack of interactive effect was observed in earlier work (Liu et al. 2000) between the same two regions, including some chemical species not tested in the current study though with a limited number of urbanized streams. Such an absence of significant differences between slopes is surprising, as interactive effects have been noted with other land use gradients. For instance, baseflow nitrate concentrations increased more per areal unit of agriculture in Piedmont streams relative to those of the Coastal Plain (Jordan et al. 1997*a*, *b*). Brown et al. (2009) and Cuffney et al. (2010) observed substantial heterogeneity in correlations between chemical responses and urbanization between nine metropolitan regions, though their findings from Piedmont watersheds generally concur with ours. The lack of province–urban gradient interaction could be attributable to the nature of the chemical data, as the majority of samples were collected during baseflow whereas most province-specific responses to urbaniza-

[9] ⟨http://www.sercc.com/climateinfo/historical/historical_md.html⟩

tion appear to be (directly or indirectly) related to high-flow events.

The diversity in physicochemical response to urbanization suggests that conservation and restoration efforts need to consider physiographic context. Heterogeneity in stream responses to urbanization among ecoregions increasingly appears to be consistent (Brown et al. 2009, Cuffney et al. 2010), therefore, a management strategy targeting one or multiple ecosystem components will likely require consideration of the physiographic context. Watershed management within a physiographic province could target those physicochemical attributes that are acutely vulnerable to change in order to increase the likelihood of maintaining a natural environment in an urban setting. Further management insight could be gained by exploring the province-specific geomorphic responses to urbanization, as many restoration efforts target habitat improvement or bank stability (Bernhardt et al. 2005) and characteristic geomorphic properties of stream ecosystems can differ substantially between physiographic provinces.

Our analyses did not consider, or treated coarsely, many factors that complicate the effects of urbanization on streams. Agricultural development may impact multiple abiotic properties in streams, especially thermal and chemical properties such as nutrients (McTammany et al. 2007), though other effects (i.e., hydrologic) are subtle in comparison to urbanization (Poff et al. 2006*a*). While excluding watersheds with $\geq$40% agriculture (at the network buffer scale for temperature analyses) where appropriate reduced the likelihood that agriculture affected our results, sites with <40% could have been affected by agriculture to some degree in addition to ISC (King et al. 2005). The disparity in watershed size distributions among data sets also precludes some conclusions. For instance, it is unclear if the patterns in hydrologic responses to urbanization would be consistently observed in small (<10 km^2) streams. Averaging hydrologic data across years was necessary to allow each site to contribute equally to statistical analyses but inevitably led to information loss. While high-flow event frequency and duration was robust to climatic variation among years (high-flow events were defined with respect to monthly medians), high-flow event magnitude was likely affected by climatic variability. Recent efforts have highlighted that impervious surfaces directly connected to stream channels, or effective impervious (EI) cover, may serve as a better predictor of degradation than total impervious cover (Walsh et al. 2005, 2009, Wenger et al. 2008). The large spatial extent of our sites and lack of EI models for the entire region negated consideration of EI as a predictor. However, our findings are meant to represent a coarse survey of comparative physicochemical responses to urbanization between regions. The consistent trends observed despite the nature of the data suggest that further investigation would support our findings and identify additional province-specific patterns.

In summary, our results highlight the need for further interregional examinations of urban land use-stream ecosystem relationships. Comparative regional approaches such as ours have consistently (Poff et al. 2006*a*, Sprague and Nowell 2008, Brown et al. 2009, Cuffney et al. 2010) found interregional variability in patterns of physicochemical response to urbanization. Similar trends are observed when other classes of land use such as row crop agriculture are examined (Jordan et al. 1997*a*, *b*, Liu et al. 2000). Considering the diversity of geoclimatic settings at continental spatial scales (for instance, the 84 delineated level-III ecoregions within the continental United States; United States Environmental Protection Agency 2010), the potential implications for watershed conservation and management are profound.

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Literature Cited

Arnold, C. L., P. J. Boison, and P. C. Patton. 1982. Sawmill Brook: an example of rapid geomorphic change related to urbanization. Journal of Geology 90:155–166.

Ator, S. W., J. M. Denver, D. E. Krantz, W. L. Newell, and S. K. Martucci. 2005. A surficial hydrogeologic framework for the Mid-Atlantic Coastal Plain. United States Geological Survey, Washington, D.C., USA.

Baker, M. E., D. E. Weller, and T. E. Jordan. 2006. Comparison of automated watershed delineations: effects on land cover areas, percentages, and relationships to nutrient discharge. Photogrammetric Engineering and Remote Sensing 72:159–168.

Baltimore County Department of Environmental Protection and Resource Management. 2008. Baltimore County national pollutant discharge elimination system: municipal stormwater discharge permit 2008 annual report. Baltimore County Department of Environmental Protection and Resource Management, Towson, Maryland, USA. ⟨http://resources.baltimorecountymd.gov/Documents/Environment/Annual%20Reports/2008npdes.pdf⟩

Bernhardt, E. S., et al. 2005. Ecology: synthesizing US river restoration efforts. Science 308:636–637.

Brown, L. R., T. F. Cuffney, J. F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A. H. Bell, and J. T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051–1069.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, New York, New York, USA.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389–1406.

Chang, H. J. 2007. Comparative streamflow characteristics in urbanizing basins in the Portland metropolitan area, Oregon, USA. Hydrological Processes 21:211–222.

Correll, D. L., T. E. Jordan, and D. E. Weller. 1999. Effects of interannual variation of precipitation on stream discharge from Rhode River subwatersheds. Journal of the American Water Resources Association 35:73–82.

Cuffney, T. F., R. A. Brightbill, J. T. May, and I. R. Waite. 2010. Response of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. Ecological Applications 20:1384–1401.

Dillow, J. J. A. 1996. Technique for estimating magnitude and frequency of peak flows in Maryland. Water-Resources Investigations Report 95-4154. U.S. Geological Survey, Towson, Maryland, USA.

Dougherty, M., R. L. Dymond, T. J. Grizzard, Jr., A. N. Godrej, C. E. Zipper, and J. Randolph. 2007. Quantifying long-term hydrologic response in an urbanizing basin. Journal of Hydrologic Engineering 12:33–41.

Goetz, S., and G. Fiske. 2008. Linking the diversity and abundance of stream biota to landscapes in the Mid-Atlantic USA. Remote Sensing of Environment 112:4075–4085.

Grable, J. L., and C. P. Harden. 2006. Geomorphic response of an Appalachian Valley and Ridge stream to urbanization. Earth Surface Processes and Landforms 31:1707–1720.

Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band, and G. T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393–403.

Herb, W. R., B. Janke, O. Mohseni, and H. G. Stefan. 2008. Thermal pollution of streams by runoff from paved surfaces. Hydrological Processes 22:987–999.

Houser, J. N. 2006. Water color affects the stratification, surface temperature, heat content and mean epilimnetic irradiance of small lakes. Canadian Journal of Fisheries and Aquatic Sciences 63:2447–2455.

Hughes, R. M., D. P. Larsen, and J. M. Omernik. 1986. Regional reference sites: a method for assessing stream potentials. Environmental Management 10:629–635.

Johnson, R. K. 2000. Spatial congruence between ecoregions and littoral macroinvertebrate assemblages. Journal of the North American Benthological Society 19:475–486.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1997*a*. Nonpoint source discharges of nutrients from piedmont watersheds of Chesapeake Bay. Journal of the American Water Resources Association 33:631–645.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1997*b*. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33:2579–2590.

Kaufmann, P. R., A. T. Herlihy, M. E. Mitch, J. J. Messer, and W. S. Overton. 1991. Stream chemistry in the eastern United States 1. Synoptic survey design, acid-base status, and regional patterns. Water Resources Research 27:611–627.

King, R. S., M. E. Baker, D. F. Whigham, D. E. Weller, T. E. Jordan, P. F. Kazyak, and M. K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15:137–153.

Klauda, R., P. Kazyak, S. Stranko, M. Southerland, N. Roth, and J. Chaillou. 1998. Maryland biological stream survey: a state agency program to assess the impact of anthropogenic stresses on stream habitat quality and biota. Environmental Monitoring and Assessment 51:299–316.

Knighton, D. 1998. Fluvial forms and processes: a new perspective. Oxford University Press, New York, New York, USA.

Konrad, C. P., D. B. Booth, and S. J. Burges. 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: consequences for channel form and streambed disturbance. Water Resources Research 41:W07009. [doi: 10.1029/2005WR004097]

Krause, C. W., B. Lockard, T. J. Newcomb, D. Kibler, V. Lohani, and D. J. Orth. 2004. Predicting influences of urban development on thermal habitat in a warm water stream. Journal of the American Water Resources Association 40: 1645–1658.

LeBlanc, R. T., R. D. Brown, and J. E. Fitzgibbon. 1997. Modeling the effects of land use change on the water temperature in unregulated urban streams. Journal of Environmental Management 49:445–469.

Lenat, D. R., and J. K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294:185–199.

Levene, H. 1960. Robust tests for the equality of variance. Pages 278–292 *in* I. Olkin, editor. Contributions to probability and statistics. Stanford University Press, Palo Alto, California, USA.

Liu, Z. J., D. E. Weller, D. L. Correll, and T. E. Jordan. 2000. Effects of land cover and geology on stream chemistry in watersheds of Chesapeake Bay. Journal of the American Water Resources Association 36:1349–1365.

Mallin, M. A., M. R. McIver, S. H. Ensign, and L. B. Cahoon. 2004. Photosynthetic and heterotrophic impacts of nutrient loading to blackwater streams. Ecological Applications 14: 823–838.

Markewich, H. W., M. J. Pavich, and G. R. Buell. 1990. Contrasting soils and landscapes of the Piedmont and Coastal Plain, eastern United States. Geomorphology 3: 417–447.

McTammany, M. E., E. F. Benfield, and J. R. Webster. 2007. Recovery of stream ecosystem metabolism from historical agriculture. Journal of the North American Benthological Society 26:532–545.

Moerke, A. H., and G. A. Lamberti. 2006. Scale-dependent influences on water quality, habitat, and fish communities in streams of the Kalamazoo River Basin, Michigan (USA). Aquatic Sciences 68:193–205.

Morgan, R. P., and S. E. Cushman. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society 24:643–655.

Nelson, K. C., and M. A. Palmer. 2007. Stream temperature surges under urbanization and climate change: data, models, and responses. Journal of the American Water Resources Association 43:440–452.

Olden, J. D., and N. L. Poff. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications 19:101–121.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Annals of the Association of American Geographers 77:118–125.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32: 333–365.

Pizzuto, J. E., W. C. Hession, and M. McBride. 2000. Comparing gravel-bed rivers in paired urban and rural catchments of southeastern Pennsylvania. Geology 28:79–82.

Poff, N. L., B. P. Bledsoe, and C. O. Cuhaciyan. 2006*a*. Hydrologic variation with land use across the contiguous United States: geomorphic and ecological consequences for stream ecosystems. Geomorphology 79:264–285.

Poff, N. L., J. D. Olden, D. M. Pepin, and B. P. Bledsoe. 2006*b*. Placing global stream flow variability in geographic and geomorphic contexts. River Research and Applications 22: 149–166.

Roy, A. H., M. C. Freeman, B. J. Freeman, S. J. Wenger, W. E. Ensign, and J. L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society 24:656–678.

Schiff, R., and G. Benoit. 2007. Effects of impervious cover at multiple spatial scales on coastal watershed streams. Journal of the American Water Resources Association 43:712–730.
Schoonover, J. E., B. G. Lockaby, and B. S. Helms. 2006. Impacts of land cover on stream hydrology in the west Georgia Piedmont, USA. Journal of Environmental Quality 35:2123–2131.
Schueler, T. R., L. Fraley-McNeal, and K. Cappiella. 2009. Is impervious cover still important? Review of recent research. Journal of Hydrologic Engineering 14:309–315.
Scott, M. C. 2006. Winners and losers among stream fishes in relation to land use legacies and urban development in the Southeastern US. Biological Conservation 127:301–309.
Sprague, L. A., and L. H. Nowell. 2008. Comparison of pesticide concentrations in streams at low flow in six metropolitan areas of the United States. Environmental Toxicology and Chemistry 27:288–298.
Stein, E. D., and D. Ackerman. 2007. Dry weather water quality loadings in arid, urban watersheds of the Los Angeles basin, California, USA. Journal of the American Water Resources Association 43:398–413.
Stoddard, J. L., D. P. Larsen, C. P. Hawkins, R. K. Johnson, and R. H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications 16:1267–1276.
Swain, L. A., T. O. Mesko, and E. F. Hollyday. 2004. Summary of the hydrogeology of the Valley and Ridge, Blue Ridge, and Piedmont physiographic provinces in the eastern United States. United States Geological Survey, Washington, D.C., USA.
Theobald, D. M., S. J. Goetz, J. B. Norman, and P. Jantz. 2009. Watersheds at risk to increased impervious surface cover in the conterminous United States. Journal of Hydrologic Engineering 14:362–368.
Thornbury, W. D. 1965. Regional geomorphology of the United States. John Wiley and Sons, New York, New York, USA.
United States Environmental Protection Agency. 1983. Methods for chemical analysis of water and wastes. National Technical Information Service, Springfield, Virginia, USA. ⟨http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=30000Q10.txt⟩
United States Environmental Protection Agency. 2004. Wadeable stream assessment: field operations manual. United States Environmental Protection Agency, Office of Water and Office of Research and Development, Washington, D.C., USA.
United States Environmental Protection Agency. 2010. Level III Ecoregions of the United States. United States Environmental Protection Agency, Western Ecology Division, Corvallis, Oregon, USA. ⟨http://www.epa.gov/wed/pages/ecoregions/level_iii.htm⟩
Utz, R. M., R. H. Hilderbrand, and D. M. Boward. 2009. Identifying regional differences in threshold responses of aquatic invertebrates to land cover gradients. Ecological Indicators 9:556–557.
Utz, R. M., R. H. Hilderbrand, and R. L. Raesly. 2010. Regional differences in patterns of fish species loss with changing land use. Biological Conservation 143:688–699.
Vannote, R. L., and B. W. Sweeney. 1980. Geographic analysis of thermal equilibria: a concept-model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. American Naturalist 115:667–695.
Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2009. Retention capacity: a metric to link stream ecology and storm-water management. Journal of Hydrologic Engineering 14:399–406.
Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706–723.
Wang, L. H., and P. Kanehl. 2003. Influences of watershed urbanization and instream habitat on macroinvertebrates in cold water streams. Journal of the American Water Resources Association 39:1181–1196.
Webb, B. W., D. M. Hannah, R. D. Moore, L. E. Brown, and F. Nobilis. 2008. Recent advances in stream and river temperature research. Hydrological Processes 22:902–918.
Wenger, S. J., J. T. Peterson, M. C. Freeman, B. J. Freeman, and D. D. Homans. 2008. Stream fish occurrence in response to impervious cover, historic land use, and hydrogeomorphic factors. Canadian Journal of Fisheries and Aquatic Sciences 65:1250–1264.
Zar, J. H. 1999. Biostatistical analysis. Prentice Hall, Upper Saddle River, New Jersey, USA.
Zipper, C. E., G. L. Holtzman, P. F. Darken, J. J. Gildea, and R. E. Stewart. 2002. Virginia USA water quality, 1978 to 1995: regional interpretation. Journal of the American Water Resources Association 38:789–802.