An Assessment of In-stream Sampling Activity of Macroinvertebrates and Stream Channelization in Dug Run, Allen County, Ohio, USA

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ABSTRACT. Macroinvertebrates are good indicators of stream quality. Changes in populations of sensitive macroinvertebrates help to show stressors to the stream. Student sampling of a section of Dug Run in northwestern Ohio has occurred since 2015. This work has been to identify how changes on the campus including construction, tree removal, and channelization may be impacting stream macroinvertebrates. Student sampling, however, also causes disturbances that may negatively impact macroinvertebrate populations. A break in student sampling—due first to the use of an adjacent off-site location in 2019 and then to COVID-19 beginning in 2020—was expected to impact the number of mayfly nymphs and caddisfly larvae captured, both considered sensitive macroinvertebrates in the stream. To measure the impact of channelization in Dug Run, the study area was split into a channelized reach, an upstream reach, and a downstream reach. Stream habitat was also studied in each reach with macroinvertebrates collected from riffles, undercuts, and pools. After a break in sampling, caddisfly larvae increased initially but have declined in the 2 following years, while mayfly nymphs increased in the last 2 years of the study. No significant differences were found in stream quality monitoring (SQM) index scores between the channelized reach compared to upstream and downstream reaches ($H = 4.15$; $p = 0.126$). There was a significant difference in taxa richness among pools, riffles, and undercuts ($H = 14.09$; $p < 0.001$). A significant difference was also found in the moderately sensitive macroinvertebrates captured in riffles between the channelized, upstream, and downstream reaches ($H = 6.82$; $p = 0.033$). A break in sampling resulted in an initial increase in mayfly nymph and caddisfly larvae samples, but it appears a variety of factors may be responsible for the numbers captured. The channelized reach had higher numbers of scuds and crayfish in riffles among the 3 reaches, which may be the result of a change in their distribution related to lack of undercuts. Both scuds and crayfish were found in significantly greater abundance in undercuts compared to pools and riffles.

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INTRODUCTION

Dug Run is a 9.5 km long 1st order tributary of the Ottawa River in Allen County, Ohio, that flows east to west through the southern portion of the University of Northwestern Ohio. Student sampling of fish and macroinvertebrates within Dug Run began in 2015 to compare the impacts of differing stormwater runoff management practices at the university (Zuwerink et al. 2020). Sampling occurred on a quarterly basis until 2019 when 2 quarters were spent on private land adjacent to the university, just downstream of the study area. In addition, sampling was discontinued during the COVID-19 pandemic due to remote classes during the initial COVID-19 outbreak and COVID-19 protocols when students returned to campus. Fish and macroinvertebrate sampling resumed in summer 2021, but only fish were sampled quarterly while macroinvertebrates were only sampled during the summer on an annual basis. During March 2022, a section of Dug Run on the western portion of campus was channelized. The stream was divided into an upstream reach above the channelized section, a channelized reach where the stream was straightened, and a downstream reach that had been previously channelized and was affected by sediment movement from the channelized section (Fig. 1). Data was collected from each reach during summer 2022.

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Stream disturbance has affected macroinvertebrate populations worldwide. Macroinvertebrate assemblages have been shown to be negatively affected by urbanization (Walsh et al. 2001; Stepenuck et al. 2002; Gál et al. 2019), agriculture (Genito et al. 2002; Hepp et al. 2010; Wang et al. 2019), channelization (Griswold 1978; Blake and Rhanor 2020), fire (Rust et al. 2019), macroinvertebrate sampling (Bossley and Smiley 2019), recreation (Hardiman and Burgin 2011), introduction of non-native species, severe weather, runoff, and erosion (Jackson and Fuereder 2006). In Dug Run, macroinvertebrate assemblages were more impacted in an urban section compared to a non-urban section managed for stormwater; however, over the course of the study there was a decline in stream quality monitoring (SQM) index scores in the reach managed for stormwater (Zuwerink et al. 2020). It appeared that sensitive macroinvertebrates had been in decline since the study began in 2015 (Zuwerink personal observation). This decline was suspected to be related to macroinvertebrate sampling, removal of trees along the bank, and/or natural cycles in insect populations.

Reductions in macroinvertebrate abundance and richness in stream riffles has been found due to student sampling (Bossley and Smiley 2017, 2019). Creating a disturbance to simulate sampling can also change macroinvertebrate communities. Doeg et al. (1989) found disturbance by kicking and raking reduced macroinvertebrate species number, species density, and individuals, but macroinvertebrates recovered by 8 days during the summer and 71 days during the winter. McCabe and Gotelli (2000) found differences in density and species richness of macroinvertebrates when they artificially created disturbance using different brushes related to frequency, intensity, and area; however, while they found lower abundance in disturbed sections, species richness increased in disturbed areas. Impacts of disturbance can be variable depending on the amount of disturbance, type of disturbance, time of year, influence of that disturbance on competitive interactions, sources of macroinvertebrates, and relative impact of the surrounding habitat.

The western section of the University of Northwestern Ohio campus, along Dug Run, has been influenced by a number of factors including channelization, construction, loss of trees, constant mowing, and student sampling. The area upstream of the sampling location is impacted by urbanization (Zuwerink et al. 2020) and the relative lack of macroinvertebrate drift from upstream may influence recovery time. Wallace (1990) indicated recolonization by macroinvertebrates may occur by migration from deeper hyporheic zones, upstream movements, downstream drift, and aerial recolonization by adults. Williams and Hynes (1976) found drift accounted for the greatest source of colonization, followed by aerial recolonization by

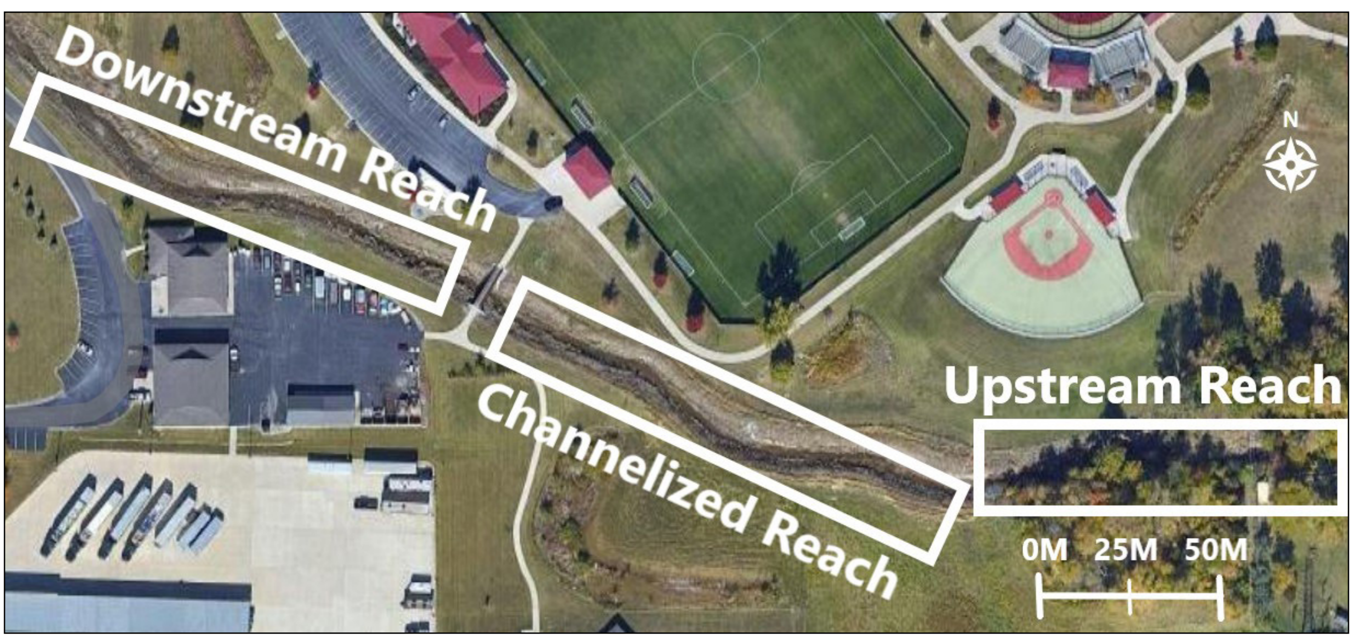

FIGURE 1. Sections of Dug Run sampled during summer 2022 after a section was channelized during March 2022. Google® maps of University of Northwestern Ohio, retrieved July 24, 2023.

adults. Recolonization in Dug Run may be more affected by aerial recolonization due to lack of macroinvertebrate drift on the campus study area.

Channelization has been found to negatively impact streams by altering stream flow, altering substrate composition, removing cover, reducing habitat diversity (Wheeler et al. 2005), and reducing habitat heterogeneity (Lau et al. 2006). Channelization also negatively impacts macroinvertebrates (Griswold 1978; Rohasliney and Jackson 2008). In some cases, macroinvertebrate recovery from channelization has been rapid (McCarthy 1981; Brooker 1985); however, Blake and Rhanor (2020) found channelization lowered taxa richness and macroinvertebrate biotic index values, and these effects could persist for many years.

The goal of the current study was to assess how macroinvertebrate populations varied on the study site due to university construction on the western half of campus, for example the Field of Dreams (a handicap accessible baseball field) which was constructed in 2017 near the stream. Other disturbance events included the removal of trees in 2016, channelization in 2022 (with the intention to quickly move storm water downstream and stabilize the bank), and the impact of student sampling. The off-site sampling downstream of the study area in 2019, and the COVID-19 pandemic that resulted in a break in sampling for over a year, provided an opportunity to see how macroinvertebrate populations changed. Because sensitive macroinvertebrates are likely to be most impacted by disturbance, it was expected that changes to the SQM index score would be tied to the capture rates of the most abundant sensitive species. In Dug Run, caddisfly larvae and mayfly nymphs were the most abundant sensitive macroinvertebrates captured from 2015 to 2018 (Zuwerink et al. 2020). With a reduction in student sampling due to both off-site collection in 2019 and COVID-19, which prevented sampling for several quarters, it was hypothesized that there would be an increase in the populations of caddisfly larvae and mayfly nymphs. It was also hypothesized that the recently channelized (2022) reach would have lower SQM index scores than areas upstream and downstream of that reach, and that channelization would negatively impact the recovery of caddisfly larvae and mayfly nymphs.

METHODS AND MATERIALS

The data in the current study was collected from Dug Run beginning in summer 2015 and ending in summer 2022. All samples were collected by students, and generally 2 samples were collected in 1 class period during the quarter—which consisted of an hour of sampling. Samples were collected at the beginning of class and students spent most of the time searching for macroinvertebrates and placing them in trays with water. In summer 2021 and 2022, students collected 3 samples per class period—which consisted of just under 2 hours of sampling per class—and sampling occurred over multiple class periods. Sampling occurred on a quarterly basis from fall 2015 to spring 2019. Fall samples were collected in September, winter samples in January, spring samples in April, and summer samples in July. During summer and fall 2019, macroinvertebrates were collected just downstream of the study site. This was because few sensitive species of macroinvertebrates had been captured in previous quarters on campus, and it provided students an opportunity to sample less disturbed stream habitats. Samples were collected again in winter 2020 on the study site, but no more sampling occurred until summer 2021 because of the COVID-19 pandemic. In summer 2021 and 2022, macroinvertebrates were sampled once for the entire year; however, fish sampling continued on a quarterly basis on the study site.

In addition, 3 reaches (upstream, channelized, and downstream) were created in summer 2022 to determine if channelization during March 2022 affected the macroinvertebrate populations. The length of each reach was measured in Google® maps. The upstream reach contained a large pool and some smaller pools, along with riffles and runs, and measured 100 m long. The area is surrounded by woods except for 50 m on the north side of the downstream end, which is mowed. There were some well-developed undercuts along with some newly formed undercuts related to erosion along the bank in the grassy area. The channelized reach contained only riffles and runs and measured 120 m long. After channelization, there was a lot of sediment in the stream and erosion along the banks (Fig. 2). The area surrounding the channelized reach was regularly mowed grass with some buildings nearby. Three pools were eliminated during channelization and all undercuts had been removed, although

FIGURE 2. Channelized section of Dug Run showing sediment deposited and erosion along the bank, April 2022

more riffles formed within the channelized reach. After 4 months, 2 undercuts had reformed in the channelized reach, but no pools had reformed. The downstream reach contained riffles and runs, with small pools reforming since the recent channelization occurred upstream, and measured 180 m long. The area surrounding this reach was regularly mowed grass with some buildings nearby. This section had been straightened at some point in its history with riffles, pools, and well-developed undercuts having reformed prior to the channelization just upstream of this reach. The qualitative habitat evaluation index (QHEI) was used to assess habitat quality in each reach (Rankin 1989).

A total of 16 macroinvertebrate kick-seine (90 cm high × 115 cm wide × 800 µm mesh) samples were collected from riffles: 5 riffles in the upstream reach, 6 riffles in the channelized reach, and 5 riffles in the downstream reach. Characteristics of each stream riffle composition were visually estimated using the categories of boulder (>254 mm), cobble (50.8 to 254 mm), gravel (6.35 to 50.8 mm), sand (0.75 to 6.35 mm), and silt (<0.75 mm). The average depth of the riffle was determined by taking 5 depth measurements across the width of the riffle and averaging them. The width of the riffle was determined using a meter tape and measuring from shoreline to shoreline at the center of the riffle. The length of the riffle was determined based on how far the stream bed altered the surface water of the stream. Sampling was initiated from the furthest downstream riffle and continued upstream to ensure sampling did not inadvertently add macroinvertebrates through drift to unsampled areas.

Kick-seine samples were collected by 2 students in each riffle. The kick-seine was placed on the downstream end of the riffle; one person would use their boots to stir up the sediment from a 1 m × 1 m section of the riffle upstream of the net. After the sediment settled out, the net was lifted, rolled up, and taken to a cloth sheet where it was unrolled. Nets were checked for organisms and placed in trays filled with water for identification. Once macroinvertebrates were collected and identified, the nets were lifted and the sheets were checked for organisms that moved through the net. Students identified macroinvertebrates using the *Identification Guide to Freshwater Macroinvertebrates* (Gill 1998) and the *Key to Macroinvertebrate Life in* *the River* (University of Wisconsin [date unknown]). Identification was confirmed by Dr. Zuwerink or Beth Seibert. Most taxa were individually counted, but the numbers of highly abundant taxa—such as midge larvae, aquatic worms, and planaria—were estimated by counting the number found in a smaller area and multiplying that count based on total area. After macroinvertebrates were identified and counted, they were released back into the stream from where they were collected. Macroinvertebrate SQM index scores were calculated using the Ohio EPA stream quality assessment form (Kopec and Lewis 1983). SQM index scores were calculated for kick-seine samples.

In addition, 3 habitats were sampled using a D-frame net (550 µm mesh): pools (n = 5), riffles (n = 8), and undercuts (n = 7) working from downstream to upstream areas in all 3 reaches. Samples from the downstream reach included 3 riffles, 3 pools, and 3 undercuts; the channelized reach included 3 riffles and 2 undercuts; and the upstream reach included 2 riffles, 2 pools, and 2 undercuts. There were no pools present in the channelized reach to sample. To capture macroinvertebrates in riffles, the D-frame net was set in the downstream section of the riffle and one person would use their boot to stir up the streambed in the riffle upstream of the net. After the sediment settled out, the net was brought up to a cloth sheet and checked for organisms by the same procedure used for the kick-seine. Since pools do not generate enough current, the net was placed downstream and a current was generated by a person kicking with their boot, which added more sediment into the net. The undercuts had a week current, slower than the riffle, so an additional current was generated by a person kicking into the root masses with the top of their boot to try to dislodge macroinvertebrates and get them to flow into a net placed on the downstream section of the undercut. Taxa richness was used to calculate differences between the 3 habitats.

Due to low sample sizes, the Kruskal-Wallis H test was used to analyze differences in SQM index scores as well as numbers of each type of macroinvertebrate collected between the upstream reach, the channelized reach, and the downstream reach . Again, because of low sample sizes and non-normality of data, the Kruskal-Wallis H test was used to analyze differences in both the taxa richness by habitat and the numbers of macroinvertebrates captured in each habitat using the D-frame net.

RESULTS

The average streambed composition of the riffles in the upstream reach was 4% silt, 9% sand, 67% gravel, 19% cobble, and 1% boulder. The average riffle depth was 6 cm with a width of 3.8 m and a length of 4.6 m. The QHEI value was 55. The average streambed composition of the riffles in the channelized reach was 8.3% silt, 8.3% sand, 66.7% gravel, 15.8% cobble, and 0.8% boulder. The average riffle depth was 7.8 cm with an average width of 2.5 m and length of 4.8 m. The QHEI value was 32 with low stream cover scores. The average streambed composition of riffles in the downstream reach was 33.8% silt, 18.8% sand, 31.3% gravel, 10% cobble, and 6.3% boulder. The average depth was 5.8 cm with an average width of 1.6 m and length of 2.8 m. The QHEI value was 32 and there was extensive embeddedness of the riffles.

While SQM index scores were higher in the channelized reach (Fig. 3), there was no significant difference among the SQM index scores of riffles between the upstream reach, channelized reach, and downstream reach ($H = 4.15$; $p = 0.126$). There was a significant difference in moderately sensitive macroinvertebrates among the riffles in the 3 reaches ($H = 6.82$; $p = 0.033$) with more taxa found in the channelized reach. Mayflies were more frequently captured in the downstream reach, while caddisflies were more frequently captured in the channelized reach (Table 1). The greatest number of caddisflies were captured in the most upstream riffle of the channelized reach. No sensitive macroinvertebrates were found in 2 of 6 riffles sampled in the channelized reach while no sensitive macroinvertebrates were found in 1 of 5 riffles sampled in the upstream reach; however, all 5 riffles sampled in the downstream reach contained sensitive macroinvertebrates.

A decrease in mayfly nymphs and caddisfly larvae per riffle was observed from 2015 through 2019, when sampling was done quarterly (Fig. 4). After a break in sampling, there was an increase in caddisfly larvae captured in winter 2020; however, despite a break in sampling from COVID-19, caddisfly larvae numbers did not increase after the break. Mayfly nymph numbers increased after a break in sampling due to COVID-19. The percentage of riffles sampled that contained mayfly nymphs or caddisfly larvae increased after a break in macroinvertebrate sampling on the study site (Fig. 5).

There was a significant difference in taxa richness among riffles, pools, and undercuts (H = 14.09; $p < 0.001$). Moderately sensitive taxa were more frequently captured in undercuts compared to riffles and pools with the D-frame net (Table 2).

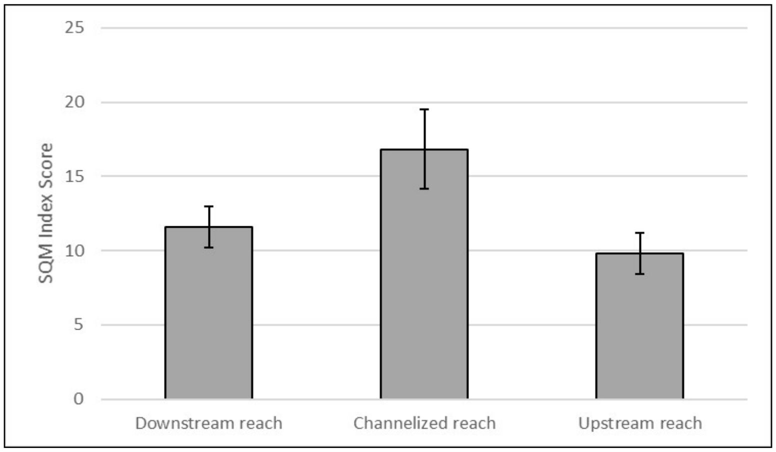

FIGURE 3. Comparison of mean stream quality monitoring (SQM) index scores among 3 reaches of Dug Run, northwestern Ohio, during summer 2022 (error bars represent SE)

Table 1
Comparison of specimens captured per sample using a kick-seine in Dug Run, northwestern Ohio, during summer 2022. Numbers in parentheses represent SE.

Macroinvertebrate	Reach sampled		
	Upstream	Channelized	Downstream
Mayfly nymph (Ephemeroptera) [a]	0.20 (0.20)	1.33 (0.49)	7.60 (2.58)*
Caddisfly larvae (Trichoptera) [a]	1.60 (0.68)	3.00 (1.55)	0.00 (0.00)
Riffle beetle (Elmidae) [a]	0.00 (0.00)	0.33 (0.21)	0.40 (0.24)
Damselfly nymph (Zygoptera) [b]	0.20 (0.20)	0.33 (0.21)	0.00 (0.00)
Dragonfly nymph (Anisoptera) [b]	0.00 (0.00)	0.00 (0.00)	0.06 (0.20)
Crane fly larvae (Tipulidae) [b]	2.40 (0.93)	2.17 (0.79)	0.80 (0.58)
Beetle larvae (*Berosus*) [b]	0.00 (0.00)	0.83 (0.31)	0.00 (0.00)
Crayfish (Decapoda) [b]	0.00 (0.00)	1.33 (0.42)	0.00 (0.00)*
Scuds (Amphipoda) [b]	0.80 (0.58)	4.50 (1.18)	0.40 (0.24)*
Clams (*Ferrissia*) [b]	14.20 (8.62)	5.17 (1.96)	4.00 (1.18)
Blackfly larvae (Simuliidae) [c]	4.40 (3.92)	0.83 (0.31)	0.80 (0.37)
Aquatic worms (Clitellata) [c]	56.00 (11.66)	91.67 (23.86)	56.00 (11.66)
Midge larvae (Diptera) [c]	170.00 (20.00)	158.33 (20.07)	146.00 (46.00)
Leeches (Hirudinea) [c]	2.00 (1.14)	6.33 (0.95)	2.40 (0.68)*
Planaria (Turbellaria) [c]	60.00 (10.00)	95.00 (43.03)	80.00 (33.02)

[a] Sensitive to pollutants (sensitivities derived from Kopec and Lewis 1983).
[b] Somewhat sensitive to pollutants.
[c] Very tolerant to pollutants.
* Significant difference among reaches ($p < 0.05$).

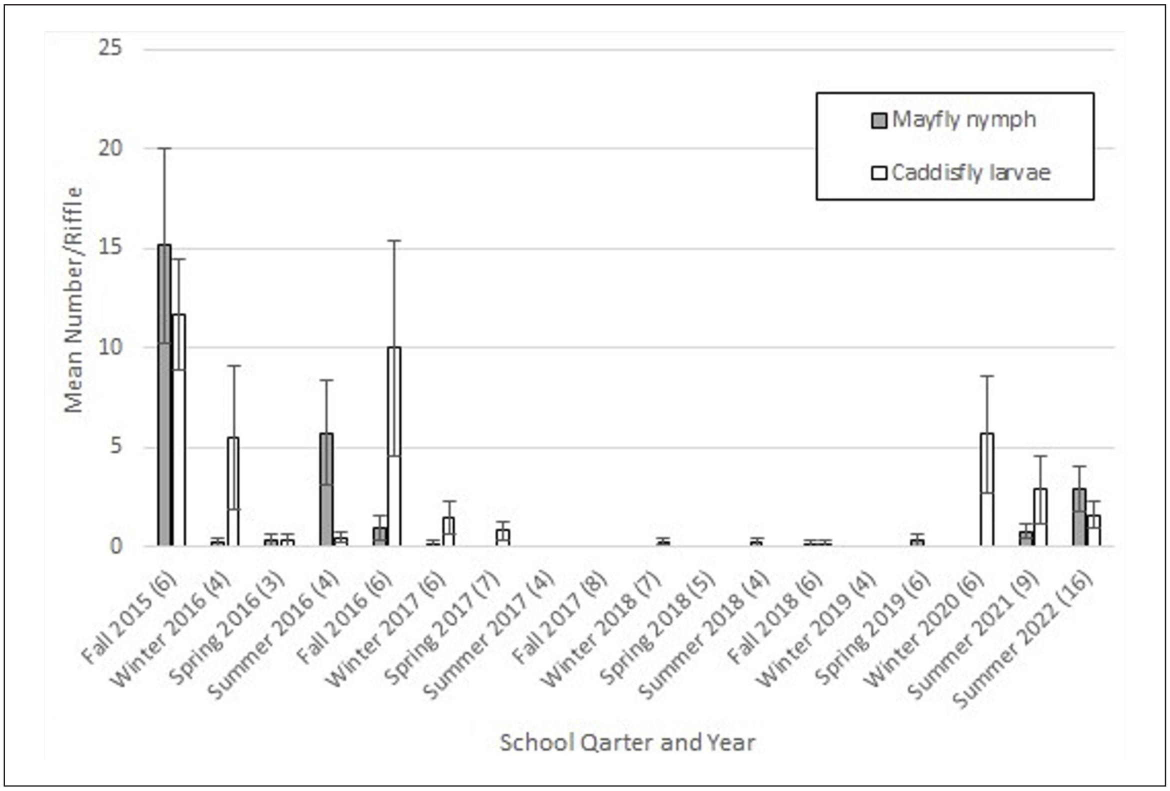

FIGURE 4. Mean number of mayfly nymphs and caddisfly larvae captured per riffle in Dug Run (error bars represent SE). Samples were collected quarterly from fall 2015 to spring 2019. Classes collected samples off-site during summer 2019 to fall 2019. No sampling occurred from spring 2019 to spring 2020 due to COVID-19. Macroinvertebrates were only collected during summer quarters of 2021 and 2022. Channelization of a reach of the study area occurred in March 2022. The number of samples collected is in parentheses after the collection quarter.

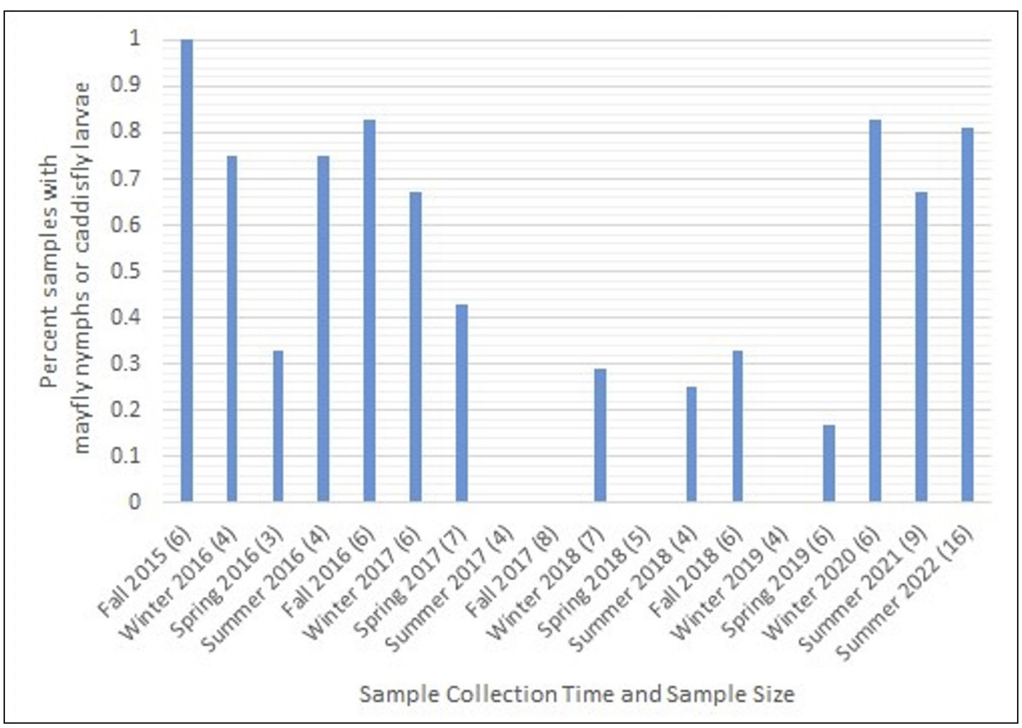

FIGURE 5. Percentage of riffles sampled with mayfly nymphs or caddisfly larvae from fall 2015 to summer 2022. The number of samples collected is in parentheses after the collection quarter.

Table 2
Comparison of specimens captured per sample using a D-frame net in Dug Run, northwestern Ohio, during summer 2022. Numbers in parentheses represent SE.

Macroinvertebrate	Habitat		
	Pool	Riffle	Undercut
Mayfly nymph (Ephemeroptera) [a]	0.00 (0.00)	0.50 (0.27)	0.14 (0.14)
Caddisfly larvae (Trichoptera) [a]	0.00 (0.00)	0.50 (0.19)	0.00 (0.00)
Damselfly nymph (Zygoptera) [b]	0.00 (0.00)	0.00 (0.00)	5.86 (2.09)*
Dragonfly nymph (Anisoptera) [b]	0.00 (0.00)	0.00 (0.00)	0.86 (0.40)
Crane fly larvae (Tipulidae) [b]	0.00 (0.00)	1.00 (0.76)	0.14 (0.14)
Beetle larvae (*Berosus*) [b]	0.00 (0.00)	0.00 (0.00)	0.14 (0.14)
Crayfish (Decapoda) [b]	0.00 (0.00)	0.13 (0.13)	5.14 (0.63)*
Scuds (Amphipoda) [b]	0.00 (0.00)	0.13 (0.13)	31.86 (11.01)*
Clams (*Ferrissia*) [b]	4.20 (2.18)	4.13 (1.51)	2.00 (0.44)
Blackfly larvae (Simuliidae) [c]	0.00 (0.00)	0.38 (0.38)	0.00 (0.00)
Aquatic worms (Clitellata) [c]	6.40 (2.66)	22.63 (5.83)	13.86 (6.62)
Midge larvae (Diptera) [c]	0.00 (0.00)	51.25 (21.58)	65.71 (17.30)*
Pouch snails (Gastropoda) [c]	0.00 (0.00)	0.00 (0.00)	0.14 (0.14)
Leeches (Hirudinea) [c]	0.00 (0.00)	1.88 (0.35)	2.29 (1.49)*
Planaria (Turbellaria) [c]	0.20 (0.20)	15.83 (4.53)	77.57 (22.86)*

[a] Sensitive to pollutants (sensitivities derived from Kopec and Lewis 1983).
[b] Somewhat sensitive to pollutants.
[c] Very tolerant to pollutants.
* Significant difference between habitats ($p < 0.05$).

DISCUSSION

Macroinvertebrate sampling disturbance may have negatively impacted caddisfly larvae in the study area as numbers improved after a break in sampling during 2019, although there was no improvement in mayfly nymph numbers. However, a longer break in sampling due to COVID-19 resulted in greater numbers of mayfly nymphs collected, but caddisfly larvae numbers decreased from 2020 to 2022. While sampling disturbance may have contributed to some of the decline observed in caddisfly larvae and mayfly nymphs over the first 3 years of the study, other factors could have also contributed to their decline: for example tree removal (that could have affected stream habitat) and aerial recolonization by adult macroinvertebrates. It is difficult to determine how much affect sampling disturbance had since the study was not specifically designed to test the impact of sampling disturbance. Robinson and Minshall (1986) found that increasing frequency of disturbance negatively impacted species richness and density of invertebrates in a stream, and the effect of disturbance was seasonal. While samples were collected in all 4 seasons during this study, it is difficult to determine if seasonal disturbance negatively impacted macroinvertebrates, and different species may respond differently to those disturbances. Sampling during the winter has been shown to negatively impact macroinvertebrate recovery (Doeg et al. 1989). Macroinvertebrate populations in Dug Run did not appear to be strongly impacted by the channelization as SQM index scores were higher in the channelized reach than the upstream and downstream reach. While channelization greatly impacted habitat—which could be seen in the removal of pools, undercuts, root wads, and woody debris—macroinvertebrate populations quickly colonized the new riffles that had formed in the channelized reach. Collier and Quinn (2003) found taxa-specific recovery rates, but those recovery rates were influenced by site characteristics. They attributed those differences to trophic resources, niche availability, and changes in community structure.

The recent channelization in Dug Run resulted in finer sediments within the downstream reach where there was a greater capture rate of mayfly nymphs. Holomuzki and Messier (1993) found greater densities of mayflies in coarser substrate. This differed from what was found in the downstream reach on the University of Northwestern Ohio campus, where there was extensive embeddedness of riffles. This may be due to the recent impacts of channelization or different life histories of the mayflies present in Dug Run. Different mayfly groups have been found to prefer different habitats depending on their life history traits (Vilenica et al. 2018). Another possibility is aerial recolonization by adults as the downstream reach of Dug Run is closer to better habitat adjacent to campus. Caddisfly larvae were also found to differ in habitat preference depending on species, with most species preferring coarser substrate (Urbanič et al. 2005). No caddisfly larvae were detected in the downstream reach of Dug Run where riffles were composed of finer sediment.

While it was predicted there would be lower SQM index scores in the channelized reach of Dug Run—as the disturbance of channelization was expected to have the greatest impact on sensitive macroinvertebrates—the greater SQM index scores actually observed in the channelized area appears to be tied to greater capture rates of moderately sensitive macroinvertebrates. Channelization has been found to impact habitat diversity (Wheeler et al. 2005; Lau et al. 2006). In addition, channelization generally results in the loss of environmentally sensitive species (Lau et al. 2006). In the current study, channelization resulted in the initial elimination of pools and undercuts. Undercuts appeared to be a preferred habitat for some moderately sensitive macroinvertebrates such as crayfish, damselfly nymphs, and scuds that were found in greater abundance on the study site. While 2 new undercuts were forming in the channelized area, the lack of this habitat may explain the greater abundance of scuds and crayfish found in the channelized reach compared to the upstream and downstream reach. Crayfish and scuds both had significantly larger numbers in undercuts compared to riffles. Differences have been found in the abundance of macroinvertebrates sampled from riffles, pools, and undercuts although these differences were seasonal (Rhodes and Hubert 1991). Channelized areas have been found with fewer undercuts and pools, and alterations in habitat were attributed to declines in certain fish species and aquatic invertebrates (Lennox 2012). In this Dug Run study, channelization appears to have affected numbers of certain taxa as a result of habitat alteration, including riffle substrate composition and loss of undercuts.

Conclusions

Considering the sections of Dug Run along the University of Northwestern Ohio campus is already under stress from tree removal in 2016, frequent mowing, and proximity to the urbanized section of the stream, this full study area would not be expected to be as resilient as a more natural area. The decline in SQM index scores observed in this stream (Zuwerink et al. 2020) may have been a combination of seasonal factors along with the decline in sensitive macroinvertebrates such as the caddisfly larvae and mayfly nymphs. While the drop in SQM index scores is noticeable from the summer of 2016 to the summer of 2017, there was much variation in the scores. The recent recovery of mayfly nymphs to the study site could be due to less sampling disturbance or to changes in stream habitat. While the recent channelization was expected to negatively impact the sensitive macroinvertebrates, the redistribution of macroinvertebrate abundances was not expected and shows the dynamic responses of macroinvertebrates after large habitat changes within the stream. One of the largest impacts of channelization appears to be the homogenization of habitat that can affect several macroinvertebrate taxa.

Further research should look at specific impacts of channelization on macroinvertebrate populations in regard to changes in stream habitat, such as pool-riffle-run sequences and a change in undercut availability. In addition, it is important to understand the role of sampling disturbance on sensitive macroinvertebrates in areas that are already under stress from urbanization pressures.

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LITERATURE CITED

Blake C, Rhanor AK. 2020. The impact of channelization on macroinvertebrate bioindicators in small order Illinois streams: insights from long-term citizen science research. Aquat Sci. 82(35):1-11. https://doi.org/10.1007/s00027-020-0706-4

Bossley JP, Smiley PC Jr. 2017. Short-term disturbance effects of outdoor education stream classes on aquatic macroinvertebrates. J Environ Prot. 8(11):1333-1353. https://doi.org/10.4236/jep.2017.811082

Bossley JP, Smiley PC Jr. 2019. Impact of student-induced disturbance on stream macroinvertebrates differs among habitat types. Sci Rep-UK. 9(1447):1-15. https://doi.org/10.1038/s41598-018-38210-1

Brooker MP. 1985. The ecological effects of channelization. Geogr J. 151(1):63-69. https://doi.org/10.2307/633280

Collier KJ, Quinn JM. 2003. Land-use influences macroinvertebrate community response following a pulse disturbance. Freshwater Biol. 48(8):1462-1481. https://doi.org/10.1046/j.1365-2427.2003.01091.x

Doeg TJ, Lake PS, Marchant R. 1989. Colonization of experimentally disturbed patches by stream macroinvertebrates in the Acheron River, Victoria. Aust J Ecol. 14(2):207-220. https://doi.org/10.1111/j.1442-9993.1989.tb01428.x

Gál B, Szivák I, Heino J, Schmera D. 2019. The effect of urbanization on freshwater macroinvertebrates–knowledge gaps and future research directions. Ecol Indic. 104:357-364. https://doi.org/10.1016/j.ecolind.2019.05.012

Genito D, Gburek WJ, Sharpley AN. 2002. Response of stream macroinvertebrates to agricultural land cover in a small watershed. J Freshwater Ecol. 17(1):109-119. https://doi.org/10.1080/02705060.2002.9663874

Gill K. 1998. Identification guide to freshwater macroinvertebrates. Avondale (PA): Stroud Water Research Center. 6p. https://sourland.org/wp-content/uploads/2020/11/MacroKey_Complete.pdf

Griswold BL, Edwards C, Woods L, Webber E. 1978. Some effects of stream channelization on fish populations, macroinvertebrates, and fishing in Ohio and Indiana. Columbia (MO): US Department of the Interior, US Fish and Wildlife Service, Office of Biological Services, National Stream Alteration Team. 64 p.

Hardiman N, Burgin S. 2011. Effects of trampling on in-stream macroinvertebrate communities from canyoning activity in the Greater Blue Mountains World Heritage Area. Wetl Ecol Manag. 19(1):61-71. https://doi.org/10.1007/s11273-010-9200-4

Holomuzki JR, Messier SH. 1993. Habitat selection by the stream mayfly *Paraleptophlebia guttata*. J N Am Benthol Soc. 12(2):126-135. https://doi.org/10.2307/1467342

Hepp LU, Milesi SV, Biasi C, Restello RM. 2010. Effects of agricultural and urban impacts on macroinvertebrates assemblages in streams (Rio Grande do Sul, Brazil). Zoologia-Curitiba. 27(1):106-113. https://doi.org/10.1590/S1984-46702010000100016

Jackson JK, Fuereder L. 2006. Long-term studies of freshwater macroinvertebrates: a review of the frequency, duration and ecological significance. Freshwater Biol. 51(3):591-603. https://doi.org/10.1111/j.1365-2427.2006.01503.x

Kopec J, Lewis S. 1983. Stream quality monitoring. Columbus (OH): Ohio Department of Natural Resources, Division of Natural Areas and Preserves, Scenic Rivers Programs. 20 p.

Lau JK, Lauer TE, Weinman ML. 2006. Impacts of channelization on stream habitats and associated fish assemblages in east central Indiana. Am Midl Nat. 156(2):319-330. https://doi.org/10.1674/0003-0031(2006)156%5B319:IOCOSH%5D2.0.CO;2

Lennox PA 3rd. 2012. Examining the impacts of stream channelization on salmonid and aquatic invertebrate communities of a fifth-order montane river [master's thesis]. [Lethbridge (AB)]: University of Lethbridge. 104 p.

McCabe DJ, Gotelli NJ. 2000. Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. Oecologia. 124(2):270-279. https://doi.org/10.1007/s004420000369

McCarthy D. 1981. The effects of arterial drainage on the invertebrate fauna and fish stocks of Irish rivers. In: O'Hara K, Barr CD. Proceedings: the second British Freshwater Fish Conference; 1981 Apr 13-15; Liverpool. Liverpool (UK): University of Liverpool.

Rankin ET. 1989. The Qualitative Habitat Evaluation Index [QHEI]: rationale, methods, and application. Columbus (OH): State of Ohio Environmental Protection Agency, Division of Water Quality Planning & Assessment, Ecological Assessment Section. 73 p. https://epa.ohio.gov/static/Portals/35/documents/QHEI_1989.pdf

Rhodes HA, Hubert WA. 1991. Submerged undercut banks as macroinvertebrate habitat in a subalpine meadow stream. Hydrobiologia. 213(2):149-153. https://doi.org/10.1007/BF00015001

Robinson CT, Minshall GW. 1986. Effects of disturbance frequency on stream benthic community structure in relation to canopy cover and season. J N Am Benthol Soc. 5(3):237-248. https://doi.org/10.2307/1467711

Rohasliney H, Jackson DC. 2008. Lignite mining and stream channelization influences on aquatic macroinvertebrate assemblages along the Natchez Trace Parkway, Mississippi, USA. Hydrobiologia. 598(1):149-162. https://doi.org/10.1007/s10750-007-9147-5

Rust AJ, Randell J, Todd AS, Hogue TS. 2019. Wildfire impacts on water quality, macroinvertebrate, and trout populations in the Upper Rio Grande. Forest Ecol Manag. 453:117636. https://doi.org/10.1016/j.foreco.2019.117636

Stepenuck KF, Crunkilton RL, Wang L. 2002. Impacts of urban landuse on macroinvertebrate communities in southeastern Wisconsin streams. J Am Water Resour As. 38(4):1041-1051. https://doi.org/10.1111/j.1752-1688.2002.tb05544.x

University of Wisconsin. [date unknown]. Key to macroinvertebrate life in the river. Madison (WI): University of Wisconsin-Extension. 1 p. Developed by the University of Wisconsin-Extension in cooperation with the Wisconsin Department of Natural Resources. https://www.bae.ncsu.edu/wp-content/uploads/2017/07/key_macro.pdf

Urbanič G, Toman MJ, Krušnik C. 2005. Microhabitat type selection of caddisfly larvae (Insecta: Trichoptera) in a shallow lowland stream. Hydrobiologia. 541(1):1-12. https://doi.org/10.1007/s10750-004-4314-4

Vilenica M, Brigić A, Sartori M, Mihaljević Z. 2018. Microhabitat selection and distribution of functional feeding groups of mayfly larvae (Ephemeroptera) in lotic karst habitats. Knowl Manag Aquat Ec. 419(17):1-12. https://doi.org/10.1051/kmae/2018011

Wallace JB. 1990. Recovery of lotic macroinvertebrate communities from disturbance. Environ Manage. 14(5):605-620. https://doi.org/10.1007/BF02394712

Walsh CJ, Sharpe AK, Breen PF, Sonneman JA. 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. Freshwater Biol. 46(4):535–551. https://doi.org/10.1046/j.1365-2427.2001.00690.x

Wang L, Gao Y, Han BP, Fan H, Yang H. 2019. The impacts of agriculture on macroinvertebrate communities: from structural changes to functional changes in Asia's cold region streams. Sci Total Environ. 676:155-164. https://doi.org/10.1016/j.scitotenv.2019.04.272

Williams DD, Hynes HBN. 1976. The recolonization mechanisms of stream benthos. Oikos. 27(2):265-272. https://doi.org/10.2307/3543905

Wheeler AP, Angermeier PL, Rosenberger AE. 2005. Impacts of new highways and subsequent landscape urbanization on stream habitat and biota. Rev Fish Sci. 13(3):141-164. https://doi.org/10.1080/10641260590964449

Zuwerink DA, Petty J, Seibert B. 2020. Effects of stormwater management and an extended culvert on stream health in Dug Run, Allen County, Ohio, USA. Ohio J Sci. 120(2):61-69. https://doi.org/10.18061/ojs.v120i2.7121