A Longitudinal Assessment of Benthic Macroinvertebrate Diversity and Water Quality along the Bronx River

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Abstract - The Bronx River is an urban waterway with a long history of anthropogenic disturbance. We conducted a longitudinal assessment of the Bronx River's water quality by measuring benthic macroinvertebrate diversity at 6 sites along the river. We integrated long-term water-quality data collected by the New York State Department of Environmental Conservation. We found that the overall water quality of the river has remained moderately impacted over different timepoints throughout the past 22 years. The study site upstream of combined sewage overflows and municipal separate stormwater systems exhibited healthier biological profiles, whereas the most-downstream sites exhibited slight declines in water quality. The most recent survey of the Bronx River (2020) revealed that high invasive species dominance was associated with benthic macroinvertebrate communities that were less healthy. Notably, one invasive species not documented in historical surveys, *Corbicula fluminea* (Asian Clam), was sampled in 5 of 6 study sites during the 2020 surveys. Moreover, no species were sampled from the order Ephemeroptera (mayflies) in 2020 despite being present in previous surveys. These results can be used to guide the management of urban rivers.

Introduction

Urban freshwater rivers are critical ecosystems for wildlife and humans (Albert et al. 2020). Many animals utilize urban rivers for sources of food, water, and living space, e.g., benthic macroinvertebrates (Wilson et al. 2021), birds (Xie et al. 2020), and fishes (Zanatta et al. 2017). In addition to providing habitats for a wide variety of nonhuman animals, urban rivers also provide different resources for humans including food, water, transportation, and recreation (Kondolf and Pinto 2017, Lerner and Holt 2012). However, over the past century, urban rivers have undergone extensive degradation and overexploitation, which has been largely attributed to increased urbanization (Beißler and Hack 2019, Bernhardt and Palmer 2007, O'Neil et al. 2016). Some factors associated with urbanization that have contributed to the decline of urban rivers include increased impervious surface cover (Bauer et al. 2007, Shuster et al. 2005), municipal and industrial discharges (Paul and Meyer 2001), and escalated human population density (Olson et al. 2016). These and other anthropogenic factors have led to rapid declines in freshwater biodiversity (Darwall et al. 2018, Fierro et al. 2018).

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Benthic macroinvertebrates are important components of urban freshwater ecosystems. These animals provide vital ecological services including nutrient cycling, decomposition, and food sources for both aquatic and land animals (Cao et al. 2018, Covich et al. 1999, Paul and Meyer 2001, Wallace and Webster 1996). Additionally, benthic macroinvertebrates have been found to be important indicators of water quality as they vary in their sensitivity to environmental stressors (Hilsenhoff 1987). Some benthic macroinvertebrate taxa, such as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), are known to be very sensitive to degraded water quality, whereas other taxa, such as Asellidae (isopods), Chironomidae (non-biting midges), and Tubificidae (sludge worms), have been found to be more tolerant of pollution (Hilsenhoff 1987). Because of their value in indicating various disturbances in aquatic habitats, benthic macroinvertebrates are frequently used in biomonitoring surveys (Bae et al. 2005; Bode et al. 1998, 2003; Deborde et al. 2016; Linke et al. 1999; Muralidharan et al. 2010; Smith et al. 2015).

Several factors, including upstream versus downstream locations (e.g., Ogbeibu and Oribhabor 2002), forested versus developed areas (e.g., Miserendino et al. 2011), and the proportion of invasive species (e.g., Francis et al. 2019), have been found to influence community composition of benthic macroinvertebrates in urban rivers. As rivers flow downstream, they tend to accumulate more pollutants, municipal discharge, and industrial waste (Alexander et al. 2007, Schertzinger et al. 2019). Accordingly, several studies have reported declines in water quality (e.g., Miskewitz and Uchrin 2013, Sun et al. 2016, Svensson et al. 2018) and shifts in benthic macroinvertebrate community composition (e.g., Azrina et al. 2006, Gray 2004) on an upstream–downstream gradient. In addition, the dominant land-cover type is another factor predictive of benthic macroinvertebrate diversity and water quality (du Plessis et al. 2015, Sponseller et al. 2001). In contrast to greenspaces, highly developed areas are comprised of high proportions of impervious surface cover, which results in predictable changes in stream ecology (Bauer et al. 2007, Paul and Meyer 2001) and major declines in benthic macroinvertebrate diversity (Paul and Meyer 2001, Utz et al. 2009). Finally, urban rivers are often dominated by invasive species (Francis et al. 2019). Two common invasive species observed in freshwater streams of North America are *Corbicula fluminea* (O.F. Müller) (Asian Clam; Ilarri and Sousa 2012, Sousa et al 2008) and *Faxonius rusticus* (Girard) (Rusty Crayfish; Wilson et al. 2004). Previous studies have shown that these invasive species can have adverse effects on other benthic macroinvertebrate taxa (Ferreira-Rodríguez et al. 2018, Haag et al. 2021, McCarthy et al. 2006, Modesto et al. 2019, Nilsson et al. 2012, Smith et al. 2019). For example, Asian Clams can negatively impact native bivalves by reducing the survival of native mussels' larva (Modesto et al. 2019) and by outcompeting native bivalves for resources (Ferreira-Rodríguez et al. 2018, Strayer 1999). Additionally, the Rusty Crayfish has been found to outcompete native crayfish (Smith et al. 2019) and reduce benthic macroinvertebrate abundance (McCarthy et al. 2006, Nilsson et al. 2012). Understanding the effects that these 3 factors—upstream versus downstream locations, forested versus developed areas, and the proportion of invasive species—have on benthic macroinvertebrate communities can be a useful way to monitor the overall health of urban rivers.

Over the past few decades, the New York State Department of Environmental Conservation's Stream Biomonitoring Unit (NYSDEC-SBU) has conducted biological assessments to evaluate the water quality of the Bronx River (Bode et al. 1998, 2003; Smith et al. 2015). The Bronx River runs through one of the largest metropolitan areas in the world. It is New York City's only freshwater river and was once a source of drinking water for Native Americans and early European settlers (de Kadt 2011). However, in the ensuing centuries, several factors contributed to the degradation of the river: the operation of mills from 1680 to 1934, the development of railroads since 1841, rapid population growth in the Bronx during the mid-1800s, and industrial development throughout the 19th century (de Kadt 2011). During the 20th century and afterwards, other factors, including the straightening and rechanneling of the river, gas and oil runoff, the dumping of automobile bodies into the river, and combined sewage overflow, also contributed to the decline in habitat quality of the Bronx River (de Kadt 2011).

The first biological assessment of the Bronx River by NYSDEC-SBU was conducted in 1998 (Bode et al. 1998), followed by 2 subsequent surveys, one in 2003 (Bode et al. 2003) and another in 2015 (Smith et al. 2015). In these 3 surveys, benthic macroinvertebrates were used as biological indicators of water quality (Bode et al. 1998, 2003; Smith et al. 2015). The survey conducted in 1998 revealed that the Bronx River exhibited moderately impacted water quality (Bode et al. 1998). The 2 subsequent studies (Bode et al. 2003, Smith et al. 2015) found similar water quality impact as the initial study. The results of these 3 biological assessments suggest that there has been no apparent change in benthic macroinvertebrate diversity between 1998 and 2015. Therefore, evaluating how the water quality of the Bronx River has changed temporally and which locations/habitats have increased, maintained, or decreased in biodiversity over time is important to better inform urban stream monitoring and restoration strategies.

The aim of this study was to conduct a longitudinal assessment of benthic macroinvertebrate diversity along the Bronx River as an indication of water quality. The study addressed 3 major research questions:

(1) How does benthic macroinvertebrate diversity currently vary based on geographical location, land cover, and proportion of invasive species?
(2) How have biodiversity indices, pH, and physical variables of the Bronx River changed over the past 22 years?
(3) How has benthic macroinvertebrate diversity varied among study sites over the past 22 years?

Based on our first research question, we predicted that habitats located upstream, surrounded predominantly by greenspace, and comprised of relatively low invasive species abundance, would harbor greater benthic macroinvertebrate diversity than habitats located downstream, surrounded predominantly by developed space, and comprised of relatively high invasive species abundance. Because previous surveys of the Bronx River have not shown any notable changes in water quality, we predicted that biodiversity indices will continue to remain similar to past values. Alternatively, since several restoration efforts have attempted to improve the health of the Bronx River, water quality might have improved, leading to increased biodiversity compared to previous study years, or a weak or absent spatial gradient.

To test each prediction, we sampled benthic macroinvertebrates at 6 study sites along the Bronx River. Additionally, we measured pH, water temperature, river depth, and river width at each site. The study sites were selected to correspond with the previous surveys conducted along the Bronx River by the NYSDEC-SBU (Bode et al. 1998, 2003; Smith et al. 2015). Because the Bronx River is a critical ecosystem for sustaining urban biodiversity, long-term monitoring of the river's water quality as measured by benthic macroinvertebrate communities can be helpful to mitigate the effects of anthropogenic disturbances, as well as to monitor and conserve benthic macroinvertebrate diversity in degraded habitats.

Methods

Field-site description

This study was conducted at 6 riffle habitat locations along the Bronx River (Figs. 1, 2). The Bronx River extends ~36 km from its source in Westchester County to its mouth, a tidal strait connected to the Long Island Sound (de Kadt 2011, Natural Resources Group 2008). Assessments conducted on the Bronx River during 1998 and 2003 include surveys at 4 locations: (1) Valhalla; (2) White Plains; (3) Tuckahoe; and (4) East Gun Hill Road (Bode et al. 1998, 2003). However, a survey conducted in 2015 excluded 1 of the 4 sites (Tuckahoe) and added 2 additional sites (Mount Vernon and East 182nd Street) for a total of 5 study locations. For the purposes of this study, we surveyed all 6 of these locations and ordered the study sites from 1 to 6 (from north to south) incorporating all sites previously surveyed by NYSDEC-SBU. The northernmost study sites (sites 1–3) were located in suburban areas, whereas the southernmost study sites (sites 4–6) were located in highly urbanized areas.

Macroinvertebrate sampling and identification

We followed NYSDEC-DOW's guidelines for benthic macroinvertebrate sampling, sorting, sub-sampling, and identification (NYSDEC-DOW 2019). We collected benthic macroinvertebrates on 12 September 2020 to correspond with sampling dates of the 3 previous studies (23 September 1998 [Bode et al. 1998], 17 September 2003 [Bode et al. 2003], 12 September 2015 [Smith et al. 2015]). To sample macroinvertebrates, we used the standardized kick-sampling method (see Bode et al. 1998, 2003; Smith et al. 2015). Following sample collection, we preserved specimens in jars containing two-thirds 95% ethanol and one-third river water. During sample sorting, we used a US No 40 standard sieve to clean any residue while rinsing samples with tap water and then placed the rinsed samples on a gridded enamel pan such that they were evenly placed across the bottom of the pan. We used a random-number generator to select samples from each 6.5 cm x 6.5 cm numbered square grid. We placed the randomly selected samples in a Petri dish and used a dissecting stereomicroscope (Zeiss Stemi 2000-C; Munich, Germany) to sub-sample 100 organisms. We sorted and counted the sub-sampled organisms and

Figure 1. The 6 study sites along the Bronx River in New York surveyed for longitudinal assessment of benthic macroinvertebrate diversity: (A) Valhalla, (B) White Plains, (C) Tuckahoe, (D) Mount Vernon, (E) East Gun Hill Road Bronx, (F) East 182nd Street Bronx. Photographs © Bobby Habig.

placed them in vials containing 70% alcohol. We identified all preserved organisms to the family level using 2 identification keys (Pennak 1978, Voshell 2002).

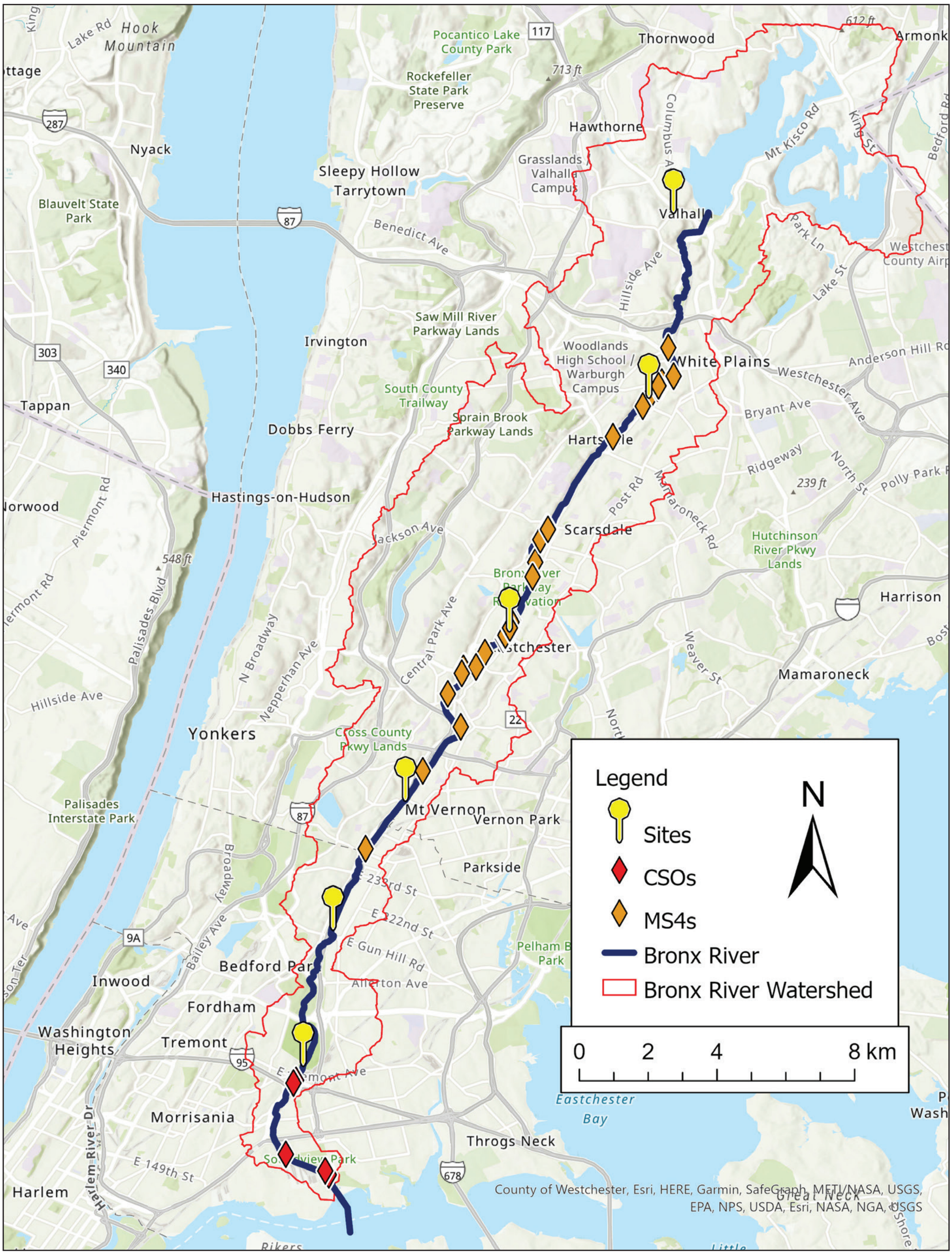

Figure 2. Map of the 6 study sites (yellow pins) surveyed for longitudinal assessment of benthic macroinvertebrate diversity. Red diamonds represent locations of combined sewage overflows and orange diamonds indicate locations with municipal separate stormwater systems. Map © Amanda Goldstein and used with permission.

Water chemistry and physical variables

At each study site, we measured water temperature (°C) using a digital thermometer (REOTEMP TM99A; San Diego, CA), river width and depth at the midpoint to the nearest centimeter using a closed reel tape, and pH to the nearest hundredth using a pH meter (YSI PRO 10 pH/ORP/temperature portable meter; Yellow Springs, OH).

Land-cover type

We calculated percent land cover using ArcGIS Pro 2.6 (Esri, Redlands, CA) and the National Land Cover Database (Dewitz 2019). We created circular buffers with 100-m radii surrounding the center of each of the 6 study sites. We categorized percent land cover into 3 distinct land-cover types by combining the percent land cover of similar groups into 3 variables: (1) developed, (2) open space, and (3) greenspace (Goldstein et al. 2022). Developed areas were largely comprised of constructed material and impervious surface cover, open spaces were comprised of homogenous vegetation in the form of lawns and golf courses, and greenspaces were dominated by trees and shrubs (Dewitz 2019).

Biodiversity indices

We used 6 family-level biodiversity indices recommended by the NYSDEC-DOW (2019) (see Supplemental Table S1 in Supplemental File 1, available online at https://www.eaglehill.us/NENAonline/suppl-files/n29-4-N1976-Habig-s1, and for BioOne subscribers, at https://www.doi.org/10.1656/N1976.s1): (1) family richness, defined as the number of distinct benthic macroinvertebrate families based on a sub-sample of 100 randomly selected organisms (Xu et al. 2014); (2) Ephemeroptera-Plecoptera-Trichoptera (EPT) family richness, the total number of mayfly (Ephemeroptera), stonefly (Plecoptera), and caddisfly (Trichoptera) larvae families in a sub-sample of 100 randomly selected organisms (NYSDEC-DOW 2019); (3) Hilsenhoff's family biotic index (FBI), an index measuring a benthic macroinvertebrate community's pollution-tolerance level (Hilsenhoff 1988, NYSDEC-DOW 2019); (4) percent model affinity (PMA), an index comparing a sample community to a model non-impacted community based on the abundance percentage of major benthic macroinvertebrate taxa (higher percentage similarity indicates a healthier community; NYSDEC-DOW 2019); (5) Biological assessment profile (BAP), an index of overall water quality based on conversion formulas that transforms the first 4 biodiversity indices above onto a common scale (NYSDEC-DOW 2019); and (6) dominant family, defined as the percentage of the most numerous family based on a sub-sample of 100 randomly selected organisms (NYSDEC-DOW 2019). We converted data from 3 previous surveys conducted along the Bronx River (Bode et al. 1998, 2003; Smith et al. 2015) into the 6 family-level biodiversity indices and compared them to the results from our survey.

We calculated BAP scores based on methods previously validated by NYSDEC-DOW (2019). Briefly, we used conversion formulas (NYSDEC-DOW 2019) to standardize 4 biodiversity indices (family richness; EPT family richness; FBI; PMA) onto a common scale of 0 to 10 (low to high water quality). The resulting BAP score is an average of these 4 standardized biodiversity indices. We classified BAP scores into 4 water quality impact categories: (1) severely impacted (BAP = 0–2.5), (2) moderately impacted (BAP = 2.5–5.0), (3) slightly impacted (BAP = 5.0–7.5), and (4) non-impacted (BAP = 7.5–10). Hereafter, BAP and overall water quality are used interchangeably.

Statistical analyses

All analyses were conducted using R version 4.03 (R Core Team 2020). We conducted Kruskal–Wallis tests using the 'stats' package to compare differences across the 6 study sites (Figs. 1, 2). We divided the Bronx River into 3 reaches—upper (sites 1 and 2), middle (sites 3 and 4), and lower (sites 5 and 6)—to classify study sites based on geographical locations. We classified study sites as having "high" invasive species dominance if more than 50 percent of a sub-sample of 100 randomly selected organisms were invasive, and "low" invasive species dominance if less than 50 percent of 100 randomly selected organisms were invasive.

We conducted mixed-effects repeated-measure ANOVAs using the 'nlme' package (Pinheiro et al. 2017) to compare metrics of biodiversity both temporally and spatially. For all temporal and spatial analyses, we included 1 of the 6 indices of biodiversity as a response variable. For temporal analyses, we also ran separate models including each water chemistry (pH) and physical (temperature, river width, and river depth) parameter as a response variable, study year as a predictor, and study site as a random effect. We ran a random intercept and random slope model accounting for site-level variability among temporal data. For spatial analyses, we included study site as a predictor variable and study year as a random effect. The temporal analyses, based on comparisons between years, only included the 3 study sites (sites 1, 2, and 5) in which there were data available for all 4 time periods (1998, 2003, 2015, 2020). Following all analyses, we used the 'multcomp' package to conduct Tukey post hoc tests. This allowed us to compare differences between years and sites.

Results

Biodiversity indices based on samples collected in 2020

Biodiversity indices, based on samples collected in 2020, varied across the 6 study sites (Table 1). Family richness varied from 6 to 10 (mean = 9, SD = 1.55); sites with the highest family richness (n = 10) were located at sites 1, 3, and 5, while the site with the fewest families (n = 6) was located at site 4. EPT family richness did not vary across the 6 study sites (mean = 1, SD = 0). FBI (pollution tolerance) varied from 5.49 to 6.06 (mean = 5.86, SD = 0.20); the study site with the highest proportion of pollution-tolerant taxa was located at site 5, and the study site with the lowest proportion of pollution-tolerant taxa was located at site 1. PMA varied from 15% to 54% (mean = 24.5, SD= 14.73); the site with the highest percent affinity (54%) to a non-impacted community was located at site 1, and the location with lowest percent affinity (15%) was found at site 4. Family dominance varied from 25% to 68% (mean = 53.5, SD =16.13). The most dominant family was Corbiculidae at 3 study sites (sites 2, 5, and 6), Gammaridae at 2 study sites (sites 3 and 4), and Hydropsychidae at 1 study site (site 1). Notably, Family Corbiculidae was comprised entirely of Asian Clams, an invasive species not previously documented in the 3 previous studies of the Bronx River (Bode et al. 1998, 2003; Smith et al. 2015).

Chemical, physical, and land-cover variables based on samples collected in 2020

Water chemistry and physical variables measured in 2020 varied across study sites (Table 1): pH varied from 7.01 to 7.28, river depth varied from 7.5 cm to 56.0 cm, river width varied from 820 cm to 1680 cm, and water temperature varied from 17.5 °C to 23.0 °C. Most study sites were dominated by developed land and open space (i.e., lawns and golf courses); on average, only 10.38% of the land surrounding the 6 study sites (100-m radii) was comprised of greenspace (Table 1).

Proportion of invasive species based on samples collected in 2020

The proportion of invasive species in the 2020 dataset varied from 0% to 68% (mean = 38.00; SD = 26.59). Specifically, site 1 harbored 0% invasive species, while site 5 was comprised of 68% invasive species. Overall, there were 2 invasive species sampled across study sites: the Asian Clam (Family Corbiculidae; mean = 36.17, SD = 26.60, min–max = 0–68) and Rusty Crayfish (Family Cambaridae; mean = 3.50, SD = 4.76, min–max = 0–11).

Table 1. Mean, minimum, maximum, and standard deviation (SD) of benthic macroinvertebrate community parameters, water chemistry and physical variables, and percent land cover (100-m radii) across 6 study sites sampled along the Bronx River in 2020.

Variable	Mean	Min	Max	SD
Biodiversity indices				
Family richness	9.00	6.00	10.00	1.55
Family richness (DEC conversion scale)	4.59	2.31	5.50	1.25
EPT family richness	1.00	1.00	1.00	0.00
EPT family richness (DEC conversion scale)	2.50	2.50	2.50	0.00
Family biotic index	5.86	5.49	6.06	0.20
Family biotic index (DEC conversion scale)	4.50	4.07	5.64	0.58
Percent model affinity	24.50	15.00	54.00	14.73
Percent model affinity (DEC conversion scale)	1.05	0.00	5.81	2.34
Biological assessment profile (BAP)	3.16	2.25	4.86	0.89
Family dominance	53.50	25.00	68.00	16.13
Water chemistry and physical variables				
pH	7.17	7.01	7.28	0.10
Depth (cm)	29.97	7.50	56.00	19.18
Width (cm)	1195.00	820.00	1680.00	328.62
Water temperature (°C)	20.18	17.50	23.00	2.00
Percent land cover				
Percent developed	46.95	12.50	86.96	28.33
Percent open space	41.29	4.35	80.00	30.59
Percent greenspace	10.38	0.00	25.00	11.08

2020 analyses

Based on Kruskal–Wallis tests, we found that study sites with low invasive species dominance exhibited higher PMA (KW = 4.80, P = 0.028) and BAP (KW = 3.43, P = 0.064) profiles than study sites with high invasive species dominance. However, we found no significant differences in biodiversity indices across study sites when comparing geographical location (reach) and dominant land-cover type.

Longitudinal changes in abundance by family from 1998 to 2020

Twenty-eight unique families (mean = 18.25; SD = 3.99; min–max = 15–22) were identified over the past 22 years, varying from a high of 22 in 1998 to a low of 15 in 2003 (see Supplemental Tables S2–S5 in Supplemental File 1). The 5 most common families sampled during the 22-year period were Chironomidae (21.16% of total samples), Gammaridae (18.58% of total samples), Hydropsychidae (17.63% of total samples), Corbiculidae (10.89% of total samples), and Naididae (8.21% of total samples) (Fig. 3). In 1998, the 2 most common families sampled along the Bronx River were Hydropsychidae (37.25% of 1998 samples) and Chironomidae (31.75% of 1998 samples) (see Supplemental Table S2 in Supplemental File 1). In 2003, the 2 most common families sampled were Chironomidae (38.75% of 2003 samples) and Naididae (29.00% of 2003 samples) (see Supplemental Table S3 in Supplemental File 1). In 2015, the 3 most common families sampled were Gammaridae (23.00% of 2015 samples), Hydropsychidae (19.60% of 2015 samples), and Chironomidae (19.60% of 2015 samples) (see Supplemental Table S4 in Supplemental File 1). In 2020, the 2 most common families sampled were Gammaridae (35.67% of 2020 samples) and Corbiculidae (34.50% of 2020 samples) (see Supplemental Table S5 in Supplemental File 1).

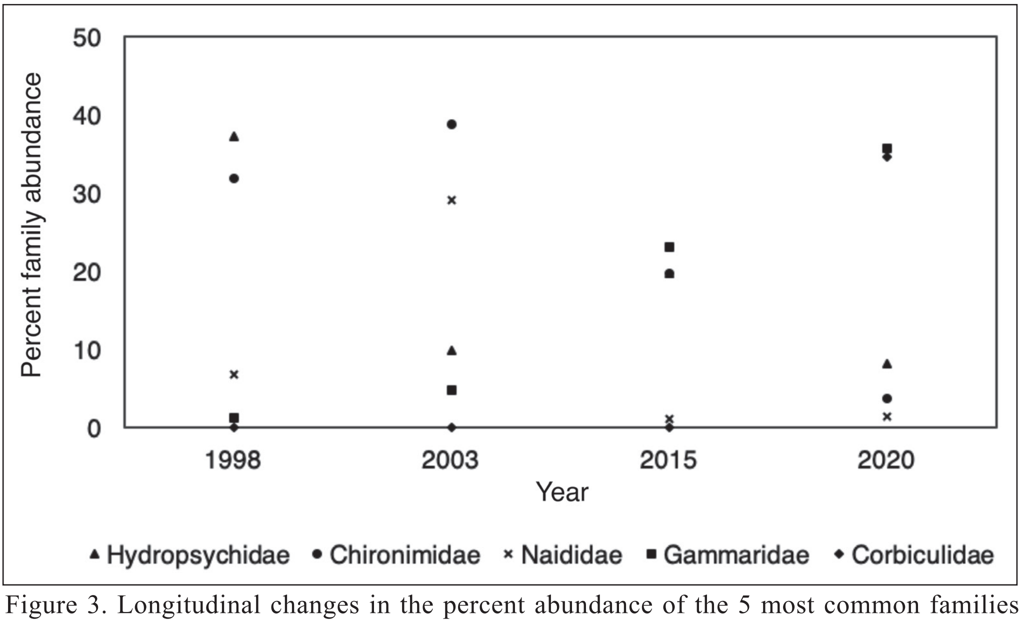

Figure 3. Longitudinal changes in the percent abundance of the 5 most common families sampled along the Bronx River from 1998 to 2020.

Temporal changes in biodiversity indices

Over the past 22 years, overall water quality along the Bronx River has varied from slightly impacted to severely impacted (Fig. 4A–F). Across all years, the average BAP score was 3.74 (SD = 1.05), which is indicative of moderately impacted water quality. The study site located farthest from the mouth of the Bronx River (site 1) exhibited the highest average BAP score (mean = 5.23, SD = 0.58, min–max = 4.84–6.07); however, the water quality impact scale at this location has steadily decreased over the past 22 years from slightly impacted in 1998 (BAP = 6.07) to moderately impacted in 2020 (BAP = 4.86) (Fig. 4A). Site 2, which is located 27.8 km from the river mouth, consistently exhibited a moderate water quality impact scale (mean = 2.99, SD = 0.48, min–max = 2.52–3.64; Fig. 4B). Site 3, located 19.6 km from the river mouth, exhibited a decline in water quality from 1998 (moderate

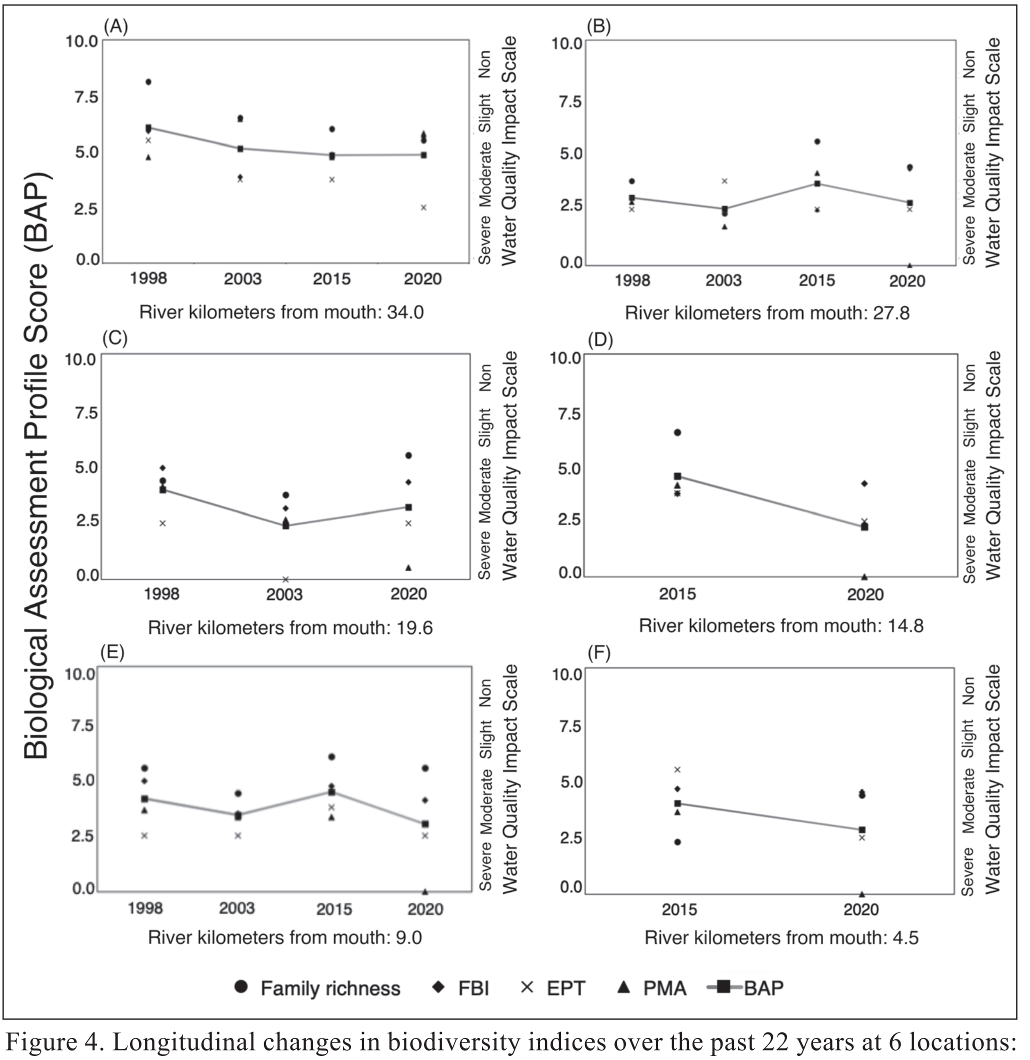

Figure 4. Longitudinal changes in biodiversity indices over the past 22 years at 6 locations: (A) site 1 (Valhalla), (B) site 2 (White Plains), (C) site 3 (Tuckahoe), (D) site 4 (Mount Vernon), (E) site 5 (East Gun Hill Road), (F) site 6 (East 182nd Street Dam).

impact) to 2003 (severe impact), but the water quality appears to have somewhat recovered in 2020 (moderate impact) (Fig. 4C); the BAP score at site 3 varied from 2.39 to 3.98 (mean = 3.19, SD = 0.80). Site 4, located 14.8 km from the river mouth, was measured at 2 different time points (Fig. 4D). The BAP score at site 4 was 4.52 (moderate impact) in 2015 and 2.25 (severe impact) in 2020. The mean BAP score at site 4 was 3.39 (SD = 1.61). Site 5 (9.0 km from the river mouth) consistently exhibited a moderate water quality impact scale over the past 22 years (mean = 3.75, SD = 0.65, min–max = 3.02–4.44; Fig. 4E). Finally, the southernmost study site (site 6), which was located 4.5 km from the river mouth, exhibited a BAP score of 4.02 (moderate impact) in 2015 and 2.85 (moderate impact) in 2020 (Fig. 4F); the mean BAP score at site 6 was 3.44 (SD = 0.83).

Changes in biodiversity and chemical/physical variables across years

For the 3 study sites (Sites 1, 2, and 5) in which there were data available for all 4 time periods (1998, 2003, 2015, 2020), mixed-effects repeated-measure ANOVAs revealed differences across years for certain biodiversity indices, including BAP and FBI (Table 2). Of note, we found that there were lower BAP scores in 2020 than in 1998 and lower FBI values in 2003 than in 1998 or 2020 (Table 2). Moreover, repeated measure ANOVAs revealed differences across years for certain chemical and physical variables, including pH, river depth, river width, and temperature (Table 2).

Changes in biodiversity across study sites from 1998 to 2020

From the 6 study sites that were sampled from 1998 to 2020, mixed-effects repeated-measure ANOVAs revealed differences between study locations for certain biodiversity indices, including BAP, Family richness, FBI, EPT, and PMA (Table 3,

Table 2. Tukey post hoc test results for differences in biodiversity indices and chemical and physical variables between years. Statistical significance: † $P < 0.1$, * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$.

	Year comparison	Estimate	SE	*z*-value	*P*
Biodiversity indices					
Biological assessment profile (BAP)	1998–2020	0.890	0.383	-2.322	0.093†
Family biotic index (FBI)	1998–2003	1.383	0.491	-2.816	0.025*
Family biotic index (FBI)	2003–2020	-1.453	0.491	2.959	0.016*
Water chemistry and physical variables					
pH	1998–2015	-0.733	0.104	7.088	<0.001***
pH	1998–2020	0.407	0.104	-3.930	<0.001***
pH	2003–2015	-0.900	0.104	8.698	<0.001***
pH	2003–2020	0.240	0.104	-2.320	0.094†
pH	2015–2020	1.140	0.104	-11.018	<0.001***
River depth	1998–2015	-33.333	12.871	2.590	0.047*
River depth	2015–2020	30.767	12.871	-2.390	0.079†
River width	1998–2015	-866.700	209.200	4.143	<0.001***
River width	1998–2020	-570.000	209.200	2.725	0.032*
Temperature	1998–2015	-3.300	1.355	2.435	0.071†
Temperature	2003–2015	-4.500	1.355	3.320	0.005**
Temperature	2003–2020	-3.833	1.355	2.828	0.024*

Fig. 5A–F). Notably, the study site located farthest from the mouth of the Bronx River (site 1) exhibited higher BAP scores and PMA than all 5 southern locations (sites 2–6).

Discussion

In a longitudinal assessment of New York City's only freshwater river, we found spatial and temporal differences in overall water quality as indicated by benthic macroinvertebrate diversity. On a spatial scale, the presence of invasive species in the Bronx River was associated with differences in water quality across study sites. Specifically, study sites with high invasive-species dominance exhibited benthic macroinvertebrate communities that were less healthy than locations with low invasive-species dominance. Moreover, in support of our prediction that upstream habitats exhibit higher diversity of benthic macroinvertebrates than downstream habitats, we found, compared to all downstream study sites, site 1 (Valhalla) exhibited the healthiest biological profiles. On a temporal scale, we found that on average, the overall water quality of the Bronx River has remained

Table 3. Tukey post hoc test results for differences in biodiversity indices between study sites (sites 1–6) based on 4 time periods (1998, 2003, 2015, 2020). Statistical significance: † $P < 0.1$, * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$.

Biodiversity indices	Site comparison	Estimate	SE	*z*-value	*P*
Biological assessment profile (BAP)	1–2	2.238	0.367	-6.089	<0.001***
Biological assessment profile (BAP)	1–3	1.889	0.402	-4.693	<0.001***
Biological assessment profile (BAP)	1–4	1.838	0.466	-3.944	0.001**
Biological assessment profile (BAP)	1–5	1.475	0.367	-4.014	<0.001***
Biological assessment profile (BAP)	1–6	1.788	0.466	-3.836	0.002**
Family richness	1–2	2.548	0.943	-2.702	0.073†
Family richness	1–6	3.188	1.155	-2.760	0.063†
Family biotic index (FBI)	1–2	2.083	0.358	-5.819	<0.001***
Family biotic index (FBI)	1–4	1.271	0.455	-2.794	0.057†
Family biotic index (FBI)	2–3	-1.132	0.392	2.886	0.044*
Family biotic index (FBI)	2–5	-1.300	0.358	3.633	0.004**
Family biotic index (FBI)	2–6	-1.441	0.455	3.168	0.019*
EPT family richness	1–3	2.157	0.825	-2.615	0.092†
Percent model affinity (PMA)	1–2	3.283	0.843	-3.892	0.001**
Percent model affinity (PMA)	1–3	2.728	0.924	-2.954	0.036*
Percent model affinity (PMA)	1–4	3.018	1.070	-2.822	0.053†
Percent model affinity (PMA)	1–5	2.883	0.843	-3.418	0.008**
Percent model affinity (PMA)	1–6	3.258	1.070	-3.046	0.027*

Figure 5 (see following page). Results of mixed-effects repeated-measure ANOVAs indicating differences in biodiversity indices across 6 study sites sampled at 4 timepoints. For these analyses, biodiversity indices were modeled as response variables, study site was modeled as a fixed effect, and year was modeled as a random effect. (A) Biological assessment profile (BAP), (B) family richness (C) family biotic index (FBI), (D) EPT family richness, and (E) percent model affinity (PMA).

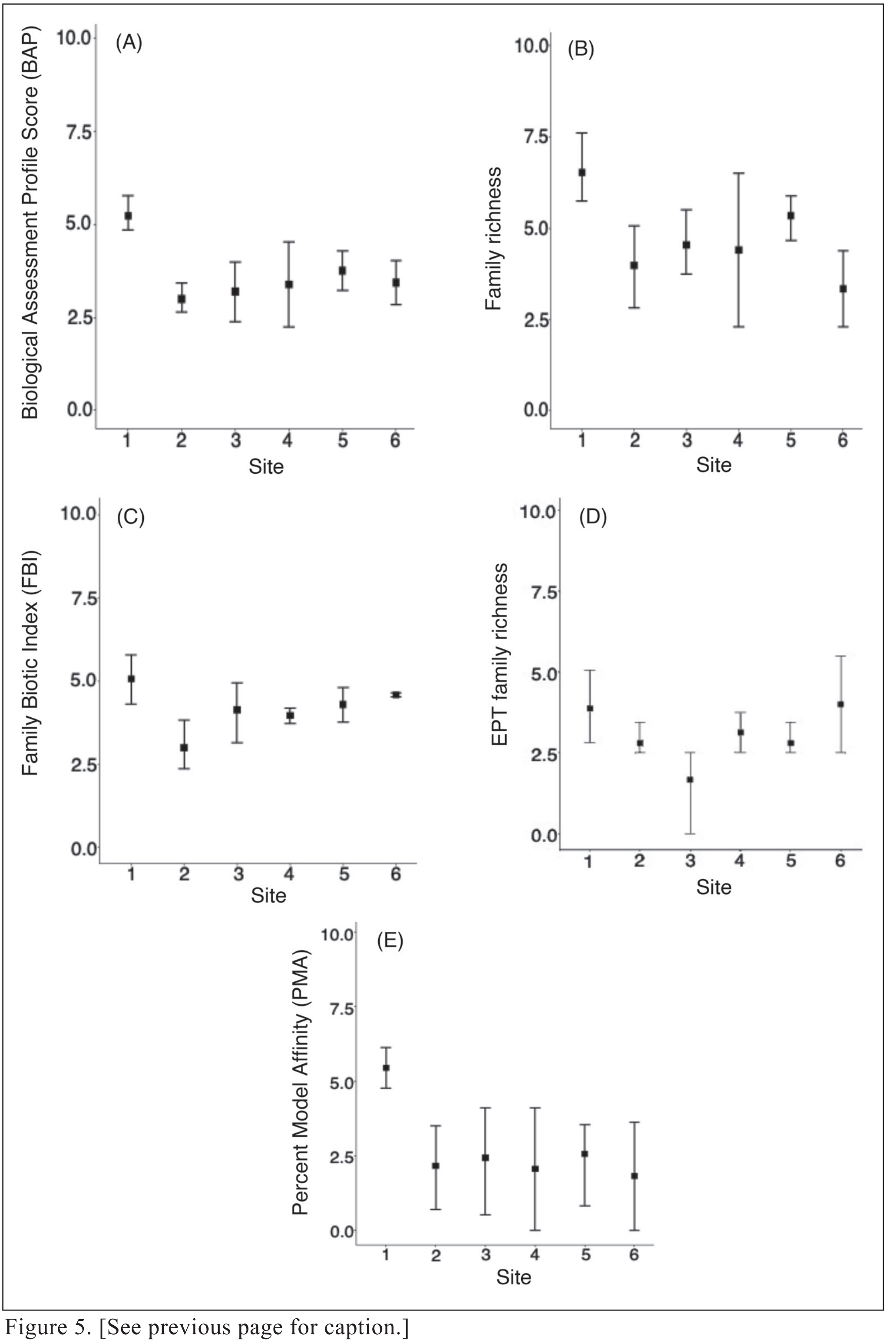

Figure 5. [See previous page for caption.]

moderately impacted over the span of 22 years, which supports our prediction that water quality along the Bronx River would remain similar to past values. However, contrary to this prediction, we also found longitudinal declines in water quality at the 3 most downstream sites: site 4 (Mount Vernon), site 5 (East Gun Hill Road), and site 6 (East 182nd Street Dam). Finally, we observed longitudinal changes in community composition as measured by benthic macroinvertebrate family dominance. Specifically, dominant benthic macroinvertebrate families in the Bronx River have shifted from Hydropsychidae (net-spinning caddisflies) and Chironomidae (non-biting midges) to Gammaridae (scuds) and Corbiculidae (Asian Clams). Results of this study highlight that temporal and spatial differences in water quality are key factors to consider in terms of urban river restoration, management, and conservation initiatives.

Invasive species impact overall water quality

The proportional abundance of invasive species was associated with 2 measures of water quality: percent model affinity (comparison to a non-impacted community) and overall water quality (BAP). Specifically, study sites with a high level of invasive-species dominance were less likely to harbor a biological community similar to a non-impacted community than study sites with a low level of invasive-species dominance. Indeed, the study sites with the most invasive species exhibited the lowest PMA, whereas the study site with no invasive species harbored a biological community most similar to a non-impacted community. Although non-impacted community indices allow for 10% "other" taxa (NYSDEC-DOW 2019), which possibly includes invasive species, we found that when invasives were present, these taxa were likely to dominate the community, hence the low PMA values. Notably, we also found that the study site with no invasive species exhibited the highest BAP score.

Two invasive species in particular were found to be established along the Bronx River: Asian Clam and Rusty Crayfish. Several studies have documented the adverse effects of these 2 species. For example, the Asian Clam has been shown to compete with native bivalves for food and habitat resources (Ferreira-Rodríguez et al. 2018, Strayer 1999). Moreover, Yeager et al. (1999) found that the Asian Clam directly impacts the mortality of native bivalves by ingesting the larva of unionid mussels. Other studies have shown that Asian Clams typically undergo large die-offs during the summer, which can release toxins into waterbodies and negatively affect native bivalve populations (Cherry et al. 2005, Cooper et al.2005). Additionally, the invasive Rusty Crayfish has been observed competing with native crayfish species for ecological resources (Olden et al. 2006, Smith et al. 2019). Several studies report a negative association between the invasive Rusty Crayfish and the density and abundance of several native benthic macroinvertebrate taxa including Ephemeroptera, Diptera, Odonata, and Gastropoda (Houghton et al. 1998, Kuhlmann 2016, McCarthy et al. 2006, Wilson et al. 2004). The adverse effects documented for these 2 invasive species might explain why study sites with high proportional abundances of invasive species exhibited lower BAP and PMA profiles.

Water quality along the Bronx River has remained moderately impacted

In support of our initial prediction, the average water quality of the Bronx River remained moderately impacted from 1998 to 2020. Study sites either experienced slight declines in overall water quality or remained relatively unchanged during this period. Some of these slight differences might reflect the changes in pH, river depth, river width, and water temperature observed across years. For example, the Bronx River exhibited significantly lower pH values in 2020 compared to the 3 historical studies. However, these results are still somewhat surprising because there have been several large-scale restoration efforts instituted over the past several decades to improve the water quality of the Bronx River (Cox and Bower 1998, de Kadt 2011, Natural Resources Group 2008, USACE 2006). For example, the Bronx River Alliance has executed several restoration initiatives, including river cleanups, litter removal, tree plantings, invasive species removal, and erosion control (de Kadt 2011). Despite these efforts, a report from the Natural Resources Group (2008) concluded that local restoration initiatives along the Bronx River have not directly benefited benthic macroinvertebrate community composition. de Kadt (2011) surmised that despite many reclamation efforts, it is difficult to improve the water quality of the Bronx River because the river is regularly inundated by combined sewage overflows, runoff, and effluent discharges. Although the water quality along the Bronx River has not significantly improved over the past 22 years, these restoration efforts might have helped to mitigate severe degradation of the river (Kail et al. 2015). Several studies have documented the limitations of stream restoration projects in urban areas (e.g., Alexander and Allan 2007, Bernhardt and Palmer 2011, Bernhardt et al. 2007, Bond and Lake 2003, Larson et al. 2001, Sundermann et al. 2011; Violin et al. 2011). For example, Violin et al. (2011) found no significant differences in physical and biological variables between restored and unrestored urban rivers in North Carolina. Several ecologists suggest that in order to restore urban streams, land managers need to take a more comprehensive approach that collectively include the following strategies: (1) the restoration of riparian vegetation, (2) instream habitat enhancement, (3) elimination of pipe stormwater treatment, (4) removal of legacy pollutants, and (5) dispersed stormwater treatment (Bernhardt and Palmer 2007; Palmer et al. 2010; Walsh et al. 2005 a, b). Of note, there are several combined sewage overflow and municipal separate stormwater system sites that feed into the Bronx River, which might explain why previous restoration efforts have had limited effects on water quality (Bernhardt and Palmer 2007). Hence, future restoration efforts might consider instream enhancement by the addition of large woody debris (Miller et al. 2010), management of wastewater effluent and legacy pollutants (Walsh et al. 2005b), improved river-catchment policies (Bernhardt and Palmer 2007), and collection of long-term pre-restoration and post-restoration data (Alexander and Allan 2007) to improve restoration utility and overall water quality of urban rivers.

Biodiversity indices along the Bronx River have changed over time

Macroinvertebrate community composition. During the past 22 years, the Bronx River has undergone several changes in community composition. One pollution-sensitive taxon that was present historically, but not in the most recent survey, was Ephemeroptera. Specifically, there was limited abundance of Baetidae (mayfly) found during surveys of 1998, 2003, and 2015. In these historical studies, the presence of mayflies was documented only at Valhalla (site 1) during the 1998 and 2003 surveys and were found at very low abundance at 5 study sites in the 2015 survey (Bode et al. 1998, 2003; Smith et al. 2015). However, mayflies were not sampled during the 2020 survey. The absence of mayflies in 2020 and the overall low proportions throughout all survey years suggest that the Bronx River is an inhospitable habitat for these pollution-sensitive taxa (Bode et al. 2003). Moreover, the invasive Rusty Crayfish might be responsible for inhibiting mayfly populations. For example, a benthic macroinvertebrate study in Wisconsin found declines in mayfly abundance in study sites and years with high Rusty Crayfish abundance (Wilson et al. 2004). Furthermore, a meta-analysis conducted by McCarthy et al. (2006) found a negative association between Rusty Crayfish and mayflies.

One invasive family that was documented in 2020 but not in the historical surveys was Corbidiculidae (represented by the Asian Clam). Interestingly, freshwater bivalves from Family Sphaeriidae were surveyed in 1998 and 2015. However, this family was not documented in the current survey of the river. One possible reason for the absence of this group could be high abundance of the invasive Asian Clam across study sites. Indeed, the Asian Clam has been found to compete with Sphaeriids for both habitat and food resources (Strayer 1999, Vaughn and Hakenkamp 2001). Despite these negative impacts, studies also suggest that Asian Clams provide ecosystem services including the provision of shelter and substrate as well as food resources for other organisms (Ilarri and Sousa 2012, Sousa et al. 2008). Overall, shifts in community composition documented along the Bronx River reflect the dynamic nature of benthic macroinvertebrate communities in an urban setting.

Family dominance. One striking result that we found when analyzing the long-term data was shifts in dominant families over the past 22 years. Two families that became more dominant over time were Gammaridae (scuds) and Corbiculidae (Asian Clam), while two families that declined over time were Hydropsychidae (net-spinning caddisflies) and Chironomidae (non-biting midges). Studies have found that Family Gammaridae is capable of thriving in polluted water (Medupin 2020, Natural Resources Group 2008), which might explain the high proportion of Gammaridae in the Bronx River. Interestingly, Family Corbiculidae was not documented in the 3 historical surveys (1998, 2003, and 2015), but became the second most dominant family in the year 2020. The invasive Asian Clam of Family Corbiculidae might have successfully invaded the Bronx River within the span of 5 years as the last survey conducted did not document the presence of this bivalve (Smith et al. 2015). Several life-history traits might have facilitated the establishment of the invasive Asian Clam along the Bronx River including rapid growth, early maturity, high fecundity rate, and rapid dispersal ability (McMahon 2002, Sousa et al. 2008).

While Gammaridae and Corbiculidae became the 2 most dominant families in 2020, Hydropsychidae (net-spinning caddisflies) and Chironomidae underwent precipitous declines over the past 2 decades. Because Trichopterans, which include the family Hydropsychidae, tend to be pollution-sensitive, the slight declines in water quality that were documented in some of the study sites might explain the longitudinal reductions in the proportion of Hydropsychidae (Bradt 2014, Bradt and Ruggiero 2017). Alternatively, the Asian Clam (Corbiculidae) might competitively exclude Hydropsychidae, but more data is required to test this hypothesis. Finally, Baladrón and Yozzo (2020) also observed declines in densities of Chironomidae at different study sites located along the Bronx River. Moreover, several studies suggest that many species from this family are sensitive to different sources of pollution (Al-Shami et al. 2010, Odume and Muller 2011, Wright and Burgin 2009). Collectively, these results provide evidence of the ephemeral nature of macroinvertebrate family dominance in the Bronx River. Whether these shifts are the results of natural variation or caused by ecological or anthropogenic disturbances remain unknown.

The most upstream location has better water quality compared to all downstream sites

Among the 6 study sites sampled across 22 years, the most upstream site (Valhalla) exhibited the highest BAP scores and PMA compared to all 5 downstream locations. Several factors, including geomorphological, biotic, and anthropogenic variables, might explain these results.

Geomorphological variables. Changes in geomorphological characteristics from upstream to downstream might result in differences in habitat quality. The river continuum concept, the idea that rivers undergo changes in geomorphological properties, including width, depth, and complexity, as the river flows from an upstream to downstream location, might explain spatial differences in macroinvertebrate diversity (Sedell et al. 1978). However, the Bronx River contains several dams over a short distance, which might impede the river's continuum (DeMarte et al. 2016). The discontinuum concept alternatively posits that dams and other barriers create a mosaic of patches that possibly disrupt allochthonous and autochthonous inputs, which result in changes in stream characteristics that might impact patterns of macroinvertebrate diversity (Doretto et al. 2020, Poole 2002).

Biotic variables. Biotic factors might also explain differences between Valhalla and the downstream study sites. In the current study, Valhalla was the only study site that harbored no invasive species. Since invasive species are known to disrupt native species abundance (Bradley et al. 2019, Gallardo et al. 2016), the absence of invasive species in Valhalla might explain why this study site exhibited higher BAP scores and PMA than all downstream sites. If the Asian Clam becomes established in Valhalla in the near future, it will be interesting to see if biodiversity indices decline correspondingly.

Anthropogenic variables. Finally, anthropogenic factors, including the downstream locations of combined sewage overflows and municipal separate stormwater systems, low human population density, and percent development, might explain differences in water quality in Valhalla compared to all downstream sites. Valhalla is located upstream of combined sewage overflow and municipal separate stormwater system sites (Fig. 2). Therefore, all study sites downstream of Valhalla are subject to discharges of organic, municipal, and industrial waste (Bode et al. 1998, 2003; Smith et al. 2015). These sources of pollution might explain why we found no differences in biodiversity indices for samples collected in 2020 when comparing upper, middle, and lower reaches of the Bronx River but instead, when comparing individual study sites, Valhalla exhibited significantly higher biodiversity indices than all downstream study sites.

Moreover, the low human population density of Valhalla might explain why this study site exhibited the highest biodiversity indices. Importantly, the establishment of invasive species is associated with high human population density (Castañeda 2012), which might account for the absence of invasive species in Valhalla. If these invasive species are spreading from the south, then it might be a matter of time before they disperse to Valhalla.

Valhalla also exhibited the lowest proportion of developed land cover compared to all 5 downstream sites, which is another reason why this study site might have exhibited the highest biodiversity indices. In support, several studies of urban streams have found a positive correlation between proportion of greenspace and macroinvertebrate diversity (Moore and Palmer 2005, Roy et al. 2003, Sponseller et al. 2001). While we found no differences in biodiversity indices of samples collected in 2020 when comparing dominant land-cover types, this result might reflect the unusual nature of the East 182nd Street study site (site 5), which had a high proportion of greenspace because of its location adjacent to the Bronx Zoo, but also the highest human population density of all 6 study sites. In contrast, the study site in Valhalla exhibited both low human population density and a high proportion of greenspace. Altogether, these findings raise the possibility that a combination of variables, an upstream location, low human population density, and percent greenspace, among other factors, might work synergistically to support macroinvertebrate diversity in an urban river.

Conclusions and future directions

A longitudinal assessment of the Bronx River over the past 22 years not only provides well-detailed information on the overall health of the Bronx River, but also indicates possible factors causing declines in water quality as measured by benthic macroinvertebrate diversity. However, the current study only surveyed 6 locations and therefore provides limited results in terms of an overall water quality assessment of the river. Moreover, the biodiversity indices used to measure water quality might have limited utility if other factors, such as climate change and geomorphological properties, substantially contribute to changes in benthic macroinvertebrate diversity. Despite these limitations, the results of the current study documented a recent invasion of the Asian Clam in 5 of 6 study locations, found relatively better biodiversity profiles at the northernmost study site, and indicated that despite restoration efforts, overall water quality of the Bronx River has remained moderately impacted. These results suggest that it is quite difficult to rectify damages to riverine ecosystems once they are inflicted with anthropogenic disturbances, and possibly the limited utility of small- to moderate-scale urban restoration projects.

These current results might be useful for state and city agencies, non-profit and conservation organizations, and other interested parties to further monitor and assess water quality and benthic macroinvertebrate diversity. Moreover, as previous studies have not documented the invasive Asian Clam, results of this current study might also be helpful for establishing invasive species management strategies along the Bronx River. Future assessment of the Bronx River should incorporate more locations along the river to evaluate the effects of abiotic, biotic, and anthropogenic factors on benthic macroinvertebrate diversity.

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Literature Cited

Albert, J.S., G. Destouni, S.M. Duke-Sylvester, A.E. Magurran, T. Oberdorff, R.E. Reis, and W.J. Ripple. 2020. Scientists' warning to humanity on the freshwater biodiversity crisis. Ambio 1–10.

Alexander, G.G., and J.D. Allan. 2007. Ecological success in stream restoration: Case studies from the midwestern United States. Environmental Management 40(2):245–255.

Alexander, R.B., E.W. Boyer, R.A. Smith, G.E. Schwarz, and R.B. Moore. 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resources Association 43(1):41–59.

Al-Shami, S.A., C.S.M. Rawi, A. HassanAhmad, and S.A.M. Nor. 2010. Distribution of Chironomidae (Insecta: Diptera) in polluted rivers of the Juru River Basin, Penang, Malaysia. Journal of Environmental Sciences 22(11):1718–1727.

Azrina, M.Z., C.K. Yap, A.R., Ismail, A. Ismail, and S.G. Tan. 2006. Anthropogenic impacts on the distribution and biodiversity of benthic macroinvertebrates and water quality of the Langat River, Peninsular Malaysia. Ecotoxicology and Environmental Safety 64(3):337–347.

Bae, Y.J., H.K. Kil, and K.S. Bae. 2005. Benthic macroinvertebrates for uses in stream biomonitoring and restoration. KSCE Journal of Civil Engineering 9(1):55–63.

Baladrón, A., and D.J. Yozzo. 2020. Macroinvertebrate assemblages, stormwater pollution, and habitat stressors in the Bronx River. Urban Naturalist 31:1–22.

Bauer, M.E., B.C. Loffelholz, and B. Wilson. 2007. Estimating and mapping impervious surface area by regression analysis of Landsat imagery. Pp. 3–18, *In* Q. Weng (Ed). Remote Sensing of Impervious Surfaces. CRC Press, Boca Raton, FL. 443 pp.

Beißler, M.R., and J. Hack. 2019. A combined field and remote-sensing based methodology to assess the ecosystem service potential of urban rivers in developing countries. Remote Sensing 11(14):1697.

Bernhardt, E.S., and M.A. Palmer. 2007. Restoring streams in an urbanizing world. Freshwater Biology 52(4):738–751.

Bernhardt, E.S., and M.A. Palmer. 2011. River restoration: The fuzzy logic of repairing reaches to reverse catchment-scale degradation. Ecological Applications 21(6):1926–1931.

Bernhardt, E.S., E.B. Sudduth, M.A. Palmer, J.D. Allan, J.L. Meyer, G. Alexander, and L. Pagano. 2007. Restoring rivers one reach at a time: Results from a survey of US river restoration practitioners. Restoration Ecology 15(3):482–493.

Bode, R.W. M.A., Novak, L.E. Abele, and D. Carlson. 1998. Bronx River Biological Assessment 1998 Survey [White Paper]. Department of Environmental Conservation. Albany, New York, NY. 36 pp.

Bode, R.W., M.A. Novak, L.E. Abele, D.L. Heitzman, and A.J. Smith. 2003. Bronx River Biological Assessment 2003 Survey [White Paper]. Department of Environmental Conservation. Albany, New York, NY. 18 pp.

Bond, N.R., and P.S. Lake. 2003. Local habitat restoration in streams: Constraints on the effectiveness of restoration for stream biota. Ecological Management and Restoration 4(3):193–198.

Bradley, B.A., B.B. Laginhas, R. Whitlock, J.M. Allen, A.E. Bates, G. Bernatchez, and C.J. Sorte. 2019. Disentangling the abundance–impact relationship for invasive species. Proceedings of the National Academy of Sciences 116(20):9919–9924.

Bradt, P.T. 2014. Changes in macroinvertebrate assemblages in a Pennsylvania trout stream over thirty-four years: Where have all the Trichoptera gone? Journal of the Pennsylvania Academy of Science 88(4):204–215.

Bradt, P.T., and G.S. Ruggiero. 2017. Biotic impoverishment and Trichoptera loss in a Pennsylvania trout stream: Benthic macroinvertebrate assemblages over 43 summers. Journal of the Pennsylvania Academy of Science 91(1):22–44.

Cao, X., L., Chai, D. Jiang, J. Wang, Y. Liu, and Y. Huang. 2018. Loss of biodiversity alters ecosystem function in freshwater streams: Potential evidence from benthic macroinvertebrates. Ecosphere 9(10):e02445.

Castañeda, R.A. 2012. Factors affecting the distribution, abundance, and condition of an invasive freshwater bivalve in a thermal plume. Ph.D. Dissertation. McGill University, Montreal, QC, Canada. 83 pp.

Cherry, D.S., J.L. Scheller, N.L. Cooper, and J.R. Bidwell. 2005. Potential effects of Asian clam (*Corbicula fluminea*) die-offs on native freshwater mussels (Unionidae) I: Water-column ammonia levels and ammonia toxicity. Journal of the North American Benthological Society 24(2):369–380.

Cooper, N.L., J.R. Bidwell, and D.S. Cherry. 2005. Potential effects of Asian Clam (*Corbicula fluminea*) die-offs on native freshwater mussels (Unionidae) II: Porewater ammonia. Journal of the North American Benthological Society 24(2):381–394.

Covich, A.P., M.A. Palmer, and T.A. Crowl. 1999. The role of benthic invertebrate species in freshwater ecosystems: Zoobenthic species influence energy flows and nutrient cycling. BioScience 49(2):119–127.

Cox, S., and P. Bower. 1998. Assessment of Bronx River Ecosystem: Pre-restoration baseline data. New York City Department of Parks and Recreation. New York, NY.18 pp.

de Kadt, M. 2011. The Bronx River: An Environmental and Social History. The History Press, Charleston, SC. 157 pp.

Darwall, W., V. Bremerich, A. De Wever, A.I. Dell, J. Freyhof, M.O. Gessner, and O. Weyl. 2018. The Alliance for Freshwater Life: A global call to unite efforts for freshwater biodiversity science and conservation. Aquatic Conservation: Marine and Freshwater Ecosystems 28(4):1015–1022.

Deborde, D.D.D., M.B.M. Hernandez, and F.S. Magbanua. 2016. Benthic macroinvertebrate community as an indicator of stream health: The effects of land use on stream benthic macroinvertebrates. Science Diliman 28(2):6–25.

DeMarte, R., J. Waldman, and M.S. Bednarski. 2016. The effects of dams on densities and sizes of American Eels in the Bronx River. Section VIII. Pp. 1–22, *In* S.H. Fernald, D.J. Yozzo, and H. Andreyko (Eds.). Final Reports of the Tibor T. Polgar Fellowship Program, 2014. Hudson River Foundation, New York, NY. 225 pp.

Dewitz, J. 2019. National Land Cover Database (NLCD) 2016 Products (ver. 2.0, July 2020): US Geological Survey data release. Available online at https://doi.org/10.5066/P96HHBIE.

du Plessis, A., T. Harmse, and F. Ahmed. 2015. Predicting water quality associated with land-cover change in the Grootdraai Dam catchment, South Africa. Water International 40(4):647–663.

Doretto, A., E. Piano, and C.E. Larson. 2020. The river continuum concept: Lessons from the past and perspectives for the future. Canadian Journal of Fisheries and Aquatic Sciences 77(11):1853–1864.

Ferreira-Rodríguez, N., R. Sousa, and I. Pardo. 2018. Negative effects of *Corbicula fluminea* over native freshwater mussels. Hydrobiologia 810(1):85–95.

Fierro, P., I. Arismendi, R.M. Hughes, C. Valdovinos, and A. Jara-Flores. 2018. A benthic macroinvertebrate multimetric index for Chilean Mediterranean streams. Ecological Indicators 91:13–23.

Francis, R.A., M.A. Chadwick, and A.J. Turbelin. 2019. An overview of non-native species invasions in urban river corridors. River Research and Applications 35(8):1269–1278.

Fu, L., Y. Jiang, J. Ding, Q. Liu, Q.Z. Peng, and M.Y. Kang. 2016. Impacts of land use and environmental factors on macroinvertebrate functional feeding groups in the Dongjiang River basin, southeast China. Journal of Freshwater Ecology 31(1):21–35.

Gallardo, B., M. Clavero, M.I. Sánchez, and M. Vilà. 2016. Global ecological impacts of invasive species in aquatic ecosystems. Global Change Biology 22(1):151–163.

Goldstein, A.J., D.C. Lahti, and B. Habig. 2022. Avian diversity and land use along the Bronx River. Urban Naturalist 50:1–22.

Gray, L. 2004. Changes in water quality and macroinvertebrate communities resulting from urban stormflows in the Provo River, Utah, USA. Hydrobiologia 518(1):33–46.

Haag, W.R., J. Culp, A.N. Drayer, M.A. McGregor, D.E. White, and S.J. Price. 2021. Abundance of an invasive bivalve, *Corbicula fluminea*, is negatively related to growth of freshwater mussels in the wild. Freshwater Biology 66(3):447–457.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. The Great Lakes Entomologist 20(1):7.

Hilsenhoff, W.L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7(1):65–68.

Houghton, D.C., J.J. Dimick, and R.V. Frie. 1998. Probable displacement of riffle-dwelling invertebrates by the introduced Rusty Crayfish, *Orconectes rusticus* (Decapoda: Cambaridae) in a north-central Wisconsin stream. The Great Lakes Entomologist 31(1):2.

Ilarri, M.I., and R. Sousa. 2012. *Corbicula fluminea* Müller (Asian Clam). A Handbook of Global Freshwater Invasive Species. Routledge, London, UK. 183 pp.

Kail, J., K. Brabec, M. Poppe, and K. Januschke. 2015. The effect of river restoration on fish, macroinvertebrates, and aquatic macrophytes: A meta-analysis. Ecological Indicators (58):311–321.

Kondolf, G.M., and P.J. Pinto. 2017. The social connectivity of urban rivers. Geomorphology (277):182–196.

Kuhlmann, M.L. 2016. Invasion-related change in crayfish density affects a stream macroinvertebrate community. Northeastern Naturalist 23(4):434.

Larson, M.G., D.B. Booth, and S.A. Morley. 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. Ecological Engineering 18(2):211–226.

Lerner, D.N., and A. Holt. 2012. How should we manage urban river corridors? Procedia Environmental Sciences 13:721–729.

Linke, S., R.C. Bailey and J. Schwindt. 1999. Temporal variability of stream bioassessments using benthic macroinvertebrates. Freshwater Biology 42(3):575–584.

McCarthy, J.M., C.L. Hein, J.D. Olden, and M. Jake Vander Zanden. 2006. Coupling long-term studies with meta-analysis to investigate impacts of non-native crayfish on zoobenthic communities. Freshwater Biology 51(2):224–235.

McMahon, R.F. 2002. Evolutionary and physiological adaptations of aquatic invasive animals: r selection versus resistance. Canadian Journal of Fisheries and Aquatic Sciences 59(7):1235–1244.

Medupin, C. 2020. Spatial and temporal variation of benthic macroinvertebrate communities along an urban river in Greater Manchester, UK. Environmental Monitoring and Assessment 192(2):1–20.

Miller, S.W., P. Budy, and J.C. Schmidt. 2010. Quantifying macroinvertebrate responses to in-stream habitat restoration: Applications of meta-analysis to river restoration. Restoration Ecology 18(1):8–19.

Miserendino, M.L., R. Casaux, M. Archangelsky, C.Y. Di Prinzio, C. Brand, and A.M. Kutschker. 2011. Assessing land-use effects on water quality, in-stream habitat, riparian ecosystems, and biodiversity in Patagonian northwest streams. Science of the Total Environment 409(3):612–624.

Miskewitz, R., and C. Uchrin. 2013. In-stream dissolved oxygen impacts and sediment oxygen demand resulting from combined sewer-overflow discharges. Journal of Environmental Engineering 139(10):1307–1313.

Modesto, V., P. Castro, M. Lopes-Lima, C. Antunes, M. Ilarri, and R. Sousa. 2019. Potential impacts of the invasive species *Corbicula fluminea* on the survival of glochidia. Science of the Total Environment 673:157–164.

Moore, A.A., and M.A. Palmer. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications 15(4):1169–1177.

Moreyra, A.K., and C. Padovesi-Fonseca. 2015. Environmental effects and urban impacts on aquatic macroinvertebrates in a stream of central Brazilian Cerrado. Sustainable Water Resources Management 1(2):125–136.

Muralidharan, M., C. Selvakumar, S. Sundar, and M. Raja. 2010. Macroinvertebrates as potential indicators of environmental quality. International Journal of Biological Technology 1:23–28.

Natural Resources Group. 2008. Urban riparian wetland restoration evaluation: A case study of the Bronx River [White Paper]. New York City Department of Parks and Recreation. New York City, NY, . 52 pp.

New York State Department of Environmental Conservation Division of Water (NYSDEC-DOW). 2019. Standard operating procedure: Biological monitoring of surface waters in New York State. Department of Environmental Conservation, Troy, NY. 188 pp.

Nilsson, E., C.T. Solomon, K.A. Wilson, T.V. Willis, B. Larget, and M.J. Vander Zanden. 2012. Effects of an invasive crayfish on trophic relationships in north-temperate lake food webs. Freshwater Biology 57(1):10–23.

Odume, O.N., and W.J. Muller. 2011. Diversity and structure of Chironomidae communities in relation to water quality differences in the Swartkops River. Physics and Chemistry of the Earth, Parts A/B/C 36(14–15):929–938.

Ogbeibu, A.E., and B.J. Oribhabor. 2002. Ecological impact of river impoundment using benthic macro-invertebrates as indicators. Water Research 36(10):2427–2436.

Olden, J.D., J.M. McCarthy, J.T. Maxted, W.W. Fetzer and M. J. Vander Zanden. 2006. The rapid spread of Rusty Crayfish (*Orconectes rusticus*) with observations on native crayfish declines in Wisconsin (USA) over the past 130 years. Biological Invasions 8(8):1621–1628.

Olson, A.R., T.W. Stewart, and J.R. Thompson. 2016. Direct and indirect effects of human population density and land use on physical features and invertebrates of Iowa (USA) streams. Urban Ecosystems 19(1):159–180.

O'Neil, J.M., D. Taillie, B. Walsh, W.C. Dennison, E.K. Bone, D.J. Reid, R. Newton, D.L. Strayer, et al. 2016. New York Harbor: Resilience in the face of four centuries of development. Regional Studies in Marine Science 8:274–286.

Palmer, M.A., H.L. Menninger, and E. Bernhardt. 2010. River restoration, habitat heterogeneity, and biodiversity: A failure of theory or practice? Freshwater Biology 55:205–222.

Paul, M.J., and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32(1):333–365.

Pennak, R.W. 1978. Fresh-water invertebrates of the United States. 2nd Edition. Wiley, Hoboken, NJ. 803 pp.

Pinheiro, J., D. Bates, S. DebRoy, D. Sarkar, S. Heisterkamp, B. Van Willigen, and R. Maintainer. 2017. Package 'nlme'. Linear and nonlinear mixed effects models version 3(1). Current verstion available online at https://cran.r-project.org/web/packages/nlme/index.html.

Poole, G.C. 2002. Fluvial landscape ecology: Addressing uniqueness within the river discontinuum. Freshwater Biology 47(4):641–660.

Roy, A.H., A.D. Rosemond, M.J. Paul, D.S. Leigh, and J.B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). Freshwater Biology 48(2):329–346.

Schertzinger, G., F. Itzel, J. Kerstein, J. Tuerk, T.C. Schmidt, and B. Sures. 2019. Accumulation pattern and possible adverse effects of organic pollutants in sediments downstream of combined sewer overflows. Science of the Total Environment 675:295–304.

Sedell, J.R., R.J. Naiman, K.W. Cummins, G.W. Minshall, and R.L. Vannote. 1978. Transport of particulate organic material in streams as a function of physical processes. Verhandlungen Der Internationalen Vereinigung Für Theoretische Und Angewandte Limnologie 20(2):1366–1375.

Shieh, S.H., B.C. Kondratieff, and J.V. Ward. 1999. Longitudinal changes in benthic organic matter and macroinvertebrates in a polluted Colorado plains stream. Hydrobiologia 411:191–209.

Shuster, W.D., J. Bonta, H. Thurston, E. Warnemuende, and D.R. Smith. 2005. Impacts of impervious surface on watershed hydrology: A review. Urban Water Journal 2(4):263–275.

Smith, A.J., S. Rickard, E.A. Mosher, J.L. Lojpersberger, D.L. Heitzman, B.T. Duffy, and M.A. Novak. 2015. Bronx River biological assessment 2015 survey [White Paper]. Department of Environmental Conservation. Troy, NY. 34 pp.

Smith, K.R., B.M. Roth, M.L. Jones, D.B. Hayes, S.J. Herbst, and N. Popoff. 2019. Changes in the distribution of Michigan crayfishes and the influence of invasive Rusty Crayfish (*Faxonius rusticus*) on native crayfish substrate associations. Biological Invasions 21(2):637–656.

Sousa, R., C. Antunes, and L. Guilhermino. 2008. Ecology of the invasive Asian Clam, *Corbicula fluminea* (Müller, 1774) in aquatic ecosystems: An overview. Annales de Limnologie-International Journal of Limnology 44(2):85–94.

Sponseller, R.A., E.F. Benfield, and H.M. Valett. 2001. Relationships between land use, spatial scale, and stream macroinvertebrate communities. Freshwater Biology 46(10):1409–1424.

Stepenuck, K.F., R.L. Crunkilton, and L. Wang. 2002. Impacts of urban land use on macroinvertebrate communities in southeastern Wisconsin streams. Journal of the American Water Resources Association 38(4):1041–1051.

Sterling, J.L., A.D. Rosemond, and S.J. Wenger. 2016. Watershed urbanization affects macroinvertebrate community structure and reduces biomass through similar pathways in Piedmont streams, Georgia, USA. Freshwater Science 35(2):676–688.

Strayer, D.L. 1999. Effects of alien species on freshwater mollusks in North America. Journal of the North American Benthological Society 18(1):74–98.

Sun, W., C. Xia, M. Xu, J. Guo, and G. Sun. 2016. Application of modified water-quality indices as indicators to assess the spatial and temporal trends of water quality in the Dongjiang River. Ecological Indicators 66:306–312.

Sundermann, A., S. Stoll, and P. Haase. 2011. River restoration success depends on the species pool of the immediate surroundings. Ecological Applications 21(6):1962–1971.

Svensson, O., A.S. Bellamy, P.J. Van den Brink, M. Tedengren, and J.S. Gunnarsson. 2018. Assessing the ecological impact of banana farms on water quality using aquatic macroinvertebrate community composition. Environmental Science and Pollution Research 25(14):13373–13381.

United States Army Corps of Engineers (USACE). 2006. Final data and documentation report, Bronx River Ecosystem Restoration Project, water quality and biological baseline data collection, Westchester and Bronx Counties, New York. Planning Division, New York District, New York, NY. 101 pp.

Utz, R.M., R.H. Hilderbrand, and D.M. Boward. 2009. Identifying regional differences in threshold responses of aquatic invertebrates to land cover gradients. Ecological Indicators 9(3):556–567.

Vaughn, C.C., and C.C. Hakenkamp. 2001. The functional role of burrowing bivalves in freshwater ecosystems. Freshwater Biology 46(11):1431–1446.

Violin, C.R., P. Cada, E.B. Sudduth, B.A. Hassett, D.L. Penrose, and E.S. Bernhardt. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications 21(6):1932–1949.

Voshell, J.R. 2002. A Guide to Common Freshwater Invertebrates of North America. McDonald and Woodward Publishing Company, Newark, OH. 442 pp.

Wallace, J.B., and J.R. Webster. 1996. The role of macroinvertebrates in stream ecosystem function. Annual Review of Entomology 41(1):115–139.

Walsh, C.J., T.D. Fletcher, and A.R. Ladson. 2005a. Stream restoration in urban catchments through redesigning stormwater systems: Looking to the catchment to save the stream. Journal of the North American Benthological Society 24(3):690–705.

Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan 2005b. The urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society 24(3):706–723.

Wilson, H.L., M.F. Johnson, P.J. Wood, C.R. Thorne, and M.P. Eichhorn. 2021. Anthropogenic litter is a novel habitat for aquatic macroinvertebrates in urban rivers. Freshwater Biology 66(3):524–534.

Wilson, K.A., J.J. Magnuson, D.M. Lodge, A.M. Hill, T.K. Kratz, W.L. Perry, and T.V. Willis. 2004. A long-term Rusty Crayfish (*Orconectes rusticus*) invasion: Dispersal patterns and community change in a north temperate lake. Canadian Journal of Fisheries and Aquatic Sciences 61(11):2255–2266.

Wright, I.A., and S. Burgin. 2009. Effects of organic and heavy metal pollution on chironomids within a pristine upland catchment. Hydrobiologia 635(1):15–25.

Xie, S., X. Wang, Y. Ren, Z. Su, Y. Su, S. Wang, and Z. Ouyang. 2020. Factors responsible for forest and water-bird distributions in rivers and lakes along an urban gradient in Beijing. Science of the Total Environment 735:139308.

Xu, M., Z. Wang, X. Duan, and B. Pan. 2014. Effects of pollution on macroinvertebrates and water quality bio-assessment. Hydrobiologia 729(1):247–259.

Yeager, M.M., R.J. Neves, and D.S. Cherry. 1999. Competitive interactions between early life stages of *Villosa iris* (Bivalvia: Unionidae) and adult Asian Clams (*Corbicula fluminea*). Pp. 253–259, *In* R.A. Tankersley, D.I. Warmolts, G.T. Watters, B.J. Armitage, P.D. Johnson, and R.S. Butler (Eds.). Freshwater Mollusk Symposium Proceedings—Part II: Proceedings of the First Freshwater Mollusk Conservation Society Symposium. Ohio Biological Survey, Columbus, OH. 274 pp. Available online at https://www.mussellab.fishwild.vt.edu/mussel/PDFfiles/Comp_interaction.pdf.

Zanatta, N., L.H.R. Pazianoto, V. de Mello Cionek, P. A. Sacramento, and E. Benedito. 2017. Population structure of fishes from an urban stream. Acta Scientiarum. Biological Sciences 39(1):27.