A LONGITUDINAL ASSESSMENT OF BENTHIC MACROINVERTEBRATE DIVERSITY AND WATER QUALITY ALONG THE BRONX RIVER

by

Maleha Mahmud

A thesis submitted in partial fulfillment of the requirements for the degree of

Master of Arts in Biology

Queens College

The City University of New York

August 2021

Approved by________David Lahti________________
Committee Chair

David Lahti
____
Signature

Date:___30 August 2021___

Approved by_________Bobby Habig______________
Research Supervisor

Bobby Habig
____
Signature

Date:___30 August 2021___

Table of Contents

Acknowledgements

I would like to thank Dr. David Lahti for inspiring me to see the beauty of biology. I never knew how fascinating the study of animals could be until I took Dr. Lahti's course in Animal Behavior during my last semester as an undergraduate. Dr. Lahti once said, "Every single living being in the universe is an interesting and unique phenomenon." His inspirational thoughts have encouraged me to appreciate all living organisms found in nature. There are countless reasons for being thankful to him, and I would especially like to express my gratitude to Dr. Lahti for always being supportive, allowing me to join the Lahti Lab, and encouraging me to see myself in the field of ecology and behavior.

There would not be a Bronx River Project if it was not for the guidance of Dr. Bobby Habig. I would need few pages to acknowledge the tremendous amount of help Dr. Habig has provided for this project! First of all, I want to thank him for allowing me to work with him on my master's project and for helping me to come up with the idea of studying benthic macroinvertebrates along the Bronx River. I would also like to thank Dr. Habig for helping me in every aspect of this project—from benthic macroinvertebrate sampling, identification, conducting analyses to a completed master's thesis. And of course, thank you for always being there whenever I needed help.

I would like to convey my appreciation to Dr. John Waldman, and Dr. Mitchell Baker for serving as members of my thesis committee, and providing advice and feedback. Dr. Waldman and Dr. Bakers' backgrounds and experiences were valuable assets to this study.

I would like to thank Cesar Castillo for providing sampling and sample sorting materials, and Dr. Jose Anadon for providing benthic macroinvertebrate identification materials.

I would like to acknowledge Amanda Goldstein for creating a beautiful map of the Bronx River, and for also calculating percent land cover data for this project. Thank you, Amanda!

A big thank you to the Bronx River team—Amanda Goldstein, Ritika Nath, Oditi Debi, Salvatore Asaro, and Catalina Tapia Boza—for helping me during field work, and being very supportive over the course of this study.

Thank you to Danielle Wasserman for helping me with benthic macroinvertebrate identification.

A heartfelt thanks to my Lahti Lab family and friends for always being so welcoming and supportive. Thank you everyone for helping and encouraging me to accomplish my goals.

I would like to thank Still-Legal Climate Grants for providing funds for this project.

I would also like to acknowledge the following individuals for providing permits and access to study sites: Brady Simmons, New York City Department of Parks and Recreation; Douglas Daniels, Kathleen O'Connor (Commissioner), and Jason Klein (Director of Conservation) of Westchester Parks Department; Paul Stringer and Greg Kozlowski, New York State Department of Environmental Conservation.

Last, but not the least, I would like to thank my Mom, and my best friends—Ritika, and Oditi—for always being there for me, and encouraging me to achieve my dreams!

Abstract

Urban freshwater rivers play a critical role in sustaining biodiversity. The Bronx River, New York City's only freshwater river, is an urban waterway that has a long history of anthropogenic disturbance. In recent years, several restoration efforts have been undertaken to improve the overall water quality of the Bronx River. To assess the efficacy of these restoration efforts, long term monitoring of the Bronx River is essential for diagnosing whether biodiversity has increased, remained stable, or decreased over time. The aim of this study was to conduct a longitudinal assessment of water quality of the Bronx River. To accomplish this aim, I conducted a study of benthic macroinvertebrate diversity at six sites along the river. I then integrated this with historical data collected by the New York State Department of Environmental Conservation over the last 22 years. I used these data to address three 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 does benthic macroinvertebrate diversity vary among study sites over the past 22 years? On a spatial scale, study sites with high invasive species dominance exhibited less healthy benthic macroinvertebrate communities than study sites with low invasive species dominance. Moreover, the study site upstream of combined sewage overflows and municipal separate stormwater systems exhibited healthier biological profiles than downstream sites. On a temporal scale, overall water quality along the Bronx River remained moderately impacted over a 22-year-period despite restoration efforts. The three most downstream sites even exhibited slight declines in water quality over time. Finally, I found temporal changes in

benthic macroinvertebrate Family dominance and fluctuations in the proportion of functional feeding groups over this 22-year period. Overall, these results demonstrate that there are both spatial and temporal differences in water quality and benthic macroinvertebrate diversity in the Bronx River. These data can be used to guide conservation and management efforts.

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., fishes: Zanatta et al. 2017; birds: Xie et al. 2020; benthic macroinvertebrates: Wilson et al. 2021). In addition to providing habitats for a wide range of nonhuman animals, urban rivers also provide different resources for humans including food, water, transportation, and recreation (Lerner and Holt 2012; Kondolf and Pinto 2017). However, over the past century, urban rivers have undergone extensive degradation and overexploitation, which has been largely attributed to increased urbanization (Bernhardt and Palmer 2007; O'Neil et al. 2016; Beißler and Hack 2019). Some factors associated with urbanization that have contributed to the decline of urban rivers include increased impervious surface cover (Shuster et al. 2005; Bauer et al. 2007), 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 (Fierro et al. 2018; Darwall et al. 2018).

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 (Wallace and Webster 1996; Covich et al. 1999; Paul and Meyer 2001; Cao et al. 2018). 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 (Bode et al. 1998, Linke et al. 1999; Bode et al. 2003; Bae et al. 2005; Muralidharan et al. 2010; Smith et al. 2015; Deborde et al. 2016).

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. First, as rivers flow from an upstream to downstream gradient, 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 from an upstream to downstream gradient (e.g., Miskewitz and Uchrin 2013; Sun et al. 2016; Svensson et al. 2018). Moreover, shifts in benthic macroinvertebrate community composition have been observed as water flows from the upper to lower reaches of urban rivers (e.g., Gray 2004; Azrina et al. 2006). Specifically, organisms that are less tolerant of pollution are more likely to be found in upstream locations whereas organisms more tolerant of pollution are more likely to be found in downstream locations (Azrina et al. 2006; Matlou et al. 2017).

Second, the dominant land cover type is another factor predictive of benthic macroinvertebrate diversity and water quality (Sponseller et al. 2001; du Plessis et al. 2015). In contrast to greenspaces, highly developed areas are comprised of high proportions of impervious surface cover, which results in predictable changes in stream ecology (Paul and Meyer 2001; Bauer et al. 2007). Specifically, areas with higher proportions of impervious surface cover have higher rates of runoff, which results in increased nutrient loading and higher rates of chemical discharge (Paul and Meyer 2001; Bauer et al. 2007). Several studies have shown that riparian habitats surrounded by higher proportions of developed space exhibit degraded benthic macroinvertebrate diversity and lower water quality compared to habitats surrounded by more greenspace (Roy et al. 2003; Miserendino et al. 2011; de Mello et al. 2018). In addition, increased impervious surface cover in developed areas is associated with major declines in benthic macroinvertebrate diversity (Paul and Meyer 2001; Utz et al. 2009).

Finally, urban rivers are often besieged by invasive species (Francis et al. 2019). Two common invasive species observed in freshwater streams of North America are the Asian clam (*Corbicula fluminea*) (Sousa et al 2008; Ilarri and Sousa 2012) and the rusty crayfish (*Faxonius rusticus*) (Wilson et al. 2004). Previous studies have shown that these invasive species can have adverse effects on other benthic macroinvertebrate taxa (McCarthy et al. 2006; Nilsson et al. 2012; Ferreira-Rodríguez et al. 2018; Smith et al. 2019; Modesto et al. 2019; Haag et al. 2021). For example, Asian clams have been found to negatively impact native bivalves in several ways: reducing the survival of native mussels' larva (Modesto et al. 2019), competing with native bivalves for resources (Ferreira-Rodríguez et al. 2018), and disrupting living spaces of native bivalves such as Sphaeriids and developing unionids (Strayer 1999). Moreover, the rusty crayfish has been found to outcompete native crayfish (Smith et al. 2019) and is associated with overall reductions in benthic macroinvertebrate abundance (McCarthy et al. 2006; Nilsson et al. 2012). Understanding the effects that these three 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.

The Bronx River runs through one of the largest metropolitan areas in the world. As New York City's only freshwater river, the Bronx River was once a source of drinking water for Native Americans and early 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-1800's, and industrial development throughout the 19^{th} century (de Kadt 2011). As the river continued to be shaped by urbanization, it was declared an "open sewer" by the Bronx River Valley Sewer Commission in the late 19^{th} century (de Kadt 2011). During the 20^{th} 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). Although the river has undergone a long history of extreme degradation, it continues to provide a significant habitat for many animals, including benthic macroinvertebrates (Bode et al. 1998, 2003; Natural Resources Group 2008; Smith et al. 2015; Baladrón and Yozzo 2020), fishes (Samaritan and Schmidt 1982; Rachlin et al. 2007), plants (Frankel 1999; Natural Resources Group 2008; Atha et al. 2016), birds (Goldstein 2021), and turtles (Aplasca et al. 2019). In recent years, several organizations, including the Natural Resources Group and the Bronx River Alliance, have facilitated multiple initiatives to reclaim and restore the water quality and biodiversity of the Bronx River (Natural Resources Group 2008; de Kadt 2011). Despite these efforts, biomonitoring surveys along the Bronx River indicate that these initiatives have yielded limited results (Bode et al. 1998, 2003; Smith et al. 2015).

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 first biological assessment of the Bronx River by NYSDEC-SBU was conducted in 1998 (Bode et al. 1998), followed by two subsequent surveys, one in 2003 (Bode et al. 2003) and another in 2015 (Smith et al. 2015). The locations that were assessed during these surveys were comprised of both suburban and urban areas. The upstream survey sites, located in Westchester County, are largely comprised of suburban habitats with more greenspace and less population density than the downstream survey sites situated in Bronx County (Bode et al. 1998, 2003; Smith et al. 2015). In these three 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 two subsequent studies (Bode et al. 2003; Smith et al. 2015) found similar water quality impact as the initial study. The results of these three 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 three 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 does benthic macroinvertebrate diversity vary among study sites over the past 22 years?

Based on my first research question, I 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, I predicted that biodiversity indices will continue to remain similar to past values. Alternatively, since several restoration efforts have attempted to improve the quality 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, I sampled benthic macroinvertebrates at six study sites along the Bronx River. Additionally, I 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

Study Area

This study was conducted at six riffle habitat locations along the Bronx River (Table 1; Figs. 1-2). The Bronx River is New York City's only freshwater river: the river extends approximately 36 kilometers from its source in Westchester County to its mouth, a tidal strait connected to the Long Island Sound (Natural Resources Group 2008; de Kadt 2011). The NYSDEC-SBU have conducted several surveys along the Bronx River over the past few decades to evaluate water quality and biological conditions of the river (Bode et al. 1998, 2003; Smith et al. 2015). Assessments conducted on the Bronx River during 1998 and 2003 include surveys at four 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 one of the four sites (Tuckahoe) and added two additional sites (Mount Vernon and East 182nd St.) for a total of five study locations. For the purposes of this study, I have surveyed all six of these locations and rearranged study sites from 1 to 6 (from north to south) incorporating all sites previously surveyed by NYSDEC-SBU.

Study sites	Location	Latitude; Longitude	Human population density (km^2)	References
Site 1	Valhalla, NY 10 m upstream of the Legion Rd. culvert	41.074170; -73.776390	3,553	Bode et al. 1998, 2003; Smith et al. 2015
Site 2	White Plains, NY 100 m downstream of the Bronx River Pkwy bridge	41.024170; -73.783060	28,288	Bode et al. 1998, 2003; Smith et al. 2015
Site 3	Tuckahoe, NY Upstream of Crestview Station	40.960833; -73.820833	20,137	Bode et al. 1998, 2003
Site 4	Mount Vernon, NY Upstream of Sherwood Ave.	40.915000; -73.849000	48,422	Smith et al. 2015
Site 5	Bronx, NY 150 m upstream of East Gun Hill Rd.	40.880000; -73.868610	107,870	Bode et al. 1998, 2003; Smith et al. 2015
Site 6	Bronx, NY Downstream of 182nd St. Dam	40.843223; -73.876689	112,838	Smith et al. 2015

Table 1: Locations, latitude, longitude, and population density of six study sites surveyed for macroinvertebrate diversity along the Bronx River. Population density is based on the zip code in which the study site is located (unitedstateszipcodes.org). References indicate previous surveys conducted along these sites.

Site 1

Site 2

Site 3

Site 4

Site 5

Site 6

Figure 1. Six study sites surveyed for longitudinal assessment of benthic macroinvertebrate diversity: (Site 1) Valhalla, NY; (Site 2) White Plains, NY; (Site 3) Tuckahoe, NY; (Site 4) Mount Vernon, NY; (Site 5) East Gun Hill Road Bronx, NY; (Site 6) East 182nd St. Bronx, NY. Photographs by Bobby Habig.

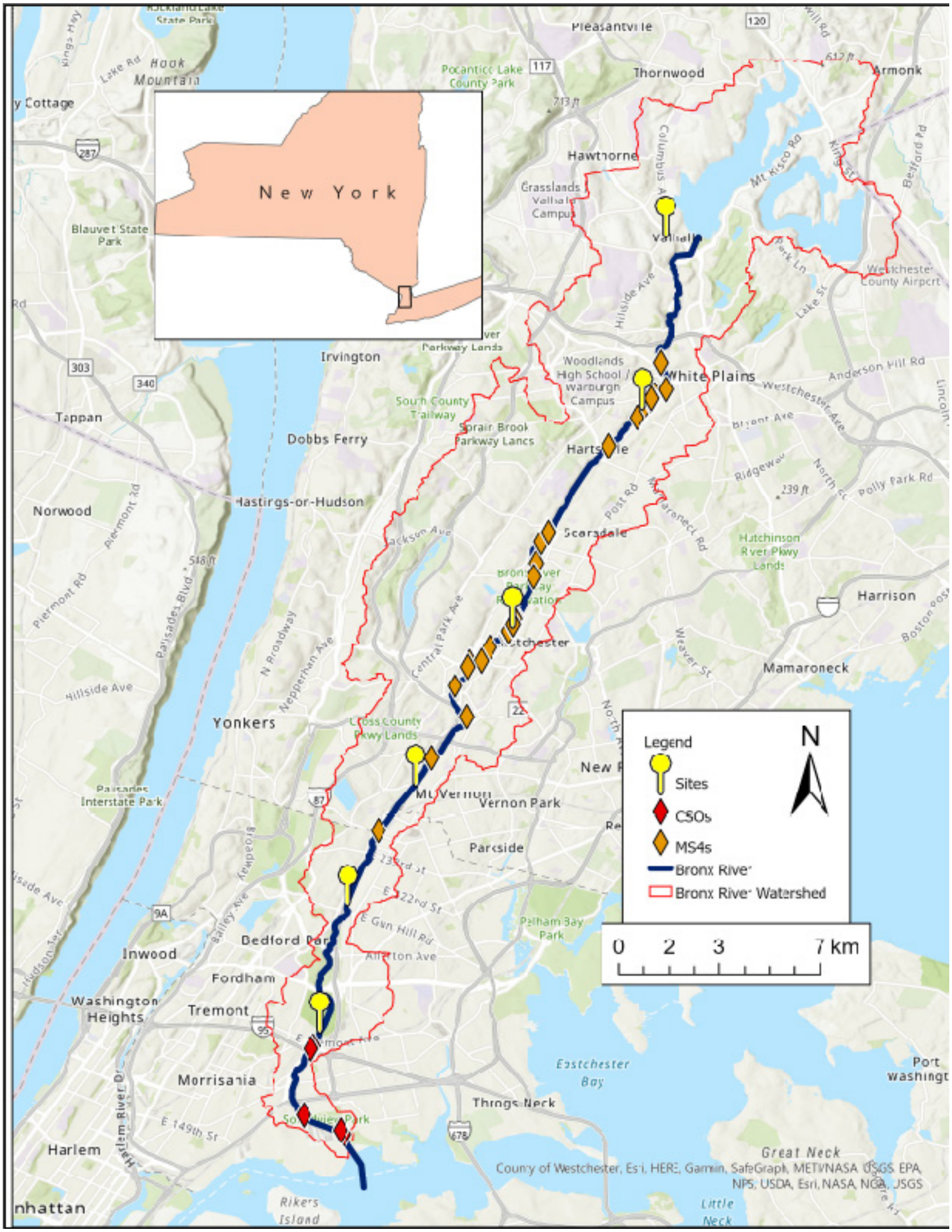


Figure 2. Map of the six study sites (yellow pins) surveyed for longitudinal assessment of benthic macroinvertebrate diversity. Red diamonds represent locations of combined sewage overflows and orange diamonds indicate locations with municipal separate stormwater systems. Map created by Amanda Goldstein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

Macroinvertebrate sampling

Benthic macroinvertebrates were collected on September 12, 2020 to correspond with sampling dates of the three previous studies (Bode et al. 1998: September 23, 1998; Bode et al. 2003: September 17, 2003; Smith et al. 2015: September 12, 2015). To sample macroinvertebrates, I used the standardized kick sampling method as described in Bode et al. (1998), Bode et al. (2003), and Smith et al. (2015). Specifically, I positioned a kick net in the river facing the flow of water, and with the support of my feet, I dislodged rocks such that organisms along with dislodged sediments were carried into the net by flow of the water. I continued sampling for five minutes over the distance of five meters. The samples that I collected were placed into an enamel pan and benthic macroinvertebrates were extracted using tweezers and then placed into labeled jars. I preserved the benthic macroinvertebrate samples in 95% ethanol with a concentration of two-thirds ethanol and one-third river water.

Sample identification

In alignment with the NYSDEC-SBU's guidelines for benthic macroinvertebrate sample sorting, sub-sampling, and identification in laboratory (Bode et al. 1998, 2003; Smith et al. 2015), I used a U.S No. 40 standard sieve to clean any residue while rinsing samples with tap water. I placed the rinsed samples on a gridded enamel pan such that the rinsed samples were evenly placed across the bottom of the pan. I used a random number generator to select samples from each 6.5 cm x 6.5 cm numbered square grid. I placed the randomly selected samples in a Petri dish and used a dissecting stereomicroscope to sub-sample 100 organisms. I sorted and counted the sub-sampled organisms and placed them in vials containing 70% alcohol. All preserved organisms were identified to the Family level using two identification keys (Pennak 1978; Voshell 2002).

Water chemistry and physical variables

At each study site, I measured water temperature, river width, river depth, and pH. To measure water temperature, I placed the probe of a digital thermometer (REO TEMP TM99A) into the river for one minute. I recorded the digital reading, which yielded an output in degrees Celsius measured to the nearest tenth. I measured the width of the river to the nearest centimeter using a closed reel tape. To do so, I held one end of the tape from one side of the riverbank while a research collaborator crossed the river and held the tape on the other side. To estimate river depth, I used a Secchi disk and a measuring tape. Specifically, I waded to the center of the river, and I allowed the Secchi disk to reach the bottom of the river while holding the string attached to the disk. Then, I measured the length of the string to the nearest tenth of a centimeter from the point where I held it at the river surface to the disk at the end of the string (river bottom). Finally, I measured pH to the nearest hundredth using a pH meter (YSI PRO 10 pH/ORP/temperature portable meter).

Land cover type

Percent land cover was calculated by a research collaborator (Amanda Goldstein) using ArcGIS Pro 2.6 (Esri Inc. 2020) and the National Land Cover Database (NLCD 2016). Circular buffers were created with 100 m radii surrounding the center of each of the six study sites. Percent land cover was simplified into land cover types by combining the percent land cover of similar groups into three variables: (1) developed; (2) open space;

and (3) greenspace (Goldstein 2021). 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 (NLCD 2016).

Biodiversity Indices

To measure benthic macroinvertebrate diversity, I used seven Family-level biodiversity indices recommended by the New York State Department of Environmental Conservation-Division of Water (NYSDEC-DOW 2019) (Table 2): (1) Family richness; (2) Ephemeroptera-Plecoptera-Trichoptera (EPT) Family richness; (3) Hilsenhoff's Family Biotic Index (FBI); (4) Percent Model Affinity (PMA); (5) Biological Assessment Profile (BAP); (6) dominant Family; and (7) functional feeding group (FFG). I compared these indices with three previous surveys conducted along the Bronx River (Bode et al. 1998, 2003; Smith et al. 2015) and converted the data from these historical surveys into the seven Family-level biodiversity indices. Each biodiversity index is defined in Table 2.

Biological Assessment Profile (BAP)

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

Biodiversity Indices	Description	References
Family richness [A]	The number of distinct benthic macroinvertebrate Families based on a sub-sample of 100 randomly selected organisms.	Xu et al. 2014
EPT Family richness [A]	A water quality index based on 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
Hilsenhoff's Family Biotic Index (FBI) [A]	An index measuring a benthic macroinvertebrate community's pollution tolerance level. The FBI score is calculated by multiplying the number of families in a sub-sample of 100 randomly selected organisms by the "undetermined" Family tolerance value (NYSDEC-DOW 2019). Calculated tolerance value scores for each Family are summed and divided by the total number of Families from the sample.	Hilsenhoff 1988; NYSDEC-DOW 2019
Percent Model Affinity (PMA)	An index comparing the sample community to a model non-impacted community based on the abundance percentage of major benthic macroinvertebrate taxa. Higher percentage of similarity indicates a healthier community. Percentage of similarity can be estimated by comparing the sample community and the non-impacted community with 40% Ephemeroptera, 5% Plecoptera, 10% Trichoptera, 10% Coleoptera, 20% Chironomidae, 5% Oligochaeta and 10% other.	Bode et al. 1998, 2003; Natural Resources Group 2008; Smith et al. 2015; NYSDEC-DOW 2019
Biological Assessment Profile (BAP)	An index of overall water quality based on conversion formulas that transforms each biodiversity index onto a common scale (NYSDEC-DOW 2019). This measure is based on the means of four standardized biodiversity indices: Family richness; EPT Family richness; FBI; and PMA. The scale is an indicator of water quality impact, 0 being severely impacted and 10 being non-impacted water quality (see Table 3).	NYSDEC-DOW 2019
Dominant Family [A]	The percentage of the most numerous Family based on a sub-sample of 100 randomly selected organisms.	NYSDEC-DOW 2019
Functional feeding group (FFG) [B]	The percentage of each feeding group within a sub-sample of 100 randomly selected organisms. Each Family is assigned to a functional feeding group based on classification by NYSDEC-DOW (2019): • Shredders: organisms that feed by chewing leaf litter, plant tissues, or wood • Scrapers: organisms that feed by grazing on rock, wood, and stems of aquatic plants • Collector-gatherers: organisms that feed by collecting particles deposited on the bottom of river • Predators: organisms that feed by capturing and engulfing other live organisms • Collector-filterers: organisms that feed by filtering particles from water bodies	Cummins et al. 2005; NYSDEC-DOW 2019

Note: [A] In cases in which there were <100 organisms sampled (Site 5 and Site 6: Smith et al. 2015), a random number generator was used to simulate a community composition of 100 organisms; [B] Families that consist of more than one FFG were assigned based on the most dominant feeding group within the Family.

Table 2: Descriptions of benthic macroinvertebrate community parameters.

Biological Assessment Profile Score (BAP)	Water Quality Impact Scale	Reference
7.5-10	Non	NYSDEC-DOW 2019
5-7.5	Slight	NYSDEC-DOW 2019
2.5-5	Moderate	NYSDEC-DOW 2019
0-2.5	Severe	NYSDEC-DOW 2019

Table 3. Biological Assessment Profile (BAP) Score and Water Quality Impact Scale established by NYSDEC-DOW.

Analyses

I conducted three sets of analyses aligned with my three research questions. All analyses were conducted using R version 4.03 (R Core Team 2020). First, to compare differences across the six study sites (Figs. 1-2) based on samples collected in 2020 (Research Question 1), I conducted Kruskal-Wallis tests using the *stats* package. I compared biodiversity indices (Table 2) based on geographical location, land cover, and proportion of invasive species. To classify study sites based on geographical location, I divided the Bronx River into three reaches: upper (Site 1; Site 2), middle (Site 3; Site 4), and lower (Site 5; Site 6) (Fig. 2). I also compared the dominant land cover type for each of the six study sites. For example, if there was more greenspace than open or developed space at a given site, the dominant land cover type was classified as “greenspace”. Finally, I classified a study site as “high” invasive species dominance if more than 50 percent of a sub-sample of 100 randomly selected organisms were invasive, and “low” invasive dominance if less than 50 percent of a sub-sample of 100 randomly selected organisms were invasive.

Second, to compare metrics of biodiversity longitudinally (Research Question 2), I conducted mixed effects repeated measure ANOVAs using the *nlme* package (Pinheiro et al. 2017). For each model, one of the seven indices of biodiversity (Table 2) was included as a response variable; year of study was included as a predictor variable. I also ran separate models that included each water chemistry (pH) and physical (temperature, river width, and river depth) parameter as a response variable. For each analysis, I ran a random intercept and random slope model accounting for site-level variability. The longitudinal analyses, based on comparisons between years, only included the three study sites (Site 1; Site 2; Site 5) in which there were data available for all four time periods (1998; 2003; 2015; 2020). Following these analyses, I used the *multcomp* package to compare differences between years using a Tukey post hoc test.

Finally, to assess whether metrics of biodiversity varied across the six study sites incorporating data from all surveys (Research Question 3), I conducted additional mixed effects repeated measure ANOVAs also using the *nlme* package (Pinheiro et al. 2017). As before, for each model, one of the seven indices of biodiversity (Table 2) was included as a response variable. For these analyses, I included data from all six study sites (Site 1; Site 2; Site 3; Site 4; Site 5; Site 6) and across all four studies (Bode et al. 1998, 2003; Smith et al. 2015; current study). For each model, study site was included as a predictor variable and year was modeled as a random effect. I used the *multcomp* package to compare differences between locations using Tukey post hoc tests.

Results

Biodiversity indices based on samples collected in 2020

Biodiversity indices, based on samples collected in 2020, varied across the six study sites (Table 4). Family richness ranged from 6 to 10 (mean = 9; SD = 1.55); sites with the highest Family richness (n = 10) were located at Site 1, Site 3, and Site 5 while the site with the fewest Families (*n* = 6) was located at Site 4. EPT Family richness did not vary across the six study sites (mean = 1; SD = 0). FBI (pollution tolerance) ranged 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 ranged from 15 to 54 percent (mean = 24.5; SD= 14.73); the site with the highest percent affinity (54%) to an undisturbed reference stream was located at Site 1; the location with lowest percent affinity (15%) was found at Site 4. Family dominance ranged from 25 to 68 percent (mean = 53.5; SD=16.13). Corbiculidae was the most dominant Family at three of the study sites (Site 2; Site 5; Site 6). Gammaridae was the most dominant Family at two of the study sites (Site 3; Site 4) and Hydropsychidae was the most dominant Family at one of the study sites (Site 1). Notably, the Family Corbiculidae was comprised entirely of the Asian clam (*C. fluminea*), an invasive species not previously documented in the three previous studies of the Bronx River (Bode et al. 1998, 2003; Smith et al. 2015). Finally, the two most common functional feeding groups sampled were collector-gatherers (mean = 47.17; SD = 20.98) and collector-filterers (mean = 42.67; SD = 21.91).

Biodiversity Indices	Mean	Min	Max	SD
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
Percent shredders	1.33	0.00	7.00	2.80
Percent scrapers	4.67	0.00	11.00	5.24
Percent collector-gatherer	47.17	28.00	78.00	20.98
Percent predators	4.17	0.00	8.00	2.99
Percent collector-filterer	42.67	14.00	69.00	21.91

Table 4. Mean, minimum, maximum, and standard deviation (SD) of benthic macroinvertebrate community parameters across six study sites sampled along the Bronx River in 2020.

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 5): pH ranged from 7.01 to 7.28; river depth ranged from 7.5 cm to 56.0 cm; river width ranged from 820 cm to 1680 cm; and water temperature ranged 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 six study sites (100 m radii) was comprised of greenspace (Table 6).

Water chemistry and physical variables	Mean	Min	Max	SD
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

Table 5. Mean, minimum, maximum, and standard deviation (SD) of water chemistry and physical variables across six study sites sampled along the Bronx River in 2020.

Percent Land Cover	Mean	Min	Max	SD
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

Table 6. Mean, minimum, maximum, and standard deviation (SD) of percent land cover (100 m radii) across six study sites sampled along the Bronx River in 2020.

Proportion of invasive species based on samples collected in 2020

The proportion of invasive species in the 2020 dataset ranged from 0 to 68 percent (mean = 38.00; SD = 26.59). Specifically, Site 1 harbored zero percent invasive species while Site 5 was comprised of 68 percent invasive species. Overall, there were two invasive species sampled across study sites: the Asian clam (*C. fluminea*; Family Corbiculidae; mean = 36.17; SD = 26.60; range: 0-68) and the rusty crayfish (*F. rusticus*; Family Cambaridae; mean = 3.50; SD = 4.76; range: 0-11).

2020 analyses

Based on Kruskal-Wallis tests, I found that study sites with a 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 a high invasive species dominance. However, I 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; range: 15-22) were identified over the past 22 years, ranging from a high of 22 in 1998 to a low of 15 in 2003 (Tables 7-10). The five 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 two most common Families sampled along the Bronx River were Hydropsychidae (37.25% of 1998 samples) and Chironomidae (31.75% of 1998 samples) (Table 7). In 2003, the two most common Families sampled were Chironomidae (38.75% of 2003 samples) and Naididae (29.00% of 2003 samples) (Table 8). In 2015, the three most common Families sampled were Gammaridae (23.00% of 2015 samples), Hydropsychidae (19.60% of 2015 samples), and Chironomidae (19.60% of total samples) (Table 9). In 2020, the two most common Families sampled along the Bronx River were Gammaridae (35.67% of 2020 samples) and Corbiculidae (34.50% of 2020 samples) (Table 10).

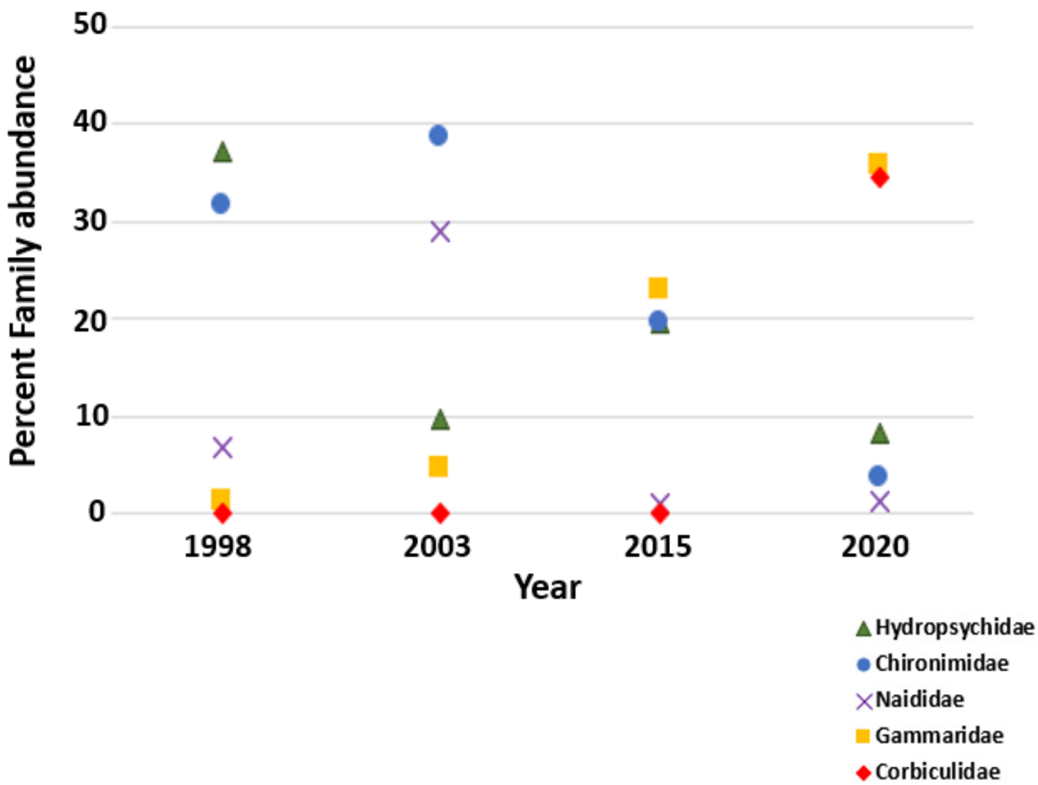


Figure 3. Longitudinal changes in the percent abundance of the five most common families sampled along the Bronx River from 1998-2020.

Family Name	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 5 East Gun Hill Rd	Percentage of total samples
AMPHIPODA					
Gammaridae	3	0	0	2	1.25
BASOMMATOPHORA					
Ancylidae	0	0	1	0	0.25
Physidae	5	0	0	0	1.25
COLEOPTERA					
Elmidae	9	1	0	0	2.50
DECAPODA					
Cambaridae	0	0	0	1	0.25
DIPTERA					
Chironomidae	13	58	22	34	31.75
Simuliidae	0	0	0	11	2.75
Tipulidae	2	0	0	0	0.50
EPHEMEROPTERA					
Baetidae	1	0	0	0	0.25
HAPLOTAXIDA					
Enchytraeidae	0	3	1	0	1.00
Naididae	3	23	0	1	6.75
Tubificidae	1	5	1	0	1.75
HOPLONEMERTEA					
Tetrastemmatidae	0	1	3	0	1.00
ISOPODA					
Asellidae	1	0	0	1	0.50
LUMBRICULIDA					
Lumbriculidae	1	0	0	0	0.25
ODONATA					
Aeshnidae	1	0	0	0	0.25
Calopterygidae	2	0	0	0	0.50
OPISTHOPORA					
Undetermined Lumbricina	0	0	2	0	0.50
RHYNCHOBDELLIDA					
Glossiphoniidae	0	1	0	2	0.75
TRICHOPTERA					
Hydropsychidae	38	8	61	42	37.25
Philopotamidae	20	0	0	0	5.00
TRICLADIDA					
Planariidae	0	0	8	3	2.75
VENERIDA					
Pisidiidae (Sphaeriidae)	0	0	1	3	1.00

Table 7. Benthic macroinvertebrate Families based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Bode et al. (1998). The Order is capitalized in the first column followed by Family name(s).

Family Name	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 5 East Gun Hill Rd	Percentage of total samples
AMPHIPODA					
Gammaridae	3	0	9	7	4.75
BASOMMATOPHORA					
Physidae	1	0	0	0	0.25
COLEOPTERA					
Elmidae	13	0	0	0	3.25
DIPTERA					
Chironomidae	50	32	59	14	38.75
Empididae	6	1	0	6	3.25
EPHEMEROPTERA					
Baetidae	12	0	0	0	3.00
HAPLOTAXIDA					
Enchytraeidae	2	0	1	5	2.00
Naididae	7	62	19	28	29.00
Tubificidae	2	1	5	1	2.25
LUMBRICULIDA					
Lumbriculidae	0	0	0	3	0.75
NEUROPTERA					
Sisyridae	0	0	3	0	0.75
OPISTHOPORA					
Undetermined Lumbricina	2	0	3	0	1.25
RHYNCHOBDELLIDA					
Glossiphoniidae	0	0	1	0	0.25
TRICHOPTERA					
Hydropsychidae	1	3	0	35	9.75
Hydroptilidae	0	1	0	0	0.25
TRICLADIDA					
Planariidae	1	0	0	1	0.50

Table 8. Benthic macroinvertebrate Families based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Bode et al. (2003). The Order is capitalized in the first column followed by Family name(s).

Family Name	Site 1 Valhalla	Site 2 White Plains	Site 4 Mount Vernon	Site 5 East Gun Hill Rd	Site 6 East 182nd St. Dam	Percentage of total samples
AMPHIPODA						
Gammaridae	31	19	40	25	0	23.00
BASOMMATOPHORA						
Physidae	0	0	0	1	0	0.20
COLEOPTERA						
Elmidae	10	5	2	2	0	3.80
DIPTERA						
Chironomidae	10	25	20	11	32	19.60
Empididae	0	0	0	1	0	0.20
Simuliidae	1	0	0	0	0	0.20
Tipulidae	0	1	0	0	0	0.20
EPHEMEROPTERA						
Baetidae	3	1	6	2	1	2.60
HAPLOTAXIDA						
Enchytraeidae	0	4	2	0	0	1.20
Naididae	0	0	5	0	0	1.00
Tubificidae	2	20	2	0	0	4.80
HOPLONEMERTEA						
Tetrastemmatidae	0	0	1	0	0	0.20
OPISTHOPORA						
Undetermined Lumbricina	2	0	8	4	0	2.80
RHYNCHOBDELLIDA						
Glossiphoniidae	1	11	0	2	0	2.80
TRICHOPTERA						
Hydropsychidae	29	4	0	36	29	19.60
Hydroptilidae	0	0	1	0	4	1.00
Philopotamidae	6	0	0	0	0	1.20
TRICLADIDA						
Planariidae	5	0	12	15	31	12.60
VENERIDA						
Pisidiidae (Sphaeriidae)	0	10	1	1	3	3.00

Table 9. Benthic macroinvertebrate Families based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Smith et al. (2015). The Order is capitalized in the first column followed by Family name(s).

Family Name	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 4 Mount Vernon	Site 5 East Gun Hill Rd.	Site 6 East 182nd St. Dam	Percentage of total samples
AMPHIPODA							
Gammaridae	23	17	64	63	21	26	35.67
BASOMMATOPHORA							
Planorbidae	0	11	1	0	0	0	2.00
Physidae	0	1	2	0	0	1	0.67
COLEOPTERA							
Elmidae	11	0	4	1	0	0	2.67
DECAPODA							
Cambaridae	0	11	8	1	0	1	3.50
DIPTERA							
Chironomidae	22	0	0	0	0	0	3.67
Tipulidae	7	0	0	0	1	0	1.33
HAPLOTAXIDA							
Enchytraeidae	1	0	0	0	1	1	0.50
Naididae	2	0	1	0	2	3	1.33
HEMIPTERA							
Veliidae	1	0	0	0	0	5	1.00
ISOPODA							
Asellidae	0	0	0	2	1	0	0.50
LUMBRICULIDA							
Lumbriculidae	1	1	3	0	3	0	1.33
ODONATA							
Aeshnidae	7	4	0	0	0	0	1.83
RHYNCHOBDELLIDA							
Glossiphoniidae	0	2	3	0	1	1	1.17
TRICHOPTERA							
Hydropsychidae	25	8	5	4	1	6	8.17
TRICLADIDA							
Planariidae	0	0	0	0	1	0	0.17
VENERIDA							
Corbiculidae	0	45	9	29	68	56	34.50

Table 10. Benthic macroinvertebrate Families based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River in 2020. The Order is capitalized in the first column followed by Family name(s).

Longitudinal changes in the percent abundance of functional feeding groups from 1998 to 2020

The percent abundance of functional feeding groups along the Bronx River varied over the past 22 years. The most common functional feeding groups sampled during the 22-year period were collector-gatherers (56.26% of total samples) and collector-filterers (31.53% of total samples) (Fig. 4). In 1998, the percentage of collector-gatherers (45.50% of 1998 samples) and collector-filterers (46.00% of 1998 samples) were nearly identical (Table 11). However, by 2003, collector-gatherers dominated the Bronx River (82.00% of 2003 samples) while the percentage of collector-filterers fell to 9.75% (Table 12). By 2015, the percentage of collector-gatherers decreased to 55.20% while the percentage of collector-filterers increased to 24.00% (Table 13). In 2020, the percentage of collector-gatherers (47.17% of 2020 samples) and collector-filterers (42.67 of 1998 samples) returned to similar proportions. The three other functional feeding groups—predators, scrapers, and shredders—remained at low levels (<5% of samples) throughout the 22-year period (Tables 11-14) with one exception: the percentage of predators sampled in 2015 was 15.80%.

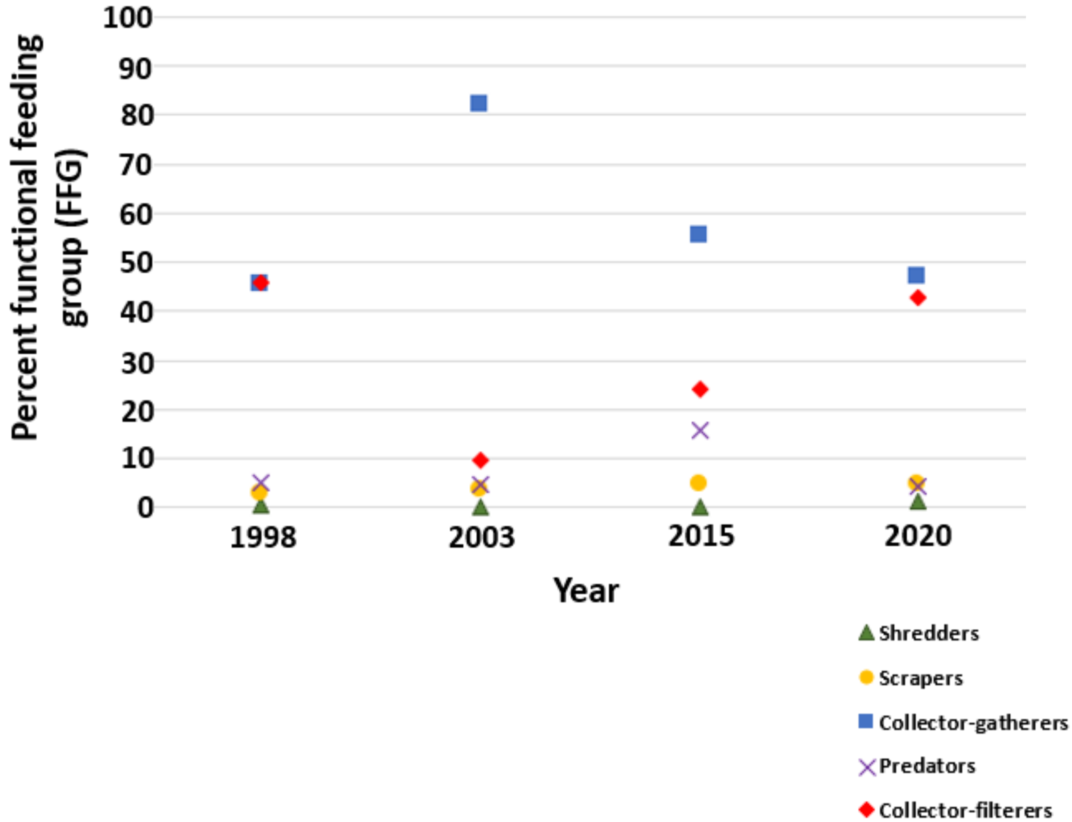


Figure 4. Longitudinal changes in the percent abundance of functional feeding groups from 1998 to 2020.

Functional feeding group (FFG)	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 5 East Gun Hill Rd.	Percentage of total samples
Shredders	2	0	0	0	0.50
Scrapers	9	1	1	0	2.75
Collector-gatherers	28	89	26	39	45.50
Predators	3	2	11	5	5.25
Collector-filterers	58	8	62	56	46.00

Table 11. Functional feeding group (FFG) composition based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Bode et al. (1998).

Functional feeding group (FFG)	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 5 East Gun Hill Rd.	Percentage of total samples
Shredders	0	0	0	0	0.00
Scrapers	13	1	0	0	3.50
Collector-gatherers	79	95	96	58	82.00
Predators	7	1	4	7	4.75
Collector-filterers	1	3	0	35	9.75

Table 12. Functional feeding group (FFG) composition based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Bode et al. (2003).

Functional feeding group (FFG)	Site 1 Valhalla	Site 2 White Plains	Site 4 Mount Vernon	Site 5 East Gun Hill Rd.	Site 6 East 182nd St. Dam	Percentage of total samples
Shredders	0	1	0	0	0	0.20
Scrapers	10	5	3	2	4	4.80
Collector-gatherers	48	69	83	43	33	55.20
Predators	6	11	13	18	31	15.80
Collector-filterers	36	14	1	37	32	24.00

Table 13. Functional feeding group (FFG) composition based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River by Smith et al. (2015).

Functional feeding group (FFG)	Site 1 Valhalla	Site 2 White Plains	Site 3 Tuckahoe	Site 4 Mount Vernon	Site 5 East Gun Hill Rd.	Site 6 East 182nd St. Dam	Percentage of total samples
Shredders	7	0	0	0	1	0	1.33
Scrapers	11	11	5	1	0	0	4.67
Collector-gatherers	49	30	78	66	28	32	47.17
Predators	8	6	3	0	2	6	4.17
Collector-filterers	25	53	14	33	69	62	42.67

Table 14. Functional feeding group (FFG) composition based on a sub-sample of 100 randomly selected organisms per study site sampled along the Bronx River in 2020.

Temporal changes in biodiversity indices

Over the past 22 years, water quality along the Bronx River has ranged from slightly impacted to moderately impacted (Fig. 5A-F). Across all years, the average BAP score was 3.74 (SD = 1.05), which is indicative of moderately impacted water quality (Table 3). 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; range: 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. 5A). 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; range: 2.52-3.64; Fig. 5B). Site 3, located 19.6 km from the river mouth, exhibited a decline in water quality from 1998 (moderate impact) to 2003 (severe impact), but the water quality appears to have somewhat recovered in 2020 (moderate impact) (Fig. 5C); the BAP score at Site 3 ranged 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 two different time points (Fig. 5D). In 2015, the BAP score at Site 4 was 4.52 (moderate impact) while in 2020, the BAP score was 2.25 (severe impact). 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; range: 3.02-4.44) (Fig. 5E). 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 a BAP score of 2.85 (moderate impact) in 2020 (Fig. 5F); the mean BAP score at Site 6 was 3.44 (SD = 0.83).

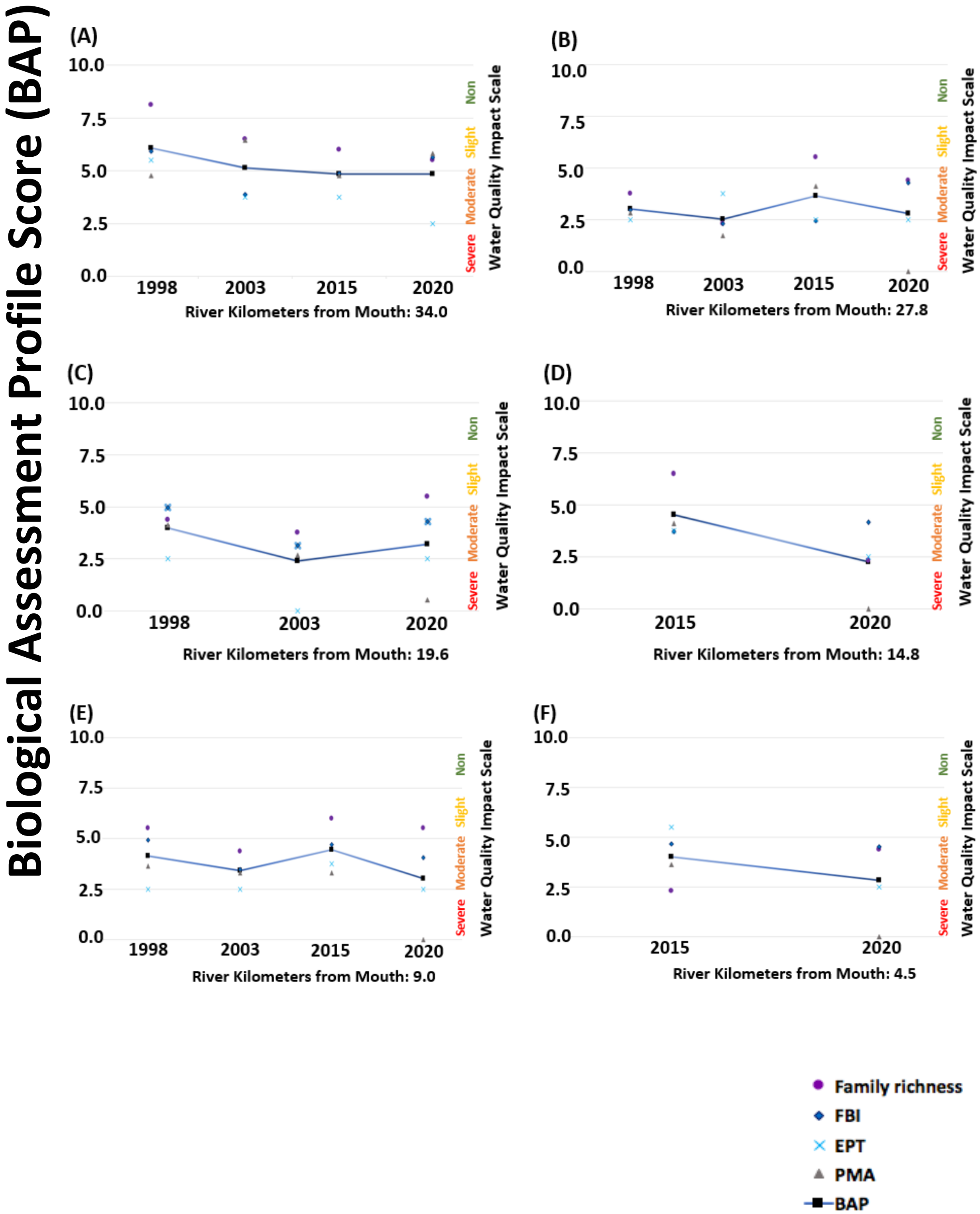


Figure 5A-F. Longitudinal changes in biodiversity indices over the past 22 years at six locations: (A) Site 1 (Valhalla), (B) Site 2 (White Plains), (C) Site 3 (Tuckahoe), (D) Site 4 (Mount Vernon), (E) Site 5 (East Gun Hill Rd.), (F) Site 6 (East 182nd St. Dam).

Changes in biodiversity and chemical/physical variables across years

For the three study sites (Site 1; Site 2; Site 5) in which there were data available for all four time periods (1998; 2003; 2015; 2020), mixed effects repeated measure ANOVAs revealed differences across years for certain biodiversity indices, including BAP, FBI, and certain functional feeding groups (Table 15). Of note, I found that there were lower BAP scores in 2020 than in 1998; and lower FBI values in 2003 than in 1998 or 2020 (Table 15). Moreover, repeated measure ANOVAs revealed differences across years for certain chemical and physical variables, including pH, river depth, river width, and temperature (Table 16).

Biodiversity Indices	**Year comparison**	**Estimate**	**SE**	***z*-value**	***P***
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 *
Percent collector-gatherer	2003 - 2020	41.667	13.832	-3.012	0.014 *
Percent collector-filterer	2003 - 2020	-36.000	14.296	2.518	0.058 ·
Percent predators	1998 - 2015	-8.333	3.162	2.635	0.042 *

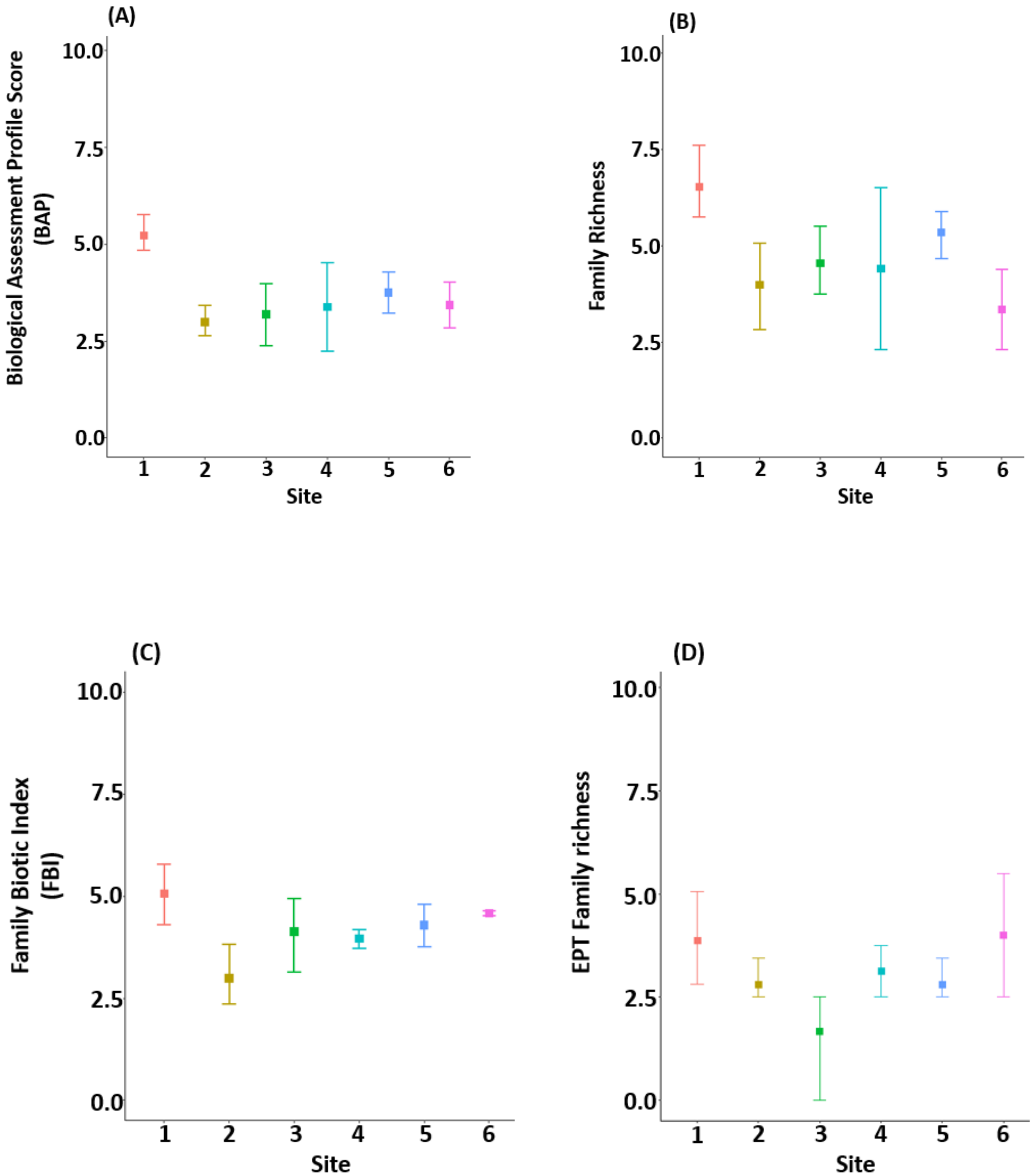
Table 15. Tukey post hoc test results for differences in biodiversity indices between years (· P>0.01; * P<0.05; ** P<0.01; *** P<0.001).

Water chemistry and physical variables	**Year comparison**	**Estimate**	**SE**	***z*-value**	***P***
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 *

Table 16. Tukey post hoc test results for differences in chemical and physical variables between years (· P>0.01; * P<0.05; ** P<0.01; *** P<0.001).

Changes in biodiversity across study sites from 1998-2020

From the six study sites that were sampled from 1998-2020, mixed effects repeated measure ANOVAs revealed differences between study locations for certain biodiversity indices, including BAP, Family richness, FBI, EPT, PMA, and percent scrapers (Fig. 6A-F; Table 17). Notably, the study site located farthest from the mouth of the Bronx River (Site 1) exhibited higher BAP scores, PMA, and percent scrapers than all five southern locations (Site 2; Site 3; Site 4; Site 5; Site 6).


(A)
Biological Assessment Profile Score (BAP)
10.0
7.5
5.0
2.5
0.0
1
2
3
4
5
6
Site
(B)
Family Richness
10.0
7.5
5.0
2.5
0.0
1
2
3
4
5
6
Site
(C)
Family Biotic Index (FBI)
10.0
7.5
5.0
2.5
0.0
1
2
3
4
5
6
Site
(D)
EPT Family richness
10.0
7.5
5.0
2.5
0.0
1
2
3
4
5
6
Site

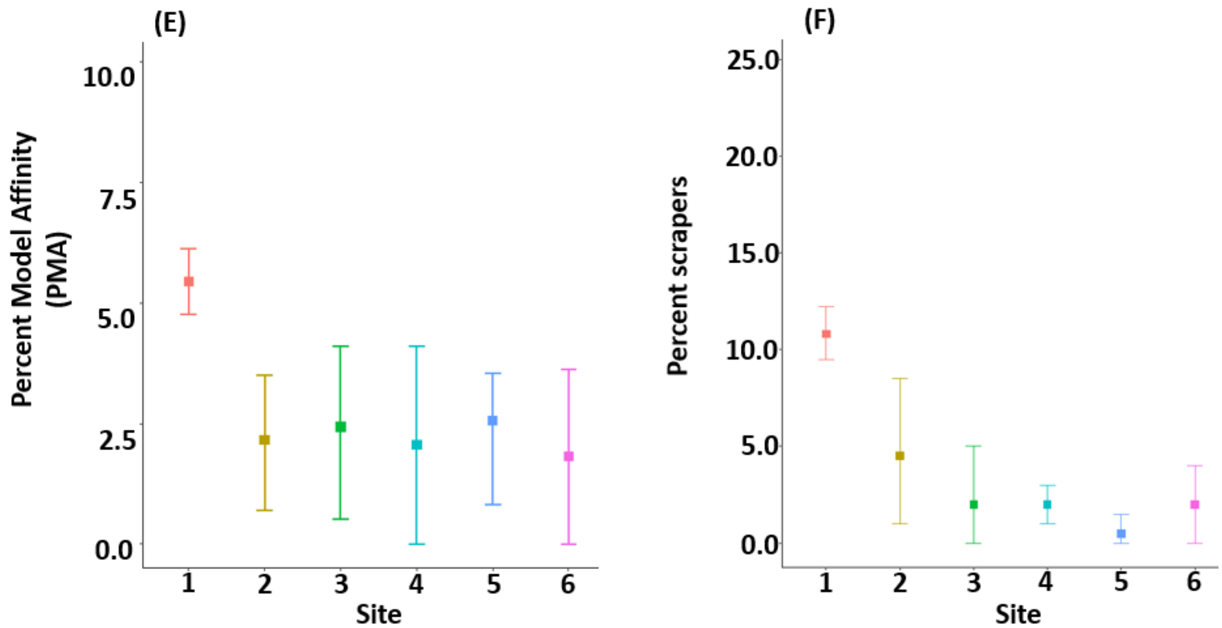


Figure 6A-F. Differences in biodiversity indices across six study sites sampled from 1998-2020. (A) Biological Assessment Profile (BAP), (B) Family richness (C) Family Biotic Index (FBI), (D) EPT Family richness, (E) Percent Model Affinity (PMA), (F) Percent scrapers.

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 *
Percent scrapers	1 - 2	6.250	1.927	-3.243	0.015 *
Percent scrapers	1 - 3	8.671	2.090	-4.149	< 0.001 ***
Percent scrapers	1 - 4	9.028	2.381	-3.791	0.002 **
Percent scrapers	1 - 5	10.250	1.927	-5.318	< 0.001 ***
Percent scrapers	1 - 6	9.028	2.381	-3.791	0.002 **
Percent predators	2 - 6	-12.145	4.614	2.632	0.088 ·

Table 17. Tukey post hoc test results for differences in biodiversity indices between study sites (Site 1; Site 2; Site 3; Site 4; Site 5; Site 6) based on four time periods (1998; 2003; 2015; 2020). (· P>0.01; * P<0.05; ** P<0.01; *** P<0.001).

Discussion

In a longitudinal assessment of New York City’s only freshwater river, I found spatial and temporal differences in overall water quality as indicated by benthic macroinvertebrate diversity. On a spatial scale, the recent introduction of invasive species into the Bronx River was associated with differences in water quality across study sites. Specifically, study sites with high invasive species dominance exhibited less healthy benthic macroinvertebrate communities than locations with low invasive species dominance. Moreover, in support of my prediction that upstream habitats exhibit higher benthic macroinvertebrate diversity than downstream habitats, I found, compared to all downstream study sites, Site 1 (Valhalla) exhibited the healthiest biological profiles. On a temporal scale, I found that on average, the overall water quality of the Bronx River has remained moderately impacted over the span of 22 years, which supports my prediction that water quality along the Bronx River would remain similar to past values. However, contrary to my prediction, I also found longitudinal declines in water quality at the three most downstream sites: Site 4 (Mount Vernon), Site 5 (East Gun Hill Road), and Site 6 (East 182nd St. Dam). Finally, I observed longitudinal changes in community composition as measured by benthic macroinvertebrate Family dominance and percent abundance of functional feeding groups. 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). Also, while the dominant functional feeding groups over the past 22 years have been collector-gatherers and collector-filterers, their proportions have fluctuated over time. 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 two measures of water quality: Percent Model Affinity (comparison to a reference stream) and overall water quality (BAP). Specifically, study sites with high invasive species dominance were less likely to harbor a biological community similar to an undisturbed reference stream than study sites with low 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 an undisturbed reference stream. Furthermore, 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: the Asian clam (*C. fluminea*) and the rusty crayfish (*F. rusticus*). Several studies have documented the adverse effects of these two species. For example, the Asian Clam has been shown to compete with native bivalves for food and habitat resources (Strayer 1999; Ferreira-Rodríguez et al. 2018). 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; Wilson et al. 2004; Mccarthy et al. 2006; Kuhlmann 2016). The adverse effects documented for these two 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 my 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 three 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; United States Army Corps of Engineers 2006; Natural Resources Group 2008; 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., Larson et al. 2001; Bond and Lake 2003; Alexander and Allan 2007; Bernhardt et al. 2007; Bernhardt and Palmer 2011; Sundermann et al. 2011; Violin et al. 2011). For example, Violin et al. (2011) found no significant differences in physical or biological variables between urban and urban restored 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 (Walsh et al. 2005 a, b; Bernhardt and Palmer 2007; Palmer et al. 2010). Of note, the Bronx River is comprised of several combined sewage overflow and municipal separate stormwater system sites, 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.

Macroinvertebrate community composition along the Bronx River has changed over time

During the past 22 years, the Bronx River has undergone several changes in community composition. Several Families identified during previous surveys were not documented in the current survey. Specifically, the following Families were documented in one or more of the surveys conducted in 1998, 2003, and 2015, but not in 2020: Ancylidae (freshwater pulmonated snails), Baetidae (Ephemeroptera: mayfly), Calopterygidae (damselflies), Empididae (dagger flies, balloon flies), Hydroptilidae (microcaddisfly), Philopotamidae (finger-net caddisfly), Sphaeriidae (fingernail clam), Simuliidae (black flies), Sisyridae (spongeflies), Tetrastemmatidae (ribbon worms), and Tubificidae (clitellate oligochaete). Conversely, the following Families were surveyed in 2020 but were not found in the three historical studies: Corbiculidae (Asian clam), Planorbidae (ramshorn snails) and Vellidae (broad-shouldered water striders).

Additionally, the Family Cambaridae was documented in 1998 (Bode et al. 1998); however, only the current survey identified the invasive rusty crayfish within this Family. 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 (proportion: 0.25%), 2003 (proportion: 3.00%) and 2015 (2.60%). 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 five 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. In support, 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 taxon that was surveyed in 2020 but not in the historical surveys was Corbiculidae (Asian clam). Interestingly, freshwater bivalves from the Family Sphaeriidae were surveyed in 1998 (proportion: 1.00%) and 2015 (proportion: 3.00%). 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 (proportion: 34.50%). 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 (Sousa et al. 2008; Ilarri and Sousa 2012). Overall, shifts in community composition documented along the Bronx River reflect the dynamic nature of benthic macroinvertebrate communities in an urban setting.

Family dominance along the Bronx River has changed over time

One striking result that I 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). From 1998 to 2020, the Family Gammaridae steadily increased in proportion from a low of 1.25% in 1998 to a high of 35.67% in 2020. Studies have found that the Family Gammaridae is capable of thriving in polluted water (Natural Resources Group 2008; Medupin 2020), which might explain the high proportion of Gammaridae in the Bronx River. Interestingly, the Family Corbiculidae was *not* sampled in the three historical surveys (1998, 2003, and 2015), but became the second most dominant Family in the year 2020 (proportion: 34.50%). The invasive Asian clam of the Family Corbiculidae might have successfully invaded the Bronx River within the span of five years as the last survey conducted did not document the presence of this bivalve (Smith et al. 2015). Richardson and Selby (2020) have suggested that successful establishment of invasive Asian clam populations can be detected when individuals of all sizes are found within a waterbody. 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 two most dominant families in 2020, two other Families underwent precipitous declines over the past two decades. In 1998, Hydropsychidae (net-spinning caddisflies) was the most dominant Family on the Bronx River (proportion: 37.25%). By 2020, the proportion of Hydropsychidae declined to only 8.17%. 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 evidence is required to test this hypothesis. Finally, Chironomidae was one of the most dominant Families in both 1998 (proportion: 31.75%) and 2003 (proportion: 38.75%); however, the proportion decreased to 19.60% in 2015 and then to only 3.67% in 2020. 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 (Wright and Burgin 2009; Al-Shami et al. 2010; Odume and Muller 2011). 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.

Collector-gatherers have dominated the Bronx River

Over the past 22 years, the most dominant functional feeding group sampled on the Bronx River was collector-gatherers (mean proportion = 56.26%). This result is consistent with other studies, which have found that collector-gatherers are capable of thriving in a wide range of habitats (Shieh et al. 1999; Stepenuck et al. 2002; Moreyra et al. 2015; Sterling et al. 2016). For example, Moreyra et al. (2015) found that among all functional feeding groups, collector gatherers were capable of surviving in undisturbed, disturbed, and even highly disturbed habitats. Indeed, studies have found a positive correlation between collector-gatherer dominance and impervious surface cover, which might explain their ubiquity in the Bronx River (Stepenuck et al. 2002; Sterling et al. 2016). In support of this result, two other macroinvertebrate studies on the Bronx River also reported high proportions of collector-gatherers (Natural Resources Group 2008; Baladrón and Yozzo 2020). The second most common functional feeding group sampled on the Bronx River were collector-filterers (mean proportion = 31.53%); however, their abundance fluctuated over time. In 1998, the percent abundance of collector-filterers was 46.00% (Bode et al. 1998). Five years later, the percent abundance was only 9.75% (Bode et al. 2003). These proportions rebounded over time (proportion in 2015: 24.00%; proportion in 2020: 42.67%). One reason that might explain the sharp decline in collector-filterers from 1998 to 2003 is the corresponding decline in overall water quality during this period. Specifically, the average BAP score in 1998 was 4.30; however, this value dropped to 3.37 in 2003. The collector-filterers that declined during this period were mainly Trichopterans, which are a pollution-sensitive taxon (Wright et al. 2018). Conversely, one reason why collector-filterers might have rebounded between 2015 and 2020 is the introduction of the invasive Asian clam (Corbiculidae). Indeed, the Asian clam comprised 34.50% of all samples collected in 2020. While collector-gatherers and collector-filterers dominated the Bronx River, there were three functional feeding groups that consistently exhibited low proportions over time: shredders (mean proportion: 0.58%), scrapers (mean proportion: 4.05 %), and predators (mean proportion: 7.53%). These functional feeding groups have been found to be more sensitive to anthropogenic disturbances (Stepenuck et al. 2002; Fu et al. 2016; Sterling et al. 2016). Overall, these results suggest that collector-gatherers are less sensitive to ecological disturbance relative to other functional feeding groups.

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

Among the six study sites sampled across 22 years, the most upstream site (Valhalla) exhibited the highest BAP scores, PMA, and percent scrapers than all five downstream locations. Several factors—including geomorphological, biotic, and anthropogenic variables—might explain these results.

First, changes in geomorphological characteristic 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 is comprised of 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 (Poole 2002; Doretto et al. 2020).

Second, biotic factors might explain differences between Valhalla and the downstream study sites. In the current study, Valhalla was the only study site that harbored zero 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.

Finally, anthropogenic factors, including the downstream locations of combined sewage overflows and municipal separate storm water 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 storm water 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). This might explain why I found no differences in biodiversity indices of 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 explain 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. Lastly, Valhalla exhibited the lowest proportion of developed land cover compared to all five 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 (Sponseller et al. 2001; Roy et al. 2003; Moore and Palmer 2005). While I 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 six study sites. In contrast, the study site in Valhalla exhibited *both* low human population density and a high proportion of greenspace. Altogether, these findings suggest that a combination of variables, an upstream location, low human population density, and percent greenspace, among other factors, 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 six 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 five of six study locations, found relatively better biodiversity profiles at the northernmost study site, and observed 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.

References

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

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

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

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

Aplasca, A. C., Titus, V., Ossiboff, R. J., Murphy, L., Seimon, T. A., Ingerman, K., ... & Iv, J. M. S. (2019). Health assessment of free-ranging chelonians in an urban section of the Bronx river, New York, USA. *Journal of Wildlife Diseases*, *55*(2), 352-362.

Atha, D. E., Forrest, T., Naczi, R. F., Pace, M. C., Rubin, M., Schuler, J. A., & Nee, M. (2016). The historic and extant spontaneous vascular flora of The New York Botanical Garden. *Brittonia*, *68*(3), 245-277.

Azrina, M.Z., Yap, C.K., Ismail, A. R., Ismail, A., Tan, S.G. (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., Kil, H. K., & Bae, K. S. (2005). Benthic macroinvertebrates for uses in stream biomonitoring and restoration. *KSCE Journal of Civil Engineering*, *9*(1), 55-63.

Baladrón, A., & Yozzo, D.J. (2020). Macroinvertebrate Assemblages, Stormwater Pollution, and Habitat Stressors in the Bronx River. *Urban Naturalist*, 31, 1-22.

Bauer, M. E., Loffelholz, B. C., & Wilson, B. (2007). Estimating and mapping impervious surface area by regression analysis of Landsat imagery, in *Remote Sensing of Impervious Surfaces*, edited by Q. Weng, pp. 31-48, CRC Press, Boca Raton, FL.

Beißler, M. R., & Hack, J. (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., & Palmer, M. A. (2007). Restoring streams in an urbanizing world. *Freshwater Biology*, *52*(4), 738-751.

Bernhardt, E. S., & Palmer, M. A. (2011). River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. *Ecological applications*, *21*(6), 1926-1931.

Bernhardt, E. S., Sudduth, E. B., Palmer, M. A., Allan, J. D., Meyer, J. L., Alexander, G., ... & Pagano, L. (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., Novak, M. A., Abele, L. E., Carlson, D. (1998). Bronx River Biological Assessment 1998 Survey [White Paper]. Department of Environmental Conservation.

Bode, R. W., Novak, M. A., Abele, L. E., Heitzman, D. L., Smith, A. J. (2003). Bronx River Biological Assessment 2003 Survey [White Paper]. Department of Environmental Conservation.

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

Bradley, B. A., Laginhas, B. B., Whitlock, R., Allen, J. M., Bates, A. E., Bernatchez, G., ... & Sorte, C. J. (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., & Ruggiero, G. S. (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., Chai, L., Jiang, D., Wang, J., Liu, Y., & Huang, Y. (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. PhD Dissertation. McGill University, Montreal, QC, Canada. 83 pp.

Cherry, D. S., Scheller, J. L., Cooper, N. L., & Bidwell, J. R. (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.

Clarke Murray, C., Gartner, H., Gregr, E. J., Chan, K., Pakhomov, E., & Therriault, T. W. (2014). Spatial distribution of marine invasive species: environmental, demographic and vector drivers. *Diversity and Distributions*, *20*(7), 824-836.

Cooper, N. L., Bidwell, J. R., & Cherry, D. S. (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., Palmer, M. A., & Crowl, T. A. (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., & Bower, P. (1998). Assessment of Bronx River Ecosystem: Pre-restoration baseline data. New York City Department of Parks and Recreation.

Cummins, K. W., Merritt, R. W., & Andrade, P. C. (2005). The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. *Studies on Neotropical Fauna and Environment*, *40*(1), 69-89.

de Kadt, M. (2011). *The Bronx River: An environmental and social history*. The History Press.

Darwall, W., Bremerich, V., De Wever, A., Dell, A. I., Freyhof, J., Gessner, M. O., ... & Weyl, O. (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., Hernandez, M. B. M., & Magbanua, F. S. (2016). Benthic macroinvertebrate community as an indicator of stream health: The effects of land use on stream benthic macroinvertebrates. *Science Diliman*, *28*(2).

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: 1- 22 pp. In S.H. Fernald, D.J. Yozzo and H. Andreyko (eds.), Final Reports of the Tibor T. Polgar Fellowship Program, 2014. Hudson River Foundation.

de Mello, K., Valente, R. A., Randhir, T. O., dos Santos, A. C. A., & Vettorazzi, C. A. (2018). Effects of land use and land cover on water quality of low-order streams in Southeastern Brazil: Watershed versus riparian zone. *Catena*, *167*, 130-138.

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

Doretto, A., Piano, E., & Larson, C. E. (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., Sousa, R., & Pardo, I. (2018). Negative effects of *Corbicula fluminea* over native freshwater mussels. *Hydrobiologia*, *810*(1), 85-95.

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

Frankel, E. (1999). A floristic survey of vascular plants of the Bronx River Parkway Reservation in Westchester, New York: compilation 1973-1998. *Journal of the Torrey Botanical Society*, 359-366.

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

Fu, L., Jiang, Y., Ding, J., Liu, Q., Peng, Q. Z., & Kang, M. Y. (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., Clavero, M., Sánchez, M. I., & Vilà, M. (2016). Global ecological impacts of invasive species in aquatic ecosystems. *Global change biology*, *22*(1), 151-163.

Goldstein, A. (2021). Predictors of avian diversity along the Bronx River (Master's Thesis, Department of Biology, Queens College, The City University of New York).

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., Culp, J., Drayer, A. N., McGregor, M. A., White, D. E., & Price, S. J. (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., Dimick, J. J., & Frie, R. V. (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., & Sousa, R. (2012). *Corbicula fluminea* Müller (Asian clam). In *A Handbook of Global Freshwater Invasive Species* (pp. 184-194). Routledge.

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

Kondolf, G. M., & Pinto, P. J. (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., Booth, D. B., & Morley, S. A. (2001). Effectiveness of large woody debris in stream rehabilitation projects in urban basins. *Ecological Engineering*, *18*(2), 211-226.

Lerner, D. N., & Holt, A. (2012). How should we manage urban river corridors? *Procedia Environmental Sciences*, *13*, 721-729.

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

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.

Matlou, K., Addo-Bediako, A., & Jooste, A. (2017). Benthic macroinvertebrate assemblage along a pollution gradient in the Steelpoort River, Olifants River System. *African Entomology*, *25*(2), 445-453.

Mccarthy, J. M., Hein, C. L., Olden, J. D., & Jake Vander Zanden, M. (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.

Miller, S. W., Budy, P., & Schmidt, J. C. (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., Casaux, R., Archangelsky, M., Di Prinzio, C. Y., Brand, C., & Kutschker, A. M. (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., & Uchrin, C. (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., Castro, P., Lopes-Lima, M., Antunes, C., Ilarri, M., & Sousa, R. (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., & Palmer, M. A. (2005). Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. *Ecological Applications*, *15*(4), 1169-1177.

Moreyra, A. K., & Padovesi-Fonseca, C. (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., Selvakumar, C., Sundar, S., & Raja, M. (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 State Department of Environmental Conservation Division of Water (NYSDEC-DOW). (2019). Standard operating procedure: Biological monitoring of surface waters in New York State.

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

Odume, O. N., & Muller, W. J. (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., & Oribhabor, B. J. (2002). Ecological impact of river impoundment using benthic macro-invertebrates as indicators. *Water Research*, *36*(10), 2427-2436.

Olden, J. D., McCarthy, J. M., Maxted, J. T., Fetzer, W. W., & Vander Zanden, M. J. (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., Stewart, T. W., & Thompson, J. R. (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., Taillie, D., Walsh, B., Dennison, W. C., Bone, E. K., Reid, D. J., 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., Menninger, H. L., & Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology*, 55, 205-222.

Paul, M. J., & Meyer, J. L. (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 ed.). Wiley.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., Heisterkamp, S., Van Willigen, B., & Maintainer, R. (2017). Package 'nlme'. *Linear and nonlinear mixed effects models, version*, *3*(1).

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

Rachlin, J. W., Warkentine, B. E., & Pappantoniou, A. (2007). An evaluation of the ichthyofauna of the Bronx River, a resilient urban waterway. *Northeastern Naturalist*, *14*(4), 531-544.

Richardson, T. D., & Selby, J. M. (2020). The nonindigenous Asian Clam, *Corbicula fluminea*, in New Hampshire. *Northeastern Naturalist*, *27*(2), 272-280.

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

Samaritan, J. M., & Schmidt, R. E. (1982). Aspects of the life history of a freshwater population of the mummichog, *Fundulus heteroclitus* (Pisces: Cyprinodontidae), in the Bronx River, New York, USA. *Hydrobiologia*, *94*(2), 149-154.

Schertzinger, G., Itzel, F., Kerstein, J., Tuerk, J., Schmidt, T. C., & Sures, B. (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., Naiman, R. J., Cummins, K. W., Minshall, G. W., & Vannote, R. L. (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., Kondratieff, B. C., & Ward, J. V. (1999). Longitudinal changes in benthic organic matter and macroinvertebrates in a polluted Colorado plains stream. *Hydrobiologia*, *411*, 191-209.

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

Smith, A. J., Rickard, S., Mosher, E. A., Lojpersberger, J.L., Heitzman, D. L., Duffy, B. T., Novak, M. A. (2015). Bronx River Biological Assessment 2015 Survey [White Paper]. Department of Environmental Conservation.

Smith, K. R., Roth, B. M., Jones, M. L., Hayes, D. B., Herbst, S. J., & Popoff, N. (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., Antunes, C., & Guilhermino, L. E. D. P. S. (2008). Ecology of the invasive Asian clam *Corbicula fluminea* (Müller, 1774) in aquatic ecosystems: an overview. In *Annales de Limnologie-International Journal of Limnology* (Vol. 44, No. 2, pp. 85-94). EDP Sciences.

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

Stepenuck, K. F., Crunkilton, R. L., & Wang, L. (2002). Impacts of urban land use on macroinvertebrate communities in southeastern Wisconsin streams 1. *JAWRA Journal of the American Water Resources Association*, *38*(4), 1041-1051.

Sterling, J. L., Rosemond, A. D., & Wenger, S. J. (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.

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

Sun, W., Xia, C., Xu, M., Guo, J., & Sun, G. (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.

Svensson, O., Bellamy, A. S., Van den Brink, P. J., Tedengren, M., & Gunnarsson, J. S. (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. (2006). Final data and documentation report, Bronx River Ecosystem Restoration Project, water quality and biological baseline data collection, Westchester and Bronx Counties, New York. United States Army Corps of Engineers, Planning Division, New York District, New York.

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

Vaughn, C. C., & Hakenkamp, C. C. (2001). The functional role of burrowing bivalves in freshwater ecosystems. *Freshwater Biology*, *46(11)*, 1431-1446.

Violin, C. R., Cada, P., Sudduth, E. B., Hassett, B. A., Penrose, D. L., & Bernhardt, E. S. (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 & Woodward Publishing Company.

Wallace, J. B., & Webster, J. R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annual review of entomology*, *41*(1), 115-139.

Walsh, C. J., Fletcher, T. D., & Ladson, A. R. (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., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (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., Johnson, M. F., Wood, P. J., Thorne, C. R., & Eichhorn, M. P. (2021). Anthropogenic litter is a novel habitat for aquatic macroinvertebrates in urban rivers. *Freshwater Biology*, *66*(3), 524-534.

Wilson, K. A., Magnuson, J. J., Lodge, D. M., Hill, A. M., Kratz, T. K., Perry, W. L., & Willis, T. V. (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., & Burgin, S. (2009). Effects of organic and heavy metal pollution on chironomids within a pristine upland catchment. *Hydrobiologia*, *635*(1), 15-25.

Wright, I. A., Paciuszkiewicz, K., & Belmer, N. (2018). Increased water pollution after closure of Australia's longest operating underground coal mine: a 13-month study of mine drainage, water chemistry and river ecology. *Water, Air, & Soil Pollution*, *229*(3), 1-20.

Xie, S., Wang, X., Ren, Y., Su, Z., Su, Y., Wang, S., ... & Ouyang, Z. (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., Wang, Z., Duan, X., & Pan, B. (2014). Effects of pollution on macroinvertebrates and water quality bio-assessment. *Hydrobiologia*, *729*(1), 247-259.

Yeager, M. M., Neves, R. J., & Cherry, D. S. (1999, March). Competitive interactions between early life stages of *Villosa iris* (Bivalvia: Unionidae) and adult Asian clams (*Corbicula fluminea*). In *Freshwater Mollusk Symposium Proceedings—Part II: Proceedings of the First Freshwater Mollusk Conservation Society Symposium* (pp. 253-259).

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