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Microplastics and Microfibers in River Sediments: A Review of Current Literature and New Data from Texas Rivers

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Abstract: Microplastics in the Earth system are now widely documented and assessed. As plastic production continues and recycling capabilities lag, however, continued monitoring of their accumulation and transport within fluvial systems is necessary for managing the ecological and geomorphic effects of their presence. In addition to plastic pellets, particles, and beads, synthetic fibers and fibers coated with synthetic dye are increasingly causing concern due to their sheer numbers in the natural environment. This study presents a comprehensive review of the current literature regarding microplastics and microfibers in the environment and their potential impact on fluvial systems. In addition, data are presented that demonstrate the presence and prevalence of microfibers in select Texas rivers. Bed sediment from the Brazos, Colorado, and Trinity Rivers was sampled and assessed during the years 2020–2021. Fibers were present in nearly all samples, most abundantly immediately downstream of urban centers. Such sampling efforts should be taken regularly in ecological and geomorphic systems to monitor the temporal accumulation of plastic particles and fibers. These may have direct or cascading effects on ecological and human health. Increasing synthetic sediment may also influence the geomorphic adjustment of river channels by alterations to biogeomorphology and hydraulic processes.

Keywords: Texas, microplastics, microfibers, river sediment, Texas rivers, rivers, water, Texas water

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Terms used in paper

Acronym/Initialism	Descriptive Name
ABS	Acrylonitrile-Butadiene-Styrene
EPS	Expanded polystyrene
FTIR	Fourier-transform infrared spectroscopy
HDPE	High-Density polyethylene
LDPE	Low-Density polyethylene
mm	Millimeter
nm	Nanometer
km	Kilometer
PA	Polyamide
PBDE	Polybrominated diphenyl ether
PBT	Polybutylene terephthalate
PC	Polycarbonate
PL	Polyester
PE	Polyethylene
PETE	Polyethylene terephthalate
PMMA	Polymethyl methacrylate
PP	Polypropylene
PS	Polystyrene
PTFE	Polytetrafluoroethylene
PVC	Polyvinyl chloride
TPU	Thermoplastic polyurethane

INTRODUCTION

Microplastics and nanoplastics—plastic remnants under 5 mm and 1-1000 nm in diameter (Gigault et al., 2018), respectively—have been found in air, water, soil, plants, and human and animal tissue (Ragusa, Svelato et al., 2021; Ragusa, Notarstefano, et al., 2022). These plastics' pervasiveness mean that they must now be considered part of the same biogeochemical cycles that transport nutrients through the natural environment (Schell et al., 2020). It is, therefore, urgent to quantify particle transport rates through the fluvial system and into food and drinking water.

Plastic production is expected to triple by 2060 (Lebreton & Andrady, 2019), while international recycling partnerships have diminished (Wen et al., 2021). It is therefore vital to continue monitoring the abundances and transport of microplastic, nanoplastic, and other synthetic particles in the fluvial system. In this study, I review the literature on plastic particles' fate and transport in fluvial systems and assess these particles' likely effects on geomorphic adjustment and ecosystem and human health. I begin by presenting an overview of plastics and plastic-related chemicals, followed by a review of how these substances affect ecological health, human health, and river hydraulics. Finally, I present data on microplastics' prevalence in select reaches of the Brazos, Colorado, and Trinity rivers in Texas. In this study, the terms "microplastic" and "nanoplastic" refer to particles of any shape, unless otherwise specified .

Microfibers

Microfibers—threadlike particles with a length between 100 µm and 5 mm and a width at least 1.5 orders of magnitude shorter—are widely recognized to be present in tandem with other microplastic particles, such as fragments (broken pieces

Table 1. Plastics detected in the natural environment by studies referenced in this work.

PLASTIC OR CHEMICAL	ABBREVIATION	DESCRIPTION
Acrylonitrile-butadiene-styrene	ABS	Provides rigidity and impact resistance. Composed of polymerized acrylonitrile, butadiene, and styrene.
Kraton g	-	Used in adhesives, sealants, and coatings. Composed of polymerized styrene, ethylene/butylene, and polystyrene.
Polyamide	PA (nylon)	Used in textiles, kitchen utensils, and automobiles manufacturing.
Polybrominated diphenyl ethers	PBDE	Used in paints, furniture, upholstery, and flame retardant.
Polybutylene terephthalate	PBT	Used as an insulator in electronics and electrical devices.
Polycarbonate	PC	High impact-resistance; often used in signs or guards. Made from the polymerization of carbonate groups.
Polyester	PL	Used widely in fabrics. Formed by esterification of various polymers.
Polyethylene	PE (PETE)	Used primarily for packaging. Formed by polymerization of ethylene.
High-density polyethylene	HDPE	Opaque PE with linear chemical structure. Often used in bottles, cutting boards, and piping.
Low-density polyethylene	LDPE	Transparent PE with a branched chemical structure. Often used in plastic bags.
Polyethylene terephthalate	PETE	Fully recyclable plastic. Made of repeating C10H8O4 units.
Polymethyl methacrylate	PMMA (acrylic)	Transparent plastic used in a variety of engineering applications. Made from methyl methacrylate.
Polyvinyl chloride	PVC	Used in pipes, doors, and windows. Made from the polymerization of the vinyl chloride monomer.
Polypropylene	PP	Used in hinges, furniture, laboratory equipment, and many other uses. Produced by chain-growth polymerization of propylene.
Polystyrene	PS	Used in packaging. Polymer made up of repeating molecules of styrene.
Expanded polystyrene	EPS	Used in packaging. PS is exposed to steam, causing expansion.
Polytetrafluoroethylene	PTFE (Teflon ®)	Used as a nonstick coating for cookware. Synthetic fluoropolymer of tetrafluoroethylene.
Thermoplastic polyurethane	TPU	Used in flexible casings, such as phone cases, and various other uses. High elasticity, transparency, and resistance to oil, grease, and abrasion.

with jagged edges), pellets (spheres), films, and foam in the natural environment, and their effects are poorly understood (De Falco et al., 2018; Liu et al., 2019; Miller et al., 2017). A subgroup of microplastics, plastic microfibers comprise up to 90% of global microplastic concentrations (Belzagui et al., 2019; Woods et al., 2018). Plastic microfibers are commonly composed of polypropylene (PP), polyester (PL), polyamide (PA or nylon), polybutylene terephthalate (PBT) acrylic, or polyethylene (PE) (Miller et al., 2017; Rochman et al., 2013; Savoca et al., 2019) (Table 1). Polytetrafluoroethylene (PTFE, known as Teflon) and Kraton G (polymer) have also been detected in plastic microfibers (Capillo et al., 2020; Table 1). Commonly associated with textiles (Belzagui et al., 2019; Chan et al., 2021; Periyasamy & Tehrani-Bagha, 2022), microfibers from the combination of personal care products, cigarette filters, carpeting, and other sources are now thought to rival textiles (Athey & Erdle, 2022).

Not all microfibers are made of plastic; microfibers made of non-plastic processed natural materials such as cotton, wool, silk, bamboo, or rayon (regenerated cellulose) persist in the environment, as well. However, these non-plastic fibers are often dyed and coated during the manufacturing process. The chemicals used in this process include flame retardants (e.g., PBDEs) and other carcinogens (Miller et al., 2017; Schreder & La Guardia, 2014).

Health concerns

Early studies of ingestion of microplastics by mammals, including humans, demonstrated that they can be transported into blood and lymph nodes and from the placenta to fetus (Hussain et al., 2001; Wick et al., 2010; Storck et al., 2015). In some animals, microplastics were linked to capillary inflammation and reproductive changes (Brown et al. 2001; Storck et al., 2015). Microplastics' constituent chemicals are also endocrine disruptors, meaning that chronic exposure can have ongoing health effects (Q. Chen et al., 2019; Campanale et al., 2020; Solleiro-Villavicencio et al., 2020).

Microplastic exposure begins and accumulates in utero (Ragusa, Matta, et al., 2022). They have been found in human placenta (Ragusa et al., 2021) and in newborn meconia (Braun et al. 2021; Ragusa, Matta, et al., 2022) at even higher concentrations than in adult stools (J. Zhang et al., 2021), breastmilk (Ragusa, Notarstefano, et al., 2022), and blood (Leslie et al., 2022). Microplastics enter human bodies through food, inhalation, and skin absorption. Once in the body, they are transported by the circulatory system and deposited in organic tissue and may spread among cells by diffusion or phagocytosis (DeLoid et al., 2021; Ragusa, Matta, et al., 2022). Within body tissues, microplastics are received as foreign bodies, triggering local immunoreactions (Ragusa, Matta, et al., 2022) and causing cascading effects.

Additives found within microplastics , specifically bisphenols and certain phthalates, are endocrine disruptors, confusing the reception and response of hormones in the host organism (Ragusa, Matta, et al., 2022). In animal studies, high exposure to microplastics in pregnant and nursing organisms is correlated with neuron and brain tissue structure alterations in progeny (Jeong et al. 2022). For example, exposure to PBDE nanoplastics caused neurochemical, olfactory, and behavioral processes very similar to those of autism spectrum disorders in humans (Kozlova et al., 2021). Other studies have demonstrated microplastics' likely effect on human placenta. Ragusa, Matta et al. (2022) reported that the endoplasmic reticula of cells within the human syncytiotrophoblast, the tissue that transports nutrients and other substances to the placenta, exhibited inflammation and roughness . This response is likely a reaction to stress, which previous studies have demonstrated can be induced by microplastics exposure in various tissues (Cortés et al., 2020; Ding et al., 2021; Gopinath et al., 2021; Pironti et al., 2021; Lu et al., 2022). Stress indicators in the endoplasmic reticulum of placental cells is concerning, as endoplasmic reticulum stress in the placenta has been linked to preeclampsia and other pregnancy complications (Burton et al., 2009; Burton & Yung, 2011 ; Sultana et al., 2017).

Although the extent to which the chemicals in plastics bioaccumulate and leach into organisms' tissues remains under investigation, other studies show similar evidence of its occurrence. Rochman et al. (2013) subjected Japanese medaka (*Oryzias latipes*) fish to concentrations of low-density polyethylene (LDPE) plastic similar to those experienced by aquatic species in the wild. After 2 months of chronic exposure to low LDPE levels, the fish exhibited higher concentrations of PBTs. The dosage was nonlethal but still induced stress in the livers of the fish, which presented as glycogen depletion, fatty vacuolations, and single-cell necrosis (Rochman et al., 2013).

Microplastics as ecological concerns in river systems

Due to the potential health effects of microplastic consumption, it is imperative to understand their fate and transport in the fluvial system. An estimated two-thirds of the U.S. population's drinking water supply comes from rivers (American Rivers, 2023), and current water treatment processes' effectiveness at removing microplastics and nanoplastics remain uncertain (Pivokonsky et al., 2018; Ma, Xue, Ding, et al., 2019; Ma, Xue, Hu, et al., 2019; Na et al., 2021; Zhou et al., 2021). Drinking water treatment generally consists of coagulation/sedimentation, filtration, and disinfection (Na et al., 2021). Disinfection is often accomplished through application of ultraviolet radiation, which has been shown to fragment microplastics and nanoplastics into progressively smaller fragments through photochemical weathering. This breakdown eases uptake of fragments by organisms. This has been shown to occur with PS fragments (Na et al., 2021), LDPE, PP, and EPS fragments (Song et al., 2017; Sun et al., 2022), TPU (Sun et al., 2022), and PVC (C. Chen et al., 2019).

Plastic detritus could be a catalyst for ecological collapse, if its release into the aquatic environment is not checked. Consumption by primary suspension feeders or sediment deposit feeders leads to plastic buildup in the digestive tract of secondary and tertiary consumers (Collard et al., 2017; Di-Méglio & Campana, 2017; Matsuguma et al. 2017; Rezania et al., 2018). This poses a clear risk to ecological health, which in turn affects geomorphic processes, because organisms influence sediment entrainment and transport. Although the role of biotic versus abiotic factors in stream morphology varies among fluvial systems, rivers with lower discharges and that are incised into soils with smaller grain sizes seem to be more heavily impacted by biotic factors compared to rivers with higher discharges and surrounded by soils with larger grain sizes (Albertson & Allen, 2015; Polvi & Sarneel, 2018).

Some fluvial organisms, such as crayfish (common in Texas rivers), are destabilizers (Albertson & Allen, 2015; Polvi & Sarneel, 2018; Statzner & Stagnes, 2008; Statzner, 2012). Microplastics have been reported to accumulate in crayfish tissues and to cause oxidative stress (Gholamhosseini et al., 2023; Silveyra et al., 2023; Yücel & Kiliç, 2022; Zeidi et al., 2022; D. Zhang et al., 2021 ; Zhang et al., 2022). Pastorino et al. (2023) proposed crayfish as a viable bioindicator for microplastic pollution, and they can be used to demonstrate a linear positive correlation between upstream urbanization and microplastic content (Dent et al., 2023), supporting the findings of an earlier study by McCormick et al. (2015). Conversely, some organic communities, such as biofilms, protect sediment from erosion by covering them with a smooth surface, acting as bed stabilizers (Polvi & Sarneel, 2018). The relationship of microplastics and biofilms is important to the ecological composition of rivers, as the structure of microorganism communities colonizing plastic are distinct from those on natural surfaces (Rummel et al., 2017). Biofilms may also aid overall ecological health by their apparent ability to reduce toxic effects of nanoplastics on aquatic organisms (Natarajan et al., 2021).

In addition to biofilms, macrophytes can also increase bed stability, in some cases doubling the force needed to dislodge sediment (Fritz & Feminella, 2003; McBride et al., 2007; Polvi & Sarneel, 2018). Mateos-Cardénas et al. (2021) have shown that microplastics and nanoplastics can affect aquatic macrophytes and terrestrial plants. When microplastics and nanoplastics are adsorbed onto the surfaces of these plants and taken into the internal structure, they can induce delayed germination, decreased root length, biomass changes, and negative changes to photosynthesis in some species (Mateos-Cardénas et al., 2021).

Sediment that is very fine to fine sand or silt is susceptible to modification by biotic factors (Albertson & Allen, 2015; Polvi & Sarneel, 2018). Biota , in turn, are susceptible to the impacts of microplastic contamination in their environments (Rummel et al., 2017; Mateos-Cardénas et al., 2021; D. Zhang et al., 2021; Zeidi et al., 2022; Zhang et al., 2022; Gholamhosseini et al., 2023; Silveyra et al., 2023; Yücel & Kiliç, 2022). Common biogeomorphic agents in the rivers assessed in this study—the Brazos, Trinity, and Colorado rivers—are presented in Table 2.

Previous studies have detected microplastics throughout the freshwater system. These contaminants have been found in:

- river sediments (Castañeda et al., 2014; Frei et al., 2019; He et al., 2020; Horton et al., 2017; Klein et al., 2015; Mani et al., 2015; Mani et al., 2019; Wang et al., 2018; Wen et al., 2021; Ramirez et al., 2019; Sarkar et al., 2019; Shruti et al., 2019; Zhang et al., 2020)
- river surface water (McCormick et al., 2015; Dikareva & Simon, 2019; Xiong et al., 2018; Irfan et al., 2020; Zhang et al., 2020)
- estuaries and lagoons (Cheng et al., 2021; Fok & Cheung, 2015; Gray et al., 2018; Pazos et al., 2018; Sadri & Thompson, 2014; Silva et al., 2018; Vianello et al., 2013; Wessel et al., 2016)
- fluvial suspended and neustonic sediment (Ding et al., 2019; Leslie et al., 2017)
- lake sediment (Anderson et al., 2017; Corcoran et al., 2015; Fischer et al., 2016; Mason et al., 2016; Mason et al., 2020; Sighicelli et al., 2018; Sruthy & Ramasamy, 2017; W. Wang et al., 2018; Xiong et al., 2018), and in
- freshwater organisms (Biginagwa et al., 2016; Collard et al., 2018; Irfan et al., 2020; Kim et al., 2018; Su et al., 2018; Zhang et al., 2019).

Even rural water bodies are susceptible to microplastic contamination, as microplastics are transported and deposited by atmospheric fallout (Dris et al., 2017).

Although studies of microplastic abundance in freshwater systems have become more common, few studies address the factors that control microplastic distribution in river systems (Corcoran et al., 2019). Microplastics present a novel twist in traditional alluvium transport studies, as their density, shape, and size diverge from the norms observed in natural river sediment, rendering traditional equations for calculating hydraulic characteristics ineffective (Waldschläger & Schüttrumpf, 2019a, 2019b). Microplastics exhibit a diversity of shapes, densities, and sizes that are distinct from natural river sediment. Natural sediment tends to be of a size and material equivalent to the nature of bedrock and velocity of flow, respectively (Nizzetto et al., 2016; Waldschläger & Schüttrumpf, 2019a, 2019b). This leads to challenges in predicting microplastic distribution in the fluvial environment, especially based on a single proxy, because standard equations for hydraulic characteristics are not always applicable (Enders et al., 2019; Waldschläger & Schüttrumpf, 2019a, 2019b).

Recent studies have attempted to develop new equations for assessing the hydraulic characteristics of microplastic particles (Enders et al., 2019; Waldschläger & Schüttrumpf, 2019a, 2019b). Enders et al. (2019) developed three equations to predict the distribution of various types of microplastic based on substrate grain size and factors such as distance from wastewater treatment plant, amount of organic matter, and distance from a marina. Waldschläger and Schüttrumpf (2019b) tested microplastic particles in a laboratory setting to assess the hydraulic characteristics of various types of plastic and developed an equation to determine the critical shear stress of different microplastic particles on natural sediment beds. They determined that critical shear stress increases with higher particle densities, and that particle size has a lower influence than density. Regression analysis also showed that critical shear stress was less dependent on particle shape because the influences of particle density, particle size, and the sediment bed compensate for the effects of particle shape (Waldschläger & Schüttrumpf, 2019b).

Although Waldschläger and Schüttrumpf (2019a, 2019b) demonstrated the importance of particle size and density on microplastic mobilization, the relative importance of each factor needs further investigation. Enders et al. (2019) compared high-density and low-density microplastic sediment abundance in estuarine sediment and found that high-density sediment was more than twice as abundant as low-density sediment. Further, the stronger the shear stress forces on a sediment bed, the larger the proportion of high-density polymers relative to low-density polymers. The density of microplastics undoubtedly plays a role in their delivery to the marine realm; approxi-

Table 2. Major biogeomorphic agents in the study areas.

River	Organism	Stabilizer	Destabilizer	Method	References
Trinity	Bowfin	X	X	Predation of destabilizers; nesting	Texas Parks & Wildlife, 2020c
Brazos, Trinity	Black Bullhead		X	Hibernation	Dehring & Krueger, 2008; Texas Parks & Wildlife, 2020a, 2020c
Brazos, Colorado, Trinity	Centrarchidae*		X	Nest excavation	Martin, 2013; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Colorado	Cichlid, Rio Grande		X	Nest excavation, burrows	Ribbink et al., 1981; Texas Parks & Wildlife, 2020b
Brazos, Colorado, Trinity	Crayfish		X	Burrowing, bioturbation	Telfair, 1981; Rice et al., 2016; U.S. Fish & Wildlife Service, 2017
Brazos, Colorado, Trinity	Cypriniformes**		X	Feeding, foraging	Texas Master Naturalist Program, 2013; Pledger et al., 2014; Huser et al., 2016; Rice et al., 2019; Gooch et al., 2012; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Colorado	Eel, American		X	Burrowing	Texas Parks & Wildlife, 2020b
Brazos, Colorado, Trinity	Freshwater Drum	X		Predation on destabilizers	Griswold & Tubb, 1977; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Brazos, Colorado, Trinity	Alligator Gar	X		Predation on destabilizers	Kennedy & Mondragon, 2013; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Trinity	Herring, Skipjack	X	X	Predation on destabilizers and stabilizers	Chandler, 2014; Texas Parks & Wildlife, 2020c
Brazos, Colorado, Trinity	Ictalurus (Catfish, blue, channel and flathead)		X	Burrowing	Harvey et al., 2019; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Brazos, Colorado, Trinity,	Mayflies, Caddisflies	X		Net-spinning	Albertson et al., 2014a; Polvi & Sarneel, 2018; Cloud, 1973; Johnson & Kennedy, 2003
Brazos, Trinity	Mussels (various species)	X	X	Embedding, burrowing	Zimmerman & de Szalay, 2007; Lash, 2011; Slye et al., 2011; Gooch et al., 2012
Trinity	Needlefish, Atlantic	X	X	Predation on destabilizers and stabilizers	Arceo-Carranza et al., 2004; Texas Parks & Wildlife, 2020c
Brazos, Trinity	Oligochaeta and Tubificinae		X	Burrowing	Lash, 2011; Slye et al., 2011
Trinity	Pacu, Redbelly		X	Consumption of lotic vegetation	Texas Parks & Wildlife, 2020c
Brazos, Colorado, Trinity	Slender Gar (Spotted and Longnose)	X		Predation on destabilizers	Kennedy & Mondragon, 2013; Texas Parks & Wildlife, 2020a, 2020b, 2020c
Brazos	Gizzard Shad		X	Feeding activities, bioturbation	Shepherd & Mills, 1996; Schaus, 2007; Texas Parks & Wildlife, 2020a
Brazos, Trinity	Rainbow Trout		X	Digging redds	DeVries, 1997; Texas Parks & Wildlife, 2020a, 2020c

*Bass, Redbreast Sunfish, Bluegill, Crappie, Warmouth

**Minnows, Chub, Carp, Carpsuckers, Smallmouth buffalo, Redhorse, Smalleye, and Sharpnose Shiner mately half of all manufactured plastics are heavier than water, meaning they can sink to the bottom of a river channel with minimal influence (Waldschläger & Schüttrumpf, 2019b).

Peng et al. (2018) tested river sediment and tidal flat sediment and found that the abundance of microplastics in river sediment was one to two orders of magnitude higher than in the tidal flat, implying that a significant amount of microplastic sediment is deposited in the river channel with natural sediment. In a study that challenges density as the dominant mobilization factor, Nizzetto et al. (2016) modeled microplastic transport through the Thames River, United Kingdom. They found that size is a determinant of mobility—smaller particles are more mobile than larger ones—but density had little or no effect on mobilization. Microplastic particle size, in turn, is determined by several other external factors. In Nizzetto et al. (2016), particle size was primarily determined by manner of weathering, rather than plastic composition. Abundances of respective particle sizes also vary within the same river channel. Corcoran et al. (2019) detected a significant difference in microplastic abundance between very fine sand and fine to medium sand sized particles, but not between fine sand and medium sand or coarse sand and granular sand sized particles in the Thames River, Canada. In a study of estuarine sediment, Enders et al. (2019) found that size distribution of microplastics was dependent on density for estuarine sediment in an estuary of the Warnow River in Germany, with high-density particles increasing in number with size, but low-density particles remaining at constant low levels without size variations. As in other studies (Nizzetto et al., 2016; Corcoran et al., 2019; Waldschläger & Schüttrumpf, 2019a, 2019b), Enders et al. (2019) found size to be a strong indicator of microplastic mobility, as median grain size (D50) was highly correlated with total microplastic abundance (the finer the sediment, the more abundant the microplastic content). They stress, however, that sediment grain size should be further evaluated with respect to potential usage as a proxy for microplastic contamination levels (Enders et al., 2019).

Corcoran et al. (2019) also found that river morphology played a role in microplastic abundance. Higher levels of microplastics were found on the cut banks of river channels than on straight reaches, but no difference was detected between the point bar and cut banks. This further differentiates microplastic sediment from natural sediment, which is often preferentially deposited on the point bars of river channels. The most significant factor controlling microplastic abundance found by Corcoran et al. (2019) was organic content. The amount of organic debris was positively correlated to microplastic content, likely a result of the similar densities of organic matter and many microplastics (Corcoran et al., 2019).

STUDY AREA

In this study, I sampled sediment from the bed and banks of three Texas rivers: the Brazos, the Colorado, and the Trinity. Specifically, I sampled sediment from upstream and downstream of urban centers. This allows for comparison of contamination level before and after urban influence. The Brazos River was sampled upstream and downstream of Waco, Texas; the Colorado River was sampled upstream and downstream of Austin, Texas; and the Trinity River was sampled upstream and downstream of Dallas, Texas (Figure 1).

The Brazos River extends 1,352 km from the confluence of its Salt Fork and Double Mountain Fork in Stonewall County to its mouth in the Gulf of Mexico near Freeport in Brazoria County (Hendrickson, 2019). The Brazos River has the highest average annual discharge of any Texas river at 7.4 km^3/s (Hendrickson, 2019). In this study, we examined a ~27 km reach beginning ~12 km upstream of the city of Waco and extending ~11 km downstream of Waco (Figure 2).

The Colorado River extends 1,387 km from its headwaters near Big Spring, Texas, to the Gulf of Mexico and has a total drainage area of 103,341 km^2 (Kammerer, 1987). Here, we assessed the river in the vicinity of Austin, Texas. Urban sprawl associated with Austin extends significantly to the northwest, following the Colorado River upstream. For this reason, observations of the river in a natural state upstream of Austin urbanization and suburbanization occurred near Smithwick, Texas, approximately 50 miles (80.5 km) northwest of Austin (Figure 3). The Colorado River was accessed downstream of Austin at the Roy G. Guerrero Park near Longhorn Dam (Figure 4).

The Trinity River originates at the confluence of West and Elm Forks northwest of Dallas, Texas and east of Arlington, Texas. It extends 1,142 km to its mouth at Trinity Bay in Chambers County, Texas. The river's average annual discharge is 7.03 km^3/s. In this study area, the Trinity River originates in Cretaceous-aged limestone, shale, and sandstone of the Eagle Ford Group before flowing southwest through the Austin Chalk, lower Taylor marl, and calcareous silt and sand of the Neylandville and Marlbrook Marl formations, both formations in the Taylor Group. The assessment began at the confluence of the West Fork and Elm Fork, marking the origin of the Trinity River's main channel (Figure 5), and ended in the Great Trinity Forest, a protected 6,000-acre bottomland hardwood forest that follows the river for 11 miles southeast of Dallas.

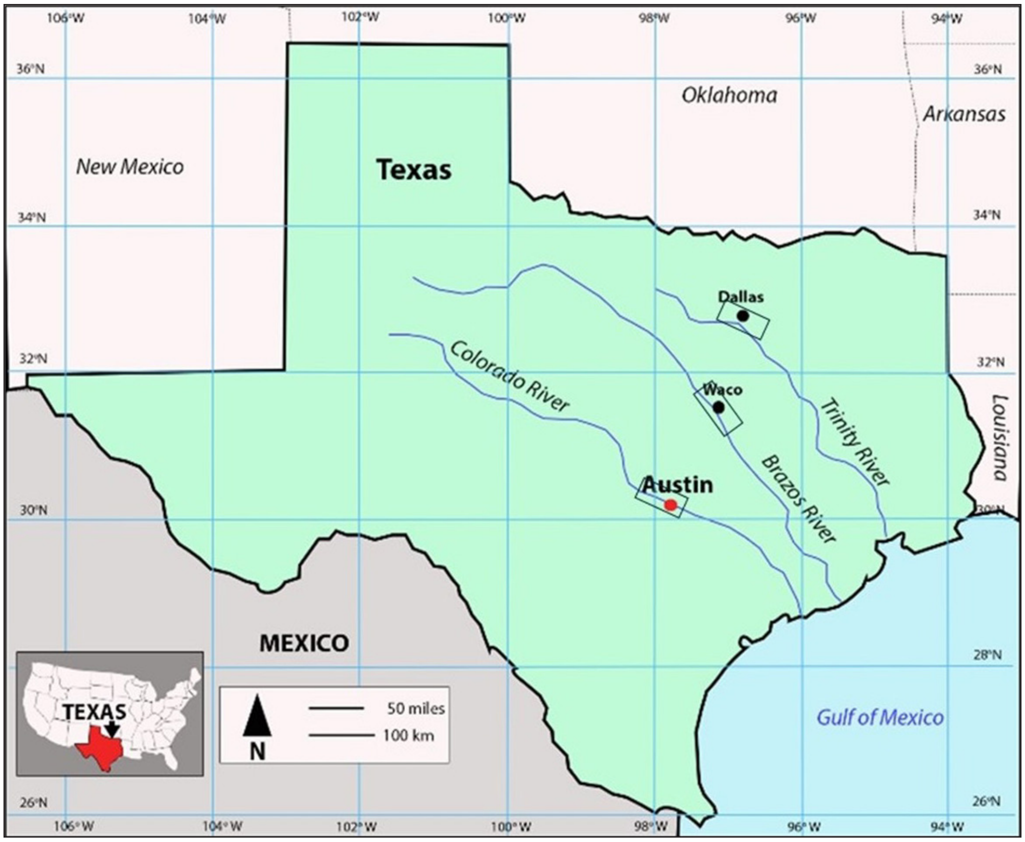

Figure 1. Location of three rivers in study area. Boxes indicate the sampling zones.

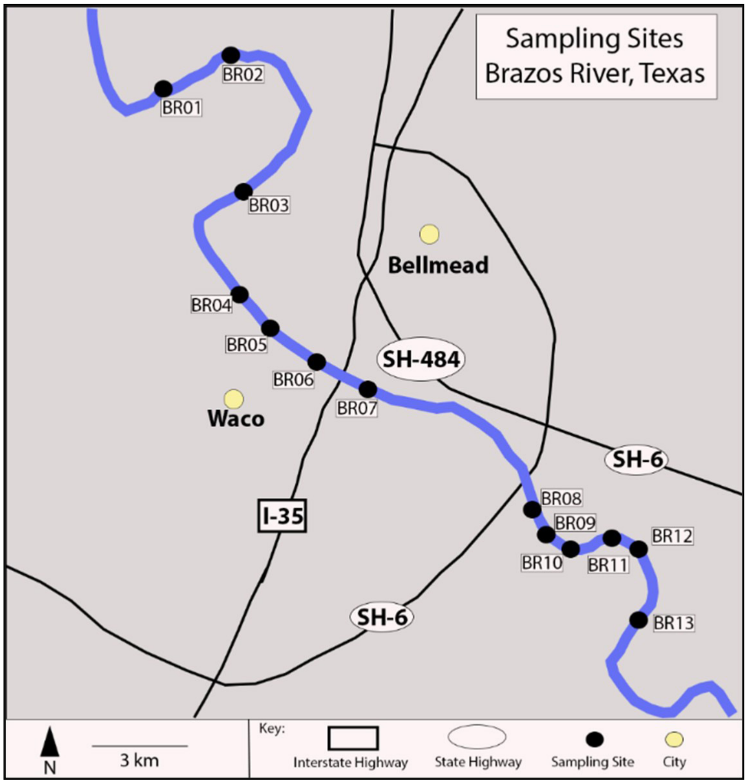

Figure 2. Sampling sites on the Brazos River near Waco, Texas.

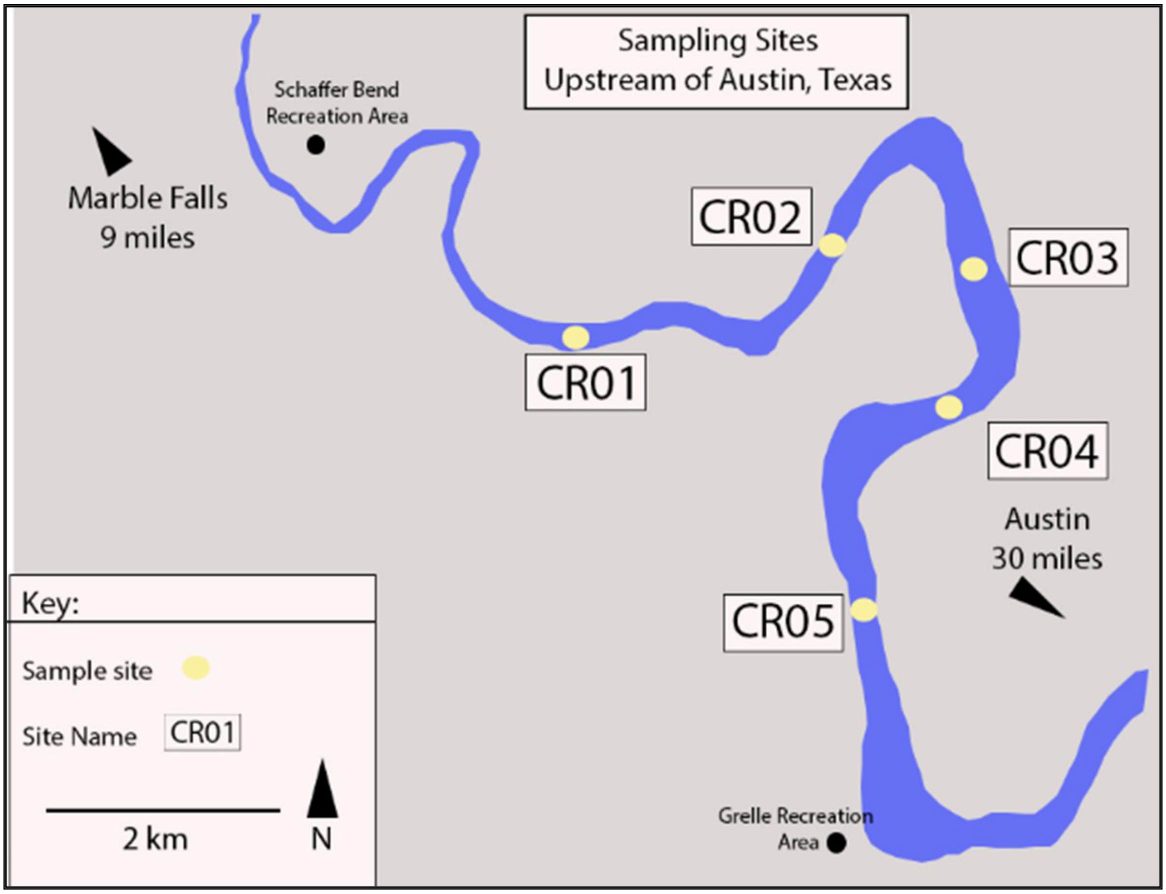

Figure 3. Sampling sites on the Colorado River, upstream of Austin, Texas.

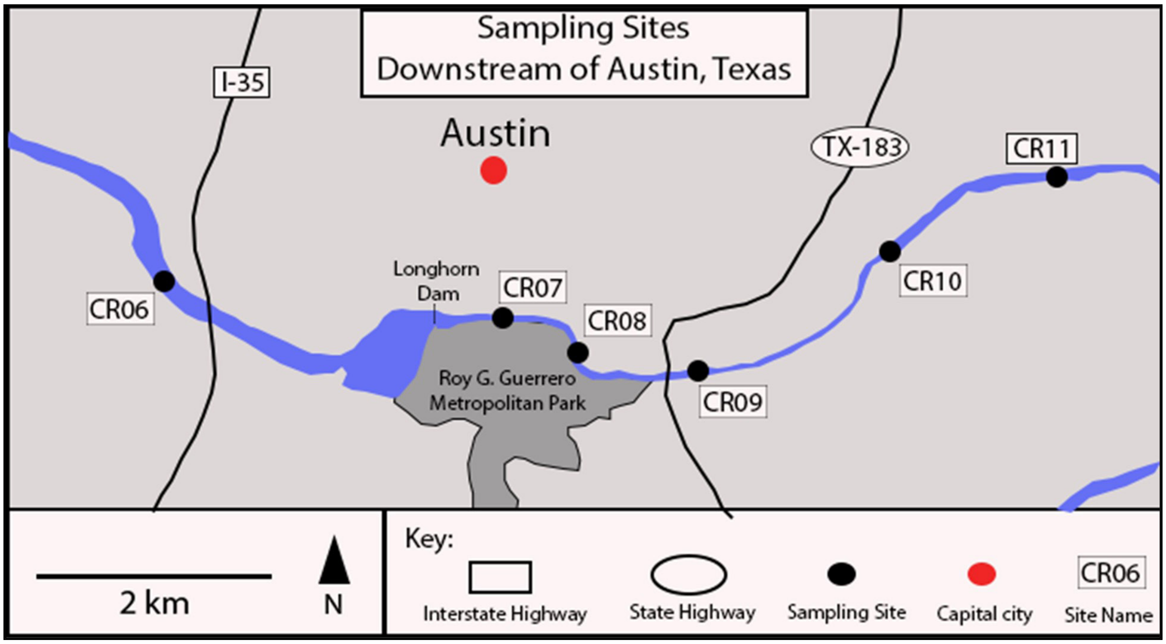

Figure 4. Sampling sites on the Colorado River, downstream of Austin, Texas.

Table 3. Average percentages of sediment sizes from the sample areas in each river assessed for this study.

River	Coarse sand	Medium sand	Fine sand	Very fine sand	Silt
	425–850 µm	250–425 µm	125–250 µm	75–125 µm	<75 µm
Brazos	0.00	0.11	0.45	0.27	0.17
Colorado	0.09	0.23	0.37	0.19	0.08
Trinity	0.02	0.12	0.39	0.21	0.26

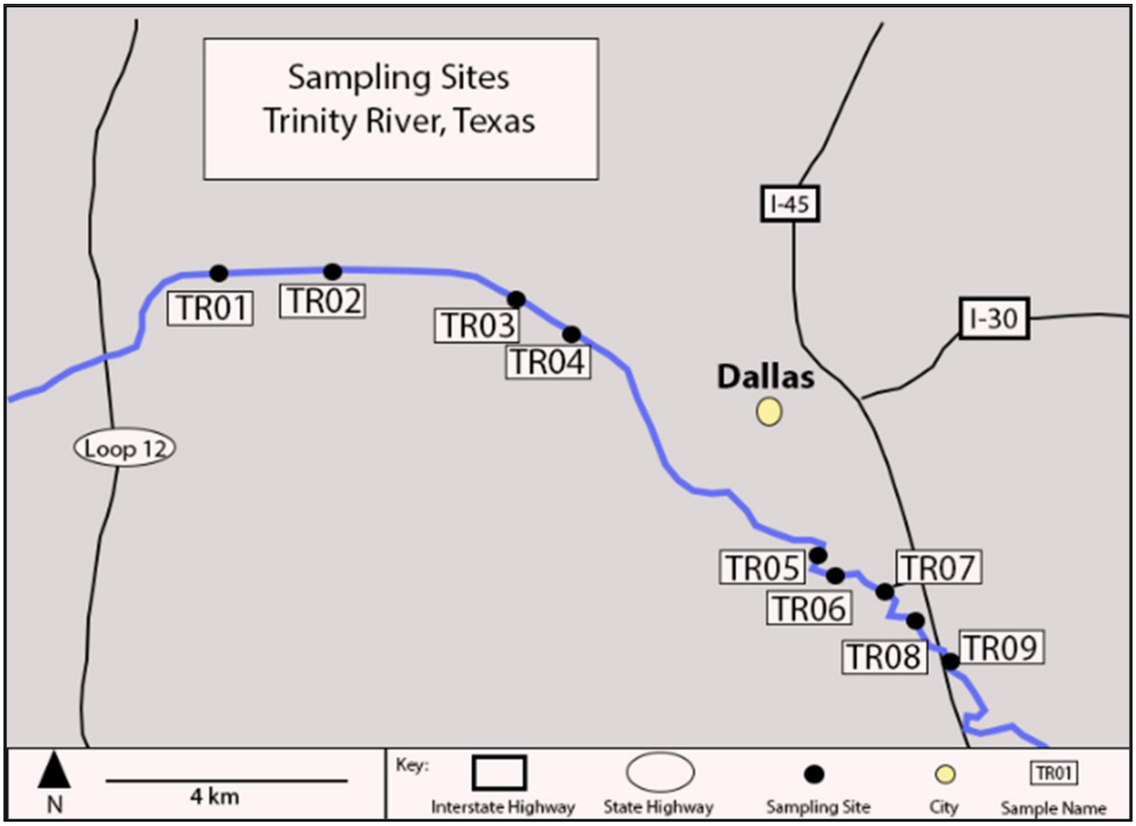

Figure 5. Sampling sites on the Trinity River near Dallas, Texas

In a prior study, Owens (2020) assessed sediment sizes in each study area and found predominant sediment sizes to be very fine to fine sand and silt, except for the Colorado River, where the predominant sediment sizes were of fine sand to medium sand (Table 3).

METHODS

As a contribution to the expansive body of work regarding microplastics in the natural environment, three major rivers in Texas were sampled and assessed for the presence of microplastics in bed sediment. The Brazos, Colorado, and Trinity rivers were selected for the presence of urban centers on their banks. The three rivers of this study were traversed by canoe through reaches between 20–27 km in length. During these field excursions, post hole diggers were used to extract bed and/or bank sediment from multiple stops in the river channel, spaced approximately 1 km apart. Bed sediment was collected at each location from depths accessible with the post hole diggers, approximately 45–60 cm (1.5–2 ft). These locations were selected for their proximity both upstream and downstream of urban settings, although river access limited sampling in some locations. Thirteen samples were taken from the Brazos River upstream and downstream of Waco, 15 samples were taken from the Colorado River upstream and downstream of Austin, and nine samples were taken from the Trinity River upstream and downstream of Dallas.

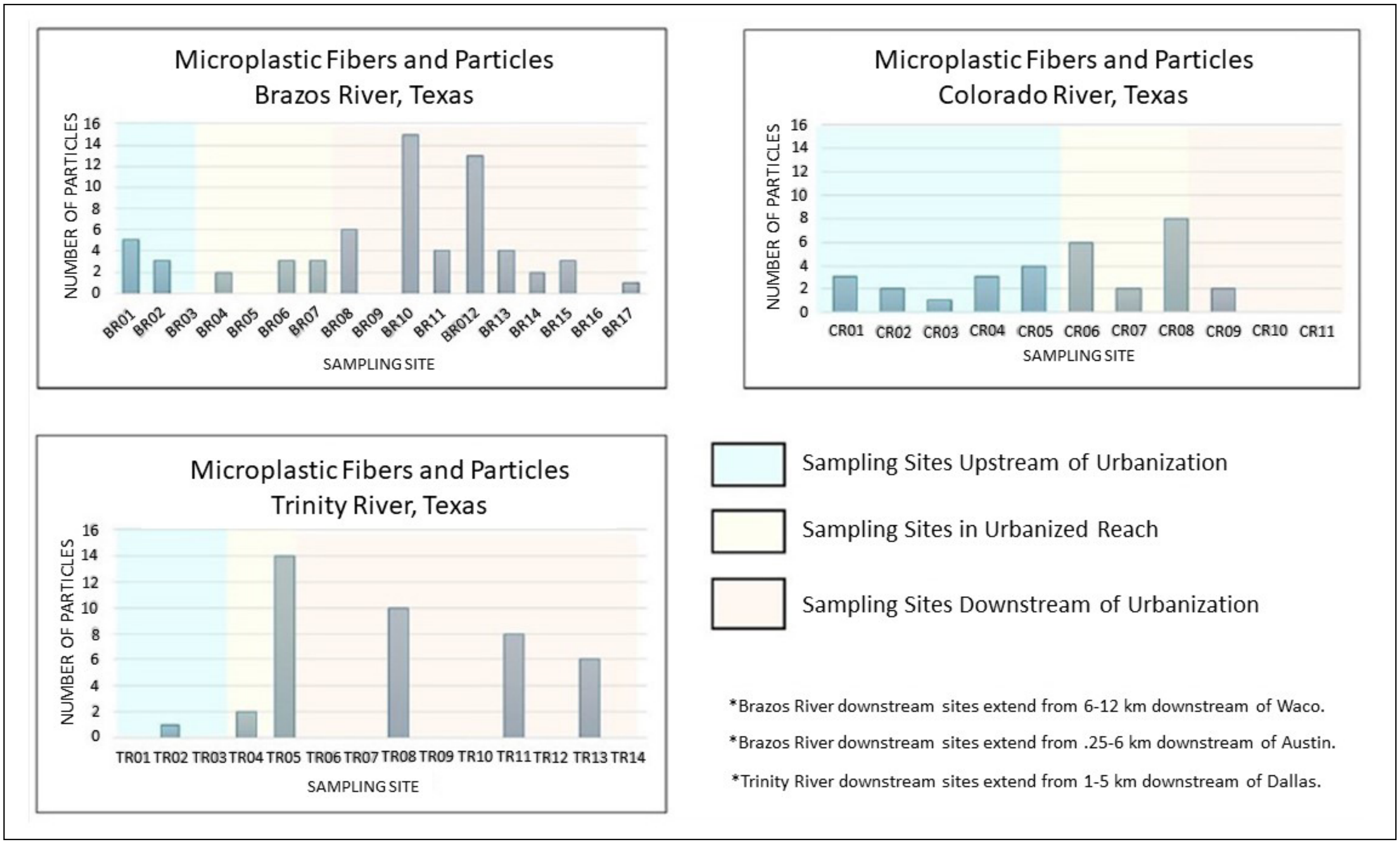

Figure 6. Numbers of synthetic or dyed particles detected by visual inspection and density separation at each sampling site.

Sediment was transported in clear polyethylene plastic bags from the field to labs in College Station, Texas and Tyler, Texas. The bags were kept sealed for the entirety of transport and storage and were stored at room temperature away from direct sunlight. Sediment was placed in a drying oven at 80 degrees °C until fully dried, at least 30 minutes. Sediment was then sieved for 20 minutes using mesh sizes 2–4.76 mm, 850 µm–2 mm, 425–850 µm, 250–425 µm, 125–250 µm, and 75–125 µm. This resulted in 178 sieved sediment samples from the 41 sample sites.

Density separation analysis was performed on the sieved samples. A subsample (7–9 g) of each sample was combined in a centrifuge tube with a 3.3 M solution of NaI, as outlined in Dikareva and Simon (2019). Centrifuge tubes were hand-shaken and then centrifuged for 5 minutes. After settling, a pipette was used to remove sediment floating at the top of the NaI solution. This was applied to a cellulose filter that was allowed to dry in an enclosed environment and protected by cellulose-based cover.

Subsamples were visually inspected under a dissecting microscope for presence of microplastics and synthetic microfibers. As visual inspection was the primary identification method in this study, a cautious approach was used to avoid overestimating microplastic numbers. Key visual indicators for likely synthetic origins included unnatural colors, lack of cellular structure, and lack of natural surface irregularities (excessive smoothness or homogeneity). Because the sediment was stored in clear polyethylene bags, no clear fibers were included in the microplastic count. Black fibers were also excluded from the final count because of the prevalence of organic black fibrous material in the natural environment. This likely creates an underestimate of the prevalence of microplastics in the environment, as it excludes fragments of tire and road wear. Underestimation, however, was deemed preferable to overestimation, when establishing a baseline of microplastic contamination in these rivers for management purposes.

RESULTS

From the 178 subsamples considered, 108 microplastics and microfibers were detected (Figures 6 and 7). The overwhelming majority (93%) were fibers and the remaining 7% were angular fragments.

The most common colors of microplastic and microfibers detected were blue (41%) and red (17%) (Figure 8). Because this study relied predominantly on visual inspection, clear fibers were not included in the final count, as they bear a strong visual

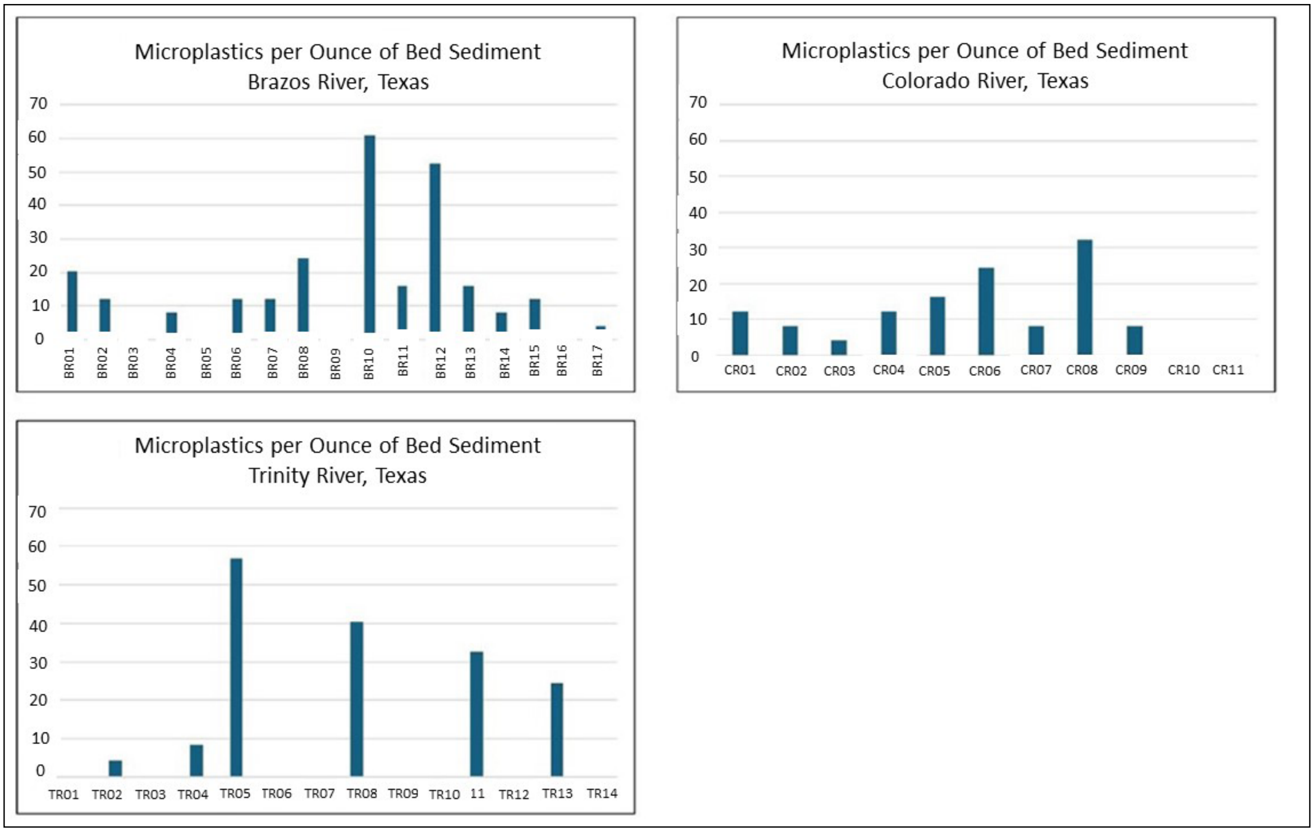

Figure 7. Microparticles per ounce of bed sediment for each river sampled.

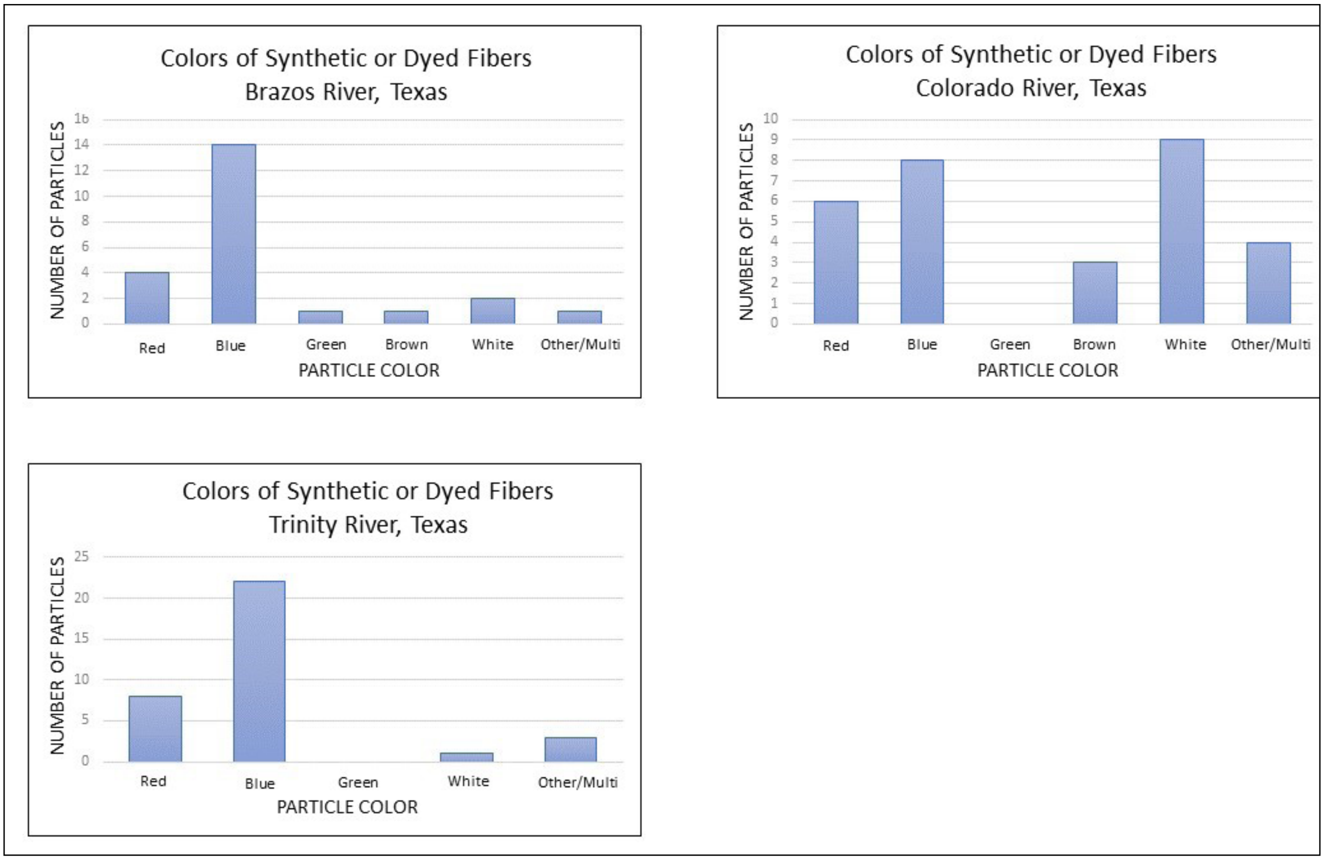

Figure 8. Color distribution of synthetic or dyed particles.

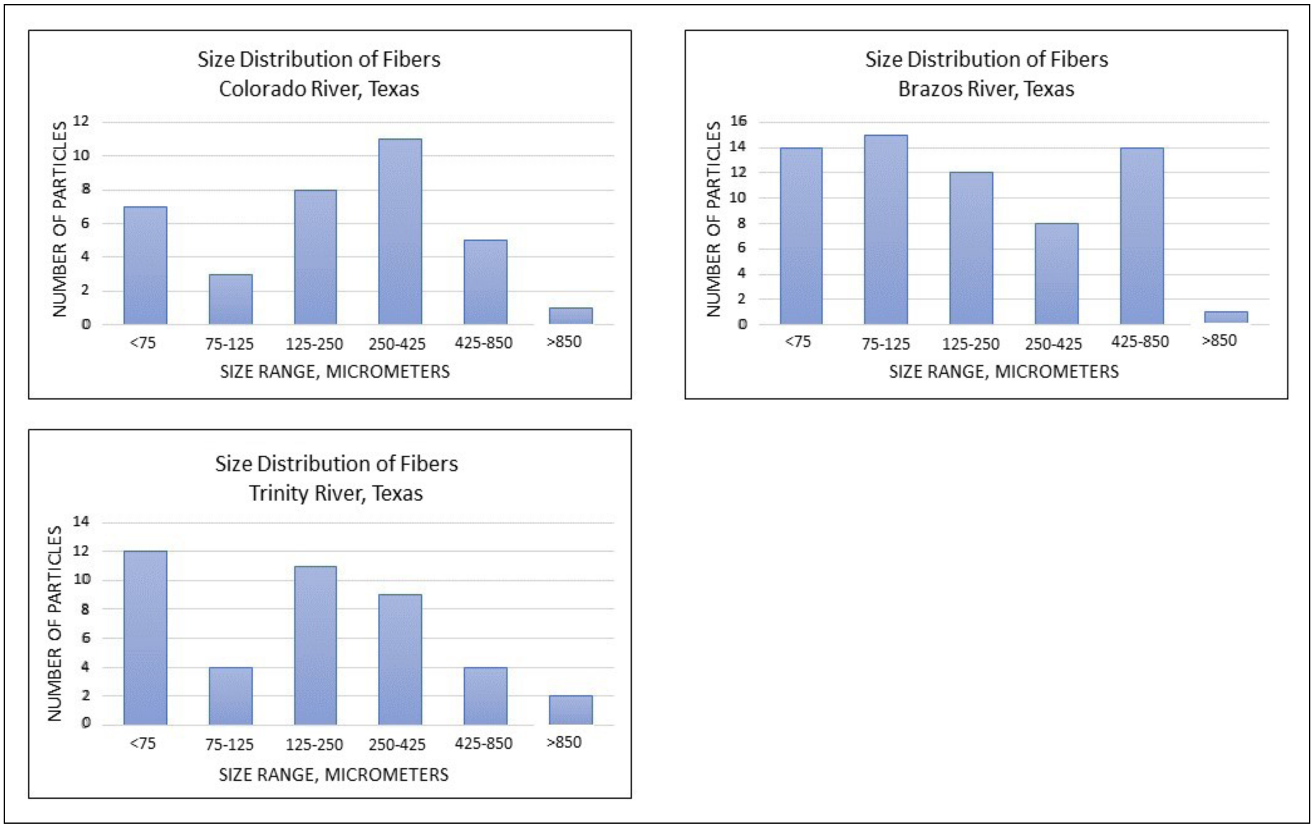

Figure 9. Size range of microplastic particles in the sample areas.

resemblance to plant lignin. Had clear fibers been included, they would have accounted for 39% of all fibers detected.

The highest levels of microfibers and fragments in all three rivers studied were close to their respective urban centers, with concentrations diminishing with distance downstream. The largest reduction for each river was between the first and second sampling sites downstream of each urban area, with occasional peaks downstream in the vicinity of picnic areas or residential areas. The size range of particles varied by location (Figure 9). In the Brazos River, the most common size range of microplastics was the 75–125 µm range (32%), followed by 425–850 µm (24%) and 125–250 µm (22%). The Brazos River showed a marked spike in microfibers immediately downstream of Waco, but this number decreased with distance downstream. In the Colorado River, the most common size range was the 250–425 µm range (31%), followed by the 125–250 µm range (23%). The Colorado River showed a gradual increase in microparticles as it flowed downstream towards Austin, with the highest amounts recorded immediately downstream of downtown Austin. In the Trinity River, the most common size range was 125–250 µm (28%), followed by the <75 µm range (25%). The Trinity River showed the highest levels of microplastics and microfibers in Dallas, which gradually dropped downstream of the city.

DISCUSSION

The prevalence of microplastic and microfiber particles in fluvial systems presents challenges to researchers and practitioners in geomorphology, river management, and ecology.

Results in this study showed that approximately 93% of suspected plastic or dyed particles were fibers, similar to estimates of worldwide microplastic pollution types (Woods et al., 2018). The most common microfiber color (41%) was blue, similar to other studies focusing specifically on microfiber pollution (Gago et al., 2018; Athey et al., 2020). Athey et al. (2020), for example, found predominantly blue or clear microfibers in sediment and fish in the Great Lakes region. Many of those microfibers were attributed to denim jeans, which release both clear, unaltered cotton fibers and blue fibers that have been treated with synthetic indigo dye.

Respective particle sizes are likely impacted by each river's discharge and level of urban sprawl or industry surrounding each urban center. The Trinity River, for example, shows higher average discharge near Dallas than the Brazos River does near Waco, which correlates with higher sediment sizes, in general. Owens (2020) found that average maximum bedload sediment sizes on the Trinity River in this region reach 300 µm, whereas in the Brazos River study area, they remain less than 250 µm. (It is worth noting that these represent sediment sizes specifically in the area of this particular study.) The Colorado River had maximum sediment sizes approximating 500 µm upstream of Austin, but this does not align with the location of the largest microplastic particles detected in this study. Rather, the largest microplastic particles on this stretch of the Colorado River were found immediately downstream of Austin, downstream of Longhorn Dam, which forms Lady Bird Lake. This may represent an accumulation of particles that were impounded in the lake and then passed through the outlet.

The data show that urban centers are clear sources of microfiber pollution in the river systems studied. In each study area, the prevalence of plastic or dyed microparticles peaked immediately downstream of urban centers and then decreased with distance downstream. The sources associated with urban areas are likely wastewater treatment plants (Mishra et al., 2022), household laundry operations (Cesa et al., 2020; Galvão et al., 2020), and direct contribution from human activities such as fishing and swimming. Fibers are released into the environment at source points such as these and then taken up by organisms and/or settle out in sediment. Waste management may play a role in the abundance of microplastics and microfibers introduced to waterways.

FUTURE RESEARCH OPPORTUNITIES

Fourier-transform infrared spectroscopy (FTIR) analysis should be used to determine the specific composition of each synthetic particle collected during this study to determine proportions of synthetic material types in the environments. The role of discharge and velocity in microplastic and synthetic particles in sediment should also be assessed. In general, further research into the role of microfibers, specifically, is needed to predict the potential impacts on ecological and human health, as well as potential changes to channel adjustment. Their small size presents novel challenges in detection and identification, but their abundance in ecosystems and overwhelming proportion of overall microplastic pollution warrant an emphasis on research into their impacts.

The role of microplastics on biogeomorphology through their impact on microorganisms also warrants further research. For example, Rummel et al. (2017) called for investigating the conditioning film on microplastics and its close link to the concept of the eco-corona, a layered structure formed by the interaction of biomolecules and nanoplastics or microplastics. Liu et al. (2022) showed that the eco-corona impacts toxicity of nanoplastics, but understanding the function of nanoplastic characteristics in the production of eco-corona remains a challenge.

Microplastic particles respond to flow characteristics in a different manner than natural sediment, making their transport from the terrestrial realm to the marine realm an opportunity for new research into sediment entrainment and mobility. With the growing abundance of microplastics and nanoplastics and difficulty in their remediation, research into the effect of their transport on the geomorphic adjustment of fluvial systems is another field for potential research.

CONCLUSION

The ubiquitousness of microplastics and nanoplastics in the environment presents a variety of concerns. The impact of their constituent chemicals on human and ecological health has been demonstrated in prior studies, as has the potential impact of microplastic and nanoplastic presence on geomorphological and sedimentological processes. In this study, sediment samples from the Brazos, Colorado, and Trinity Rivers were analyzed for the presence of microplastic particles. Microplastics or microfibers were found in the majority of the sediment samples analyzed and were most abundant immediately downstream of urban centers. Particles were overwhelmingly in the form of microfibers, especially those dyed blue. Future research should determine the proportions of different composition types of the microfibers and particles detected, so that regulations and wastewater operations may respond to the growing concern of the impacts of these particles. The health and management of Texas rivers will increasingly rely on our understanding of the distribution of these particles. Plastic production is not predicted to decrease in the near future, and until domestic recycling capability and availability are significantly improved, these particles will become an increasingly prominent component of our stream networks.

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