An assessment of the condition of flowing waters in predominantly urban areas of the conterminous U.S. and its relationship to measures of environmental justice

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Abstract

Urbanization often leads to environmental degradation and there is a growing concern that these impacts are inequitably distributed. We assessed the condition of urban flowing waters across the conterminous US using data from EPA's National Rivers and Streams Assessment and tested whether degradation was related to metrics of environmental justice (EJ). We found that urban flowing waters are more degraded than their non-urban counterparts. Indeed, the proportion of the length of the nation's urban flowing waters in poor condition, based on common environmental quality indicators, was often nearly twice as high as the proportion for the nation's flowing waters as a whole. The majority of urban waters were in poor ecological condition for water quality integrity, nutrient concentrations, and riparian disturbance although, most were in good ecological condition for riparian vegetation, instream cover, bed sediment, enterococci, and dissolved oxygen. For biological indicators, urban flowing water was mostly in poor condition for both fish (52% of total length) and macroinvertebrate biotic integrity (80% of total length). Despite widespread degradation, we did not find that flowing water degradation was strongly related to the two EJ measures we analyzed (% low income and % minority). The highest correlations we observed (|r|=0.3) were between fish biotic integrity and % low income, and between riparian disturbance and % low income. To our knowledge, this is the first study to assess the pervasiveness of urban flowing water degradation and its relationship to EJ on a national scale. While this study did not uncover a compelling association between the studied environmental parameters and income and minority status in the surrounding human population, more research is needed to assess access to healthy rivers and streams for all communities.

Keywords Urban streams · Urban rivers · Environmental assessment · Environmental justice · Ecological condition

Introduction

Society and the environment are inescapably linked, from the instrumental and intrinsic services ecosystems provide to the impact of anthropogenic exploitation and pollution, which ultimately means that social inequities and inequitable access to clean environments can be harmful to everyone (Crushing et al. 2015; Murray et al. 2022). Recently, the need to address inequitable distribution of ecosystem services and disproportionate burden of anthropogenic impacts on low-income and communities of racial and ethnic minorities has come to the forefront of the agenda of environmental groups. The concept of environmental justice, however, is neither new nor unstudied. Over two decades of interdisciplinary literature document the reality and impacts of environmental racism and the need for

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actions toward environmental justice (Mohai et al. 2009). The United States (US) Environmental Protection Agency (EPA) defines environmental justice as: [t]he fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies. Fair treatment means that no population, due to policy or economic disempowerment, is forced to bear a disproportionate share of the negative human health or environmental impacts of pollution or environmental consequences resulting from industrial, municipal, and commercial operations or the execution of federal, state, local and tribal programs and policies.

Starting in the 1970's, research began focusing on environmental inequalities in the US (Lave and Seskin 1970) and, by the late 1970s and early 1980s, a national movement for environmental justice began as different groups raised environmental justice concerns in the US (Brulle 2000; Gottlieb 1993). Current and historical research indicate that poor people and people of color tend to live near environmental hazards and are impacted to a greater extent from exposure to environmental hazards (Bryant and Mohai 1992; Brown 1995; Bullard 2000; Brulle and Pellow 2006). In more recent work, studies have found that the poor and especially nonwhite poor bear a disproportionate burden of exposure to suboptimal, unhealthy environmental conditions in the US (Evans and Kantrowitz 2022). The causes of environmental injustice are complex and intersectional, stemming from current and historical inequities at systemic levels of human societal structures (Schell et al. 2020; Lane et al. 2022; Morello-Frosch and Obasogie 2023). For example, economics, sociopolitical dynamics, and racial discrimination are all factors that can contribute to the siting of toxic facilities where land is cheap, more low-income people and minorities reside (sometimes due to past industrial zoning laws intended to segregate racial groups), and communities lack social capital to mount effective opposition to the siting of hazardous or polluting facilities (Mohai et al. 2009). In addition, "insidious loops" exemplify a disproportionate impact on indigenous people due to compounding effects of environmental injustice stemming from historical settler industries (Whyte 2018).

Inequitable distribution of air pollution and chemical exposure from contaminated air, soil, and drinking water is well documented; however, more research is needed to identify disparities in access to healthy aquatic resources for recreation and subsistence. Existing literature on stream health suggests relationships between community race/ethnicity, economic status and pollution loads where adverse exposures are greatest for poor or minority groups (Sanchez et al. 2014; Angermeier et al. 2021; Horvath et al. 2022). Exposure to contaminants via fish consumption has also been documented to vary widely among ethnic groups (Stevens et al. 2018; Hitt and Hendryx 2010). Some studies have also examined biological indices as indicators of stream health. In a study conducted in Virginia, the Virginia Stream Condition Index was more impaired in counties with a lower percentage of white residents and correlated negatively with percent black population (Angermeier et al. 2021). In a study simply looking at the number of monitoring stations around the country, unmonitored subwatersheds had a moderately lower median household income than monitored subwatersheds, although the most influential factor associated with number of monitoring stations was population size (Bryson and Johnson 2022). While more studies are emerging regarding stream health and social justice indicators, many are conducted on a local or regional scale exemplifying a gap in research in this field at the national scale.

A national analysis between measures of environmental justice and aquatic condition is greatly complicated by the large number of stressors affecting flowing waters (e.g., agriculture, urbanization, logging, mining). These stressors all affect surface waters in different ways, and they all have different relationships with EJ metrics. Thus, any associations between EJ and condition will be entangled and hard to see across all sites in the nation. A good place to start examining disparities in aquatic ecosystems would be in those already recognized as degraded and dominated by a single stressor and where socio-ecological interactions are pronounced (Pickett et al. 2016; Murray et al. 2022). Urban Stream Syndrome is defined as the "consistently observed degradation of streams draining urban land" (Meyer et al. 2005). Urban streams are affected physically through altered hydrology, geomorphology and piping and channeling, chemically with increased concentrations of toxins, ions, and nutrients, and biologically with reduced biological richness and a dominance of tolerant species (Paul and Meyer 2001; Meyer et al. 2005; Walsh et al. 2005; Wenger et al. 2009; Roth et al. 1996; Wang et al. 2003). While symptoms of urban stream syndrome appear to occur consistently across regions (Walsh et al. 2005), biological responses to urbanization vary regionally (Brown 2009). In addition, prior land use has also emerged as a compounding factor in how ecosystems responded to urbanization (Sprague et al. 2008; Brown 2009; Walsh et al. 2005). A national assessment of urban flowing waters such as in our study, plays a necessary role in understanding relationships between socioeconomic factors, urbanization, and ecosystem processes.

Due to the difficulties and expense of conducting large-scale surveys, there have been many more studies of urban waters within single cities or across small regions than across larger regions. To study and assess surface waters at national scales, the EPA's National Aquatic Resource Survey (NARS) began in 2004 and was designed to estimate

the condition of surface waters throughout the US. A large number (~ 1000) of randomly selected lakes, streams, rivers, wetlands or near coastal sites are visited each year during a defined index period (e.g., summer baseflow for streams/ rivers). Each of the five water body types are visited once every 5 years. For logistical reasons, stream and river sampling were combined into one survey (the National Rivers and Streams Assessment or NRSA) and done over 2-year period every 5 years. At each site, fish and macroinvertebrate assemblages, water quality, and physical habitat data are collected during a 1-day sampling visit. Thus, the NRSA data provide a unique opportunity to assess urban flowing waters and their relationships to EJ issues across the conterminous United States (CONUS) through use of consistently collected data. In this study, we had two main objectives: one, assess the ecological condition of all urban flowing waters across the CONUS; and two, evaluate the relationships between metrics of EJ and ecological condition in urban flowing waters.

Methods

Survey design

Field crews for the National Rivers and Streams Assessment made 6,722 sample visits during the summers of 2008 and 2009, 2013 and 2014, and 2018 and 2019 across the CONUS (Fig. 1). The NRSA used a probability-based design to select the sites (Stevens and Olsen 2004; Olsen and Peck 2008; USEPA 2016a) with a target population of all streams and rivers with flowing water during the June-September index period. Sites were selected from the National Hydrography Dataset (USGS 2013), which generally reflects the blue-line network at the 1:100,000 map scale. The NRSA is representative of a target population of 1,981,000 km of flowing waters ranging from the Mississippi River to tiny headwater streams. The design was spatially balanced and stratified by state, ecoregion, and stream order to even out the sample site distribution across areas and stream sizes.
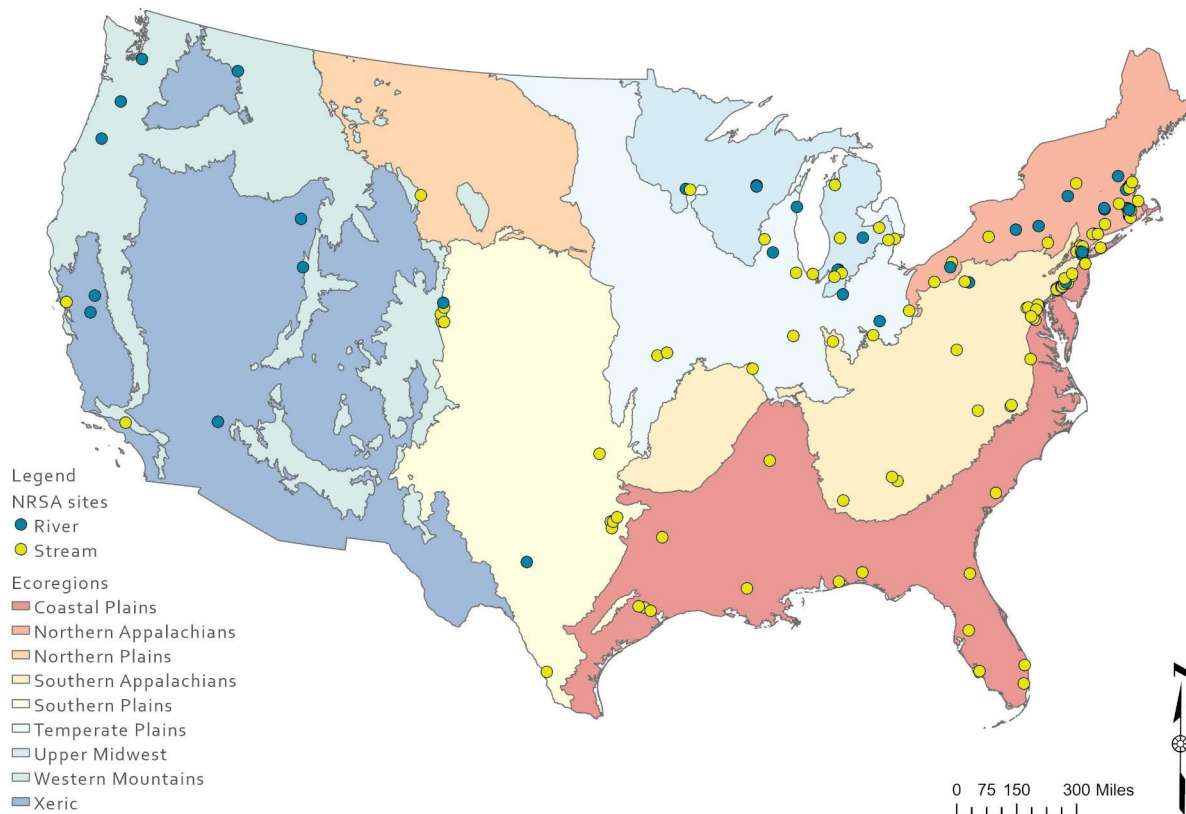

Fig. 1 Location of the 149 NRSA Urban sites across the conterminous United States. NRSA assessments are based on the nine ecoregions shown here that are aggregations of Omernik and Griffith (2014) level-III ecoregions, aggregated as described in Herlihy et al. (2008)

Field data collection

Biological data were collected as described in detail in USEPA (2009, 2013a, b). Briefly, a sample site was established around the randomly chosen sample point of sufficient extent to characterize the assemblages within the site (Reynolds et al. 2003; Hughes and Peck 2008). Nearly all the sites were sampled for fish by backpack or boat electrofishing. In wadeable sites < 13 m wide, a reach length equal to 40 channel widths, or a minimum of 150 m for headwater streams, was sampled. For wadeable sites > 13 m wide and boatable sites, the minimum reach length sampled was the longer of 500 m or 20 channel widths. For macroinvertebrate sampling, eleven subsamples were taken in a systematic zig-zag pattern at each of 11 equidistant transects along the sample site through use of a D-frame kick net (500-µm mesh, 0.09 m^2 area). For wadeable streams, samples were collected in a left, center, right alternating order; at boatable sites, samples were collected at alternating left and right bank locations from the wadeable margins of the river. The 11 subsamples were combined, preserved in ethanol, and shipped to the laboratory, where a fixed laboratory count of 500 individuals were identified to the lowest possible taxon through use of multiple local, regional, and national keys (USEPA 2012). National ecological condition scores for NRSA based on multi-metric indices (MMI) have been developed for both macroinvertebrates (Stoddard et al. 2008) and fish (USEPA 2016a). These MMIs are based on summing 6–8 different metric scores that capture different aspects of biotic integrity (e.g., native species richness, % intolerant individuals). MMI scores have also been converted to good/fair/poor condition classes for both fish and macroinvertebrates based on the percentile distribution of least-disturbed reference sites (USEPA 2016a).

For water quality variables, one water grab sample was collected from within the sample reach (USEPA 2009). Samples were shipped by overnight courier to a central analytical laboratory except for a few states that used their own state laboratories. The water quality variables were analyzed in the lab using meters to measure pH and conductivity. Sulfate and chloride concentrations were measured by ion chromatography, total phosphorus and total nitrogen were measured by acid persulfate digestion and colorimetry, and turbidity was measured with a nephelometer. Lab methodologies are described in USEPA (2012).

Physical habitat condition and substrate variables were collected as described in Hughes and Peck (2008); USEPA (2009, 2013a, b); and Kaufmann et al. (1999). Multiple measurements were made at the 11 evenly spaced transects along the sample site. Woody riparian vegetation cover, human disturbances, fish cover, substrate composition, and wetted width and depth data were collected at each transect through use of standardized field forms. Between transects, crews determined slope and collected depth, width, substrate, and habitat unit data at systematic intervals. Habitat metrics were calculated from this data as described in Kaufmann et al. (1999).

GIS data collection

For each sample site, we calculated the percentage of agricultural and developed (urban) land use/land cover based on US National Land Cover Data (NLCD, Dewitz 2019) at two spatial scales, the entire watershed above the sample site, and a 1 km radius circular buffer around the sample site. Agricultural land cover was defined as the sum of two NLCD classes: Pasture/Hay (areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops), and Cultivated Crops (areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards). Developed land cover was defined as the sum of the four NLCD classes: Developed, Open Space (areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses), Developed, Low Intensity (areas with a mixture of constructed materials and vegetation), Developed, Medium Intensity (areas with a mixture of constructed materials and vegetation, impervious surfaces account for 50–79% of the total cover), and Developed, High Intensity (highly developed areas where people reside or work in high numbers (Dewitz 2019). We used the NLCD data from the nearest year preceding each NRSA sampling (e.g., 2016 NLCD data for 2018–2019 NRSA). Watershed data were taken from StreamCat (Hill et al. 2016), and the circular buffer data by clipping out a 1 km buffer using ARC/INFO.

Socioeconomic data for the EJ metrics were acquired from EPA's EJ Screen GIS data layer (USEPA 2023). There were a large number of possible EJ metrics that could be calculated from the EJ Screen data layer. Many of them are highly correlated with each other. We chose two EJ metrics for this analysis, % minority and % low income as they were not highly correlated with each other at our sites (r < 0.5), and they capture two important EJ gradients. The minority metric is defined as the percent of individuals in a census block who list their racial status as a race other than white alone (not multiracial) and/or list their ethnicity as Hispanic or Latino. The low income metric is defined as the percent of a census block group's population in households where the household income is less than or equal to twice the federal poverty level (USEPA 2023).

The two EJ metrics were each calculated for three different spatial scales; the entire census block where the sample site was located, and both a 1 and 10 km radius circular

buffer around the sample site. To merge the EJ Screen GIS data which is given at the census block level with the circular buffers, a "summarize within" function was done in Arc-GIS and used to approximate the population metrics within the neighborhood created by the circular buffers. This calculation weights the EJ census block data according to the proportion of the census block actually inside the circular buffer.

Data analyses

We are unaware of any commonly accepted definition of what makes a stream or river "Urban" For purposes of this study, we used a predominantly urban definition. Any flowing water in the NRSA target population that had a watershed > 50% developed land **or** had land cover in a 1 km radius circle around the sample site > 50% developed land was considered urban. Similarly, there is no accepted definition differentiating streams from rivers. For this analysis, we defined rivers as sites with watershed areas > 1000 km^2, versus streams that had watershed areas < 1000 km^2. There were sufficient sample sizes in each to make robust population estimates.

There were 149 unique sample sites in predominantly urban areas in the NRSA database sampled between 2008 and 2019 (Fig. 1). NRSA does include repeat sampling to a subset of sites. When a site was visited multiple times, we used the first visit to the site, in the most recent sample year, in our analyses to avoid double counting sites. Of the 149 urban sites, 142 were probability sites. Each NRSA probability site has a sample weight, calculated as the inverse of its inclusion probability from the randomized probability design (Olsen and Peck 2008). We used weighted analysis of the 142 predominantly urban probability sites to make inference to the condition of the entire population of urban flowing water length in the CONUS.

NRSA routinely releases an assessment of the condition of CONUS flowing waters after each survey cycle (e.g., USEPA, 2016b). For our assessment of urban waters, we used the condition indicators developed for these assessment reports. Specifically, we used two biological condition measures (fish MMI and macroinvertebrate MMI), five water quality condition measures (water quality integrity index (WQII), total nitrogen, total phosphorus, dissolved oxygen, and Enterococci), and four physical habitat condition measures (riparian disturbance, riparian vegetation, instream cover, and bed sediment). The metrics used to define condition and the thresholds used to define categories of good/fair/poor condition are described in the NRSA technical report (USEPA 2016a).

Differences between the means of the flowing water samples were tested with a standard t-test. Associations of

the EJ metrics at all three spatial scales with the landscape disturbance metrics, and flowing water condition metrics were analyzed using Spearman rank correlation as some of the metrics were not normally distributed. We also built two regression models (one for fish and one for macroinvertebrates), using both stream and river data combined, following the all subsets-based approach described in Burnham and Anderson (1998). All the disturbance variables were considered as potential predictors to predict both the fish and macroinvertebrate MMI scores. We defined the important model predictors as those that were in over half of the possible models and then constructed a multiple regression model just using the important predictors. A Spearman correlation analysis of the residual MMI score from these disturbance models versus the EJ metrics was then done to check for any possible associations that would not be related to the level of flowing water disturbance. For all our statistical analyses, as we were analyzing many possible associations, we adjusted the p-value for all these multiple tests using a Bonferroni approach (dividing 0.05 by the number of tests to get an adjusted p-value).

We also conducted a relative risk analysis using the flowing water condition classes and EJ metrics. Relative risk is commonly used in the medical literature and it's adaptation for use in the analysis of NRSA data is detailed in Van Sickle and Paulsen (2008). Relative risk is calculated using class data and a 2×2 contingency table. To obtain just two condition categories for use in the contingency table, we compared a not-poor condition class (i.e., the combination of good and fair condition) to the poor condition class. For both the % minority and % low income EJ metrics, we divided the sites into two groups, advantaged and disadvantaged using a threshold of above/below 50% for % minority and above/below 33.3% for % low income. We used 33.3% as the threshold for % low income to have a sufficient number of disadvantaged low income sites in the contingency table. To calculate the 2×2 contingency table matrix, the population sample weights were used to calculate the proportion of the urban flowing water population (by length) that is in each of the cells of the matrix (e.g., poor condition and disadvantaged). The relative risk ratio (RR) is then calculated as the ratio of two proportions,

$$RR = \frac{Pr\,(Poor\ Condition,\ given\ Disadvantaged\ EJ)}{Pr\,(Poor\ Condition,\ given\ Advantaged\ EJ)}$$

where Pr is the proportion of total urban stream/river length. A relative risk value of 1.0 indicates that there is no association between EJ status and condition, while values greater than 1.0 suggest greater relative risk. For example, if 30% of the nation's urban flowing water population is in poor condition but it is equally divided among sites with

Table 1 Characteristics of urban flowing water length in CONUS

Size Category	Sample Size	Length (km) {% of Urban}	Urban Only Watershed Dominated (%)	Urban Only 1 km circle Dominated (%)	Both Watershed and Circle Dominated (%)
Small Streams	75	50,510 {76%}	62%	8.7%	30%
Large Streams	30	12,070 {18%}	21%	39%	40%
Small Rivers	32	2618 {4.0%}	0%	100%	0%
Large Rivers	12	1105 {1.7%}	0%	100%	0%
ALL	149	66,310	50%	19%	30%

advantaged and disadvantaged status (15% in each) then the RR = 0.15/0.15 = 1 and there is no association between condition and EJ status class. Conversely if the 30% of the nation's urban flowing waters in poor condition was observed as 25% in sites that were disadvantaged and 5% in sites that were advantaged, then the RR = 0.25/0.05 = 5.0. The higher the relative risk value, the greater the risk of poor site condition. A relative risk of 5 indicates that we are 5 times more likely to see a stream/river in poor condition when the site is disadvantaged than when it is in the advantaged category. Relative risk and the statistical confidence intervals around each relative risk ratio were calculated using the spsurvey package in R (Van Sickle and Paulsen 2008; Dumelle et al. 2023). When the lower 95% confidence interval for any given relative risk ratio falls below 1.0, we did not consider relative risk to be statistically significant.

Results

The urban flowing water population

There were 149 unique sites in the NRSA database sampled between 2008 and 2019 that met our definition of urban flowing waters (Fig. 1). Of these, 105 were streams and 44 were rivers (Table 1). When broken down into classes of where the urban land is located (watershed, local area, or both), the sampled small streams almost all have predominantly urban watersheds. Less than 40% of them have predominantly urban cover in their 1 km buffer. On the other hand, rivers, due to their size, do not have predominantly urban watersheds but are considered urban because of the predominantly urban land cover in their local area. Of the 149 sites, 142 were probability sites selected using the NRSA random sampling design and used to make inference to the whole urban population estimated to consist of 66,310 km of flowing water, 94% of which are streams.

Urban flowing waters were located in all ecoregions but were relatively rare in the Northern Plains, Western Mountains, and Xeric West (Table 2; Fig. 1). Urban waters were

Table 2 Urban flowing water population by ecoregion and names of rivers sampled

Ecoregion	Sample Size	Urban Length (km)	Urban length as % of Total Population	Urban Rivers Sampled
CPL	26	21,480	7.2	Schuylkill
NAP	33	4947	3.0	Connecticut, Merrimack, Mohawk, Chemung, Chenango, Chicopee, Shenango, Blackstone
NPL	1	125	0.3	None
SAP	38	23,200	6.2	Allegheny, Passaic
SPL	12	3635	4.6	North Concho, Cache la Poudre
TPL	11	5220	2.3	Rock, Great Miami, Wabash, Manitowoc
UMW	17	5939	5.6	Mississippi, Wisconsin, Saint Joseph, Grand, Elkhart
WMT	4	573	0.2	Willamette, Spokane, White
XER	7	1192	1.3	Colorado, San Joaquin, American, Weber, Portneuf
All CONUS	149	66,310	3.9	

most dense in the Coastal Plain (7.2% of ecoregion length), Southern Appalachians (6.2%) and Temperate Plains (5.6%). Major urban rivers in the NRSA sample include the Schuylkill, Connecticut, Allegheny, Wabash, Mississippi, Willamette, and Colorado.

Urban flowing water condition

Based on macroinvertebrates assemblages, urban flowing waters were mostly in poor condition (80%, Table 3). This was true for both rivers and streams; only 3% of streams and 14% of rivers were in good condition (Fig. 2). Fish assemblages in urban flowing waters were also mostly in poor condition (52%) but not to the degree that macroinvertebrates showed. For fish, however, streams had a higher

% of the length in good condition (25%) than did rivers (10%). Urban waters have a large proportion of waters in poor biological condition, but good waters are more likely to be found for fish in streams versus rivers whereas the opposite is true for macroinvertebrates, where good macroinvertebrate condition is more likely to be found in rivers than streams.

Urban flowing water quality is mostly in poor condition for water quality integrity and nutrients (Table 3). The WQII is only good for 9% of the nation's urban flowing water length but higher in rivers (27% good) than streams (8%, Fig. 2). Other water quality metrics have more length in good condition. Enterococci are good (below criteria) in 56% of the flowing water length and dissolved oxygen is good in 69%. Only 13% of the length is poor for dissolved oxygen, all of which were in streams and none in rivers.

None of the urban sites had good physical habitat based on riparian disturbance and the vast majority (75%) were in poor condition (Table 3). This was true for both rivers and streams (Fig. 2). On the other hand, the majority of the urban flowing water length was in good physical habitat condition based on riparian vegetation, instream cover, and bed sediment. Riparian vegetation condition was similar in streams (29% poor) and rivers (23% poor) as was instream cover. Streambed sediment, however, was in worse condition in rivers (38% poor) than streams (15% poor).

Relationship between flowing water condition and EJ metrics

EJ metric values varied widely among the urban flowing water sites, % minority ranged from 0 to 100% and % low income ranged from 3 to 85%. At the 1 km EJ spatial scale, median % low income was 26.5% (IQR = 17.2–43.9), and median % minority was also 26.5% (IQR = 15.0-47.1). Among the three EJ spatial scales, EJ metrics were highly correlated with Spearman correlations between the block and 1 km scale, and 1 km scale with 10 km scale having r > 0.7. Correlations between the block and 10 km scale were somewhat weaker (r = 0.54–0.58). The two EJ metrics were also significantly correlated with each other, although at much weaker levels, with correlations ranging from 0.37 at the block scale to 0.22 at the 10 km scale. For % minority, there were no differences, by t-test, between stream and river samples (Fig. 3). For % low income, however, values were significantly lower in streams than rivers (Fig. 3).

Correlations between metrics of flowing water condition and EJ metrics were generally low (Table 4). Only the correlation between fish MMI and % low income, and between riparian disturbance and both % minority and % low income was significant after adjusting for multiple tests. In these three cases, as % minority and low income increased, fish condition declined, and riparian disturbance increased. Only those correlations with EJ metrics at the 1 km radius buffer scale were significant, relationships at other spatial scales were not significant. The scatterplot between fish MMI and % low income (Fig. 4) shows the significant negative

Table 3 The ecological condition of urban flowing waters based on NRSA condition classes. Weighted estimates based on 142 probability sites representing 66,310 km of flowing water length. A small fraction of the population was not assessed due to sampling issues, especially for fish. Good/fair/poor thresholds are described in USEPA (2016b) or as footnoted. The total national %Poor length across CONUS using the 2018–2019 NRSA data is shown for comparison

Condition Indicator	Urban %Good	Urban %Fair	Urban %Poor	National %Poor
Biological				
Macroinvertebrate MMI	3.3	16.6	80.1	47.3
Fish MMI	24.1	14.6	51.9	29.4
Water Quality				
Water Quality Integrity Index[a]	9.2	24.1	66.7	37.5
Total Phosphorus	23.0	17.5	59.5	41.8
Total Nitrogen	17.5	25.2	57.3	43.6
Enterococci[b]	56.4	---	42.4	20.3
Dissolved Oxygen[c]	69.4	16.7	13.7	5.9
Physical Habitat				
Riparian Disturbance	0	24.5	74.6	21.5
Riparian Vegetation	55.2	15.2	28.6	26.8
Instream Cover	56.6	25.0	17.4	9.9
Bed Sediment	53.1	27.4	15.8	19.7

[a] good was defined as WQII ≥ 85 and poor as WQII < 70.

[b] enterococci had no Fair category, good length was below criteria level, poor length above it

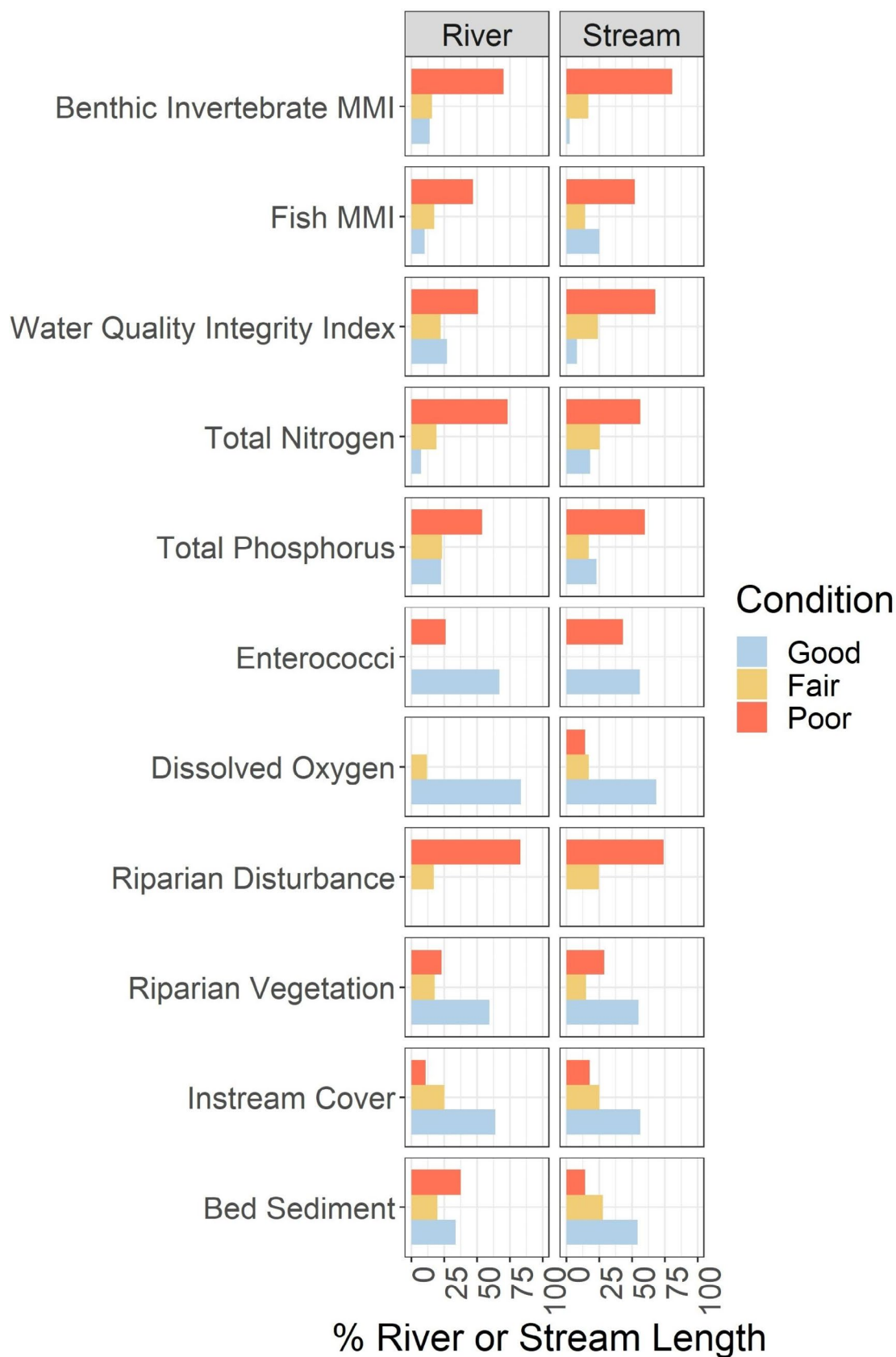[c] good was defined as DO ≥ 5 mg/L and poor as DO < 3 mg/L.

Fig. 2 NRSA population estimates of the percent of the length of flowing water in different condition classes in predominantly urban flowing waters of the United States. Results are broken down by stream and river classes
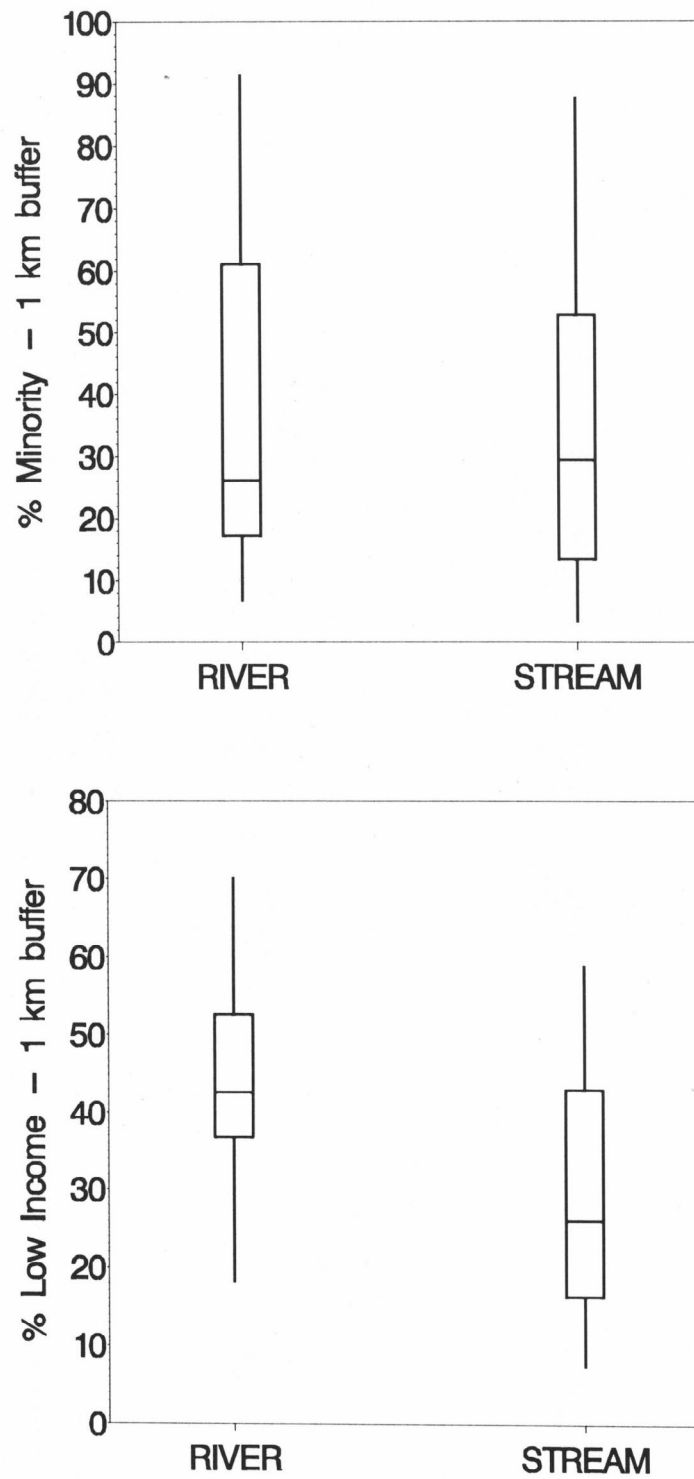
Fig. 3 Weighted box and whisker plot comparing stream and river population percentiles of % Minority and % Low Income in predominantly urban waters. The box shows the 25th /75th percentiles, whiskers the 5th /95th percentiles and the line in the box the median

Table 4 Spearman rank correlation coefficients (top) and *p*-values (bottom) of the association between flowing water condition metrics and EJ metric scores at varying EJ spatial scales (Census Block, 1 km radius buffer, and 10 km radius buffer)

Indicator	Minority Block	Minority 1 km	Minority 10 km	LowIncome Block	LowIncome 1 km	LowIncome 10 km
Macroinvertebrate MMI	-0.0915	-0.175	-0.171	-0.109	-0.116	-0.127
	0.267	0.0327	0.0368	0.186	0.159	0.123
Fish MMI	-0.0600	-0.189	-0.134	-0.198	-0.319*	-0.167
	0.498	0.0314	0.129	0.0242	0.0002	0.0572
Water Quality Integrity Index	-0.0284	-0.0574	-0.0940	-0.0408	-0.0714	-0.0979
	0.732	0.487	0.254	0.621	0.387	0.235
Riparian Disturbance Index	0.175	0.271*	0.106	0.230	0.303*	0.1558
	0.0336	0.0009	0.202	0.0048	0.0002	0.0586
% Fine Sediment	0.0803	0.135	0.0922	0.0888	0.0340	-0.0137
	0.335	0.104	0.268	0.286	0.684	0.870

* significant at $p < 0.05$ after Bonferroni correction for multiple tests

relationship between the two with similar slopes for streams and rivers.

There were stronger and more significant correlations between the landscape disturbance metrics and EJ metrics (Table 5). Relationships with % minority were positive and significant for watershed % developed land, road density, and population density at the 10 km radius buffer EJ spatial scale. The strongest relationship was with population density ($r = 0.44$). The relationships between EJ metrics and % low income were significant and positive at the 1 km EJ buffer scale for % developed land (watershed and local), riparian disturbance, and population density. Relationships were positive for riparian disturbance and %developed land in the 1 km buffer but negative for population density and %developed land in the watershed. The strongest correlation was with %developed land in the 1 km buffer ($r = 0.41$). Scatterplots of this relationship (Fig. 4) show that the relationships are mostly driven by streams in that the gradient in river samples was curtailed by the fact that there were no river samples that had %developed land in the 1 km buffer below 50%. For the other EJ spatial scales, there were no significant relationships with % low income and landscape disturbance at the 10 km scale and only two weakly significant relationships at the census block scale (Table 5).

We built an all subsets-based regression model for fish, and another one for macroinvertebrates, using all of the landscape and flowing water disturbance variables in Tables 4 and 5 to predict their MMI scores. The important disturbance variables for the fish model were WQII, watershed %agriculture, watershed %developed, and %fine sediment. As a number of sites were not sampled or did not have fish, the model had an $n = 128$, with an $r^2 = 0.227$ and RMSE = 13.5. For the macroinvertebrate MMI model ($n = 146$), the important predictor variables were WQII, %fine sediment, riparian disturbance index, and watershed %developed, and the model had an $r^2 = 0.197$ and RMSE = 13.7. The residual MMI scores from these models

represent variation in MMI not related to disturbance. Correlations of these residual fish and macroinvertebrate MMIs showed no significant correlations with EJ metrics at any spatial scale (Table 6). The largest correlation coefficient was r=-0.194 with the residuals from the fish MMI. Thus, there does not appear to be any strong EJ relationships with MMI scores after pulling out the disturbance signal. So, the EJ-biology relationship appears to be weaker than the disturbance-biology relationship.

Relative risk analysis

The relative risks of having poor flowing water condition when there was disadvantaged EJ condition were low, below 1.5 for all indicators of condition (Table 7). The lower 95% confidence interval for most tests was below 1 indicating that there was not a statistically significant risk at a 0.05 confidence level. The only risks that were significant were macroinvertebrate MMI and % minority and WQII and % minority. The highest observed relative risk (1.47) was for poor fish condition and low income at the 1 km EJ spatial scale, but the lower confidence limit was 1.0 so it was not significant. Relative risk results were very similar among the three EJ spatial scales for each EJ metric. Risks were generally higher for disadvantaged % minority than for % low income.

Discussion

Continental-scale urban flowing water degradation

The negative effects of urbanization on flowing water condition are well documented and reviewed (Paul and Meyer 2001; Walsh et al. 2005; Wenger et al. 2009; Zerega et al. 2021). These negative effects include increased sedimentation, increased nutrient inputs, flashy hydrology, and altered
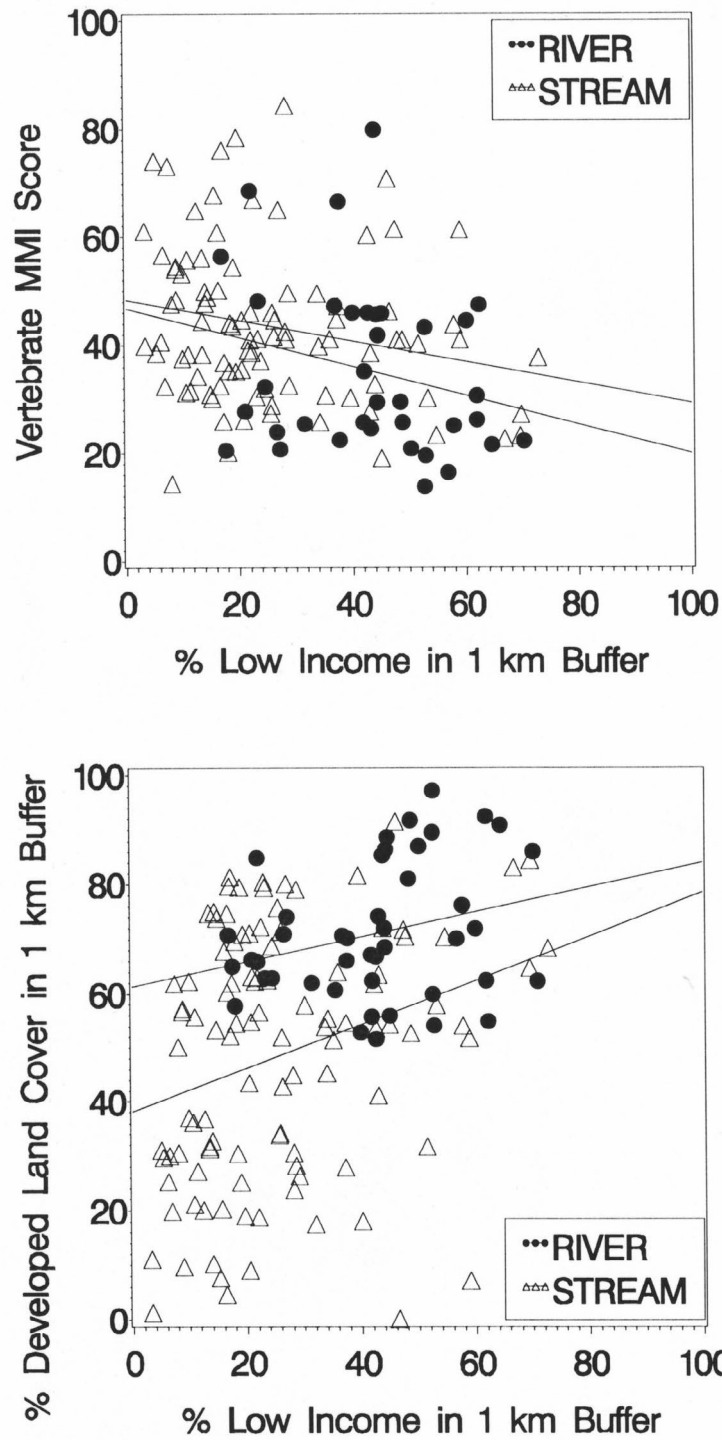
Fig. 4 Scatterplots of vertebrate MMI score (top) and % developed land (bottom) versus % Low Income in the 1 km buffer for predominantly urban NRSA sites

Table 5 Spearman rank correlation coefficients (top) and *p*-values (bottom) of the association between landscape disturbance cover metrics and EJ metric scores at varying EJ spatial scales (Census Block, 1 km radius buffer, and 10 km radius buffer)

Indicator	Minority Block	Minority 1 km	Minority 10 km	LowIncome Block	LowIncome 1 km	LowIncome 10 km
% Watershed in Agriculture	-0.108	-0.0953	-0.224	0.248	0.270*	0.238
	0.188	0.248	0.0061	0.0023	0.0009	0.0034
% 1 km radius circle in Agriculture	-0.205	-0.197	-0.265	-0.0455	-0.0314	0.00123
	0.0122	0.0159	0.0011	0.582	0.704	0.9882
% Watershed in Developed Land	0.0989	0.108	0.303*	-0.271*	-0.336*	-0.218
	0.230	0.191	0.0002	0.0008	<0.0001	0.0076
% 1 km radius circle in Developed Land	0.106	0.203	0.0387	0.254	0.405*	0.208
	0.200	0.0130	0.640	0.0017	<0.0001	0.0109
Watershed Road Density (km/km^2)	0.0949	0.143	0.339*	-0.209	-0.256	-0.122
	0.250	0.0819	<0.0001	0.0106	0.0017	0.137
Watershed Population Density (#/km^2)	0.144	0.200	0.440*	-0.269*	-0.310*	-0.200
	0.0795	0.0146	<0.0001	0.0009	0.0001	0.0142

* significant at $p < 0.05$ after Bonferroni correction for multiple tests

Table 6 Spearman rank correlation coefficient for residual disturbance model fish and macroinvertebrate MMI scores with EJ metrics at three spatial scales

	% Minority			% Low Income		
Response	Block	1 km	10 km	Block	1 km	10 km
Residual Fish MMI	-0.035	-0.165	-0.163	-0.095	-0.194	-0.077
Residual Macroinvertebrate MMI	-0.001	-0.070	-0.052	-0.039	-0.070	-0.090

[a] None of the associations were significant at $p < 0.05$ after correcting for multiple tests

Table 7 Relative risk of having poor flowing water condition with disadvantaged EJ metric scores at varying EJ spatial scales

Flowing Water Condition	EJ Disadvantage	Scale	Relative Risk	95% Conf. Int
Macroinvertebrate	Minority	Block	1.17	0.996–1.37
Macroinvertebrate	Minority	1 km	1.21*	1.03–1.42
Macroinvertebrate	Minority	10 km	1.26*	1.09–1.46
Macroinvertebrate	Low Income	Block	0.823	0.654–1.04
Macroinvertebrate	Low Income	1 km	0.874	0.710–1.07
Macroinvertebrate	Low Income	10 km	1.01	0.842–1.22
WQII	Minority	Block	1.10	0.780–1.56
WQII	Minority	1 km	1.39*	1.07–1.82
WQII	Minority	10 km	1.33*	1.03–1.73
WQII	Low Income	Block	1.04	0.783–1.39
WQII	Low Income	1 km	1.03	0.752–1.40
WQII	Low Income	10 km	0.996	0.720–1.38
Fish	Minority	Block	1.04	0.651–1.66
Fish	Minority	1 km	1.45	0.975–2.16
Fish	Minority	10 km	1.46	0.992–2.16
Fish	Low Income	Block	1.19	0.817–1.73
Fish	Low Income	1 km	1.47	1.00–2.15
Fish	Low Income	10 km	1.15	0.757–1.76

* indicates a significant risk in that the 95% CI does not overlap a relative risk of one

riparian zone and instream habitat (Walsh et al. 2005; Kupilas et al. 2021). Changes in riparian zone and instream water quality, hydrology, and habitat ultimately affect the biological communities that can persist in urban flowing water (Vinson and Hawkins 1998). Our study indicates that these negative effects of urbanization on biological communities are not isolated incidences but are widespread across the nation. Indeed, we found a much higher percentage of flowing water length in poor condition in urban flowing waters than were found in all flowing waters of the CONUS (Table 3) for both fish (52% vs. 29%) and especially macroinvertebrates (80% vs. 47%). Our results are consistent with existing research that finds sensitive species are usually absent or less abundant in urban streams and urban ecosystems are more likely to be dominated by tolerant taxa (Walsh et al. 2005). Declines in fish assemblage richness and diversity have also been observed in urban flowing water (Wenger et al. 2009; Engman and Ramírez 2012; Stranko et al. 2012) as have increases in invasive and tolerant fish species (e.g., Antoniazzi et al. 2023).

Four water quality and physical habitat indicators also showed a much higher percentage of urban flowing water length in poor condition than the national estimates. Riparian disturbance estimates were over three times higher (75 vs. 22%), and WQII (67 vs. 38%), enterococci (42 vs. 20%), and dissolved oxygen (14 vs. 6%) were twice as high (Table 3). The much higher percentage of urban flowing water in poor condition for these indicators helps explain

the high percentage of urban flowing water in poor condition for benthic macroinvertebrate and fish communities. Furthermore, enterococci can cause a host of negative human health effects and observing double the percentage of urban flowing water in poor condition for enterococci than the National equivalent is therefore also a human health issue.

In our analyses, some differences did exist when comparing river and stream condition. One difference that stood out between rivers and streams was that all river sites had > 50% developed land cover within the 1 km buffer, but many stream sites had % developed land cover within the 1 km buffer well below 50% (Fig. 3). This difference was due to our definition of urban flowing water and the fact that rivers have very large watersheds compared to streams. All of the river sites met our definition of urban flowing water because they had > 50% developed land cover within the 1 km buffer, but these river sites did not have > 50% watershed developed because the watersheds were so large. In contrast, many stream sites met our definition of urban because the watershed was > 50% developed, but these stream sites did not always have > 50% developed within the 1 km buffer. Therefore, despite all stream and river sites meeting our definition of urban, stream sites skewed toward being less developed and more suburban than river sites, which has implications for the differences we observed between rivers and streams. For example, several indicators showed very different estimates of poor condition when broken out into streams and rivers, separately. Urban rivers had over twice as much length in poor condition for bed sediment compared to streams (38 vs. 15%). However, the biological indicators showed the opposite with slightly higher estimates of poor condition for streams compared to rivers for both the macroinvertebrate MMI (81 vs. 70%), and fish MMI (52 vs. 47%). The distribution of % low income communities also differed somewhat between rivers and streams (Fig. 3). Urban rivers generally had higher % low income communities living nearby than streams, but no difference was observed based on % minority.

The way that urban flowing water is defined has implications for the results and interpretation of results from urban flowing water studies. Furthermore, there is no commonly accepted definition of what constitutes an urban flowing water, and many different definitions are used in the urban waters literature (Paul and Meyer 2001). We used a predominantly (i.e., > 50% developed) urban definition that likely captured the higher range of urbanized sites while omitting other less urbanized sites from our analyses. However, a different % developed threshold, or different urbanization metrics altogether, could be used to define urban areas. For example, a synthesis of urban streams research by Wenger et al. (2009) considered a broad definition of urban to include any landscape with more than 1 residential structure per 2 hectares. They used such a broad definition because they pointed out that even low-density urban structures can have large negative effects on aquatic ecosystems (Wenger et al. 2009). We decided to use a more stringent definition of urban because one of our research objectives was to assess if environmental justice communities in urban areas were disproportionally exposed to flowing water degradation, which we thought would be most evident in the most urbanized areas. Furthermore, we used this definition of urban flowing waters because it is sufficient to capture the mechanisms that broadly, and often severely, cause urban flowing water degradation.

Like any well designed poll or statistical survey, the NRSA probability design provides robust estimates of the condition of a target population with known confidence bounds. Thus, our estimates of the condition of CONUS urban waterways provide a robust picture of the overall status of that population of flowing waters. Like any survey, however, these estimates do have limitations. For one, it only applies to the defined target population. In our case, that is the blue-line network of streams and rivers that are mapped in the GIS data layer from which the sites were randomly selected. As such, buried streams or filled in streams that are not depicted on the maps in the data layer are not in our sample. Unfortunately, there are a number of streams in urban areas that have been shunted underground or buried and we can make no inference about their condition in this study. Another limitation of the NRSA data is a temporal one, in that by just collecting one summer index sample, it provides a snapshot of condition during summer baseflow. Our estimates are specific to conditions during summer baseflow and not directly inferable to what might be occurring during storm episodes or winter flows. In an ideal world, it would be great to sample thousands of sites multiple times during the year but that is well beyond the capability of current resources.

Historical land use can also affect results of studies conducted on flowing waters. For example, previous studies conducted on urban flowing waters have found that the effects of urbanization on water quality are reduced or confounded when urbanization occurs on land previously used for agriculture (Fitzpatrick et al. 2004; Van Sickle et al. 2004; Heatherly et al. 2007; Wenger et al. 2009). In an analysis of nine metropolitan areas across the US, Brown et al. (2009) attributed a lack of responses of water quality (nitrogen and herbicides), algae, and fishes to urbanization to previous agricultural uses suggesting sensitive species were eliminated before urbanization occurred (Wenger et al. 2009). This phenomenon may be especially true for nutrients (nitrogen) as urban streams have been shown to have similar nitrogen levels to agricultural areas (Grimm et al. 2005; Kaushal et al. 2006; Mueller and Spahr 2006). Some

studies suggest that the combined effects of human disturbances rather than a threshold response to urbanization was responsible for a decline in fish assemblages (Brown 2009; Wenger et al. 2009). Furthermore, as illustrated in Sanchez et al. (2014), agricultural areas that are often located in more rural environments can be equally as degraded as their urban counterparts. The diverse nature of the CONUS adds a complexity to this study in that some areas of low income and high minority were more rural with differing environmental challenges than low income or high minority areas near cities. More research needs to be done to assess environmental justice in both urban and rural landscapes.

Urban flowing water degradation and environmental justice

We found only weak or no associations between degree of flowing water degradation and % low income or minority, with only a few of these associations being statistically significant. These findings are in accordance with those reported by Sanchez et al. (2014), who assessed whether or not several common fish and benthic indicators were negatively associated with several metrics representing marginalized communities in Saginaw Bay basin, Michigan. They found only weak correlations between benthic indicators and marginalized communities, the strongest of which was a Spearman correlation coefficient of -0.18 between benthic Hilsenhoff Biotic Index (HBI) and household size. A weak positive correlation was also found between HBI and total minority population (0.15) and between fish Index of Biological Integrity and a measure of low income (0.14). Except for fish MMI and % low income inside the 1 km buffer in our study, which had a significant correlation of -0.32 (Table 4), the strengths of the relationships in Sanchez et al. (2014) were in line with those found in our study, which were consistently <|r|=0.2 for correlations between ecological indicators and EJ metrics at all three assessment units (block, 1 km buffer, 10 km buffer). Additionally, the weak Pearson's correlations (|r|<0.20) of residual fish and macroinvertebrate MMI scores with EJ metrics after factoring out disturbance (Table 6) further supported the lack of association between flowing water condition and EJ communities. In contrast, Horvath et al. (2022) reported evidence that their spatial stream network model was better able to predict stream quality index score when poverty was included as a predictor in their model. Furthermore, they noted a negative relationship between poverty and stream quality index where an increase of 1% in poverty resulted in the model predicting a 1% decrease in stream quality index score, suggesting that low income communities may be exposed to disproportionate risk. This study, however, took place on a smaller scale, and in a more targeted area than our study,

which may explain why they observed a clearer relationship between EJ and environmental condition.

Our relative risk analysis further supported that low income or minority communities were not generally at a higher risk of being located near degraded urban flowing waters. For example, the only statistically significant relative risks occurred for benthic MMI and % minority at the 1 and 10 km assessment units and the same for the WQII (Table 5). This stronger relationship between benthic MMI and % minority was also in accordance with the Spearman rank correlations, which also showed significant relationships between the benthic MMI and % minority at the 1 and 10 km assessment units. Thus, it appears that in general there was a slightly stronger signal with % minority than for % low income and associations with benthic macroinvertebrates were slightly stronger than with fish. Similar studies have attributed differing results between invertebrates and fish to spatial scale as macroinvertebrates represent more local conditions and respond more quickly to stress than fish, which often migrate further distances and are therefore often exposed to a broader range of conditions (Allan et al. 1997; Flinders et al. 2008; Sanchez et al. 2014). These findings could suggest relationships between socioeconomic factors and environmental condition are more easily observed in benthic macroinvertebrates than fish due to scale.

There are some unique considerations that need to be made when trying to assess relationships between flowing water condition and EJ communities. For example, flowing waters are inherently connected and influenced by local as well as upstream factors (Hynes 1975; Booth et al. 2014). Therefore, local benthic and fish communities are affected by cumulative watershed processes occurring upstream, such as sedimentation and nutrients inputs, as well as by local habitat condition, such as of the riparian and instream area (Vinson and Hawkins 1998; Markovic et al. 2019). It is likely that the large effects from cumulative watershed processes occurring upstream make it difficult to observe relationships between EJ communities and local flowing water condition. Furthermore, the local effects of urbanization associated with lower income communities may not be materially different from those associated with higher income communities. In both communities, the stream is likely facing the same local habitat degradation (e.g., destruction of the riparian zone and elevated inputs of sediment). For example, the resulting riparian zone and instream habitat degradation caused by an expensive condominium complex may not be distinguishable from the degradation caused by a neglected, rundown riverfront.

Assessment unit and spatial scale can also have large effects on the outcome of environmental justice studies (Baden et al. 2007; Mohai et al. 2009; Daneshvar et al.

2016). We considered three assessment units as the census block and as the 1 and 10 km radius circles around the sampling location, but other methods exist for delineating assessment units. For example, Maantay and Maroko (2009) used a Cadastral-based Expert Dasymetric System method (CEDS) to represent the uneven distribution of a population more realistically in a given area. They found that the population estimated to be at risk from flooding based on the CEDS method was much higher than the population estimated to be at risk based on a centroid-containment method (73% less estimated to be affected by flooding) and a filtered areal-weighting interpolation method (37% less). Furthermore, within urban areas, low income / high income and low minority / high minority communities are often distributed patchily and the extremes of low and high are often located in close proximity, which further complicates finding relationships based on coarse assessment units which assume homogeneity within the unit. This complication may explain why many EJ studies focus on looking for associations between EJ communities and environmental degradation in the areas around discrete and isolated sites, such as toxic waste sites (Bowen et al. 1995; Bullard et al. 2008). In our study, at a national scale and with the coarseness of census bureau demographic data at the block level, it was not possible to obtain the resolution that may have been necessary to capture the effects of certain local factors on an association between EJ communities and flowing water degradation. The scale used in this study may not be sensitive enough to capture inequity and a measure of segregation or unevenness may provide more insight into relationships between socioeconomic parameters and environmental degradation (Morello-Frosch and Jesdale 2006).

Summary and conclusions

Human activities often lead to environmental degradation and there is a growing concern that this degradation is distributed inequitably in society, particularly in urban areas. Mitigating degradation and addressing inequities first requires identifying where, and to what extent, these activities occur. To this end, we assessed the pervasiveness of urban river and stream (flowing water) degradation across the CONUS and then assessed whether the degree of degradation was related to metrics of environmental justice. We found that urban flowing water degradation is indeed pervasive across the CONUS with the proportion of urban waters being in poor condition often twice as high as the nation as a whole.

Based on both fish and macroinvertebrates assemblages, urban flowing waters were mostly in poor condition, with fish (52% poor) being in somewhat better condition than macroinvertebrates (80% poor). Urban flowing water

quality is also mostly in poor condition for water quality integrity and nutrient concentrations, whereas the majority of the length was in good condition for enterococci and dissolved oxygen. None of the urban sites had good physical habitat, and 75% of the length was in poor condition based on riparian disturbance. The majority of the urban flowing water length, however, was in good physical habitat condition based on riparian vegetation, instream cover, and bed sediment.

Despite this widespread degradation of urban flowing waters, we found no strong evidence that flowing water degradation was highly related to the two EJ measures we analyzed. Results did vary among the assessed ecological indicators and EJ metrics, with the strongest associations being observed for fish MMI, and riparian disturbance with % low income at the 1 km buffer scale. There were stronger and more significant correlations between the landscape disturbance metrics and EJ metrics and these relationships varied by spatial scale. The % low income was more strongly related to landscape disturbance at the 1 km scale whereas % minority was more strongly related at the 10 km scale. To our knowledge, this is the first study to assess the pervasiveness of urban flowing water degradation at the CONUS-level using a consistent database, and then assess whether the degree of degradation is related to common environmental justice metrics. A number of factors may obscure relationships between flowing water condition and environmental justice indicators. Analyses could be extended in the future to small lakes and wetlands in urban areas that may be better related to local conditions and less influenced by watershed scale factors. More research is needed to assess the potential inequitable access to healthy rivers and streams.

Declarations

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