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The scaling of urban surface water abundance and impairment with city size

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Urbanization alters surface water compared to nonurban landscapes, yet little is known regarding how basic aquatic ecosystem characteristics, such as the abundance and impairment of surface water, differ with population size or regional context. This study examined the abundance, scaling, and impairment of surface water by quantifying the stream length, water body area, and impaired stream length for 3520 cities in the United States with populations from 2500 to 18 million. Stream length, water body area, and impaired stream length were quantified using the National Hydrography Dataset and the EPA's 303(d) list. These metrics were scaled with population and city area using single and piecewise power-law models and related to biophysical factors (precipitation, topography) and land cover. Results show that abundance of stream length and water body area in cities actually increases with city area; however, the per person abundance decreases with population size. Relative to population, impaired stream length did not increase until city populations were >25,000 people, then scaled linearly with population. Some variation in abundance and impairment was explained by biophysical context and land cover. Development intensity correlated with stream density and impairment; however, those relationships depended on the orientation of the land covers. When high intensity development occupied the local elevation highs (+15 m) and undeveloped land the elevation lows, the percentage of impaired streams was less than the opposite land cover orientation (−15 m) or very flat land. These results show that surface water abundance and impairment across contiguous US cities are influenced by city size and by biophysical setting interacting with land cover intensity.

1. Introduction

Over the last 50 years, urban land cover spread over large areas of the United States and the surface waters therein (Lubowski et al., 2006; Schneider et al., 2009). Urbanization changed the chemistry, hydrology, geomorphology, and ecology of surface waters, in addition to burying and removing many urban headwater streams, which were replaced by grey infrastructure and reduced the functional capacity of urban watersheds (Elmore and Kaushal, 2008; Roy et al., 2009; Beaulieu et al., 2015; Broadhead et al., 2015). This widespread degradation of urban aquatic ecosystems, known as *urban stream syndrome*, has common characteristics; however, the causes and solutions are not homogenous (Paul and Meyer, 2001; Walsh et al., 2005; Booth et al., 2016; Hale et al., 2016). To date, the majority of studies on urban surface waters only include one or a few large cities (Boone et al., 2012), with few examining patterns in abundance across macroscales or multiple cities (Steele and Heffernan, 2014; Hale, 2016), and none examining the patterns of surface water impairment among cities. The social, economic, and biophysical processes that control the resilience of urban ecosystems operate at local and broad scales (Alberti and Marzluff, 2004). Therefore, a broader assessment of surface water abundance and impairment is essential to understanding the consequences of urban population growth and land cover change (Foley et al., 2005; Grimm et al., 2008) and to developing management and remediation strategies that create resilient urban aquatic ecosystems across a wide range of geophysical settings (Violin et al., 2011; Pickett et al., 2014).

Despite the broad geomorphic and hydrographic restructuring of urban watersheds, how variation in watershed hydrography and city size contributes to changes in water quality remains unclear. Water quality of urban surface waters is generally degraded compared to undeveloped watersheds because of changes in impervious surface, hydrology, and chemical inputs (Paul and Meyer, 2001; Meyer et al., 2005; Walsh et al., 2005; Steele et al., 2010). Yet few studies directly compare water quality among cities at the regional to continental scale (Brown et al., 2009). Factors that influence water quality vary with city size and interact with differing geophysical setting. For example, stream burial and stormwater infrastructure have been shown to influence water quality (Pennino et al., 2014; Beaulieu et al., 2015; Hale et al., 2015); however, stormwater infrastructure can vary with

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city size and age of development (Hale, 2016). Despite the ubiquitous degradation, geographically extensive analysis of urban water quality across city sizes is not available.

The best available data on degradation of urban waters comes from the regulatory process. Section 303d of the Clean Water Act (CWA) of 1972 mandates the identification and remediation of impaired waters by the states. In general, for water bodies to be listed, states monitor water quality; and when a certain threshold of water samples exceed limits of what is defined as impaired, an action plan is created to address sources of pollution causing the impairment. It is important to recognize, however, that the specifics of each step in this process are variable by state (Keller and Cavallaro, 2008). While the 303d listed streams are certainly filtered through this sociopolitical process and not inclusive of all degraded waters, it does provide the only available aggregated data at the national scale to begin investigating these larger patterns of impairment. Most waters are impaired because of one of five issues: pathogens and metals, excess nutrients, sediments or organics (Sudduth et al., 2007). Patterns of listed streams can provide insight into the geophysical affecting water quality and sociopolitical processes that leads to official listing as impaired.

Degradation and impairment of surface waters in urban watersheds is often linked to the building of infrastructure and impervious surfaces. Urban developers use relatively homogeneous construction and design standards to transform agricultural and undeveloped ecosystems into cities (Alberti, 2005; Roach et al., 2008). As a result, at the national scale, many components of urban ecosystems are more similar to each other than are their native landscapes (Groffman et al., 2014; Hall et al., 2015), including the abundance of surface water and the geomorphology of restored streams (Steele et al., 2014; Doyle et al., 2015). The abundance of surface water in cities tends to converge on relatively dry conditions; and the more intense the development, such as commercial and multifamily housing, the drier the conditions (Steele et al., 2014). Multiple studies have found *stream deserts* within large cities, where the streams have been buried or removed (Elmore and Kaushal, 2008; Roy et al., 2009; Napieralski and Carvalhaes, 2016). Given that larger cities have larger areas of intense development, the relative abundance of surface water for the city as a whole would decrease as population size increased. However, no research has examined how surface water abundance may differ from smaller cities to large cities. Other parts of cities - such as green space, houses, or the gross domestic product - scale in a nonlinear relationship with population size (Bettencourt et al., 2007; Fuller and Gaston, 2009). The scaling of surface waters with population also could provide insights into how surface waters differ with city size and into the organizing processes that lead to the restructuring of urban watersheds.

In addition to population size and land cover, biophysical factors such as climate and topography influence urban surface waters. Urban geomorphology has received considerable attention from engineers and urban planners because the angle of slope affects the identification of sites for development and the landscaping (the reshaping of earth's land surface) required for urban development (Csima, 2010). During construction, developers often alter the local topography to create more suitable surfaces for construction and residential/commercial lands (Tenenbaum et al., 2006; Csima, 2010; Jones et al., 2014). These development decisions and construction may differentially alter the surface water hydrography of varying topography forms. For example, in a rolling-hill piedmont landscape (mean slopes ranged from 8 to 13%), Jones et al. (2014) observed drastic geomorphic alterations in two urban watersheds compared to a forested reference watershed. Development reduced smoothly varying hillslopes and sharply increased topographic breaks between flat and steep slopes (Jones et al., 2014). Jones et al. (2014) proposed that the changes in topography alone were substantial enough to disconnect riparian areas from stream waters.

The local geomorphology also influences the location and extent of land covers and green space within cities. High intensity urban development, with stricter engineering requirements, is preferentially located on the local flat land and on the elevation highs (ridges) or lows (valleys) (Csima, 2010). Steeper slopes are more expensive to develop for residential or commercial uses; however, they may be more suitable for parks and natural preserves. In Sheffield, England, elevation was positively correlated with the extent of green space (Davies et al., 2008). In flatter landscapes, however, urban open area and green space may be preferentially located around local lows because of the increased wetness and surface water features, which hinder other types of land uses.

The objective of this study is to examine the broad patterns and quantify the abundance of total streams, waterbodies, and impaired streams for all cities of the United States. Specifically, I i) determined how surface water abundance and impairment scales with population and city area, and ii) examined how biophysical factors (climate, topography), land cover, and their interaction relate to the impairment of streams waters.

2. Methods

2.1. City selection, surface water, and impaired streams

This study included 3520 cities of the continental United States. Cities were defined by the U.S. Census Bureau as *urban areas* (UAC), which include all continuous census block groups for a city; and TIGER shapefiles were downloaded from the U.S. Census Bureau. By this definition, a single *city* could encompass multiple political cities within one contiguous unit. This delineation was selected because it allows for analysis of the morphological city in its entirety, without including large rural areas that are included in Metropolitan Statistical Areas. Population for each city was calculated by summing the 2010 population of all the census blocks intersecting the urban areas. The population ranged from 2500 to 18 million people. Population density was calculated by dividing the total population by the total area of the city. AcrGIS 10.2 was used for all spatial analyses.

Data for total stream length and water body area (lakes, ponds, wetlands) for each city were attained from the National Hydrography Dataset (NHD) (USGS, 2012a). The flowline data layers were intersected with the urban area to determine the total stream length and area of water bodies within each city. The national coverage of high resolution data was produced at a minimum of 1:24,000 and maximum of 1:5000 scales, which varies by state. Although at the national scale the NHD is the best available data, notably stream length could be missing. The number of total streams missing from the medium resolution NHD (1:100,000 scale) may be greater than half in some cities (Roy et al., 2009), with the majority of missing streams being intermittent and ephemeral. The high resolution improves this margin of error, but a number of streams are still likely to be missing. In addition, the NHD does not often include buried streams and many other human constructed flow paths that similarly shed water from the landscape like streams. While certainly important to characterizing urban watersheds, the lack of data at this scale inhibits the inclusion of these nontraditional types of streams in these estimates at this time. Given the wide geographic extent and the very large number of cities included in this study, the absolute number of streams and water bodies may differ, but the relationships between them still provides useful insights into how urban hydrography and impairment differs with city size.

To characterize stream impairment, the EPA's 303(d) list of impaired streams was obtained from the EPA as shapefiles (https://www.epa.gov/waterdata/waters-geospatial-data-downloads#303dListedImpairedWaters, current as of 1 May 2015). These data do not represent all streams impacted by urban development, but only a subset officially listed on the 303(d). The language of the CWA is general enough that criteria and implementation of the guidelines varies considerably among states; however, some of that flexibility allowed states to develop criteria tailored to the geophysical conditions within the state (Keller and Cavallaro, 2008). Despite the heterogeneity at the state level, the 303(d) list is the best database currently available that aggregates water quality issues at the national level. While identification of impaired waters varies by state, no evidence suggests that the processes lead to a population size bias. Impaired stream lengths were intersected with the urban areas to calculate the total length of impaired flowlines for each city. Urban development affects downstream waters beyond city borders, but the extent to which any city contributes to downstream impairment is unknown. Accounting for externalized impairment is beyond the scope of this study.

2.2. Land cover, topography, precipitation

Land cover in each city was also characterized using the 2011 National Land Cover Data (NLCD). The NLCD categories are based on percent impervious surface area and land use (Fry et al., 2011). The original 15-category NLCD was reclassified into seven classes: urban open area (NLCD = 21), urban low intensity (NLCD = 22), urban medium intensity (NLCD = 23), urban high intensity (NLCD = 24), agriculture (NLCD = 81, 82), wetlands (NLCD = 90 and 95), and undeveloped (all remaining NLCD). Urban open area land cover includes parks, golf courses and other spaces that are developed (i.e., the natural vegetation removed/altered) but not necessarily built up, as well as very low density housing. Low intensity land cover included residential single family homes and other low density development. Medium intensity development includes multifamily residential development and some business districts. High intensity development includes commercial, industrial, and other highly developed areas. The undeveloped category includes regions designated as forest, scrub, or desert, depending on the region and climate. The whole city land cover composition was calculated using the *zonal statistics* tool. In addition to the whole city, the land cover composition of a 100-m buffer on either side of all streams and impaired streams was calculated. A recent review found that buffers from 25 to 40 m on stream headwaters are needed to protect water quality, habitat, and organisms (Sweeney and Newbold, 2014). The choice of 100 m for this study was a compromise between analysis constraints and effective buffers. Streams and rivers in the NHD are vector lines with no width. For wider features, a smaller buffer width would have not been able to capture the development surrounding the rivers.

The topography was characterized using 30 m digital elevation model (DEM) from the 3DEP seamless DEMs (USGS, 2012b). The NLCD raster for the contiguous U.S. was clipped by the UAC boundaries. Because of processing constraints, each land cover was separated into individual files using the raster calculator. Then the DEMs were divided by the land cover raster to isolate the elevations for each land cover. The zonal statistics tool was then used to calculate the mean and standard deviation of elevation for each land cover in every city. Mean slope for the city was calculated using the slope tool in the Spatial Analysis Toolbox from the 30-m DEM, and zonal statistics were used to calculate the mean slope. The slope tool calculates the slope of each 30 × 30 m pixel based on the maximum rate of change in values from one cell to its neighbors (Burrough et al., 2015). Precipitation data was calculated using the 1981–2010 Mean Annual Precipitation (MAP) published in 2012 by the Prism Climate group (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu) made available by USDA/NRCS National Geospatial Center of Excellence. Vector data was converted to a raster, then mean precipitation calculated from the zonal analysis tool using the city boundary as the zone.

2.3. Data analysis

Data was visualized first on linear and on log-log scale plots. In a preliminary analysis, a number of different model fits to data were examined (linear, logistic, exponential, quadratic, power law). Power law and linear models provided the best fit; however, the fit of linear models was driven by the results of the largest cities (data not shown), therefore the power-law model was selected. Previous studies found that power law distributions best characterize many attributes of cities (e.g. population, total area and green space), landscapes, and watersheds (Batty, 2008; Fehr et al., 2011; Fuller and Gaston, 2009; Goodchild, 1988; Klinkenberg and Goodchild, 1992). Therefore, total stream length, water body area, and impaired stream length were scaled with city population and city area using power-law models:

$$A = \alpha x^{\beta} \tag{1}$$

where A is the cumulative length of stream or water body area, x is the urban population or total urban area, α is the intercept, and β is the scaling exponent. The β exponent expresses the scaling relationships between surface water and population or area. The parameter β is either <1 (sublinear), $=1$ (linear or unity), or >1 (superlinear). To calculate the model, I fit a linear regression model to the log-transformed data (log(A) > log (x)). The scaling analysis for total stream length and water body area included all cities; however, the scaling of impairment was only performed on cities with greater than zero impaired streams ($n = 1868$). A representative number of cities across the entire population range had some impaired waters. This is a limitation of the approach; however, given the large number of cities still included, the approach provides valuable information on how impairment differs with population size. To test for population size bias by states with differing approaches in the 303d listing, two states with known aggressive listing strategies (Ohio and Michigan) were compared with two states with more relaxed approaches (Mississippi and North Carolina).

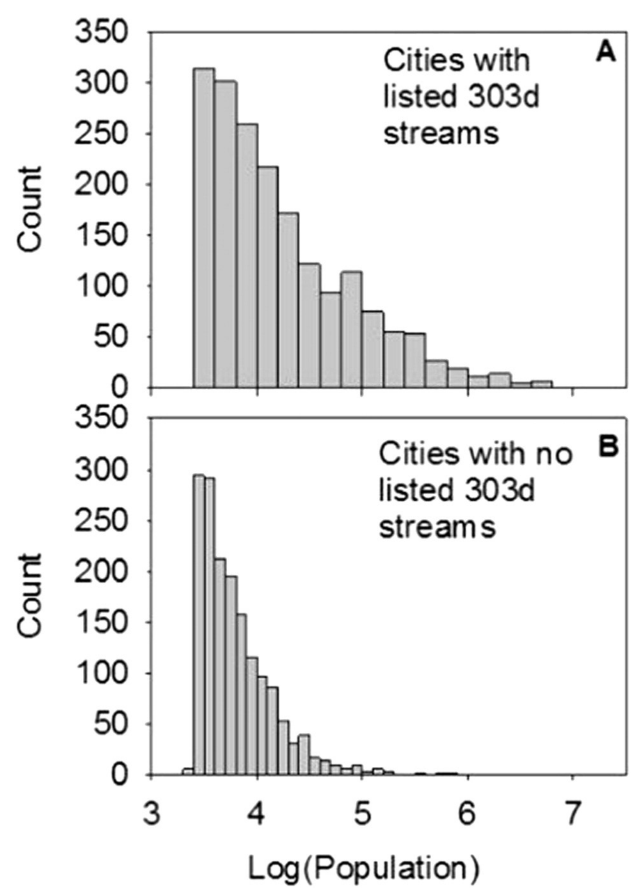

Fig. 1. Histogram of the number of cities across the population range from 2500 to 18 million (A) with streams listed on the EPA's 303d list and (B) with zero stream length listed on the 303d list.

A breakpoint (or piecewise) regression on log-log transformed data was used to assess the potential for slope changes in the surface water–city size relationship. This approach uses least mean squares to determine four parameters: the value of the predictor variable (in this case, population or area) at which the slope break occurs, and the estimated value of the response variable at the minimum, maximum, and threshold values of the predictor variable. These parameters allow for calculation of slopes (i.e., power-law scaling exponents) above and below the threshold populations. Two-segment and three-segment breakpoint models were assessed for each variable combination. Goodness of fit was assessed by evaluation of the R^2 parameter and visual inspection of the residuals.

To examine the relationship between biophysical factors and impairment, a regression approach was used to test for relationships between stream density and percent impairment with population density, percentage of land covers, precipitation, slope of the land, and

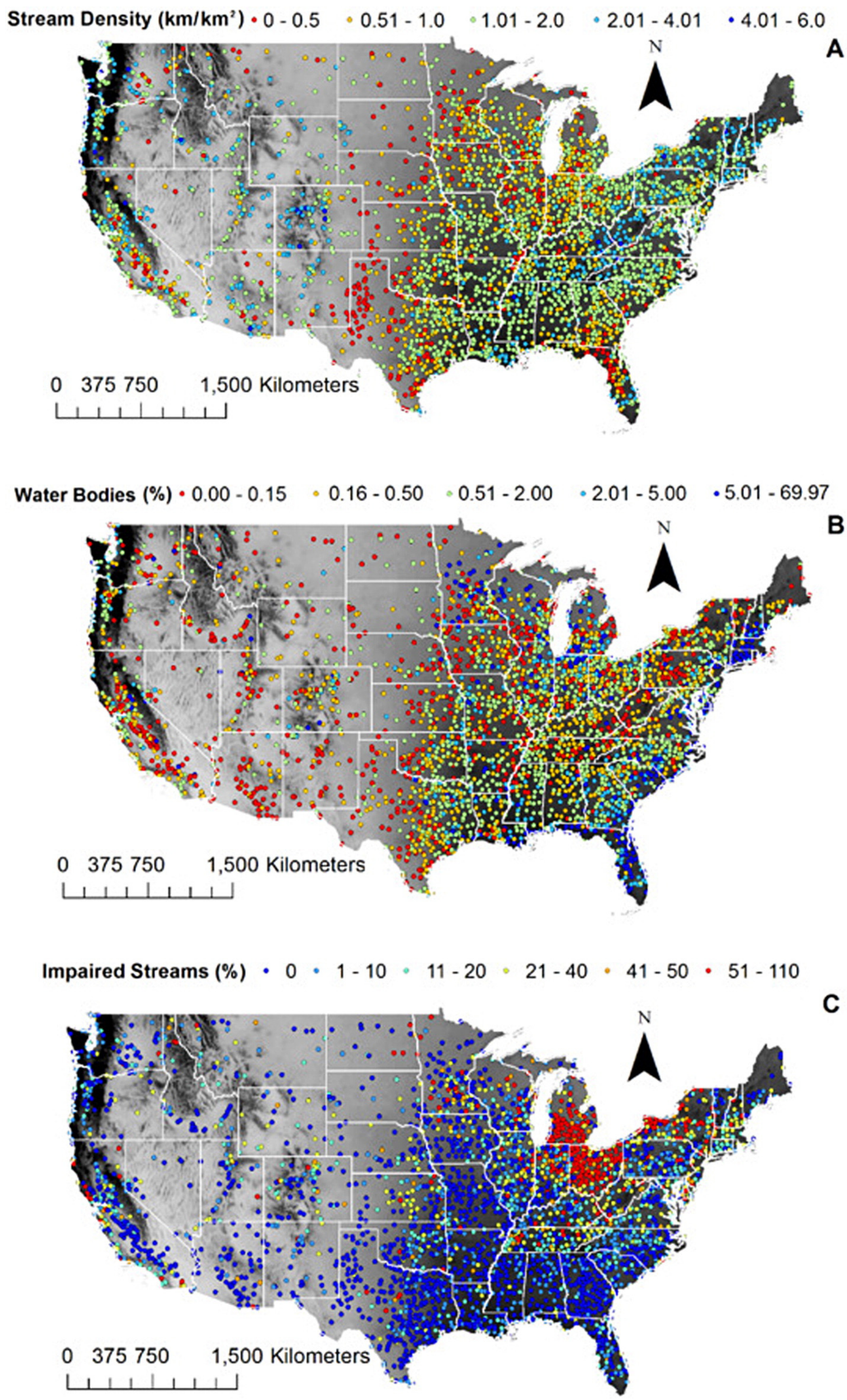

Fig. 2. Geographic distribution of cities (color points) with (A) total stream density, (B) percent area of water bodies, and (C) percentage of stream length impaired. Grey scale background is precipitation with state outlines.

the land cover composition of a 100-m buffer around the streams. In addition, the effects of the topographic position of land cover were analyzed using a moving window technique. Given the high number of observations, variance, and an expectation of nonlinear trend, simple plotting was ineffective for observing the response patterns. A binning approach would have introduced arbitrary delineations in expected outcomes. The moving window provided the clearest results of central tendency without creating artificial binning boundaries. The difference in elevation between the high intensity land cover and all other land covers was calculated for each city; however, only the difference between high intensity land cover and undeveloped land cover was included in this study. The data were then sorted by the difference in elevation. Next, for a window of 100 observations starting with the lowest, the mean percent impaired stream length and mean elevation difference were calculated. The window was moved by one observation until all 3520 observations were included. This created a series of means across the entire differential elevation range. This approach generated a finer grade examination of how topographic position affected the occurrence of impaired streams. All data analyses were conducted using SigmaPlot v12.

3. Results

Total urban stream length in the contiguous U.S. urban areas was 413,000 km, of which 66,800 km (16%) was identified as impaired. Of the 3520 cities included in this study, 53% of the cities included some length of impaired, 303d-listed stream waters. Smaller population cities were less likely to have impaired streams than larger cities (Fig. 1).

Urban stream density was geographically related to the slope of the land, with higher densities associated with mountainous regions (Fig. 2A). Water body area followed climatic and topographic gradients, with greater abundance in wetter and flatter locations. The length of impaired streams was also geographically distributed; however, these were less associated with biophysical gradients (Fig. 2C). The variation in the states approach to listing water bodies as impaired clearly influenced the geographic distribution of impaired streams across the country. States with greater percentages of impaired stream length include Michigan, Ohio, and New York. While cities of the southern U.S. (Georgia to Texas) have relatively little impaired urban stream length. This pattern may reflect evaluation processes that favor (or not) listing streams as impaired. However, southern cities tend to also have higher precipitation and a more diffuse development style that lead to lower population densities. The combination of climate and urban development style may lessen the impact of urbanization on water quality in this region.

3.1. Scaling of stream length, water bodies, and impaired streams

In general, larger cities have greater stream length per unit area than smaller cities. Population and urban area were significant predictors of stream length; however, in both cases the relationship was nonlinear (Table 1, Fig. 3). The relationship between total stream length and urban area was best described by a single power law model with a superlinear slope ($\beta = 1.07$; (Fig.3A). The superlinear relationship indicates that larger cities actually have a greater stream density than smaller cities do, by ~7% for every order of magnitude increase in city area. For population, the best model fit was a two transition, piecewise power-law model (T1 ~ 30,000 and T2 ~ 1 million people) and three scaling domains that transitioned from a linear ($\beta_1 = 1.02, \beta_2 = 0.95$) to a sublinear relationship ($\beta_3 = 0.67$; Fig. 3B). In other words, the length of stream per person does not differ much with city size until cities reach >1 million people, when the per-person length starts to decrease.

Similar to the scaling of streams, larger cities also have greater densities of lakes, ponds, and wetlands than smaller cities, but more variability among cities. Compared to streams, the area of water bodies increased faster with population and with city area (Fig. 4A, B). A significant transition point in both scaling relationships occurred at ~350 km^2 and 400,000 people. In smaller cities, water body area scaled superlinearly with area ($\beta = 1.40$) and with people ($\beta = 1.18$). In other words, the areal density of and the per-person area of water bodies increased with city size. In large cities, however, water bodies scaled linearly with area and sublinearly with population.

Like streams and water bodies, impaired stream length also increased as city size increased once populations are >25,000. Impaired stream length was best fit by piecewise power-law models with a single transition point (T1 = ~40 km or 25,000 people) for relationships with area and population (Fig. 5). In small towns below the threshold, the length of impaired streams scaled sublinearly such that population and area were both increasing faster than the length of impaired streams, but substantial variation occurred within this range. Above 25,000 people, the length of impaired streams increased linearly with population ($\beta = 0.98$) and super-linearly with area ($\beta = 1.10$). In other words, the per-person length of impairment remained similar between 25,000 and 18 million people, but the areal density of impaired streams increased.

Table 1
Statistical estimates of scaling relationships between population or area and the stream length, water body area, and impaired stream length for contiguous US urban areas.[a]

	Impaired streams		Total stream length		Water body area	
	Population	Area	Population	Area	Population	Area
n	1868	1868	3453	3453	3245	3245
Power-law						
Adj r2	0.428	0.427	0.721	0.814	0.430	0.535
a	0.224	−2.729	0.372	−3.483	0.522	−4.571
SE(a)	0.094	0.173	0.042	0.064	0.095	0.160
β	0.819	0.862	0.979	1.076	1.146	1.338
SE(β)	0.022	0.023	0.010	0.009	0.023	0.022
Piecewise (two segment)						
Adj r2	0.434	0.439	0.721	0.814	0.431	0.536
a	5.189	−0.674	0.200	−6.782	0.408	−4.870
β (lower)	0.612	0.572	1.024	1.564	1.175	1.380
β (upper)	0.978	1.105	0.917	1.043	0.805	0.956
y1	3.104	2.914	3.680	3.030	4.401	3.785
SE(y1)	0.034	0.054	0.012	0.043	0.022	0.029
y2	3.723	3.683	4.853	3.886	7.007	6.9258
SE(y2)	0.130	0.095	0.276	0.045	0.450	0.3191
y3	6.514	6.291	7.346	7.178	8.334	8.291
SE(y2)	0.113	0.099	0.067	0.027	0.289	0.202
T1	4.410	7.616	4.544	6.819	5.615	8.548
SE(T1)	0.162	0.109	0.280	0.039	0.394	0.241
Piecewise (three segment)						
Adj r^2	0.433	0.434	0.722	ns	ns	ns
a	0.444	0.664	2.158			
β (lower)	−0.305	−0.379	1.017			
β (middle)	1.027	0.718	0.948			
β (upper)	0.785	1.097	0.669			
y1	3.087	3.104	3.682			
SE(y1)	0.031	0.033	0.012			
y2	3.948	3.761	4.823			
SE(y2)	0.136	0.126	0.465			
y3	5.361	5.360	6.269			
SE(y3)	0.452	0.605	0.350			
y4	6.202	6.232	7.084			
SE(y4)	0.322	0.333	0.187			
T1	4.703	4.471	4.520			
SE(T1)	0.160	0.158	0.466			
T2	5.992	6.032	6.045			
SE(T2)	0.443	0.617	0.387			

[a] Statistical outputs are shown for single power-law ($\log[y] = A + \beta \log[x]$) and for power-law scaling with one- and two-segment piecewise regressions. Piecewise models estimated values of abundance at the minimum, maximum, and threshold values of population (y1, y2, y3, y4). Upper and lower slopes were calculated from piecewise model parameters. All model parameters are shown with error as (SE(x)). Models significant unless noted (ns).

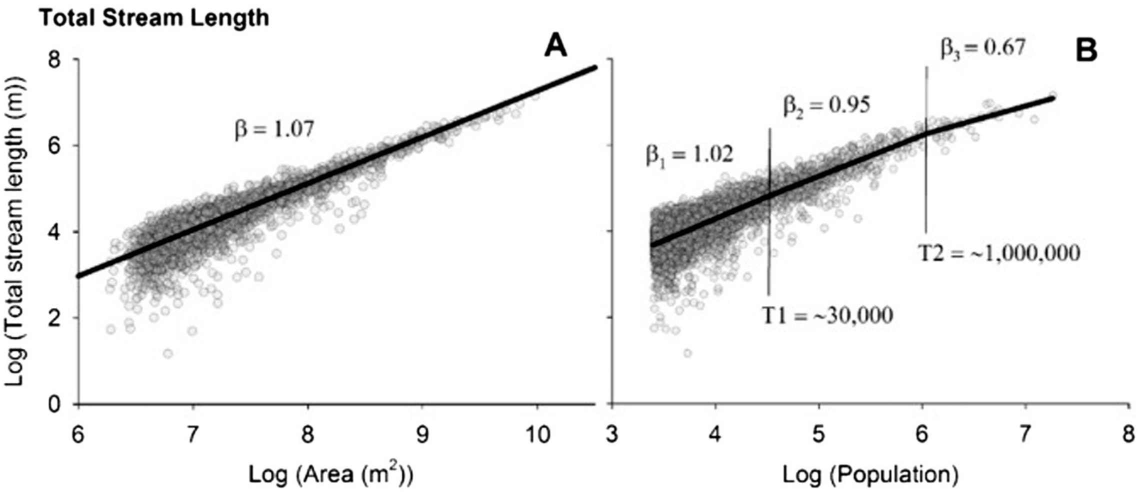

Fig. 3. Power-law scaling models ($y = ax^{\beta}$) describe a nonlinear relationship between the total stream length in cities and the (A) urban area and (B) population. The stream length–area relationship was best fit by a single power-law model. The stream length–population relationship was best fit by a three-segment, piecewise model with two transition points (T1 and T2), and three slopes (β_1, β_2, and β_3).

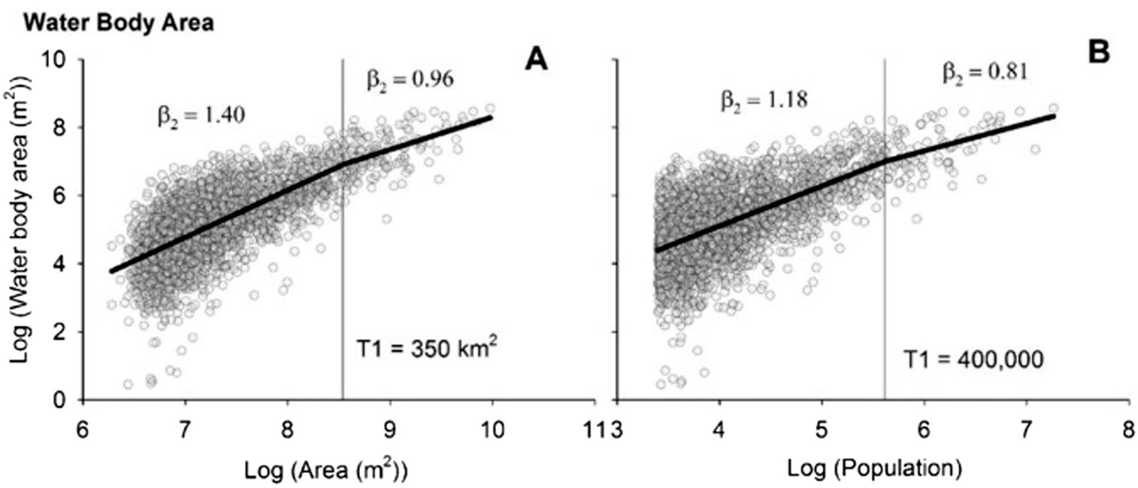

Fig. 4. Power-law scaling models ($y = ax^{\beta}$) describe a nonlinear relationship between total water body area in cities and (A) urban area and (B) population. Both relationships were best fit by a two-segment, piecewise model with one transition point (T1–350 km^2 or ~400,000 people), and two slopes (β_1, and β_2).

The scaling of impaired waters among states with differing approaches to the 303d listing was similar. Model slopes of aggressive programs (Michigan: $\beta = 0.96$; Ohio: $\beta = 0.98$) were similar to states with less aggressive programs (Mississippi: $\beta = 0.98$; North Carolina: $\beta = 0.97$). What differed was α (the y-intercept), where aggressive programs (Michigan: $\alpha = 1.00$, $R^2 = 0.54$; Ohio: $\alpha = 1.29$, $R^2 = 0.73$)

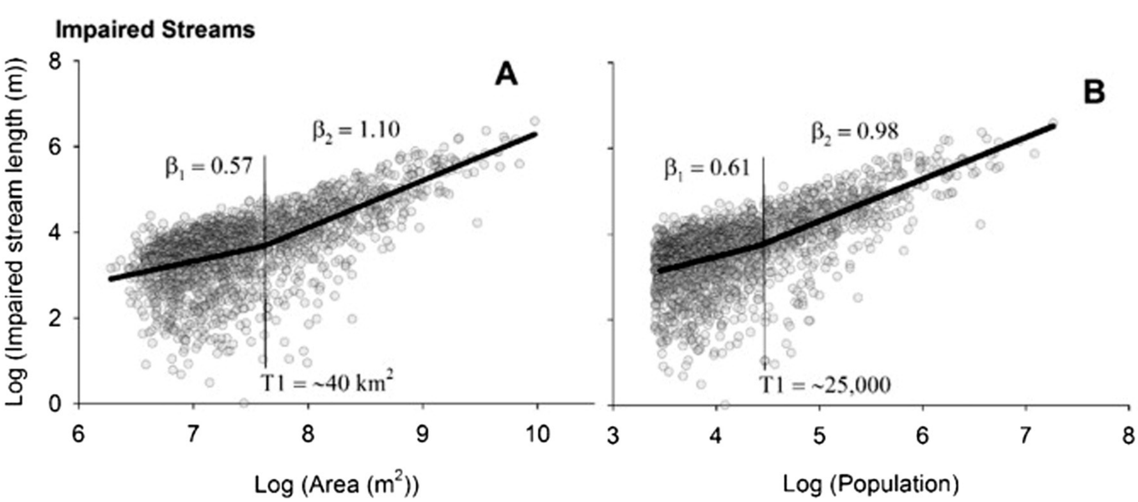

Fig. 5. Power-law scaling models describe a nonlinear relationship between the impaired stream length in cities and (A) urban area and (B) population. The relationships were best fit by a two-segment, piecewise model with one transition point (T1 ~ 40 km^2 or ~25,000 people) and two slopes (β_1, and β_2).

Table 2
Relationships between stream density and the percentage of stream length impaired and the population density, percent land cover, biophysical attributes, and land cover composition of a 100 m stream buffer (all relationships are significant unless otherwise noted, ns).

	Stream density	Adj R^2	*P* value	% Impaired streams	Adj R^2	*P* value
Population density	= 1644.9 − 0.37x	0.0180	<0.001	= 11.546 + 0.01x	0.0023	0.003
Land covers						
High intensity	= 1603.4 − 50.8x	0.0351	<0.001	= 12.1 + 0.57x	0.0042	<0.001
Medium intensity	= 1525.4 − 980.4x	0.0109	<0.001	ns		
Low intensity	= 1865.9 − 1708.2x	0.0793	<0.001	ns		
Open area	= 1594.7 − 911.9x	0.0141	<0.001	= 18.1 − 16.8x	0.0046	<0.001
Agriculture	ns			= 12.1 + 18.0x	0.0067	<0.001
Undeveloped	= 1218.3 + 944.5x	0.0330	<0.001	= 17.1 − 14.0x	0.0070	<0.001
Biophysical						
Slope	= 1122.8 + 117.2x	0.0979	<0.001	= 13.3 + 0.46x	0.0013	0.019
Precipitation	= 1290.7 + 2.9	0.0026	0.001	ns		

were larger than less aggressive programs (Mississippi: $\alpha = 0.0983$, $R^2 = 0.57$; North Carolina: $\alpha = 0.32$, $R^2 = 0.58$). Clearly, the aggressive states were capturing more degraded stream length, which explains some of the variability in the cumulative model. However, the similarity in slopes suggests that the trajectory of impairment with city size is robust despite the variation in approaches to impaired listing.

3.2. Land cover, biophysical properties, and stream impairment

Land cover composition was significantly related to the density and impairment of streams. Total stream density was positively related to the percentage of undeveloped land and negatively related to developed land covers (Table 2). Similarly, the percent of impaired streams increased with population density, high intensity and agricultural land covers, and slope (Table 2). Although these relationships were significant, the land covers alone accounted for very little of the variation across the 3520 cities, in most cases <5%. The land cover composition of the stream buffers reflected similar results as the composition of the whole city.

Compared to all streams, impaired streams tended to have larger proportions of high intensity land cover within the buffer; however, they also tended to have greater proportions of undeveloped land within the buffer than the broader stream network (Fig. 6). While greater proportions of high intensity land are likely to lead to worse impairment, a greater proportion of undeveloped land in the buffer may reflect restoration efforts to improve stream quality.

Regional biophysical factors, precipitation and topography, had a complex relationship with stream density and impairment. Overall, total stream density was also positively related to the slope of the terrain and precipitation (Table 2). Likewise, the percent stream impairment increased as the slope of the land increased (Table 2). But slope and precipitation explained very little of the variation in stream density or impairment (<10%).

The interaction between topography and location of land covers influenced the frequency of impaired streams. Across the U.S. cities, 685 cities had a *mound* orientation, where the high intensity development is located on the local highs (>5 m) and the undeveloped land occupies the local lows. While 1532 were *flat* cities, the difference between the high intensity and undeveloped land was <±5 m, and 1235 were *bowl* cities, where the high intensity land cover is located on the local lows and undeveloped land on the local highs. The other land covers generally fell between the two extremes (data not shown). The mean percentage of impaired streams tended to be lower for mound cities (5 to 10% impaired) compared to the flat (5 to 20% impaired) or bowl (10 to 20% impaired) cities (Fig. 7). The flat cities had the most variation, but whether this is because of actual effects of topography or other factors and variability in the data is uncertain.

4. Discussion

4.1. Surface water abundance from towns to cities

Despite their intense development, larger cities actually have greater densities of streams and water bodies than smaller cities. At local scales, cities are highly heterogeneous (Cadenasso et al., 2007) and previous studies document substantial losses of streams through burial and removal that create stream deserts in cities (Elmore and Kaushal, 2008; Roy et al., 2009; Napieralski and Carvalhaes, 2016). However, the results presented here indicate that larger cities have greater surface water per unit area compared to smaller cities. This does not contradict the presence of stream deserts in cities but rather suggests: (i) stream density is complex and heterogeneous at local scales in large cities with some areas that are water poor and others with relatively larger amounts of surface waters; and (ii) that stream deserts also occur in small cities, and as a whole, smaller cities may be more surface water poor than

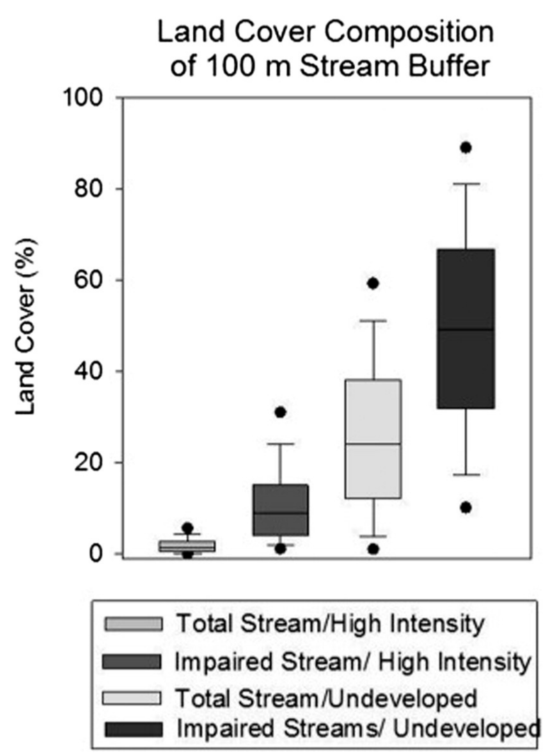

Fig. 6. The percentage of high intensity and undeveloped land cover in a 100-m buffer (200-m total) around the total stream network and impaired streams.

larger cities. Surface water from Indianapolis and surrounding towns illustrates the broader pattern (Fig. 8). The areal extent of Indianapolis is many times that of the small surrounding towns, with numerous stream reaches. The decrease in stream density in downtown Indianapolis and other highly developed suburbs is visible, but the wider city encompasses an extensive stream network. The smaller towns as a whole are much more water poor, as they are located within the stream network such that any burial or conversion to sewer systems removes a much larger percentage of the stream length.

Urban hydrographic patterns of small and large cities likely reflect the interaction of two mechanisms: site selection and alteration. Rivers and surface water have long influenced where people have settled and which cities grow (Cronon, 1992). Access to water for domestic, agricultural, commercial, and transportation activities creates the need to be close to water. However, flooding and other water-associated hazards, as well as the slope of the land, influence the positioning of small towns. Cities selectively locate on the best local *dry* land with fewer surface water features. So if small cities start with less surface water, the removal of streams would have a greater impact on the surface water density. As cities grow, they must contend with less desirable (aka more surface water) conditions, which they must either build around or remove. Therefore, even though cities remove many features (Elmore and Kaushal, 2008; Roy et al., 2009), large cities may end up incorporating more surface water into the urban matrix. The distribution of water body types in small, medium, and large cities suggests that selection may be more important in the initial stages of cities and that alteration becomes more important as cities grow outward from their original settlements (Steele and Heffernan, 2014). Larger populations and tax bases may increase demand for drinking water reservoirs and recreational opportunities and the need for stormwater management ponds and channels, all of which increase the relative density of water bodies in large cities.

Discussion of the role of the NHD data used here and errors of omission is important. Because of the very large scale of analysis, not all errors in the NHD could be corrected and how these errors may vary with city size has not been systematically analyzed. If mapping omissions increase in large cities because of the density of built structures and irregular flow paths, larger cities may have even more stream length and water body area than predicted here, which still contradicts some basic assumptions about surface water in cities. Development of large-scale databases at higher resolutions that encompass the wide diversity of nontraditional flow paths and hydrographic features would greatly improve our ability to characterize urban watersheds and estimate the ecological and ecosystem processes at larger scales.

The difference in surface water abundance may have important implications for the ecological and ecosystem processes in large and small cities. The abundance of surface water mediates fluxes of carbon, nitrogen, and phosphorus (Cole et al., 2007; Fraterrigo and Downing, 2008; Downing, 2010), as well as species abundance and distribution (Dunham and Rieman, 1999; Dahlgren and Ehrlen, 2005). The removal

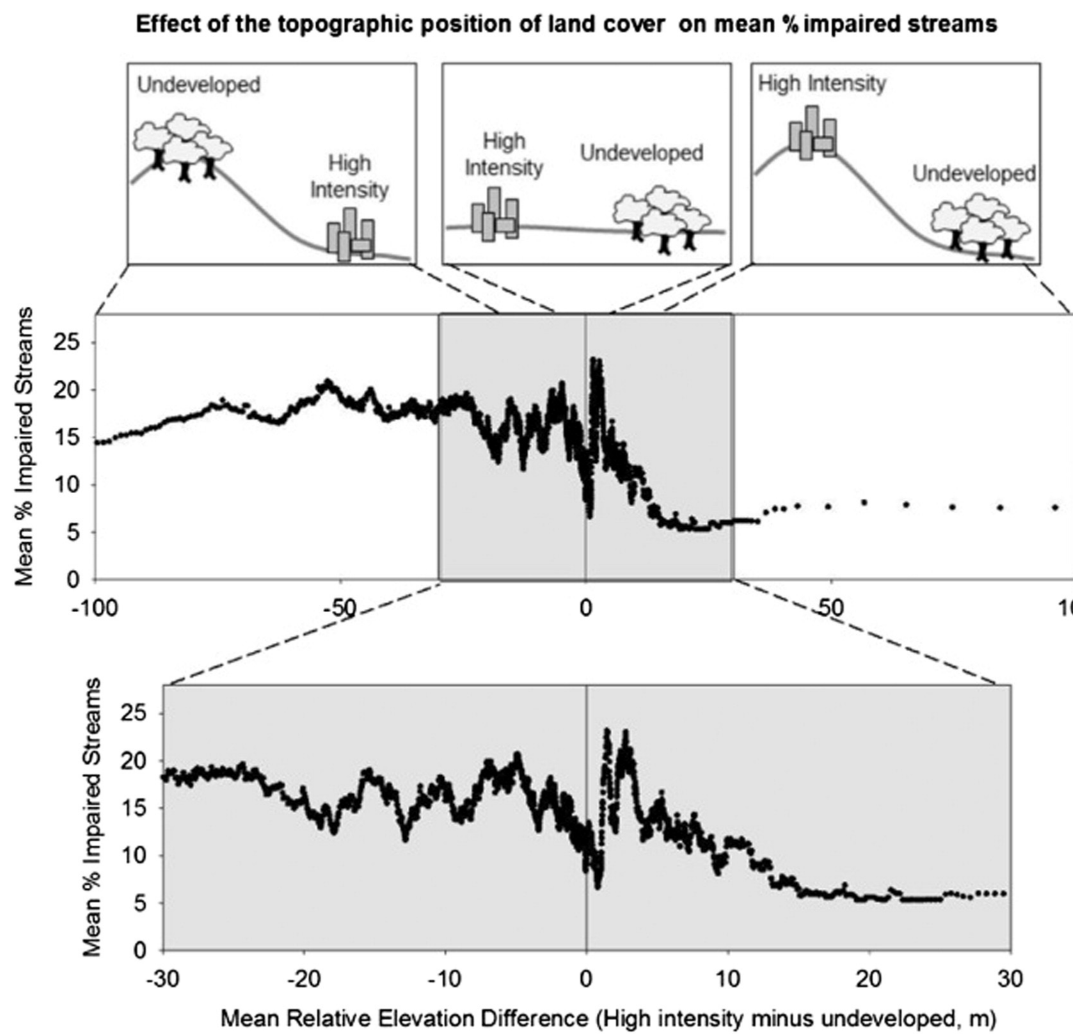

Fig. 7. Relationship between topographic position of undeveloped and high intensity land cover. Each point represents the mean percentage of impaired streams and mean difference in elevation for a window of 100 observations.

of streams can affect biogeochemical fluxes at small scales (Larson and Grimm, 2011). Further research is needed to determine if the difference in surface water abundance with city size leads to variation in biogeochemical or ecosystem processes across population gradients.

4.2. The role of surface water in cities

The scaling patterns of surface water in the U.S. indicate that steams and water bodies fulfill different roles in the urban matrix and that those roles change with city size. Bettencourt et al. (2007) proposed that the scaling properties of urban characteristics reflect the different functional roles within the greater urban system. For example, urban infrastructure scales with a linear or sublinear relationship to population reflect an economy of scale and use, which makes larger cities more efficient (Bettencourt et al., 2007). Alternatively, certain urban characteristics - such as patents, wealth, and green space - scale superlinearly with population, suggesting that social goods are generated at an increased pace as population increases (Bettencourt et al., 2007; Fuller and Gaston, 2009). Here, the scaling of streams and water bodies reflects these two roles in cities. The scaling patterns of water bodies suggest that they are valued as a social good and that population growth creates demand for water bodies within cities. Water bodies are added to urban landscapes for aesthetic and recreational purposes and increase the monetary value of the surrounding land (Walsh et al., 2011; Abbott and Klaiber, 2013). The social value of water bodies increases with water body size (Steele, unpublished data) and the average urban water body is often larger than the natural water body size (Steele and Heffernan, 2014). This value likely does not extend to all types of water bodies. Wetlands, for example, are also included in this analysis but generally are removed from cities (Kentula et al., 2004; Jiang et al., 2012). In contrast, the scaling of streams suggests that streams are treated and valued as functional components, like infrastructure, of cities. Small streams are easily removed and do not provide as many recreational opportunities as larger lakes and ponds.

An important aspect of the scaling patterns of surface water is the presence of a transition point that creates two, or more, scaling domains with city size. In self-organized systems, presence of scaling domains suggests that the organizational processes differ between smaller and larger cities (Ludwig et al., 2000). For surface water, the processes that created the current hydrography (e.g. the demand for surface water and space, development policies and styles, and the biophysical limitations) differ between smaller and larger cities. The shift with population suggests that large cities become denser, decreasing the abundance of surface water per person. Yet the abundance of surface water relative to area does not decline. This suggests that the change in slope is more likely caused by a change in the density of people rather than greater surface water removal. However, why this occurs as a transition rather than incrementally across the population range is unknown but might be related to changes in land use policies and population density that are only implemented in larger cities.

The abundance of impaired streams also had a significant transition and increased faster than population and city area in medium and large cities. The transition population (~25,000 people) occurs at approximately the same population as the initial transition in the abundance of stream per person (~30,000 people). The increasing number of people per length of stream likely increases the frequency and intensity of disturbances and results in more degraded stream length. This transition may result from differences in identifying impaired streams in cities. Streams in larger cities may be more intensely screened and more likely listed as impaired. However, given that there are also significant

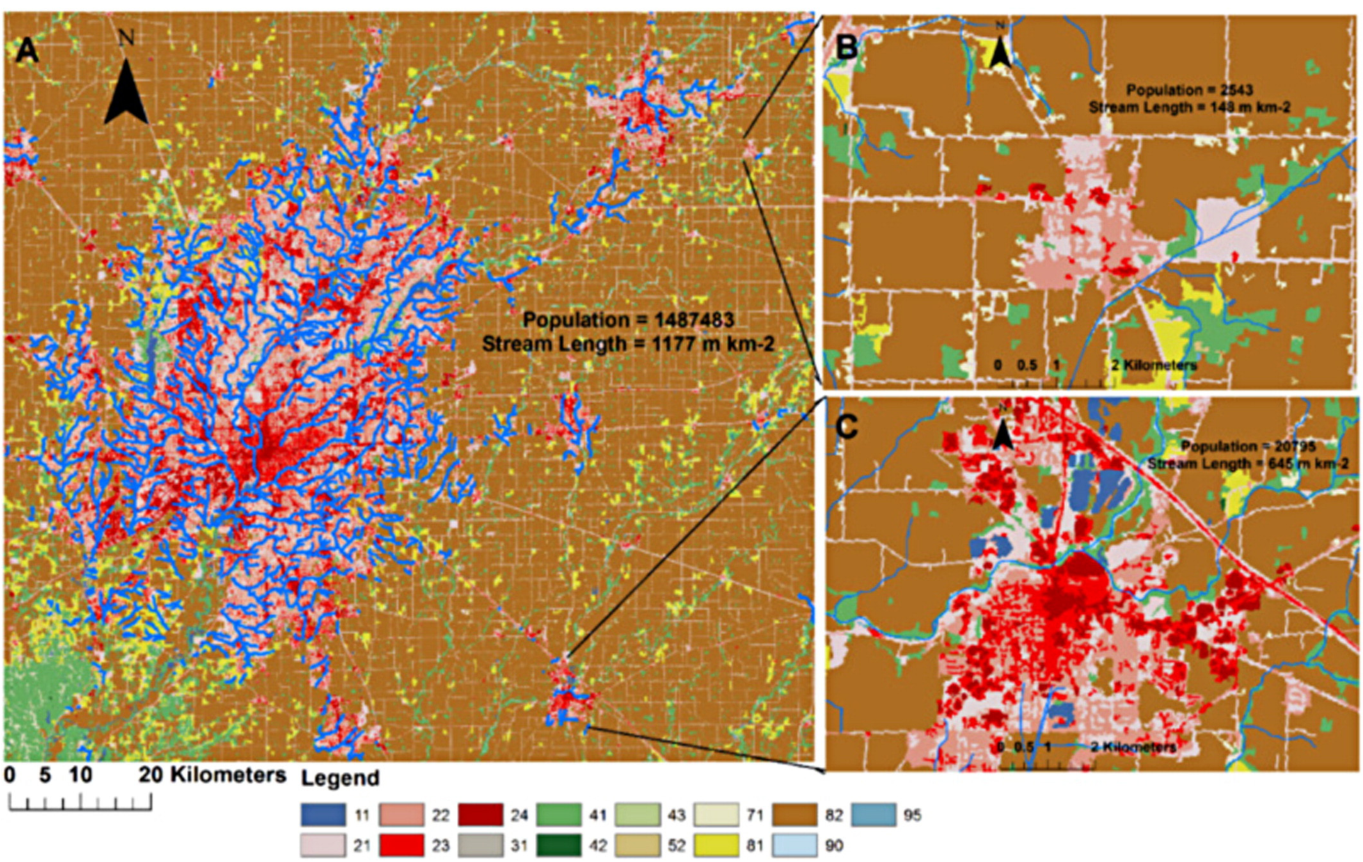

Fig. 8. Illustration of the stream water in Indianapolis, IN, USA (A) population 1,487,483 and two smaller towns with populations of 2543 (B) and 29,795 (C). The density of stream length increases with population size. Blue lines are the stream length from the National Hydrography Data. The background is the 2011 National Land Cover Dataset (11 = Open water; 21 = Developed – open space; 22 = Developed – low intensity; 23 = Developed – medium intensity; 24 = Developed – High intensity; 31 = Bare; 41, 42, 43 = Forested; 52 = Shrub; 71 = Grassland; 81, 82 = Agriculture; 90, 95 = Wetlands.

transitions in total stream length and water body abundance, a transition in impaired streams likely reflects actual conditions in surface water quality. A transition in surface water abundance and impairment has important implications for urban watersheds. Other attributes of urban aquatic ecosystems may also have similar transitions with city size and differing responses to disturbances and resiliency. Therefore, we may need to develop watershed planning and restoration efforts tailored to population size.

4.3. Urban structure, topography, and the impairment of streams

As expected, intense development was correlated with impaired streams, and less/undeveloped area with less impaired streams. However, land cover alone explained very little of the variance. Impaired streams were actually more likely to have undeveloped buffers surrounding the reaches than the rest of the unimpaired stream network. The NLCD land cover intensity classes are correlated with the amount of impervious surface cover. Impervious surface area and stormwater runoff are generally considered to be the major cause of stream degradation (Arnold and Gibbons, 1996). However, the lack of strong relationships between land cover intensity suggests that impervious surface is only weakly related to stream impairment at the national scale. A lack of strong relationships suggests that other factors contribute to stream density and impairment, such as historical land use, types of industry, or simply better identification of impaired waters. Therefore, improving stream water quality and preventing impairment is likely not as simple as increasing the amount of pervious surface area in cities.

A wide assortment of stressors affect urban streams (Walsh et al., 2005), and here results show that the relative topographic position of land cover may influence the impairment of streams. In *bowl* cities, the highest intensity development is more likely to be located on or much closer to the local surface waters. This orientation likely shortens the surface and subsurface flow paths between water moving from city to the local water ways. In *mound* cities, the distance to the surface water is longer. In addition, the local highs are usually the borders between watersheds; therefore, urban seepage and stormflow is more likely to drain into several different watersheds, dispersing the impact over a greater area. The differences in distance and dispersion of urban storms flows may impact the movement of sediments, nutrients, and other pollutants. Therefore, the interaction between land cover and local geomorphology may influence the resilience of urban streams to disturbances.

5. Conclusions

Differences in the abundance and scaling of surface water reveal the variation in hydroscapes with city size and functional role of surface water in cities. Observations of distinct scaling domains indicate shifts in the organizing processes of cities and the development of hydroscapes. Though the intensity of land cover was related to the abundance and impairment of streams, the interaction with topography suggests a more complex relationship between the urbanized development and surface waters.

Acknowledgements

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