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MICROPLASTIC POLLUTION IN INDIANA'S WHITE RIVER: AN EXPLORATORY STUDY

Lindsay Hylton, J.L. Ghezzi[1] and **B. Han**: Ball State University, Department of Natural Resources and Environmental Management, 2000 University Avenue, WQ 110, Muncie, IN 47306 USA

ABSTRACT. Plastic material is now a ubiquitous source of aquatic pollution. Microplastics, tiny plastic pieces often not visible to the naked eye, are a growing environmental concern in both marine and freshwater ecosystems. While many studies have documented the abundance and danger of microplastics in global oceans, little research is available on microplastic presence and impact in riverine ecosystems. This exploratory study aims to build on the findings of recent freshwater microplastic studies by reporting on the occurrence and types of microplastic pollution found in the West Fork White River in central Indiana. Fifteen surface water samples were collected from three bridge sites along the river over a four-month period (August - November 2015) and analyzed using established NOAA laboratory methods. Analysis revealed various microplastic particle types in the White River, with synthetic fibers being the predominant type collected. A total of 146 plastic pieces were collected across all sites and collection periods, with an average microplastic concentration of 0.71 items m^{-3}. Surprisingly, there were no significant differences in microplastic concentrations among sites of differing population density. Further, a local wastewater treatment plant had no effect on particle type or abundance. These findings contribute to current gaps in microplastic research on freshwater, especially fluvial, environments. This contribution may guide researchers in better understanding the extent to which these synthetic particles are polluting U.S. surface waters as a whole.

Keywords: Microplastics, White River, freshwater ecosystems, microbeads

INTRODUCTION

Microplastics are primary- and secondary-sourced plastics smaller than 5 mm in size (Arthur et al. 2008; Ivar do Sul & Costa 2014). Primary microplastics are those manufactured to be microscopic in size for industrial and domestic use. Secondary microplastic debris can be derived from various classes of plastics that come from land-based sources, especially plastic packaging (including disposable single-use items), as well as fishing industry litter (Andrady 2011). Primary microplastics can come from microplastics used in air-blasting technology to remove rust and paint from machinery, boat hulls, and engines (Browne et al. 2007; Cole et al. 2011). Primary microplastics also include plastic beads (or spherules) from exfoliating facial cleansers, body washes, hand soaps, and toothpastes. Polyethylene pieces, or "scrubbers," have been utilized in personal care products to replace natural exfoliates, such as pumice or apricot husks (Zitko & Hanlon 1991; Gregory 1996; Fendall & Sewell 2009). After product use, these plastic pieces wash down the drain with the product and end up in city wastewater systems. Here, they can bypass removal by the initial coarse treatment screens (Derraik 2002; Vesilind 2003), potentially making their way into final effluent and sewage sludge (Fendall & Sewell 2009; Cole et al. 2011).

Synthetic textiles can contribute to microplastic pollution concerns due to possible release of fibers into sewage systems when laundered. Synthetic fibers, such as nylon, Orlon, dacron, and spandex, were first used by the textile industry over 50 years ago to supplement natural fibers such as wool, cotton, and linen (Habib et al. 1998). Aerobic or anaerobic bacteria used in sewage treatment do not readily decompose synthetic fibers, allowing them to concentrate in sewage sludge or be discharged with effluents (Habib et al. 1998). Rivers are a likely source of transport for these synthetic fibers through the aquatic environment. The application of digested and dewatered sludge as a low-grade fertilizer (a common agricultural practice), in addition to atmospheric deposition, are also likely sources of synthetic fibers throughout a watershed.

While Habib, et al. (1998) found a prevalence of synthetic fibers in sewage sludge, Rochman et al.

[1] *Corresponding author:* Jessi Ghezzi; 765-285-8845 (phone); 765-285-2606 (fax); JLGhezzi@bsu.edu.

(2015) conservatively estimates that approximately 8 trillion microbeads per day are emitted into U.S. aquatic habitats. As the issue has become more publicized by the media, especially with the recent discovery of microbead abundance in the Great Lakes (Eriksen et al. 2013; Baldwin et al. 2016; Mason et al. 2016b), some companies have pledged to remove these plastics from their "rinse-off personal care products" (Rochman et al. 2015). To ensure their removal, President Obama signed a bill, the Microbead-Free Waters Act of 2015, which required manufacturers to eliminate the pollutant from their products by 2017. However, this does not solve the accumulation of microplastics from other sources.

Despite being a likely source for microplastic flow to the ocean, the literature on riverine microplastic concentrations is minimal (Gaspari et al. 2014; Lechner et al. 2014; McCormick et al. 2014; Mani et al. 2015; Baldwin et al. 2016). With recent studies indicating microplastics in the Great Lakes (Eriksen et al. 2013; Baldwin et al. 2016; Mason et al. 2016a), it is timely to quantify microplastic types and concentrations in unstudied riverine systems, such as those within the White River in Indiana. The objectives of this exploratory study were to (i) identify and (ii) compare the abundance and types of microplastics at three sites along the West Fork of the White River in central Indiana. Microplastics were identified as fragments (broken down larger plastics), beads (spherules from personal care products, bead blasting, etc.), fibers (from synthetic textiles), films (plastic wrapping and bags), foam (foam packaging and cups), and pellets (preproduction pellets 5 mm and larger). The findings could provide information on spatial differentiation of plastics between locations along a river with varying watershed characteristics, in addition to characterizing the extent of this pollutant in an unexamined freshwater system. We hypothesized that (i) microplastics at three sites along the West Fork of Indiana's White River differed in abundance, and (ii) microplastic concentrations downstream from a waste water treatment plant and those sampled in areas with greater population density were greater than those upstream or from areas with lower population density.

METHODS

Site description.—The White River of central Indiana was chosen because there are currently no studies that have sampled for microplastics.

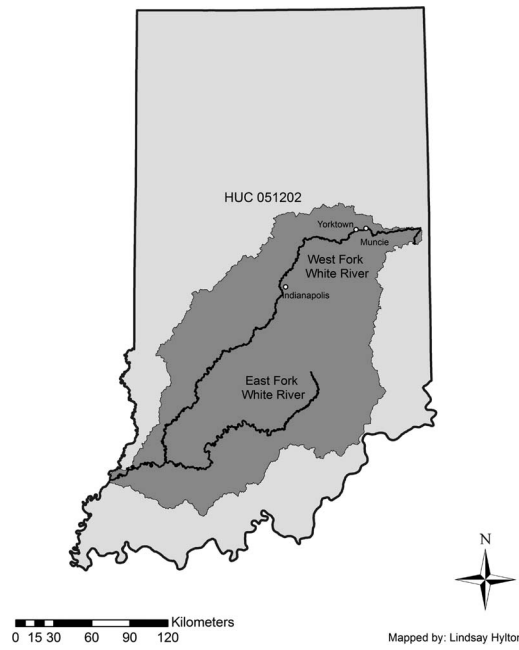


Figure 1.—Site locations within the White River watershed are shown along the Indiana White River. Site 1 (Muncie), Site 2 (Yorktown), and Site 3 (Indianapolis) are marked with white points.

This study was conducted at three locations within the watershed of the West Fork White River basin in Delaware and Marion Counties, Indiana (Fig. 1). Sampling took place from August to November of 2015. Sites 1 and 2 were chosen based on their relative locations upstream and downstream, respectively, of the Muncie Water Pollution Control Facility, and Site 3 was chosen for its more urban and highly populated watershed. Feasibility and safety also played a role in choosing the specific bridges used for the sampling locations. All three sampling locations are located along the West Fork White River, which fall within the Upper White (River) Watershed (HUC 05120201). This watershed has a drainage area of 7055 km^2 and 573 km of flowing water. Its land use is approximately 60% agriculture and 25% developed land (US Census Bureau Data 2015; USGS StreamStats 2017). Further, the three sites fall within two subwatersheds (HUC-12). Sites 1 (Muncie) and 2 (Yorktown) are located in York Prairie Creek-White River, which has 36 km^2 of developed land, 53% of its total area. Site 3 (Indianapolis) is located in Pogues Run-White River, which has 60 km^2 of

Table 1.—Watershed characteristics (US Census Bureau Data 2015; USGS StreamStats 2017).

Site	Population	Sampling point drainage area (km^2)	Drainage % urban development
Muncie	70,087	632.7	4.4
Yorktown	11,231	636.9	4.8
Indianapolis	853,173	4,228.7	10.7

developed land, contributing to 99% of its total area (Wikiwatershed 2017) (Fig. 1).

Sampling locations.—The first sampling site is the furthest upstream, located at the S. Nichols Avenue Bridge in Muncie, IN (40° 11′ 6″ N 85° 24′ 42″ W) (Fig. 1). According to USGS StreamStats data, the drainage area contributing flow to the point sampled is 4.4% urban and drains 632.7 km^2 (Fig. 1; Table 1). The water here is generally shallow, consisting of riffle, glide, and some pool habitats, and is divided by a sand bar. The substrate appears to be sandy silt with cobble. Site 1 is located 2.4 km upstream of the Muncie Water Pollution Control Facility.

The second sampling site, located at the S. Nebo Road Bridge in Yorktown, IN (40° 11′ 9″ N 85° 27′ 43″ W), is located 1.8 km downstream of the Muncie Water Pollution Control facility. The total drainage basin area at this point is 4.8% urban and drains a total of 636.9 km^2 (Fig. 1; Table 1; USGS StreamStats 2017). There is a large rocky island in the middle of the river and the water is generally shallow, with silt and cobble substrate. The habitat consists of riffles, runs, and pools.

The third sampling site is located at the Oliver Avenue Bridge in Indianapolis, IN (39° 45′ 30″ N 86° 10′ 25″ W) (Fig. 1), making it the site furthest downstream. At this sampling point, the total drainage basin area is 10.7% urban and drains 4,228.7 km^2 (Fig. 1; Table 1; USGS StreamStats 2017). The water is significantly deeper than at the first two sites, consisting of primarily slow glide and run habitats. During the study period, Indianapolis had an estimated population of 853,173 (2015), making it the most densely populated watershed of the three sampling sites (US Census Bureau 2015; Table 1).

Sampling procedure.—At each of the three sites, surface water samples were collected on five dates between August through November 2015 (N = 3 sites × 5 events = 15 total events). During each event, two sequential (duplicate) 10 min surface water samples were collected from the same point from the site bridge, the values from these duplicate samples were later averaged to provide one value for each site for each sample event. (While 30 samples were collected, duplicates were averaged, thus N = 15). All samples were collected during daylight hours and not within 48 h of a runoff event (an event resulting in combined sewer overflow). Samples were collected using a Wildco stationary stream drift net (99.06 cm length, 45.72 cm wide, and 30.48 cm tall, 363μm mesh) with a detachable mesh dolphin bucket (368μm mesh). This mesh size falls within a commonly used size range in other microplastic studies (Hidalgo-Ruz et al. 2012; Baldwin et al. 2016). The net was modified to be deployed from a bridge and for flotation.

A digital mechanical flowmeter (2030R, General Oceanics, Miami, FL) was attached across the mouth of the net to measure the velocity of the water entering. The total volume of water being filtered through the net was calculated using the width and height of the net, the duration of the sample, and the velocity of flow (Lechner et al. 2014; Baldwin et al. 2016). Due to low velocity (< 10 cm s^{-1}) at all three sites, a low-flow rotor was used. After 10 min, the net was rinsed with a pressure sprayer into a 200 ml dolphin bucket at the end of the net (Baldwin et al. 2016). The plastics and organic debris collected were rinsed from the bucket into a sealed glass jar with deionized water, and then placed on ice for transport to the laboratory (McCormick et al. 2014). This process was immediately repeated to obtain the duplicate sample. A water sample also was collected at each sampling event using a standard grab sampler deployed from the bridge to assess water temperature at the time of sampling. Air temperature was noted and estimated river discharge for each location were collected from USGS steam flow measurements at sites 03347000 (White River at Muncie) and 03353000 (White River at Indianapolis) (http://waterdata.usgs.gov/nwis/rt).

Laboratory analysis.—Processing and laboratory analyses of samples were completed using a modified version of the methods

developed by the National Oceanic and Atmospheric Administration (NOAA; Masura et al. 2015; Baldwin et al. 2016). Samples were first wet sieved through two stacked stainless steel sieves (mesh sizes 500 μm and 250 μm). These sizes were chosen to highlight the smaller spectrum of microplastic pollution (250–500 μm) and to adhere to a commonly used sieve size (500 μm; Hidalgo-Ruz et al. 2012). In this study a sieve to exclude plastics greater than the upper microplastic size limit (> 5 mm) was not utilized, in order to decrease processing time and avoid overall plastic loss by the use of an additional sieve.

After sieving, mass of dried solids was determined (Masura et al. 2015). Wet peroxide oxidation was then used to degrade any organic material prior to density separation (Masura et al. 2015). Floating solids and plastics were drained into either 250 μm or 500 μm mesh size custom-made nylon sieves (Masura et al. 2015). After a 24 h drying period, visual sorting of the samples was conducted with the use of a stereoscope (dissecting microscope) at 40× magnification or higher (Hidalgo-Ruz et al. 2012; McCormick et al. 2014; Masura et al. 2015). Tweezers were used to remove all identifiable plastics from the sieves for both size categories and transfer them to labeled glass vials for storage. Each sample was examined under the microscope using the Marine & Environmental Research Institute (MERI) identification guidelines (Hidalgo-Ruz et al. 2012; MERI 2015). Plastic particles were counted and categorized into six categories based on their morphology: fragments (broken down larger plastics), beads (spherules from personal care products, bead blasting, etc.), fibers (synthetic textiles), films (plastic wrapping and bags), foam (foam packaging and cups), and pellets (preproduction pellets 5 mm and larger) (Lechner et al. 2014; Zbyszewski et al. 2014; Baldwin et al. 2016). Additionally, the "hot needle test" (*sensu* De Witte et al. 2014) was used in distinguishing between plastic and non-plastic particles, especially for fibers. The total plastic count for each sample was recorded, along with type (from one of the six categories mentioned previously), and color.

Quality assurance and control.—Precautions were taken during this study to avoid contamination. Samples were processed under a fume hood and always remained covered when not in use. Other equipment and tools used in the laboratory also were washed and covered after use. Further, samples were collected and analyzed in duplicates to increase precision.

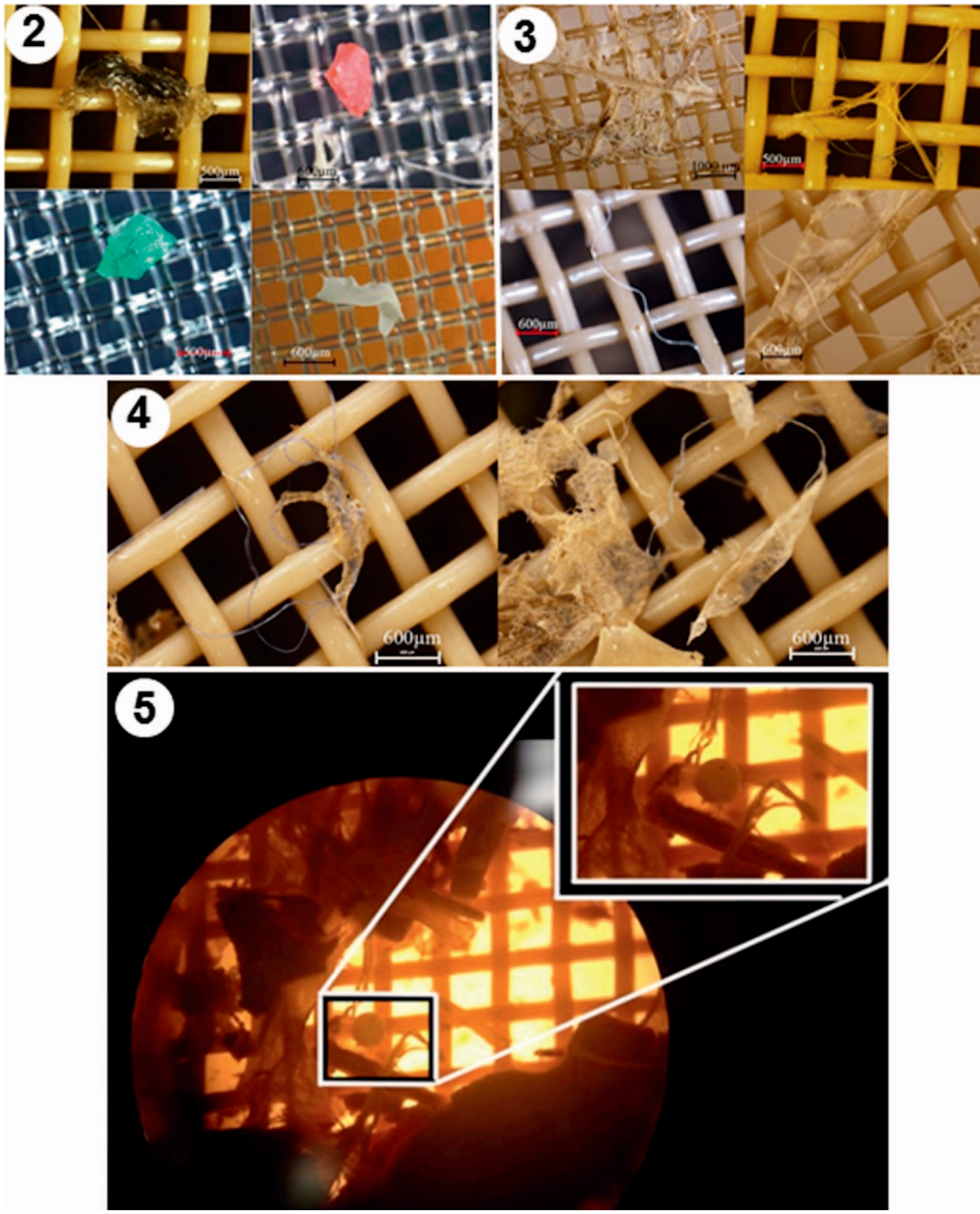
Statistical data analysis.—Plastic concentrations were reported in particles, or items, per cubic meter (item m^{-3}) (Hidalgo-Ruz et al. 2012; Baldwin et al. 2016). Data analyses were conducted using SPSS software (IBM Corp., Armonk, NY) with statistical significance reported at $\alpha = 0.05$. Because of the limited sample size ($N = 15$), a Kruskal-Wallis test by ranks (equivalent to a non-parametric one-way ANOVA test) was used to evaluate differences in plastic concentrations among sites. Kruskal-Wallis was also used to compare concentrations among months. All analyses were completed for plastics collected only on the 250 μm sieve (the smaller particles) and for plastics collected on both the 250 and 500 μm sieves combined (total plastics). This study aimed in part to capture microplastics on the smaller end of their size range (250–500 μm), which are often underestimated (Baldwin et al. 2016).

RESULTS

Total pieces and types collected.—Microplastics of numerous types, colors, and sizes were collected from all three sites sampled (Figs. 2–5). Translucent, white, black, and red plastics were the most prevalent colors collected (Fig. 2). Across all samples, a total of 146 plastic pieces were collected from the White River over the duration of this study. Of those pieces, 40 (27 %), were in the 250–500 μm size range (Figs. 6 & 7).

The 146 microplastic pieces collected in this study were comprised of 16 fragments, one spherule, 112 fibers, three films, 13 foamed plastics, and one pellet (Fig. 7). Synthetic fibers were the predominant plastic type collected (~80 % of the total plastics collected) (Fig. 6). Fragments and foamed plastics constituted the next largest plastic type (11–13% and 4–9%, respectively; Fig. 6), while films, beads, and pellets ranged from 1–2% of items collected (Fig. 6).

Average concentrations.—Site 1 (Muncie) and Site 3 (Indianapolis) resulted in very similar average concentrations of smaller microplastics (0.24 and 0.23 items m^{-3}, respectively; Fig. 8). However, for the average *total microplastic* concentration, Site 1 (Muncie) had an average of 0.75 items m^{-3}, compared to Site 3 (Indianapolis) which had the highest of the three sites (0.93 items m^{-3}; Fig. 8). Site 2 (Yorktown) had the lowest average concentra-



Figures 2–5.—Various microplastics, fibers, and spherules collected. 2. Microplastic fragments of various colors and sizes. 3. Various synthetic fibers. 4. Non-synthetic fibers (which did not melt during the hot needle test) and are likely made of cotton or rayon. 5. Spherule (microbead) among organic debris.

tion for smaller microplastics, with 0.15 items m^{-3} (Fig. 8). Site 2 (Yorktown) also had the lowest average *total microplastic* concentration, with 0.44 items m^{-3} (Fig. 8). The overall average *total microplastic* concentration for the White River, based on the three sites sampled, was 0.71 items m^{-3} (Fig. 8).

Concentration differences between sites.—There was no difference in plastic concentration among sites for either small plastics ($p =$

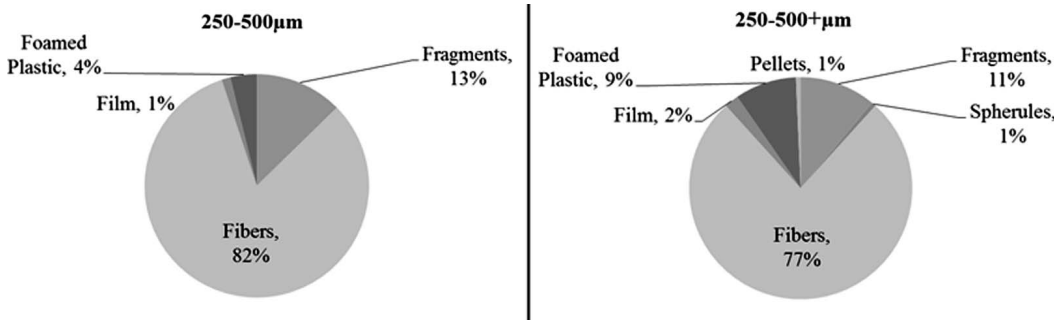


Figure 6.—Graph of average plastic concentration percentages by microplastic type across all three sites for 250–500 µm (left) and 250–500+ µm (right).

0.961) or total plastics ($p = 0.395$) concentrations. Further, there was no difference in plastic concentration among sampling month for either the smaller ($p = 0.849$) or total ($p = 0.753$) plastics.

DISCUSSION

Total pieces and types collected.—The lack of beads (spherules) and pellets found in the White River, along with the prevalence of fibers (~80% of the average particles collected; Fig. 6) and fragments (11% of the average particles collected; Fig. 6) are relatively consistent with the results of recent fluvial/tributary studies, but not of lacustrine/lake studies. Pellets and beads also were scarce in recent tributary Great Lake samples, despite large portions of discoveries in lake samples (Baldwin et al. 2016). Studies along the Danube and European Rhine rivers that found large concentrations of plastic pellets (as well as beads) were attributed to their proximity to vast plastic manufacturing facilities along these rivers. It also should be noted that those watersheds were much larger in scale (Lechner et al. 2014; Mani et al. 2015). The prevalence of fibers in this study (Fig. 6) mirrors the large-scale study on 29 Great Lake tributaries, where Baldwin et al. (2016) observed a similar dominance of fibers and fragments (71% and 17% on average, respectively). Other fluvial studies done on the Seine River and Chicago's North Shore Channel found fibers were the most abundant plastic type collected (Gasperi et al. 2014; McCormick et al. 2015), followed by fragments (McCormick et al. 2015). Fragments were also in high quantities in the Rhine, while fibers were not accounted for in the Danube studies (Lechner et al. 2014; Mani et al. 2015).

Other tributary samples from freshwater, non-fluvial, lacustrine/lake studies (namely the Great Lakes) found a variety of particle types but consistently show substantially higher abundance of fibers (Free et al. 2014; Baldwin et al. 2016). However, in non-tributary Great Lake samples, only 2% of particles were fibers compared to 20% in Mongolian lakes (Baldwin et al. 2016). The authors attribute this difference in fiber contents between fluvial and lacustrine environments in part to analytical methods, but also to the actual physical properties of different plastic types and

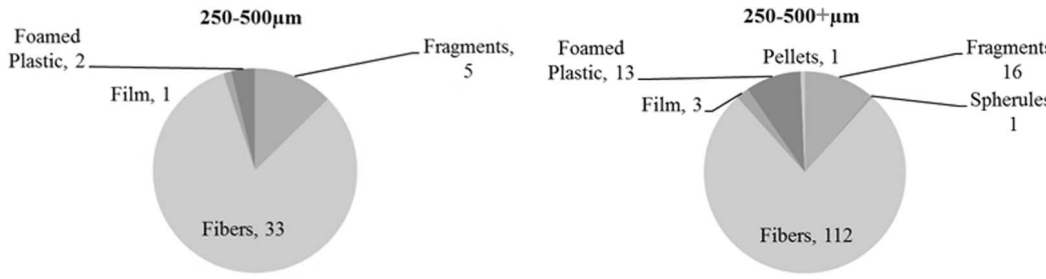


Figure 7.—Graph of average total microplastics counted across all three sites by type for 250–500 µm (smaller plastics; left) and 250–500+ µm (total plastics; right).

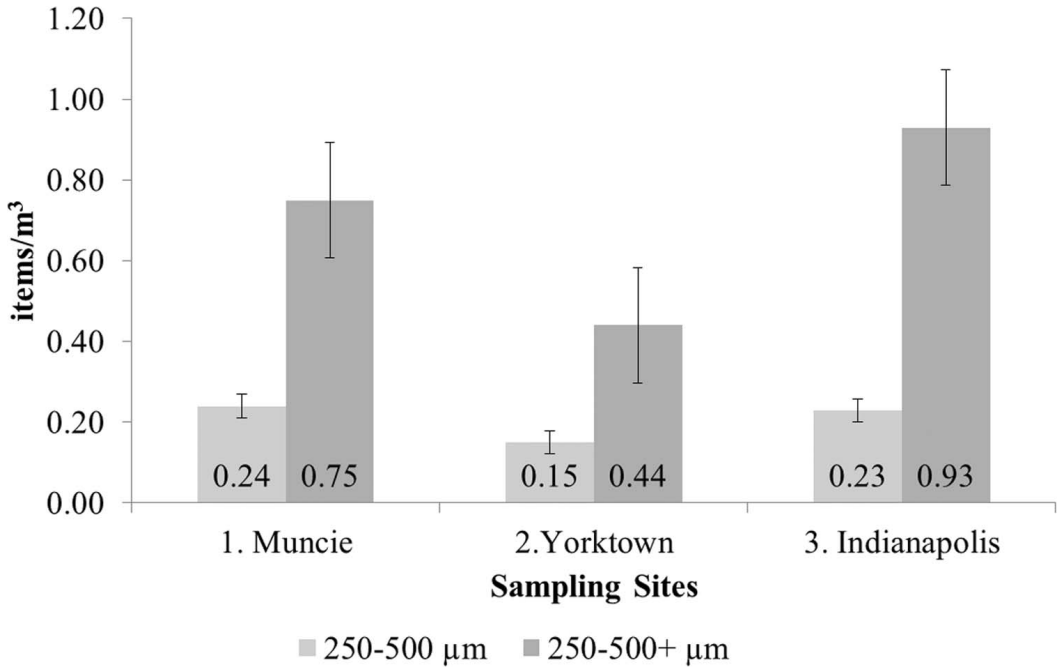


Figure 8.—Graph of average plastic concentration by site (items m^{-3}).

the unique hydraulics of river systems versus those of the Great Lakes (Baldwin et al. 2016). Calmer lake currents allow for easily sunk fibers (i.e., rayon, polyester, nylon, and cellulose acetate) to settle into the sediment, while more tumultuous river currents may keep these fibers in suspension where they are easier to capture during surface sampling events (Baldwin et al. 2016; Ballent et al. 2016). This may explain the lower fiber quantities found in lakes compared to tributaries. Conversely, most films, pellets, or foams are made of polymers that tend to float (polypropylene, polystyrene, and polyethylene) until changes in density, due to biofouling, cause them to sink. This increases the likelihood of finding them in higher concentrations in lake systems compared to other plastic types (Baldwin et al. 2016; Ballent et al. 2016). This relative lack of fibers found in lakes, due to settling, suggests the need for further research into microplastic abundance within lakebed sediments and the possible effects on the organisms living in that habitat.

Average concentrations.—The plastic concentrations measured from three sites on the White River (0.44–0.93 items m^{-3}, mean 0.71 items m^{-3}; Fig. 8) are comparable to the limited literature available for river microplastic studies. Baldwin et al. (2016) summarized notable studies on Chicago's North Shore Channel and Paris' Seine River with mean concentrations of 1.9 to 17.9 and 0.28–0.47 pieces m^{-3}, respectively, while Great Lakes tributaries reported a mean of 4.2 items m^{-3}.

Concentration differences between sites.—Although plastic concentrations in this study did not differ significantly by sampling site (Fig. 8), the quantities and types of plastics found are consistent with those of other recent fluvial studies (Baldwin et al. 2016) and contribute to the overall understanding of microplastic abundance and behavior in this unique environment (Figs. 2–5). Additionally, the results build on the understanding of the effect of watershed attributes, such as urban development, on microplastic pollution. Although not statistically significant, the average total plastic concentration at the Indianapolis site was the highest of all three sampling locations (0.93 items m^{-3}; Fig. 8), likely attributable to a densely populated, urban subwatershed (Yonkos et al. 2014). Muncie had the next highest, followed by Yorktown (0.75 and 0.44 items m^{-3}, respectively; Fig. 8). Greater quantities of impervious surfaces and combined storm sewers in urban watersheds enhance prevalence and mobility of plastic

litter into receiving water bodies (Baldwin et al. 2016).

Another point of interest in this study was in determining whether samples taken at Site 2 (Yorktown), which is just downstream of the Muncie Water Pollution Control Facility (a waste water treatment plant, or WWTP), would have higher microplastic concentrations than those taken at Site 1 (Muncie), which is just upstream. Since microplastic concentrations were not found to differ significantly between any of the sites and the Yorktown site showed the lowest microplastic concentrations (Fig. 8), there was no evidence that this WWTP was discharging microplastic pollution at higher rates. The Muncie site had nearly doubled the average total plastic concentration as the Yorktown site (0.44 vs. 0.75 items m^{-3}; Fig. 8). Some recent studies have suggested that WWTPs can be relatively efficient at microplastic removal and can ultimately act as both a sink and a source for this pollutant (Carr et al., 2016; Mintenig et al. 2017).

WWTP processes can be effective at removing microbeads (Carr et al. 2016; Murphy et al. 2016) but less so with other forms of microplastics such as fibers or fragments (Mason et al. 2016a). The lack of beads (spherules) found in this study (only 1 total; Fig. 7) is consistent with the findings of Murphy et al. (2016), who explain that most microbeads in face washes contain positively-buoyant polyethylene and tend to float on the surface of wastewater, where they are easily skimmed off during grease removal. Conversely, Mason et al. (2016a), while studying wastewater effluent from 17 different U.S. wastewater treatment facilities, found fibers (59%) and fragments (33%) to be the most common microplastic types, which is consistent with the current study (Figs. 6 & 7). Likewise, McCormick et al. (2014) who sampled in Chicago's North Shore Channel, both upstream and downstream of a WWTP, found a high abundance of microplastic downstream of the facility, in comparison with the upstream site.

Heavy precipitation events represent another mechanism through which WWTPs could contribute microplastic pollution, even microbeads, to waterways (Murphy et al. 2016). Many Midwestern states, including Indiana, have numerous combined sewer overflow outfalls along waterways, where untreated combined waste and stormwater is discharged when the volume of influent to a facility exceeds the treatable volume. This untreated wastewater has potential to greatly affect the amount of microplastic entering the environment (Murphy et al 2016), yet no studies have investigated stormwater overflow as it relates to microplastic pollution.

Considerations for future microplastic studies.—This study aimed to capture microplastics on the smaller end of their size range (250–500 μm). These are often underestimated since one finds an inverse relationship between particle size and plastic concentration (Baldwin et al. 2016). Our results from limited sampling suggest that Indiana's White River contains quantities of microplastic particles, especially of synthetic fibers, comparable to other fluvial studies. The widespread use and laundering of synthetic clothing and the land application of treated sewage sludge, in addition to atmospheric deposition and overland runoff, are all likely sources of synthetic fibers found in the White River and similar systems.

Overall, the prevalence of microplastics in aquatic systems worldwide suggests a considerable source of pollution with unknown long-term consequences. In both marine and freshwater environments, they can easily be mistaken for food by aquatic life, or, if heavy enough, eventually make their way into sediment. Here, they could become available to benthic feeders or remain in the environment for years to come. Future studies on freshwater organisms and sediments will help researchers gain a better understanding of the fate of increasing microplastic pollution.

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