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Water quality impacts of small-scale hydromodification in an urban stream in Connecticut, USA

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Abstract

Introduction: Construction activities in and along urban streams increase the sediment input into surface waters, causing an overall decline in water quality and aquatic ecosystems. In this case study, we investigate the water quality impacts of local hydromodification in an urban stream (discharge 0.4 m^3/s). At the site of interest, workers removed a stream crossing consisting of an embankment with culverts and replaced it with a small bridge (single span of 25 m) in an effort to improve flow capacity.

Methods: Water samples were taken at four sites along the North Branch Park River in Connecticut, Northeastern United States. Turbidity and dissolved oxygen (DO) were measured in situ, and nitrate and total phosphorus (TP) were measured in the laboratory. Benthic macroinvertebrate samples were also collected and analyzed for taxon richness and Shannon-Weaver species diversity. Data were compared between upstream and downstream sites and before, during, and after hydromodification. We used one-way ANOVA combined with the post hoc Turkey test to derive statistical significance.

Results: During construction, turbidity increased temporarily by 60.9% [from 2.48 Nephelometric Turbidity Units (NTU) over ambient to 4.00 NTU]. Once construction was completed, DO increased locally from 11.0 to 13.0 mg/L. Benthic macroinvertebrate taxon richness and species diversity declined by 61.6 and 32.6% respectively, with no recovery observed in the year following construction. Water quality was only affected within 50 m downstream. Nitrate and TP concentrations were unaffected.

Conclusions: Small-scale hydromodification temporarily increased the turbidity as a result of increased sediment input, approaching the maximum level for clean water (5 NTU). Benthic macroinvertebrate communities declined in the immediate downstream vicinity of construction but are expected to recover soon given that turbidity recovered to pre-construction levels, and DO increased. These outcomes emphasize that environmental assessment is important not only for large-scale hydromodification but also for smaller scale stream modifications.

Keywords: Urban streams, Pollution, Hydrologic services, Water quality, Benthic macroinvertebrates, North Branch Park River

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Introduction

Construction activities and hydromodification are recognized as major sources of water quality impairment in urban streams, because they increase sediment input into waterways and downstream aquatic ecosystems (Brabec et al. 2002; Meyer et al. 2005; Chen et al. 2009; Houser and Pruess 2009; Hassan et al. 2015). Highway construction can lead to a rapid decline in stream water quality (e.g., suspended solids, iron, chloride, sulfate, nitrogen, and pH), not only during construction but also in the period that follows (Chen et al. 2009; Purcell et al. 2012). The pollution from this type of large construction projects degrades aquatic ecosystems and has severe impacts on food webs (Bennett et al. 2001). Evidence suggests that aquatic ecosystems in small streams are equally susceptible to pollution by sediment (Lemly 1982); however, the local water quality impacts in these small streams are extremely variable (Berger et al. 2017).

A healthy aquatic ecosystem depends on minimum standards for water quality, defined to guarantee sufficient dissolved oxygen (DO) concentrations (> 5 mg/L) and limited phosphorus and nitrogen concentrations (Bennett et al. 2001; Le et al. 2010). Reduced water quality is often first visible in higher turbidity caused by light scattering coming from particles present in the water column. Turbidity increases the temperature of water, which can then hold less oxygen. Oxygen is further reduced when eutrophication resulting from excess phosphorus and nitrogen input causes an algal bloom that consumes the oxygen (Correll 1998; Patil and Deng 2012; Hobbie et al. 2017). The effects of water quality impairment on food chains, including benthic macroinvertebrate and fish communities, are often immediate and profound (DaSilva et al. 2013; Hassan et al. 2015). Fish experience physiological effects from high sediment loads, affecting reproductive behavior (Bilotta and Brazier 2008). The decline of benthic macroinvertebrate communities affects recreational fishing and aquaculture, decimating the local value of these ecosystem services (Patil and Deng 2012). Benthic macroinvertebrates have specific requirements for DO, nutrients, and light penetration (Covich et al. 1999; Mundahl and Hunt 2011). This explains why they are an important proxy for water quality and ecosystem health (e.g., Quinn et al. 1997; Weigel et al. 2002; Chen et al. 2009; Mundahl and Hunt 2011).

Increased concern for what is collectively known as the urban stream syndrome, attributed to urban infrastructure, has paved the way for a shift in how urban streams are valuated in the USA. More emphasis is now placed on optimizing the balance between on the one hand efficient storm drainage—an important market ecosystem service—and on the other hand biodiversity, support for urban ecosystems, habitat for plant and animal life, and social services (esthetic and recreational), which are critical nonmarket ecosystem services (Wilson and Carpenter 1999; Sun et al. 2017). Frequent flooding disrupts this balance, as is the case for the North Branch Park River in Connecticut, Northeastern United States. In there, a combination of increased paved surface and sewer overflows in the upstream watershed carries excess sediment and adsorbed pollutants into the stream (Fuss and O'Neill 2010). Flooding and water quality impairment have prompted a local shift to low impact development and river restoration in the watershed (Fuss and O'Neill 2010). Nonetheless, impacts on sediments, nutrients, oxygen, and biotic assemblages are difficult to predict due to the complexity of urban hydrologic systems (Meyer et al. 2005).

The objective of this study was to evaluate the water quality impacts of flow improvement works in the urban North Branch Park River (drainage area 74.1 km^2) in Connecticut, Northeastern United States. In the studied location, a stream crossing in the form of an embankment with culverts was replaced with small single-span bridge to increase flow capacity and reduce flood occurrence in the immediate upstream vicinity. We evaluated the impacts of this flow improvement project on water quality by comparing turbidity [a common indicator of suspended solids], dissolved oxygen (DO), total phosphorus (TP), and benthic macroinvertebrate abundance, taxon richness and diversity (Weigel et al. 2002; Chen et al. 2009; Hassan et al. 2015) measured before, during, and after construction.

Methods

North Branch Park River

The North Branch Park River in Connecticut, Northeastern United States, drains a 74.1-km^2 urban watershed that includes part of the city of Hartford and adjacent communities. This watershed is part of the Connecticut River basin (Fig. 1a) and home to approximately 48,000 residents (Fuss and O'Neill 2010). Temperatures exceeded 32.2 °C during 19 days/year on average, and descended below − 17.8 °C during 6 days/year (Fuss and O'Neill 2010). With a precipitation of 1172 mm/year and snowfall of 1151 mm/year between 1971 and 2000 (at Bradley International Airport, National Climatic Data Center), climate is characterized as hot-summer humid continental (Dfa).

The studied section of the North Branch Park River runs through the middle of the University of Hartford campus. Median discharge was 0.43 m^3/s (2015–2017 data for gauging station 1,191,000, US Geological Survey in cooperation with the City of Hartford), and flow velocity varied between 0.3 and 3.0 m/s (Fuss and O'Neill 2010).

Hydromodification

There are three stream crossings along the studied stream section, including (along the direction of flow;

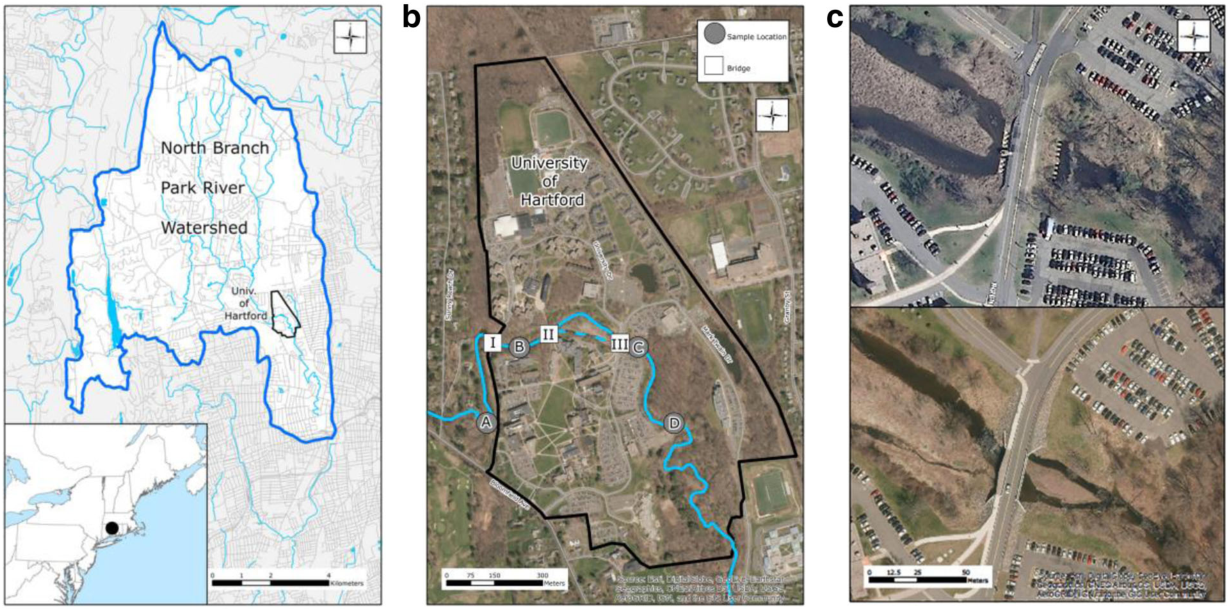

Fig. 1 **a** Location of the North Branch Park River watershed in Connecticut, USA. **b** Location of the four sampling sites. Sites A and B were located upstream of the construction site, and sites C and D were downstream of the construction site. Stream crossing I is a concrete single-span road bridge, stream crossing II is a pedestrian truss bridge, and stream crossing III is the new concrete single-span road bridge. The dashed line represents a stream channel that only conducts water during high flow; only the northern channel conducts water during normal flow. **c** Satellite view of the previous structure (embankment with culverts) in March 2012 (top) and the new structure (bridge) after it was completed in February 2013 (bottom)

Fig. 1b) a road bridge (I) and a pedestrian bridge (II), and another road bridge (III). The upstream single-span concrete bridge (I) and pedestrian truss bridge (II) have sufficiently wide spans to allow flow to pass unimpeded—27 and 72 m, respectively, the latter supported by a pier along the stream bank. The site of interest was the third stream crossing (III) (41.7986° N, 72.7132° W). A bridge with a single span of 25 m was built at this location, to replace a stream crossing in the form of an embankment with culverts that severely impeded throughput of high water flows (Fig. 1c).

The former structure supported a paved road and allowed water to pass through eight culverts, each 76.2 cm in diameter and 21 m long, with two culverts submerged during normal flow. The combined flow capacity of these culverts was insufficient during fullbank discharge—the average channel width at this location is approximately 12 m, with a bankfull width of 75 m—and frequently caused water to back up behind the embankment. This resulted in repeated flooding of the adjacent parking lots, and a solution was urgently needed.

The site was modified in an effort to increase flow capacity and reduce flood risk in the direct upstream vicinity. First, the stream was temporarily diverted, and the embankment demolished. Next, the construction of a single-span bridge started in the same location. Concrete foundations were laid, and abutments were created on top. The superstructure was suspended and secured in place, and the deck was paved with asphalt to create the road. Finally, the stream was rerouted to its original pathway. The building project started in June 2012, and the bridge was completed in February 2013. The 25-m span of the bridge substantially increased the flow capacity. No new occurrences of flooding were reported after this.

Water sampling

Four sampling sites were selected along a 1200-m reach that comprises all three stream crossings (Fig. 1b). In the direction of flow, site A was the furthest upstream sampling site, followed by stream crossing I, then site B, stream crossing II, and stream crossing III, where construction took place. Site A and site B were located 680 and 260 m upstream of the construction site, respectively, and site C and site D were located 50 and 365 m downstream of the construction site, respectively. Discharge was comparable along this stream section, with no tributaries, and the channel and stream bank characteristics in these sites were as follows:

Site A: Rocky streambed, gentle bank slope, bank vegetation largely absent
Site B: Gravel streambed, gentle bank slope, open bank vegetation
Site C: Sandy streambed, gentle bank slope, open bank vegetation

Site D: Sandy streambed, steep bank slope, open bank vegetation with plant debris

Water samples were collected on a monthly basis over the course of three 6-month periods, each of which included the fall and winter seasons: prior to construction work during the period between September 2011 and February 2012, during construction from September 2012 to February 2013, and after construction from September 2013 to February 2014. The sampling periods were selected based on the construction period and accessibility of sampling sites. No samples were collected at site D in February 2014 when the stream was frozen. To obtain baseline data, samples were collected on non-rainy days at least 5 days after a storm event or intense snowmelt. Triplicate samples were collected on each site and at each sampling date directly below the water surface in the center of the stream channel (standard protocol, US Geological Survey National Water-Quality Assessment Project) (number of samples $n = 204$).

Turbidity, DO, nitrate, and total phosphorus determination

During water sampling, turbidity was measured in situ with the 2100Q Portable Turbidimeter (Hach Company, Loveland, Colorado), and DO was measured with the Orion 4 Star pH/DO Portable Meter (Thermo Scientific, Madison, Wisconsin). In the laboratory, water samples were prepared for nitrate determination using the cadmium reduction method and samples for TP analysis were prepared using the acid persulfate digestion method. Nitrate and TP concentrations were determined by using the Genesys 10s UV-Vis spectrophotometer (Thermo Scientific, Madison, WI) following the EPA-approved standard procedures (Eaton and Franson 2005).

Benthic macroinvertebrate sampling and analysis

Benthic macroinvertebrates were also collected on a monthly basis during the same three 6-month periods. A Turtox D-Shaped dip net (45 × 23 cm, Wildco, Buffalo, New York) was dragged across the streambed over a length of 15 m between the left and right banks, while stirring up the streambed sediment with feet. The materials collected in the net were placed in a 250-μm sieve (Will Scientific Inc., Rochester, New York), rinsed several times, and then transferred to a collection bottle for further analysis. Three samples were collected at each site. Again, no samples were collected at site D in February 2014 when the stream was frozen.

In the laboratory, benthic macroinvertebrates were identified using the keys provided by Thorp and Covich (2009). After classifying the organisms, specimens were preserved in 95% ethanol. Taxon abundance was estimated as the mean number of specimens per taxon divided by the surveyed area. The taxon richness (number of unique taxa) and Shannon-Weaver index of taxonomic diversity were also calculated (Zhu et al. 2006; Mundahl and Hunt 2011; Zhu et al. 2015).

Statistical analysis

Prior to statistical analysis, water quality data were transformed to reduce heteroscedasticity by taking the natural logarithm $[ln(x+1)]$. One-way analysis of variance (ANOVA) was performed to test the equality of means for individual water quality indicators (turbidity, nitrate, TP, and benthic macroinvertebrate taxon richness and species diversity) between sampling sites (A, B, C, and D) (Kuehl 2000). This was done for the periods prior to construction, during construction, and after construction, respectively. A second ANOVA was performed to test the equality of the same means between these periods for each sampling site. In other words, the first ANOVA evaluates along-stream differences in water quality for each period, and the second ANOVA identifies time-dependent changes at individual sampling sites.

Because ANOVA only indicates whether or not the evaluated means are presumed equal, the post hoc Tukey test was performed to compare all possible pairs of means and precisely identify the differences that are greater than the standard error (SE) (Kuehl 2000). All statistical analyses were conducted using IBM SPSS Statistics 21 (IBM Corporation, Armonk, New York). Statistical significance was determined for $\alpha = 0.05$.

Results

Turbidity

No difference in turbidity was observed between sites prior to construction (df = 3, $F = 0.13$, $p = 0.944$) or after construction (df = 3, $F = 1.45$, $p = 0.238$). But, during the construction period, a significant difference started to develop (df = 3, $F = 4.05$, $p = 0.010$; Table 1 and Fig. 2), when turbidity increased immediately downstream of the construction site during construction period and returned to prior levels after construction. This is reflected by measurements for site C (Fig. 2), where values first increased from 2.48 ± 0.36 Nephelometric Turbidity Unit (NTU) to 4.00 ± 0.94 NTU (+ 60.9%) and then decreased to 2.4 ± 0.15 NTU (df = 2, $F = 46.9$, $p < 0.001$).

A similar but less extreme pattern was observed further downstream of construction at site D, where turbidity increased from 2.55 ± 0.32 NTU to 3.32 ± 0.36 NTU during construction and then decreased to 2.64 ± 0.18 NTU (df = 2, $F = 12.4$, $p < 0.001$). By contrast, turbidity did not change significantly at upstream site A (df = 2, $F = 0.62$, $F = 0.542$) whereas turbidity at site B decreased slightly from 2.23 ± 0.23 NTU to 1.82 ±

Table 1 Results of one-way ANOVA

Water quality indicator		ANOVA between sampling sites			ANOVA between periods			
		Before construction	During construction	After construction	Site A	Site B	Site C	Site D
Turbidity	*F*	0.127	4.051	1.445	0.623	6.346	46.927	12.429
	p value	0.944	0.010	0.238	0.542	0.004	< 0.001	< 0.001
DO	*F*	0.105	0.149	0.118	2.881	1.588	5.552	2.896
	p value	0.956	0.929	0.948	0.090	0.239	0.017	0.091
Nitrate	*F*	1.138	0.479	0.044	47.000	59.305	48.666	44.006
	p value	0.340	0.698	0.988	< 0.001	< 0.001	< 0.001	< 0.001
TP	*F*	0.067	0.410	0.502	14.038	142.671	26.721	61.170
	p value	0.977	0.746	0.682	< 0.001	< 0.001	< 0.001	< 0.001
Macroinvertebrate taxon richness	*F*	17.459	5.687	1.113	2.442	5.000	46.731	2.016
	p value	< 0.001	0.006	0.370	0.121	0.022	<0.001	0.173
Shannon-Weaver macroinvertebrate species diversity	*F*	5.705	1.711	2.189	3.782	3.029	5.239	0.500
	p value	0.005	0.197	0.125	0.049	0.081	0.020	0.618

Left: Comparison of sampling site means of turbidity, DO, nitrate, TP, macroinvertebrate taxon richness, and Shannon-Weaver macroinvertebrate species diversity, evaluated for the periods before (BE), during (DU), and after construction (AF). Right: Comparison of periodic means (BE, DU, AF) of the same variables per sampling site

0.22 NTU during construction and then declined again to 2.30 ± 0.15 NTU (df = 2, $F = 6.35$, $p = 0.004$; Table 1).

DO, nitrate, and total phosphorus

DO concentrations increased significantly at site C after construction (df = 2, $F = 5.55$, $p = 0.017$; Table 1), from less than 11.0 to 13.0 mg/L (Fig. 3a). Modest increases were also observed at the other sites, but these increases were not significant ($p > 0.05$), and similar across the sites before, during, and after construction (Fig. 3a; all $p > 0.05$). DO concentrations were highest during the colder months (maximum of 16.4 mg/L measured in January 2014) and lowest during the warmer months (5.70 mg/L in September 2011).

Although there were significant temporal differences in nitrate and TP concentrations (Fig. 3b, c), values were always similar for the four sampling sites (all $p > 0.05$; Table 1). Nitrate concentrations were highest before construction (0.38 mg/L), declined during construction, and finally increased again after construction to a value below the pre-construction reading (0.34 mg/L) (Fig. 3b). TP was also similar across sampling site and decreased markedly throughout the observed period, from a value of approximately 50 μg/L before construction, to 35 μg/L and 25 μg/L during and after construction, respectively (Fig. 3c).

Benthic macroinvertebrate abundance, taxon richness, and diversity

Benthic macroinvertebrate taxon richness and species diversity showed the greatest change directly downstream of the construction site ($p < 0.02$ for site C; Table 1) and became more similar across the four sampling sites after construction was complete ($p > 0.05$).

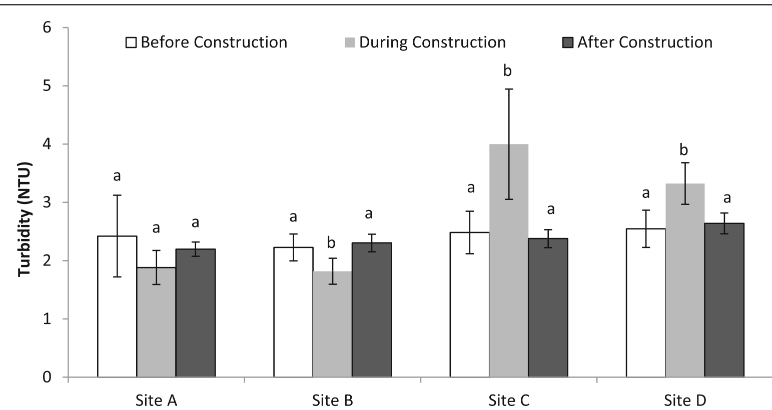

Fig. 2 Turbidity at the four sampling sites in the North Branch Park River. Bars indicate the mean ± 1 standard error (SE). The (a, b) markings above the columns indicate the groups identified with the Tukey test ($\alpha = 0.05$)

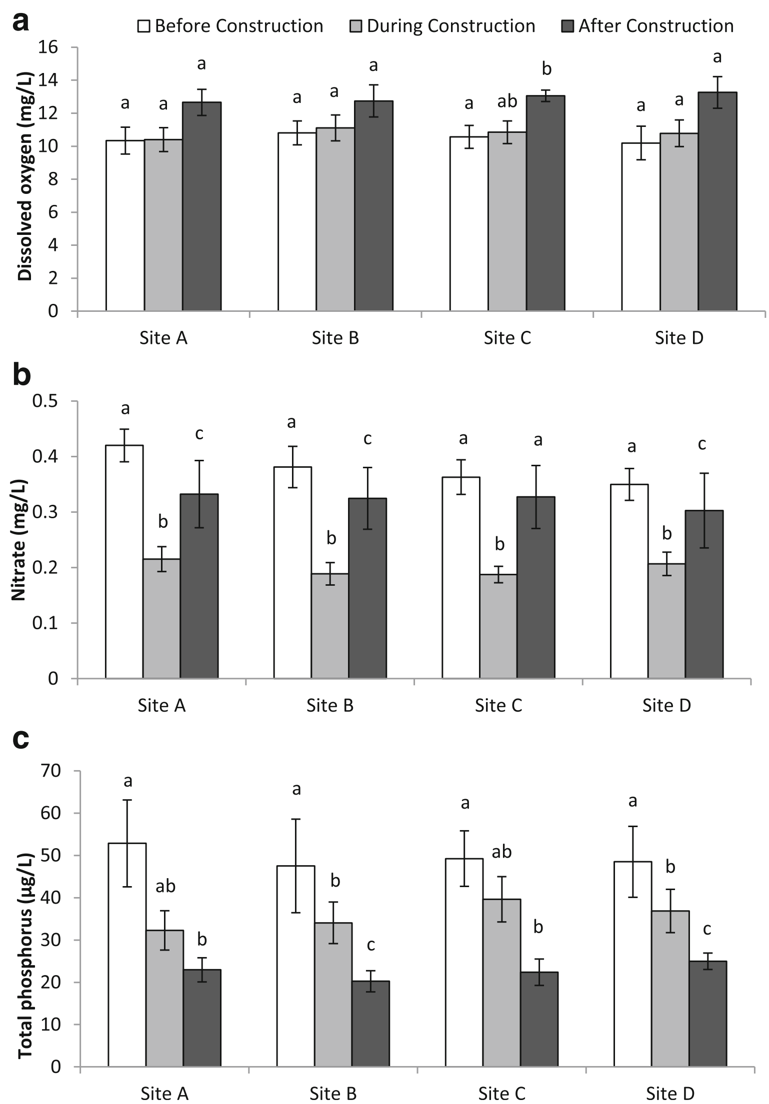

Fig. 3 Dissolved oxygen (**a**), nitrate concentration (**b**), and total phosphorus concentration (**c**) at the four sampling sites in the North Branch Park River. Bars indicate the mean ± 1 SE. The (a, b) markings above the columns indicate the groups identified with the Tukey test ($\alpha = 0.05$)

Before construction, the most abundant organism at all sampling sites was the aquatic worm, ranging from 6.5 individuals/m^2 at site A to 13.8 individuals/m^2 at site C (Table 2). The second most abundant organism at all sites was the Mayfly larva, with between 1.9 and 8.0 individuals/m^2. Midge was also found at all sites.

Before construction, abundance at downstream site C and site D was greater than at upstream site A and site B. But numbers declined drastically during construction. For example, the number of aquatic worms decreased from 13.8 individuals/m^2 before construction to 0.7 individuals/m^2 during construction and 0.9 individuals/m^2 after construction. In contrast, horsehair worm and mayfly larva populations increased during and after construction at downstream site C and site D. Caddisfly larvae were observed at site C and site D before and after, but not during construction. Other taxa including planarian, soldier fly larva, and stonefly larva seem to have disappeared altogether after construction.

Taxon richness also differed significantly between the four sites before construction (df = 3, $F = 17.46$, $p < 0.001$; Table 1 and Fig. 4a, b). During construction, taxon richness was lowest at site C whereas site B had the highest richness (df = 3, $F = 5.69$, $p = 0.006$). Taxon richness did not change significantly at site A (df = 2, $F = 2.44$, $p = 0.121$) and site D (df = 2, $F = 2.02$, $p = 0.173$), but decreased slightly at site B after construction (df = 2, $F = 5.00$, $p = 0.022$). An even more substantial decrease was observed at site C, where taxon richness declined from 7.3 ± 0.4 before construction to 2.8 ± 0.7 during

Table 2 Composition and mean abundance (individuals/m^2) of benthic macroinvertebrates at the four sampling sites during the periods before (BE), during (DU), and after construction (AF). Missing values indicate that no specimens were identified

		Site A			Site B			Site C			Site D		
Organism	Classification	BE	DU	AF	BE	DU	AF	BE	DU	AF	BE	DU	AF
Aquatic worm	*Oligochaeta*[d]	6.5	3.4	0.4	7.3	6.7	0.3	13.8	0.7	0.9	9.6	1.0	0.1
Backswimmer	*Notonectidae*[b]										0.4		
Caddisfly larva	*Trichoptera*[c]			0.6	0.3	0.4	2.8	6.2		5.9	4.0		1.2
Crane fly larva	*Tipulidae*[b]		0.1			0.1				0.1			
Crayfish	*Astacidea*[c]	0.3	0.1		1.0		0.1	0.7	0.3		0.1	0.1	
Damselfly larva	*Zygoptera*[c]		0.1	0.1		0.1	0.1					0.3	0.7
Freshwater clam	*Veneroida*[c]							3.7	0.3		1.2	0.3	
Horsehair worm	*Nematomorpha*[e]		0.4	0.6		0.1	0.3		2.5	5.6		2.1	0.3
Leech	*Hirudinea*[d]							0.6					
Lunged snail	*Fisherola*[a]	0.1	0.1		0.1			0.1	0.1	0.1		0.7	
Mayfly larva	*Ephemeroptera*[c]	1.9	0.4	0.1	3.0	10.4	4.4	8.0	0.9	0.1	5.5	1.5	1.3
Midge	*Nematocera*[c]	0.1	0.9		3.0	1.2		6.2			5.0	0.1	
Predaceous diving beetle	*Dytiscidae*[b]		0.1	0.6			0.1	1.0			0.1	0.1	0.1
Planarian	*Dugesiidae*[b]					0.1	0.1	0.1			0.1		
Riffle beetle larva	*Elmidae*[b]		0.1		0.7	1.0	0.1	1.3	0.3		0.4	0.4	
Scud	*Amphipoda*[e]	0.4	0.7	2.7	0.7	1.3	0.7	0.4	0.3		0.1	0.4	
Soldier fly larva	*Stratiomyidae*[b]							0.1			0.1		
Sowbug	*Isopoda*[c]					0.1							
Stonefly larva	*Plecoptera*[c]				0.1	0.6		0.3			0.1		
Water boatman	*Corixidae*[b]		0.1							0.1			
Water bug	*Nepomorpha*[b]				0.1								
Water flea	*Cladocera*[d]	0.1						0.1					
Water penny beetle	*Psephenidae*[b]				0.1	0.6		0.3			0.3	0.1	0.1
Water snipe fly larva	*Athericidae*[b]					0.7			0.3				

[a]Genus
[b]Family
[c]Order (including infra-, sub-)
[d]Class (including sub-)
[e]Phylum

construction, and 2.8 ± 0.8 after construction (df = 2, F = 46.73, $p < 0.001$; Fig. 4b). This is equivalent to a − 61.6% decrease. After construction, taxon richness had become similar across the four sites (df = 3, F = 1.11, $p = 0.370$).

Site A had a lower Shannon-Weaver species diversity compared to the other sites before construction (df = 3, $F = 5.71$, $p = 0.005$; Table 1); however, this diversity increased during construction (Fig. 4c) and became more similar across all four sites during (df = 3, $F = 1.71$, $p = 0.197$) and after construction (df = 3, $F = 2.19$, p = 0.125; Table 2). Species diversity did not change significantly with time at site B and site D ($p > 0.05$). The species diversity trend at site C was opposite to that observed at site A, declining from 1.3 ± 0.2 before construction to 0.9 ± 0.1 during construction and 0.7 ± 0.1 after the construction (df = 2, $F = 5.24$, $p = 0.020$; Fig. 4c).

Discussion

Increased turbidity during construction, improved DO after construction

The stream diversion created before the start of flow improvement work in the North Branch Park River accelerated the water flow around the site, causing increased sediment detachment and turbidity during construction. A secondary cause of increased turbidity was the materials used for construction, such as coarse sand. Even during low intensity rainfall, these materials were rapidly carried off into the stream by surface runoff generated on paved surfaces (Meyer et al. 2005). After the work was completed, rerouting the stream to its original flow path stirred up even more sediment. Although water samples were highly transparent, turbidity approached the upper limit of class AA/A/B water

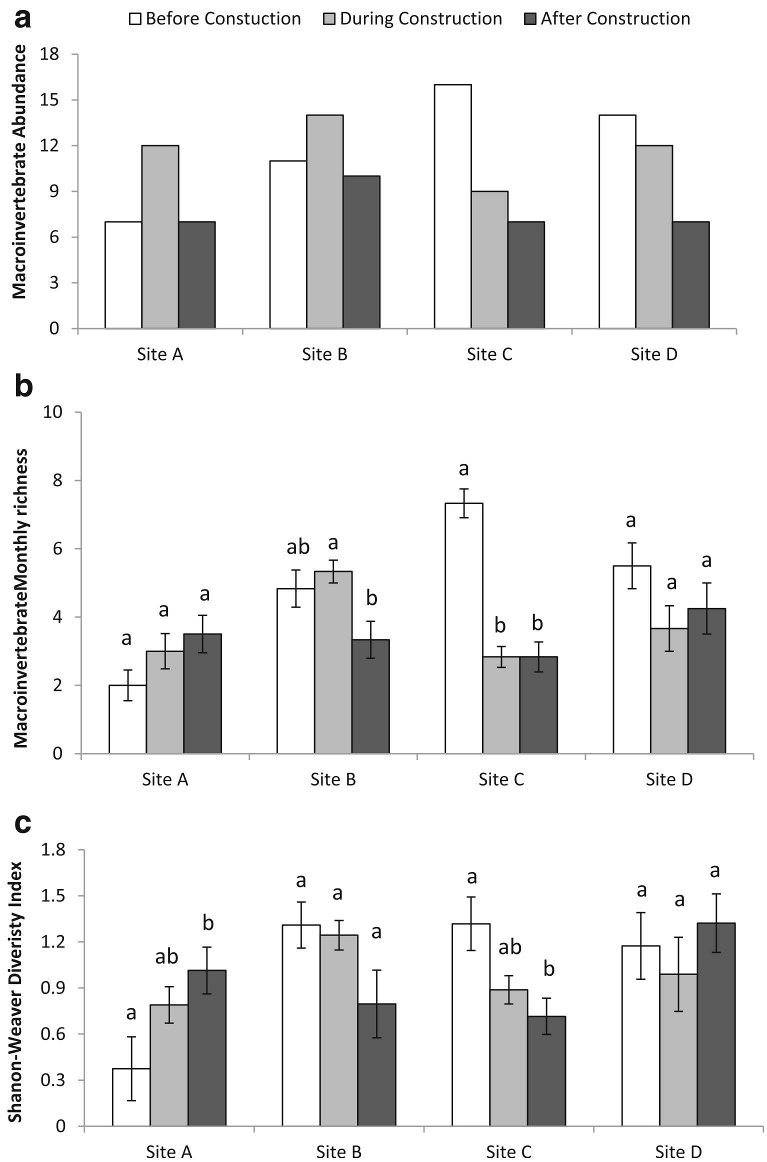

Fig. 4 Taxon abundance (**a**), monthly mean taxon richness (**b**), and Shannon-Weaver Diversity index (**c**) at the four study sites in the Park River. Data in panels **b** and **c** were represented as mean ± 1 SE. The (a, b) markings above the columns indicate the groups identified with the Tukey test ($\alpha = 0.05$)

quality standards in Connecticut (5 NTU over ambient levels; State of Connecticut Department of Energy and Environmental Protection 2013 regulations for Clean Water Act purposes). The increase in DO at the modified site indicates that vertical mixing, and likely also horizontal mixing, improved now after the water flow was no longer confined to culverts.

Limited impact on nitrate and TP

Changes in nitrate and TP were similar across the four sampling sites, and no evidence was found pointing to a link between nutrient concentrations and hydromodification at site C. Although not evaluated, the changes in nutrients are likely connected to changes in nitrate and TP input attributed to combined sewer overflows (Fuss and O'Neill 2010). Input variations can result from a variety of temporal patterns in precipitation, rain storm frequency, snow accumulation, and snow melt. Given the small size and urban character of the watershed, in-stream conversion of nutrients has a much smaller impact on nutrient concentrations than input variations. The lack of evidence for hydromodification impact on in-stream nutrient concentrations is in line with studies involving larger scale bridge construction projects (Purcell et al. 2012).

Decreased taxon richness and species diversity

Increased turbidity was accompanied by a decline of taxon richness and species diversity at the construction site, but not further downstream. The 61.8% decline in taxon richness immediately downstream of the construction site (site C) was substantial, and the benthic macroinvertebrates that are sensitive to minor changes in water quality disappeared altogether (Covich et al. 1999). Nonetheless, construction did not appear to affect taxon richness and diversity further downstream (at site D), while taxon abundance recovered within the year. The local baseline of taxon richness and species diversity is, however, dependent on local environmental factors. The greater taxon richness and species diversity for sites downstream of the construction site vs. upstream sites has no link to hydromodification, but is explained by the type of streambed material. Benthic macroinvertebrates prefer habitats with fine sediments, including sand, to rocky streambeds (Quinn and Hickey 1990). Site C and site D downstream of the construction site have sandy streambed material and therefore generally greater taxon richness and species diversity compared to upstream site A and site B that have rock and gravel stream beds, respectively.

Likewise, the regression in water quality ascribed to the flow improvement work was local and short-lasted. Increased turbidity and decreased taxon richness and species diversity was most significant at site C, limiting the negative impacts of this small-scale construction project to a distance of less than 50 m downstream of the site where work was conducted. The effects further downstream showed signs of recovery within the timespan of 1 year; however, the increase in local flow velocity under the bridge is permanent. Therefore, this location may not recover to pre-construction taxon richness and species diversity but instead reach a dynamic equilibrium corresponding with levels observed at upstream sites.

Conclusions

Flow improvement work was carried out in the urban North Branch Park River, where a stream crossing in the form of an embankment with eight culverts was replaced with a bridge (25-m span). This small-scale hydromodification affected the water quality as follows:

(1) Turbidity increased by 60.9% within 50 m downstream of the construction site and returned to the pre-construction level after the work was completed. During construction, turbidity approached the maximum level for clean water of 5 NTU over ambient levels (State of Connecticut Department of Energy and Environmental Protection 2013 regulations for Clean Water Act purposes). Dissolved oxygen increased locally during the last stage of our survey, indicating that hydromodification resulted in better vertical mixing.

(2) There was no evidence that hydromodification affected nitrate and TP concentrations in stream water. Respective changes were similar at upstream and downstream sites and therefore most likely caused by variations in nutrient input associated with combined sewer overflows.

(3) Benthic macroinvertebrate taxon richness and species diversity declined by 61.6 and 32.6%, respectively, during construction. This negative effect on the aquatic ecosystem was limited to within 50 m downstream of construction. But neither taxon richness nor species diversity recovered within 1 year after flow improvement was completed. Nonetheless, benthic macroinvertebrate communities may yet recover given that turbidity subsided and DO increased in the year following construction.

Urban aquatic ecosystem and water quality-related services are very sensitive to the impacts of small-scale hydromodification projects, and our results underline the importance of environmental assessments prior to modification of flow structures.

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Availability of data and materials
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Authors' contributions
All authors helped review and revise the manuscript. BZ was the project leader. BZ, DSS, and APB designed the study, conducted the data analysis, and wrote the manuscript. DSS, APB, DMR, BMK, MLY, and ASP collected and processed the data. NRB created the maps and provided project guidance and expertise. All authors have read and approved the final manuscript.

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Competing interests
The authors declare that they have no competing interests.

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