Microplastic contamination in the Avon/Ōtākaro River, Christchurch, New Zealand

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A thesis submitted in fulfilment of the requirements for the degree of

Master of Science

In

Environmental Science

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by

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March 2020

Abstract

Microplastics (< 5 mm in diameter) are detrimental to the health of aquatic organisms and freshwater ecosystems. Freshwater research in New Zealand has been limited to date with no investigations being conducted into the concentrations of microplastics in urban freshwater systems in Christchurch. Thirty river sediment samples were collected from six zones along the Avon/Ōtākaro River, resulting in the identification of 156 microplastic particles. The mean abundance was 5.2 particles per 100 g^{-1} of sediment. The frequency of polymers was PET (41) > PP (33) > PS (29) for the lower reach of the river. Polyester (15) and nylon (14) were present primarily in higher population locations in the middle and upper reaches of the Avon/Ōtākaro River. Fragments (47%) were the most abundant morphotype. Microplastic size on average was between 100 – 300 µm (43%). Red (27%) and blue (26%) were the most commonly identified colours. Microplastic accumulation zones were influenced by flow rate, land elevation and tributary infiltration.

In stormwater drain sediment samples, microplastics were present at a lower mean abundance (3.3 particles per 100 g^{-1} of sediment). From the fifteen samples, 45 microplastics were identified from stormwater drain sediments and thirteen polymers were isolated. The frequency of detection for polymers was nylon (8) > rubber (6) > PP (6) in all reaches of the Avon/Ōtākaro River. Fragments (66%) and microplastics between 500 – 1000 µm (34%) were most abundant. White (20%) was the most commonly identified colour. A range of external coatings were identified on microplastics including polymers used in transportation (ABS, rubber, PC), additives, building materials, and recreational activities. There was a relationship between land use and the type of microplastic isolated from stormwater drain sediments. Overall, there was no relationship between river and stormwater drain sediment microplastics. These results indicate land-based sources such as land overflows and road runoff as well as stormwater drains influence river microplastic accumulation. The implications of this study provided baseline data for the Avon/Ōtākaro River which can be utilised for future Christchurch research and as a reference for future New Zealand freshwater microplastic research.

Acknowledgements

I would like to acknowledge the support of my two supervisors: Associate Professor Sally Gaw and Dr. Olga Pantos for going the extra mile as supervisors for me. I would also like to thank the support of Helena Ruffell, Kimberley Kovacs-Wilks, and the Environmental Science team for their help in the lab and technical assistance. Thank you to Nick Oliver and Wayne Mackay for their installation and expert knowledge on running mechanical equipment. Thank you to Michaela Bould for the use of Environment Canterbury's GIS layers. Thank you to Michele Stevenson of Environment Canterbury and Kirsty Patten of the Christchurch City Council for their expertise around the history of the Avon/Ōtākaro River. Additional gratitude needs to be given to Fraser Doake and the team at ESR for their support, time and expertise during sampling analysis. I would also like to thank Roger Creswell at Lincoln University for his assistance regarding total organic carbon analysis.

Thank you to the Waterways Centre for Freshwater Management for financial support during my research and for the opportunity to present at the annual Waterways Post-graduate conference. I would also like to thank the AIM^2 – Aotearoa Impacts and Mitigation of Microplastics team for allowing me to present at the annual conference in Nelson. Lastly, thank you to my family for their continual support throughout my university education and to my partner Emily for her unconditional support and assistance during my sampling.

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1. Introduction

1.1. Urban rivers

Urban rivers are highly valued both in New Zealand and internationally for their environmental, visual, and cultural significance (Morris & Ruru, 2010). For Maori, waterways are a source of life and are regularly used for the transportation of goods (Morris & Ruru, 2010). All rivers have a distinctive catchment, ecosystem, and hydrological processes. For example, the spring-fed Avon/Ōtākaro River provides high-quality water for aquatic ecosystems as well as being a source of food and material for local iwi (White, 2009; Main and Taylor, 2010; Watts, 2011). Rivers are used for a multitude of activities including recreational swimming, fishing and boating. Culturally, rivers have always been at the heart of global cities as a trade route, source of food and a sink for the drainage of water from viable land (Coates, 2013). Pollution in rivers have detrimental effects on aquatic ecosystems including loss of biodiversity and changes to the food chain (Auta et al., 2017). Microplastics are one example of pollutants entering urban waterways (Ding et al., 2019). This thesis focuses on the concentration and types of microplastics within river sediments and stormwater drain networks adjacent to the Avon/Ōtākaro River, Christchurch.

1.2. Plastics

Plastics are a synthetic or semi-synthetic polymer compound that are malleable hence, various shapes can be formed for different applications. Plastics are identified as any synthetic polymer compound but can also include chemically altered natural materials. There are two forms of microplastics; primary and secondary (Blair et al., 2019; Jaikumar et al., 2019). Primary microplastics refer to plastics that are deliberately designed to be smaller than 5 mm in size, such as microbeads found in cosmetics. Secondary microplastics are plastics originally greater than 5 mm in size which have been fragmented through friction, biodegradation or chemical alteration in the water system (Laskar and Kumar, 2019). The lifespan of plastics is variable depending on its composition and environmental factors such as sunlight, wind and rain but can be decades to centuries (Barnes et al., 2009; Imholf et al., 2013., He et al., 2019; Slootmeakers et al., 2019). Plastics can contain additives such as metals and UV stabilisers (Cantwell et al., 2015; Pacheco-Juarez et al., 2019) or antioxidants which can also be added to plastics to prevent oxidation, however, numerous

non-intentional added substances (NIAS) can form from the antioxidant reaction to oxygen in the fragmenting process (Bartsch et al., 2018).

1.2.1. Plastic polymers

Polymers are large molecules made of simplistic repeating units called monomers (Figure 1.1). Several polymers such as polyethylene terephthalate, polyethylene, polypropylene, polyvinyl chloride, polystyrene, and polycarbonates are forms of synthetic resin-based thermoplastics (Azapaic et al., 2003). Other plastics such as polyurethane and epoxy resins are deemed as thermoset plastics (Burns and Boxall, 2018). Many of these plastics were discovered accidentally during heating and chemical mixing phases in the early 20th century (Billmeyer, 1972). Polymers tend to vary in composition, colour and material. This is influenced by the chemical additives and organic composition used during production or environmental conditions such as sunlight exposure (Mascia, 1982; Haave et al., 2019).

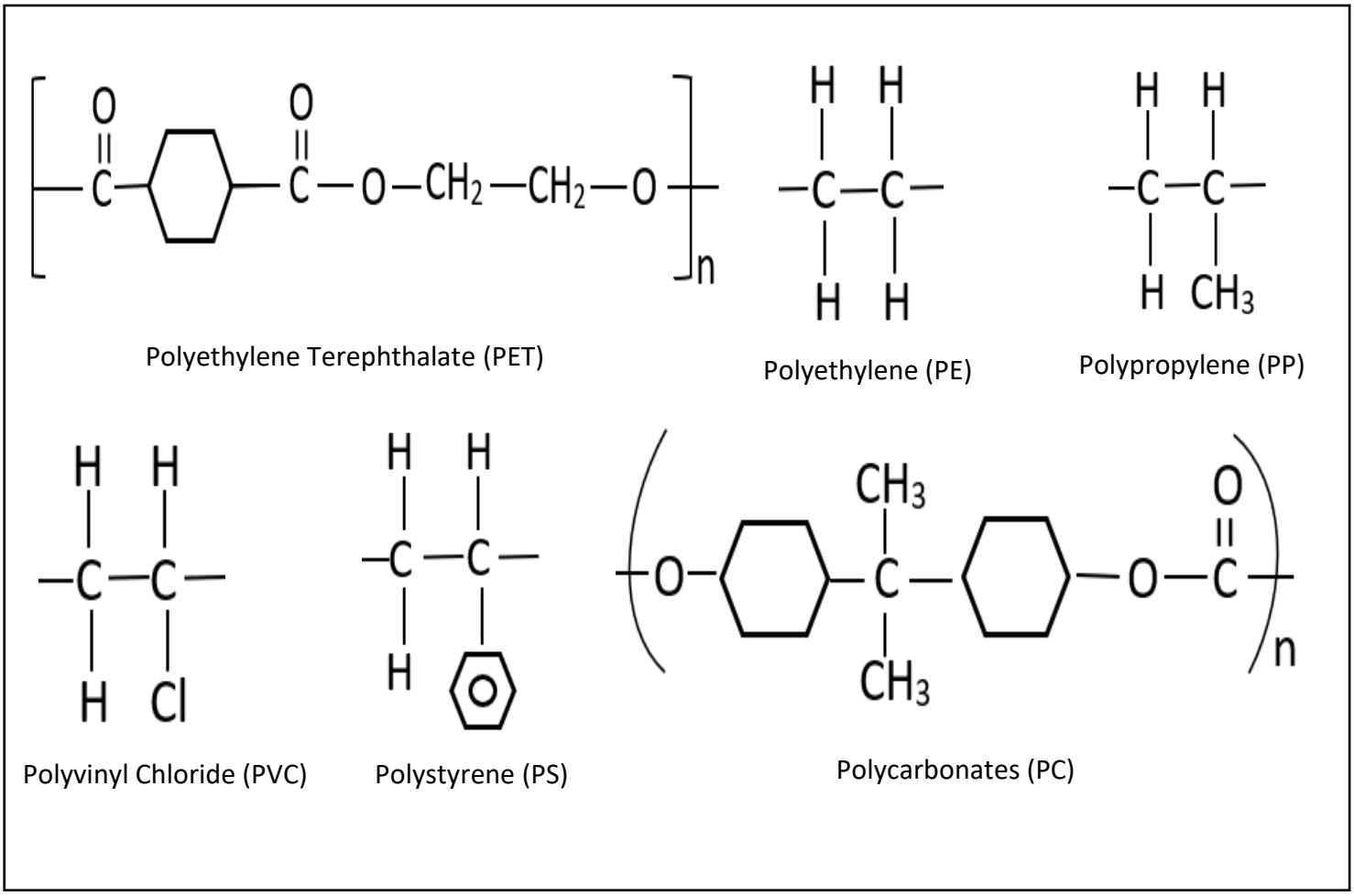

Figure 1.1. Chemical structures of monomers for PET, PE, PP, PVC, PS, and PC thermoplastics.

1.2.2. Polyethylene Terephthalate (PET)

Polyethylene terephthalate (PET) is formed through the composition of petroleum monomers ethylene glycol and terephthalic acid (Westerhoff et al., 2008). PET is commonly used for containing soft drinks, juice, cooking oils and other liquids. It is designed for single-use applications as it can be recycled but not reused due to potential carcinogenic material that may be released during leaching (Martin et al., 2017). Internationally, polyethylene terephthalic is regarded as one of the highest contributors of microplastics due to its abundance and wide use in packaging (Lacerda et al., 2019). It is the most common form of microplastic present within the Atlantic Ocean and along North America's eastern coastline (Morét-Ferguson et al., 2010).

1.2.3. Polyethylene (PE), High and Low-density Polyethylene (HDPE and LDPE)

Polyethylene (PE) is one of the most frequently used plastics in production. It is formed from the opening and linking of the ethylene molecule [C_2H_4]. Ethylene is composed of two carbon alkane units [CH_2CH_2] (Liu et al., 2019). Traditionally, polyethylene can be divided into high-density and low-density formulations. The composition of HDPE and LDPE are determined by where the hydrogen atoms attach to the carbon atoms (Hazrat et al., 2015). They tend to be light in tensile strength compared to most thermoplastics (Table 1.1).

High-Density Polyethylene is a tightly compact plastic which has minimal branching within the polymer chain allowing for less movement and more cohesive strength. The high tensile strength allows for the plastics to be used in outdoor furniture, plastic chairs and other long-lasting products (Barletta et al., 2017). Due to its density, these plastics will likely directly sink to the riverbed causing issues for benthic creatures.

Low-Density Polyethylene when compared to HDPE, is more flexible and ductile due to its side branching approach (Liu et al., 2019; Rodriguez-Seijo et al., 2019). It is the second-highest polluting plastic producing 64 million tonnes of waste annually (Geyer et al., 2017; Prakesh and Goel, 2019) Being of lighter density, LDPE will float on the surface of a water column. As it is also difficult to degrade within the freshwater ecosystem, it becomes a prevalent plastic polymer (Estoppey et al., 2015).

1.2.4. Polypropylene (PP)

Polypropylene plastic has the molecular formula C_3H_6 and next to ethylene, it is the simplest alkene hydrocarbon (Malpass and Band, 2012). Polypropylene contains an extra carbon alkane unit [$–CH_2(CH_3)CH_2$-] (Azapagic et al., 2003; Foroozan et al., 2019). It is used in a variety of industrial, household and commercial uses due to its lightweight, strength and durability. It does not absorb water like PE and PVC, nor is it degraded by bacteria or mould (Zhu et al., 2017). It is more buoyant than PE plastics in a water body as it has a lower density than water (1g/cm^3) (Table 1.1). Polypropylene plastic is easily mobilised in waterways and can travel further than PE plastics before fragmenting (Hoellein et al., 2019).

1.2.5. Polyvinyl Chloride (PVC)

Polyvinyl Chloride [CH_2=CHCl], is manufactured by reacting ethylene with hydrogen and oxygen (Skórczewska et al., 2018). PVC comes in two formulations: either as rigid / unplasticised PVC (rPVC/uPVC) or, flexible PVC. Flexible PVC is most common and is widely used in household electrics, construction and insulation. PVC is widely used in the building applications (spouting, tubing) and health industries (blood bags, sterilisation equipment) due to its economical and versatile qualities. PVC has a density of 1.3 to 1.7 g/cm^3 which is considerably heavier than PE and PP densities of 1g/cm^3. Lower concentrations of PVC are measured in water when compared to PE and PP polymers due to its greater density (Koelmans et al., 2019). PVC can leach toxic chemicals such as phthalates, dioxins, and cadmium when melted but will generally maintain its structure in water or cooler conditions (Diepens and Koelmans, 2018).

1.2.6. Polystyrene (PS)

Polystyrene is created through suspension polymerisation where ethylene is mixed with benzene in the presence of aluminium chloride to create styrene CH_2=CHC_6H_5 (Wan et al., 2019). Polystyrene can be used in foam formation as an absorber in delivery packaging and in solid formation for food packaging. Whilst having useful implications for everyday life, it is easily disintegrated. As it is inert, neither basic nor acidic conditions will destroy the material. Polystyrene is primarily identified within European and Asian freshwater ecosystems (Cole et al., 2015) and is prevalent within stormwater retention basins (Liu et al., 2019).

1.2.7. Polycarbonates (PC)

Polycarbonates are a product of Bisphenol A (BPA), a liquid form of benzene and phosgene (Ranade et al., 2007). Applications of Polycarbonates include vehicle body panelling, light production, bulletproof glass, medical equipment and CD/DVD disc production. Due to the use of bisphenol A (BPA) during production, when PC plastics are introduced to the environment, chemicals can easily leach into waterways.

1.2.8. Morphotype

Morphotypes of microplastics are based on five classes: Fibre, Foam, Fragment, Pellet, and Film. Fibres are a common form of microplastic globally, particularly synthetic materials like nylon and polyester (Figure 1.2). They are primarily derived from washing of clothing but are also found in toys, building materials and sports equipment such as fishing nets (Pico et al., 2019). Fibres can be sub-categorised into natural and synthetic. Natural fibres are extracted from living organisms and are grouped based on plants (cotton, hemp, linen) or animal sources (fur, silk, wool). Synthetic fibres (acrylic, fibreglass, polyamide, nylon, polyester, latex) are produced through chemical alterations and are commonly found within the environment, especially in wastewater treatment plants (WWTP's) (Conley et al., 2019). Fibres such as polyester, when released by washing machines can be fragmented by the influence of detergents with warm waters allowing for a faster breakdown of materials compared to other synthetic fibres (Yang et al., 2019). Polyester garments such as fleece can release > 1900 fibres per washing cycle whereas cotton and rayon will release >100 (Browne et al., 2011; Zambrano et al., 2019).

Fragments are mostly derived from larger macroplastics (greater than 5 mm) which have been broken down within the freshwater system. Next to fibres, fragments are the most commonly isolated morphotype in microplastic investigations consisting of PE, PP, PS and PET polymers (Barnes et al., 2009). The location of fragmented microplastics is dependent on the land use, urbanisation and watershed characteristics of tributaries and the main freshwater system (Farenfeld et al., 2019). Foam, pellets and film morphotypes are present in lower concentrations internationally. Developing nations are more likely to see foam, pellet and film microplastics from textile and industrial production (Stranton et al., 2019).

Table 1.1. Polymer density and tensile strength based on international literature on common microplastic polymers (adapted from Billmeyer, 1972 and Bond et al., 2018).

Polymer	Density (g / cm^3)	Tensile Strength (MPa)
Polyethylene Terephthalate	1.38 – 1.45	690
Polyethylene	0.88 – 1.00	20
High-Density Polyethylene	0.94 – 0.98	28
Low-Density Polyethylene	0.91 – 0.94	10
Polypropylene	0.94 – 1.00	28
Poly-vinyl Chloride	1.30 – 1.65	59
Polystyrene	0.96 - 1.04	42
Polycarbonates	1.20 – 1.22	70

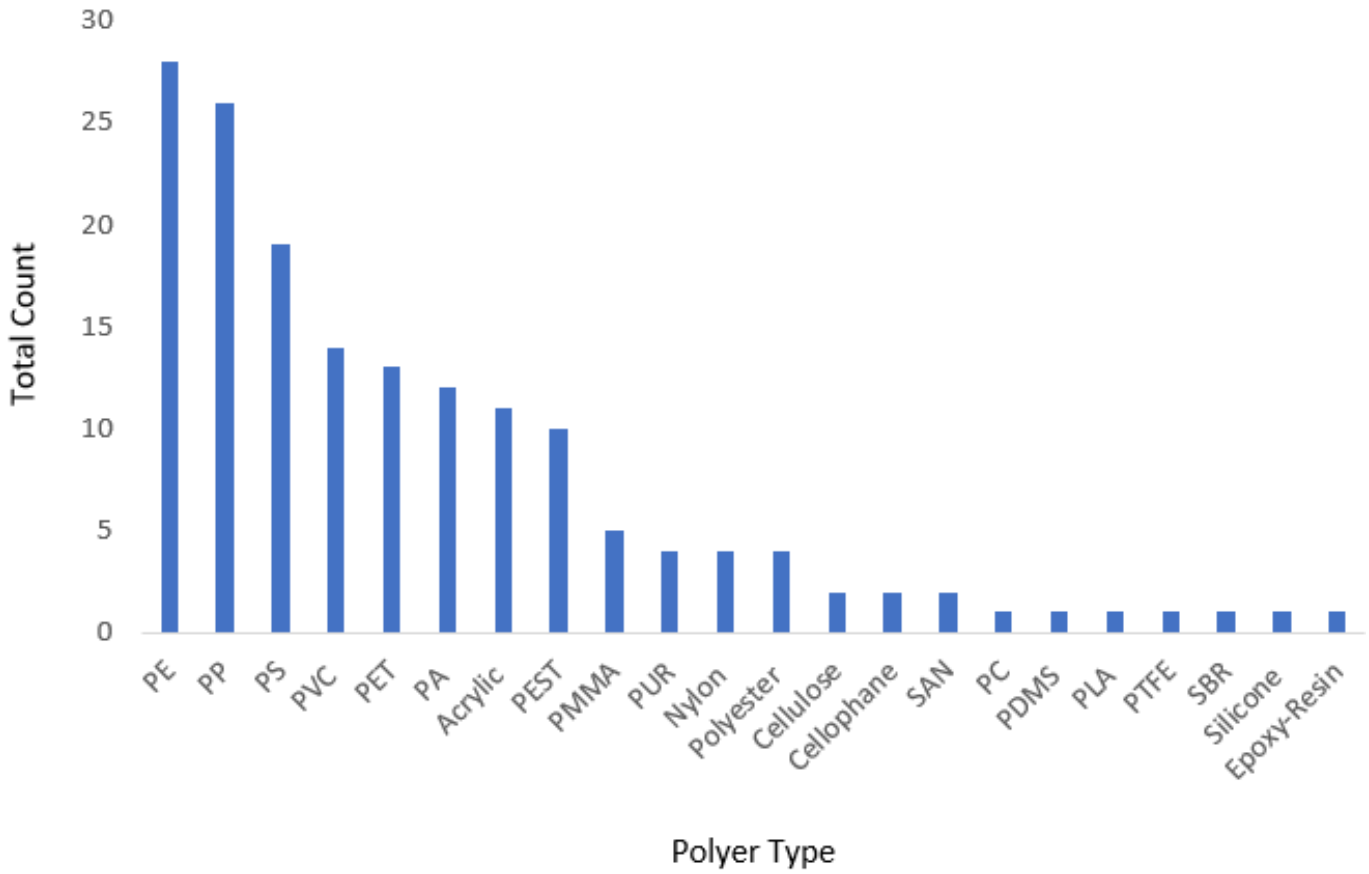

Figure 1.2. Identification of different microplastic polymers based on international scientific research (adapted from Fahrenfeld et al., 2019 and Koelmans et al., 2019).

1.3. Microplastics in rivers

Rivers are a potential source for microplastics to enter coastal environments because of the multiple land-based inputs that distribute surface and wastewater (Hoellein et al., 2019; Schirinzi et al., 2019). Microplastics enter urban waterways through a series of networks ranging from stormwater drains, wastewater disposal, river confluences and aeolian processes. Urban rivers are regarded as the predominant pathway of microplastics from urban settlements to the oceans. These microplastics become significant environmental hazards once in the rivers due to their biological and chemical interactions, ability to be allochthonous in behaviour and risk of submersion under river sediment (Hoellein et al., 2019). Globally, it is estimated that 1.2 and 2.4 million tonnes of plastic per year flow from rivers into oceans and estuaries (Bruge et al., 2018) especially in locations with higher microplastic production and poor waste management plans such as in Asia, South America and the east coast of Africa (Figure 1.3) (Sadri and Thompson, 2014; Shruit et al., 2019).

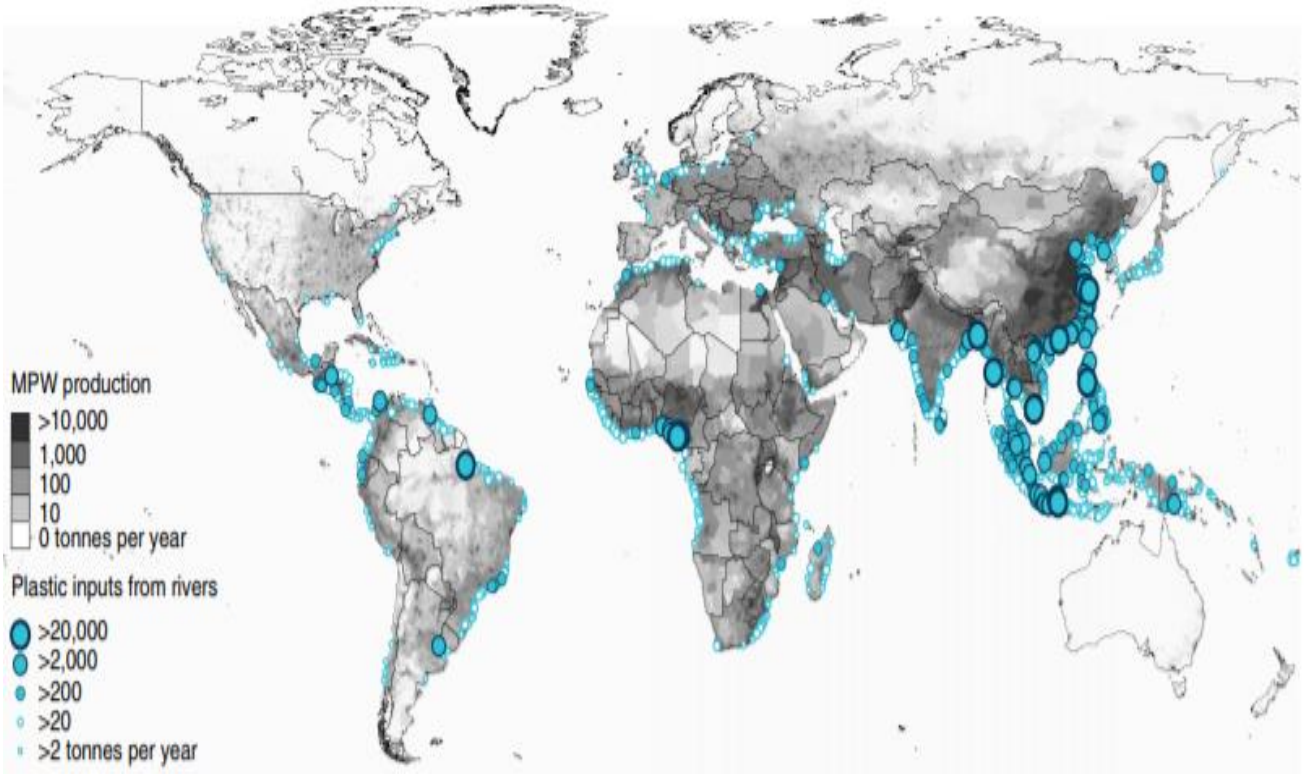

Figure 1.3: Total mass of river plastics entering the oceans (tonnes/yr^{-1}) compared to total microplastic waste production (tonnes/yr^{-1}) (adapted from Lebreton et al., 2017).

1.4. Sources of microplastics

International studies on microplastics have primarily focused on marine environments (Galloway and Ceri, 2017) and the implications on marine organisms (Wright et al., 2013). Only recently have studies shifted towards urban rivers and streams (Jambeck et al., 2015; Schmidt et al., 2017). Microplastics are comprised of synthetic thermoplastics (polyethylene, polypropylene, polystyrene and polyethylene terephthalate), vehicle waste (vehicle rubber, brakes, lights and external body panelling), synthetic fibres from clothing and other forms of apparel, abrasives and personal care products (microbeads) (Van Wijien et al., 2019). Urban runoff from stormwater drains or wind are a considerable source of microplastics as there is limited to no dilution of water or treatment conducted allowing for water to flow directly into the river system (Sutton et al., 2016; Lasee et al., 2017).

1.4.1. Urban sources

Research into microplastic quantities in urban areas across Europe, Asia, and America has gained significant attention due to the impacts on the aquatic environment and health of people residing in urban locations. These investigations have primarily focused on urban settlements, wastewater treatment plants and on rural settlements downstream of cities. Most research on urban sources has been within major Asian and European cities' river networks (Wang et al., 2017; Wu and Xiong, 2017; Mendoza et al., 2019). Results have reported a universal trend of microplastic concentrations decreasing from urban settlements (Alam et al., 2019). This shows the highest identifiable sources of microplastics are from road run off, stormwater drains, greywater and wastewater treatment plants in urban environments. Generic human-made litter (cigarette butts, food packaging), fibrous material from domestic washing, breakdown of fishing nets, and recreational aquatic equipment were the most commonly detected microplastics (Jambeck et al., 2015).

1.4.2. Road wear particles

Road wear particles or 'road dust' are plastic fragments derived from vehicles that fracture from mechanical abrasion, excessive usage, heating and weathering (Kole et al., 2017). External vehicle fragments such as acrylonitrile butadiene styrene (ABS) are a commonly identified plastic pollutant within the environment. Plastic from vehicles are the highest form of non-exhaust pollutants to impact the surrounding freshwater environment. (Keuken

et al. 2010). Road wear particles can include fragments from external body panelling, brake dust, tyre rubber, road paint, traffic lights and headlights (Oper and Friedler, 2010). The concentration of road wear particles can be correlated to the density and volume of traffic conditions (Panko et al., 2012) but is also impacted by road material, surfaces, and organic decomposition (Lee et al., 2013). Plastic usage externally on vehicles has steadily increased since the 1960's (Motte, 1982) and accounts for 8% of the total global plastic production (Azapagic et al., 2003; Galafassi et al., 2019). Compressed moulded thermoset polyester (such as SMC or Fibreglass) are used on external panelling of most vehicles due to their relative light weight and price difference to steel (Motte, 1982). PP and ABS (used on bumpers) and HDPE (fuel tanks) are most likely to be found within road dust due to their exposure to atmospheric conditions and other vehicles (Azapagic et al., 2003). The previous research into microplastics in road wear particles is summarised below (Table 1.2).

Table 1.2. Summarised microplastic morphotypes, size, and concentrations based on international road dust literature.

Location	Morphotype	Size	Concentrations	Sampling method
Bushehr city, Iran Abbasi et al., (2019)	Fibres, fragments, pellets	10 – 140 µm	210 – 1658 MP /10 g^{-1} dust	Plastic dustpan and brush
Tehran, Iran Dehghani et al., (2017)	Fragments, fibres, spheres, pellets	230 – 500 µm	240 – 1934 MP/ 10 g^{-1} dust	Plastic dustpan and brush
Netherlands Siegfried et al., (2017)	Fibres, spheres, fragments	10 – 100 µm	1.09 – 120 g/cm^3	Vacuum
Germany Sommer et al., (2018)	Irregular shaped fragments	20 – 250 µm	.08 – 10.2 g/cm^3	Passive sampler device Sigma-2
Norway Siegfried et al., (2017)	Fragments, fibres, pellets, spheres	30 – 350 µm	1 – 10.4 g/cm^3	Vacuum

Brake dust, compared to tyre wear particles travel further from their source due to their lighter weight and morphotype (Budai and Clement, 2018). Polypropylene is used in most vehicle brake discs but is prone to disintegrating upon heat and friction resulting in fragments being released onto the road (Maxwell, 1994). Of the total road dust identified within one metre of stormwater drains on average, brake dust is a contributor of 20% of all dust found (Adachi and Tainosho, 2004; Kole et al., 2017). Exhaust and combustion input account for 12% of road dust (Kreider et al., 2010; Gunawardana et al., 2012; Gulia et al., 2019).

Micro-rubber fragmentation depends on traffic density and breaking strength. Micro-rubber particles will vary in sizes but are definitively angular in shape (Abbasi et al., 2017). Globally, 15% of all microplastics detected have been derived from micro-rubber which fragments from road tyres during mechanical abrasion (Sutton et al., 2019; van Wijinen et al., 2019). The removal of micro-rubber is dependent on surrounding environments. Cities with high-rise buildings and narrow streets can accumulate 16-17% more micro-rubber than towns and rural locations (Amato et al., 2014). Most micro-rubber samples globally are dominated by black spherical, fragments, fibres and film-like particles. (Hwang et al., 2016; Abbasi et al., 2019; Hüffer et al., 2019).

1.4.3. Airborne plastics

The deposition of airborne microplastics is an unquantified source of microplastics to freshwater. They can travel over vast horizontal and vertical distances through atmospheric transport and have been identified several kilometres from their original source and within mountainous regions (Fox, 2019). Airborne microplastics have been reported within elevated sources such as glaciers and alpine lakes where freshwater runoff feeds river systems (Allen et al., 2019; Ambrosini et al., 2019). Airborne microplastics consist of polyethylene terephthalate (PET), polyethylene (PE), polyester (PES), polyacrylonitrile (PAN), poly (N-methyl acrylamide) (PAA), rayon (RY), ethylene-vinyl acetate (EVA), epoxy resin (EP), and alkyd resin (ALK). Sizes can range from 10 µm to 4 mm (Foekma et al., 2013; Naji et al., 2017; Li et al., 2018; Liu et al., 2019).

Airborne microplastics concentrations are based on surrounding environmental conditions such as wind transport (speed, direction and duration) which can become the primary

distribution pathway of airborne microplastics, accounting for 6-20% of total plastics (Prata, 2018). Precipitation and temperature are also controllers of airborne distribution. As microplastics can become airborne indoors and outdoors, these microplastics are seldomly found in their source location (Alzona et al., 1979). They are isolated from ground level rather than at higher altitudes as buildings and other tall structures block the wind transportation process (Liu et al., 2019).

Airborne microplastics are a significant human health concern as most plastics are released from human activity and generally have some toxicity potential associated with them (Liu et al., 2019). The inhalation of airborne plastics is commonly identified within the literature. Ingestion rates are estimated to be 5-15 particles per year in dense European cities (Alves et al., 2018; Sommer et al., 2018). Globally, a total of 12.6 g/kg^{-1} and 13.7g/kg^{-1} of plastic additives were identified within dust samples being inhaled by humans in average indoor and outdoor conditions from 2014 (Liu et al., 2019a).

Synthetic fibres are the main source of airborne microplastic distribution due to the unravelling of fibres from clothing (Dries et al., 2016; Prata., 2018). Other sources can include traffic and household dust (Karbalaei et al., 2018). Synthetic garments have been responsible for the release of approximately 1900 fibres per wash (Browne et al., 2011) and 1200 – 3000 fibres in the atmosphere (Li et al., 2020). Fibres both synthetic and natural are of most concern to the environment as they have a considerable smaller surface area in relation to other microplastics (Lusher et al., 2013; Stanton et al., 2019).

1.4.4. Wastewater treatment plants (WWTP's)

Wastewater treatment plants are a known source of microplastics due to the incomplete removal during treatment (Magnusson and Noren, 2014; Estahbanati and Fahrenfeld, 2016). Fibrous material from washing and clothing are commonly identified which enter waterways through either effluent discharge or stormwater runoff (Wang et al., 2017), accounting for 20% of the global microplastic levels (van Wijnen et al., 2019). Wastewater treatment plants, depending on the processes implemented, have variable efficiency in the removal of microplastics (Sun et al., 2019). Techniques involving grease and grit have been utilised in the study of microbead movement through wastewater treatment plants. These reports have concluded that while levels of microplastics can decrease from 15.7 particles /L^{-1} to 0.3

particles /L^{-1}, the volume of water being treated will determine the level of microplastics contained (Murphy et al., 2016).

During storm events or heavy rainfall, sewer overflows cause microplastic pollution through the direct mixing of stormwater from urban and road surfaces, untreated wastewater, and re-suspended sewer sediments (Bollmann et al., 2019). As this water does not undergo any treatment during release, microplastic quantities from sewer overflows can be double that of regular wastewater treatment plants. In cities with lower sewer systems standards, higher impervious surfaces, and non-separated sewer and stormwater pipelines, the microplastic release can be even greater in volume (Dris et al., 2018).

1.5. Environmental fate of microplastics

As environmental fate refers to the degradation of a chemical or pollutant once within the environment, it is critical to comprehend the sources (land-based), infiltration locations (stormwater and runoff), transportation routes (drains and tributaries), and depositional locations (confluences, river mouth and estuaries) of microplastics (Mani et al., 2015; Carr et al., 2016; Shruit et al., 2019). Once in the river network, microplastics can be transported several kilometres either through water flow or wind and become deposited anywhere in the aquatic ecosystem (Rezaei et al., 2019). Lighter-density plastics will remain suspended on the water surface whilst heavier plastics (greater than 1g/cm^3) will be deposited into the sediment (Hidalgo-Ruz et al., 2012; Foekema et al., 2013; Yonkos et al., 2014; Mani et al., 2016; Huang et al., 2017).

1.5.1. UV degradation

Ultra-violet (UV) degradation is a response to the altering of a polymers' physical structure due to sunlight exposure. The chemical structure of plastic begins to alter with UV damage. Microplastics that have been degraded by UV radiation have higher sorption capability of heavy metals than regular polymers (Wang et al., 2020). An increase in surface area can lead to an increase in the surface roughness, change topography and alter the chemical characteristics of the plastics (Cooper and Corcoran, 2010). Some polymers (PP, ABS, PVC) can have UV stabilisers (quenchers, hindered amine light-stabilisers) added to protect or minimise the impacts of UV damage or 'photo degradation' (McKeen, 2013).

1.5.2. Environmental parameters

Environmental fate is determined by the properties of the microplastic as well as environmental conditions such as the aquatic conditions in which the microplastic enters (water depth, flow rate, salinity) (Enders et al., 2019). Once in the environment, the composition of microplastics can be influenced by fragmentation, chemical breakdown or bleaching from UV exposure. This can alter the surrounding aquatic environment through an increase in pH level, decrease in oxygen levels, and an increase in total phosphorous, total nitrogen or total carbon. Human influences should also be considered as environmental parameters such as land use adjacent to and upstream of rivers, recreational activities, and human modifications to rivers (dams and drains) which all have implications on microplastic transportation and abundance (Kataoka et al., 2019). The environmental parameters can have temporal, spatial, short-term, and long-term influences depending on the duration of microplastic saturation, stage of fragmentation, buoyancy, and the density of the plastic. If a polymer begins to sink, benthic organisms can become infected. Polymers that float will only impact the water surface but are more prone to weathering and accelerated breakdown exacerbating the leaching process. (Bayo et al., 2020).

1.6. Environmental fate on aquatic organisms

Microplastics can have detrimental impacts on marine and freshwater organisms. As microplastics have complex features (different polymers, densities, additives, morphotypes and sizes), once in the freshwater ecosystems they can float on the surface water, within the water column or sink into the benthic sediments (Wang et al., 2019). Particle density can be altered by fouling as chemical alterations are undertaken by biota which leads to negatively buoyant plastics depositing into the sediments (Karami, 2017). Low- density polymers like LDPE and PP will float within the water column and high-density polymers like PET and PVC will sink and become embedded within the benthic sediments (Franzellitti et al., 2019). As these polymers can settle or be transported throughout the river system, they have the potential to impact on small invertebrates through to lager predatory creatures (Cole et al., 2013; Sanchez et al., 2014).

1.6.1. Feeding behaviour

The uptake of microplastics occurs accidentally due to the inability of organisms to distinguish between plastic polymers and natural food sources. Many aquatic creatures can suffocate or die from the resulting blockages. Fragmented and spherical microplastics are most likely to be consumed by aquatic organisms based on the geometric similarity to regular food (Wang et al., 2019). Feeding behaviours of organisms that live close to river mouths, estuaries and deltic coastlines can be impacted from accidental ingestion due to the high proportion of land based microplastics (Gu et al., 2019). Transparent polyethylene microbeads are most prevalent in organisms who wash up near the mouth of urbanised rivers which transport plastics downstream to the coastline (Wijesekara et al., 2018). The effects on riverine creatures are still not as well understood when compared to marine organisms. The mechanism of uptake depends on the morphotype, size and physical colour of the polymers. Aquatic creatures will consume any fragment so long as it can fit inside the mouth or, can be grasped by tentacles (Mohsen et al., 2019). The dimensions of the stomach and intestines also contribute to the digestion capability of organisms (Lithner et al., 2012; Wu et al., 2019). Creatures with more coiled, complex intestines and a narrower stomach will retain more plastics within the gut potentially leading to mortality of organisms (Browne et al., 2015; Cole and Galloway, 2015; Jabeen et al., 2017).

1.6.2. Toxicity

Toxicity has implications on all organisms that use freshwater ecosystems as a food source. The uptake and ingestion of microplastics and microfibres can lead to subsequent transportation of toxic chemicals which have negative implications on the wider ecosystem such as organ and reproduction failures (Carbery et al., 2018; Caron et al*.,* 2018; Zhae et al., 2018; Berglund et al., 2019). Many of these microplastics which are ingested are from upstream sources and have been transported either by aeolian processes or from stormwater drain / land overflow infiltration (Blettler et al., 2019: Bordos et al., 2019). Due to the chemical properties of plastic polymers, they can act as a vector of chemical contaminants (resins, heavy metals and organic toxins) into the aquatic environment (Fred-Ahmandu et al., 2020). The absorption of these chemical contaminants from microplastics can occur through weathering and level of crystallinity.

Microplastics can also contain chemical additives. Additives are a series of chemical substances designed to prolong the life expectancy of a plastic product (Teuten et al., 2009; Bond et al., 2018; Hahladakis et al., 2018). These additives include plasticisers, flame retardants, antioxidants, lubricants and thermal stabilisers, bisphenol A (BPA), polybrominated diphenyl ethers (PBDE), and nonylphenols (NP). Whilst providing critical functionality to polymer development, these additives can be toxic to freshwater and aquatic organisms (Fries et al., 2013; Luo et al. 2019). Chemical leaching of monomers and additives occurs due to changes in the localised environment such as pH changes, light exposure and breakdown. The release of chemical additives and contaminants is dependent on the polymer pore space and the size of the molecule (Teuten et al., 2019). Once in the ecosystem, phthalates and other harmful endocrine disruptors can cause changes to water chemistry and the surrounding ecosystem. While chemicals are found and used in daily products, once leached, they have the potential to cause significant health issues for aquatic organisms and humans (Wagner and Oehlmann, 2009; Grindler et al., 2018).

1.6.3. Addition of organic carbon

Organic carbon is a natural process resulting from the breakdown of plant and animal remains (Rummel et al., 2017) but can be enhanced by the release of plastic polymers into waterways (Arias-Andres et al., 2018., 2019). Polymers begin to release carbon into waterways through photodegradation via sunlight (Figure 1.1) (Zhu et al., 2020). Leaching of carbon is common for polymers as most polymers are based on the carbon atom (Zhu et al., 2020). Once in the environment, polymers immediately leach carbon. Rates of leaching are dependent on the monomer structure, duration of saturation, and sunlight exposure. The release of carbon can lead to the decreasing capacity of oxygen leading to a decrease in aquatic life (Kopf et al., 2015).

1.7. Microplastic regulations

As a result of the toxicity and environmental degradation of microplastics, many nations have implemented regulations around the use of plastics. Plastic regulations do not apply to natural polymers which are chemically altered (Nova Institute, 2019). Microbeads (µBs), are a form of primary microplastic comprised of LDPE plastic (Rochman et al., 2015). They are difficult to detect due to their size (> 1mm) which makes their true environmental impacts

unknown (Zarfl and Matthies, 2010; Guerranti et al., 2019). Microbeads have been widely banned or regulated across the western world, including New Zealand, due to their considerable lifespan (Nelson et al., 2019). Other plastics such as single-use plastic bags, styrofoam and other polyethylene plastics have been banned or regulated due to environmental impacts, durability, and dispersal of the plastic (Dauvergne, 2018; Ryan, 2018).

Within New Zealand, policies and regulations are in progress to mitigate the impacts of microplastics. International policies such as the USA's Microbead Free Water Act of 2015 and the UK's Environmental Protection Regulations 2017 (Burns and Boxall 2018) have been the base for the single-use plastic bag ban. New regulations are removing the use of plastic single-use cutlery, straws and containers (Tremblay et al., 2019).

1.8. International urban freshwater networks

Global microplastic investigations have primarily focused on the role rivers and other freshwater systems have as a transportation mechanism to oceans (Law and Thompson, 2014). Investigations have been undertaken to understand the implications on estuaries and tidally influenced rivers (Yonkos et al., 2014; Fok and Cheung, 2015; Naidoo et al., 2016), lakes (Fischer et al., 2016; Anderson et al., 2017), wastewater treatment plants (Berglund et al., 2019; Bayo et al., 2020), and freshwater organisms (Feng et al., 2019; Fernández Severini et al., 2019; Masia et al., 2019).

1.8.1. Europe

Microplastics have been identified in several European freshwater networks (Table 1.3). Microplastic pollution has been correlated to population density along river embankments in France (River Rhone and Seine), Britain (River Thames), Italy (River Tevere) and the socio-economic development of sewage systems (Dris et al., 2015; Siegfried et al., 2017). Microbeads have been extensively researched in western European rivers such as the Rhine where flow rates of below 104 m/s^{-1} allow for the deposition of microbeads and lighter polymer plastics. (Mani et al., 2019). Confluence locations of feeding tributaries were also identified as sources of all microplastics stating that urban development inland was the principle source of fibre and fragment pollution (Mani et al., 2015). The Thames was identified as a significant microplastic sink. Fibres account for 76% of all microplastics along

all reaches of the river (McGoran et al., 2017). The total debris count of material within the Tamir/Tavy Estuary was 82% microplastic. Mining and sewage were the meaningful contributors of these microplastics (Miller, 1999; Sadri and Thompson, 2014). Across all European studies, pellets and foam were seldomly isolated.

Table 1.3: Total river microplastic yield of Continental Europe from 2000 - 2014 (adapted from Siegfried et al., 2017).

Location within Europe	Total microplastic yield (kg/km^2
United Kingdom	5 - 30
Scandinavia	0.1 – 0.5
Western Europe	0.5 - 20
Southern Europe	0 - 5
Eastern Europe	0.1 - 5
Baltic States	0 - 5

1.8.2. Central and Southern Asia

Asia's microplastic situation is vastly different from Europe as Asian microplastics consists of predominantly fibres from the textile industry. Across Asia, microplastic concentrations have significantly risen since the development of recreational activities and redevelopment of small towns into large cities (Campanella, 2008). These developments has resulted in microplastics become common within community drinking water supplies such as Lake Taihu; China's third-largest lake (Su et al., 2016), tourism locations, (Mohamed and Obbard, 2014) and within mountainous lakes (Free at al., 2014).

Wastewater treatment plants are a considerable source of microplastics throughout eastern and southern Asia due to the mismanagement of recycling and poor handling of sewer and stormwater lines (Wu et al., 2017). The cities of Suzhou release 50,000 m^{3-1} of waste material per day subsequently releasing 1000 – 13,000 microplastics (Sui et al., 2016). Fibres (nylon and polyester) were the most isolated microplastic due to the abundance of textile factories that line major river tributaries of the lake such as the Yangtze (Yang et al., 2019).

The influence of textile factories is highlighted within Vietnam, specifically the Saigon River. Ho Chi Min City's microplastic issues are primarily derived from synthetic fibres such as polyester although, fragments account for 42% of microplastics identified (Figure 1.4) (Lahens et al., 2018). Within Japan, half of all microplastics identified within Honshu rivers were polyethylene fragments (Kataoka et al., 2019). Compared to research in China and Vietnam, a wider distribution of polymers was found such as acryl, nylon, ethylene-vinyl acetate, and polyacetylene. Very few fibres (10%) and no pellet or foam microplastics were identified. The sources of microplastics were identified as homes not connected to sewer systems and sewer overflow during heavy rainfall events. This trend was also identified in previous research throughout China, Taiwan and Malaysia (Eerkes-Medrano et al., 2015; Wu and Xiong, 2017).

Figure 1.4. Synthetic fibres from the River Saigon under sterio-microscope at 200 µm (Lahens et al., 2018).

1.8.3. Central and North America

Central and North American researchers have measured microplastics in urban, alpine and conservation land parcels (Sutton et al., 2016). Similar to Asia, the textile industry is a significant source of microplastics within North America as significant quantity of fibres (65%) and primary sourced microbeads (15%) were isolated from sediments in the River Atoyac, Puebla City, Mexico (Shruti et al., 2019). These were the direct result of 732 textile factories operating within the Atoyac basin, Mexico. Several vehicle plastics were also isolated from vehicle manufacturing processes (Karlsson et al., 2018).

Sediment cores were taken from Lake Ontario (Corcoran et al., 2015) and Lake Winnipeg (Anderson et al., 2017) where microplastics were identified in lake cores 3 metres deep and in the centre of the lake which were derived from anthropogenic activities on the lake shoreline. The total microplastic yield within the Canadian Great Lakes is estimated to be 60,000 particles per km^2 at the northern and south-eastern basin (Anderson et al., 2017).

The vast quantity of microplastics is correlated to the location of Norway House and Winnipeg; two towns on the lakefront. Similar concentrations of polyethylene and polypropylene microplastics from lake townships in Michigan, USA have been identified in Lake Eire from Cleveland (Leite et al., 2014; Mason et al., 2016).

The impact of dams has shown to reduce microplastic concentrations downstream (Watkins et al., 2019). Research on six dams in Ithaca, New York determined the presence of dams were able to stop 93% of plastics within the reservoir from transporting further downstream past the dam. The 7% of microplastics that did breach the dam, such as fibres, polyethylene fragments and polystyrene beads, were then found to settle into the sediment below rather than float in the water column or travel downstream (Watkins et al., 2019).

Mixed land-use rivers, such as the River Gallitan, Montana found no correlation to seasonality or discharge showing that stormwater microplastic discharge is diluted or, is derived from other sources (Barrows et al., 2018). The dispersal of fragmented microplastics from urban centres upstream accounted for 40% of the total plastics observed. Microplastics were only noted once downstream of urban settlements highlighting the impact of anthropogenic activity. Microplastics identified could be traced to fishing netting (polypropylene), flippers for swimming (polyethylene) and cutlery (polystyrene) (Peters and Bratton, 2016).

1.8.4. Australia

Compared to European, Asian and American research, only limited investigations have been conducted into microplastics in freshwater systems within Australia. Research to date has focused on pathways and implications on marine ecosystems and the transportation of plastics through ocean current patterns (Hitchcock and Mitrovic, 2019; He et al., 2020).

Microplastics were identified in several coral reef systems along Australia's eastern coastline (Hall et al., 2015) and have been reported in aquatic organisms which have travelled from marine ecosystems to feed within freshwater rivers. Polypropylene fibres from netting, lures and fishing line accounted for 63% of microplastics and were commonly identified within the study, especially in summer but still identifiable during winter (Halstead et al., 2018).

Three freshwater estuaries along Australia's New South Wales coastline were studied to investigate the impacts of human influence and interaction (Hitchcock and Mitrovic, 2019). Microplastics reported a correlation to human influence and impact, increasing in total volume with an increase in human development. The research identified that land use has a significant impact on microplastic dispersal. In the Clyde Estuary (98 particles per m^3) microplastic accumulation was limited to fibres and pellets under 500 µm. This was because the estuary is within protected forest land with few settlements along the river path. Estuaries within commercial or industrial land such as the Bega Estuary (246 particles per m^3) and the Hunter Estuary (1032 particles per m^3) identified higher volumes of fragmented microplastics and an increase in the abundance of polyethylene, polypropylene, polyvinyl chloride, and polystyrene. The size distribution also increased as the mean microplastic size was recorded at 650 – 800 µm. These microplastics were derived from mining activities and settlements upstream.

Within urban freshwater river sediments, polyethylene was the most commonly identified microplastic polymer. The relationship between population density, artificial land use and the abundance of microplastics was examined within the Brisbane River and Yarra River (He et al., 2020; Su et al., 2020). All thirty-two samples confirmed microplastics with fibres being the most identifiable morphotype (74% in Brisbane and 82% in the Yarra). These fibres were correlated to the high foot-traffic population and anthropogenic activities within Brisbane and Melbourne city. Of the fragments identified (22% and 16%) all were smaller than 500 µm. The highest accumulation zones of microplastics were near the mouth of the Brisbane and Yarra Rivers which is where breeding grounds of aquatic organisms occur. No pellets or spheres were isolated from either river sediment samples. The remaining microplastics were polystyrene foam (Hall et al., 2015).

1.9. Summary of freshwater studies into microplastics

International research into microplastics regarding study locations, sampling methodology, and microplastic identification are summarised in Table 1.4. An apparent issue internationally is that several alternative units are utilised for reporting microplastic abundance. These units have included microplastic particles being measured in particles L^{-1}, particles m^3, particle/1000 m^{-3}, particles/1000 m^2, and particles km^{-2} (Mendoza and Belcher, 2019). This can lead to the misrepresentation of the true microplastic concentration within the environment and makes comparisons between studies problematic.

As sampling methodologies have been developed to be implemented on the specific sampling area, certain methods will suit specific locations. Investigations within large lakes and rivers have required manta trawls as this sampling method has been more suitable for a wider catchment analysis of microplastic quantities. Sampling at a smaller scale in embankment sediments, creeks, or rivers at a township-scale have required more precision hence sieving and filtering have been utilised to collect smaller particles (Mason et al., 2016).

Fragments and fibres were the most frequently isolated morphotype having been identified within wastewater treatment plants and river sediments. (Horton et al., 2017; Koelmans et al., 2019). Pellets and films were identified across most surface water investigations suggesting the source of pellets internationally are not derived from wastewater treatment plant sources. Foam was only identified within surface water studies supporting the evidence of foam plastics being positively buoyant within the water column (Faure et al., 2015). Pellets and films are by-products of plastic production and packaging. As they are not identified within wastewater treatment plants shows they are most likely introduced to the environment from poor disposal methods (Murphy et al., 2016).

Polyethylene and polypropylene microplastics were identified as the most common polymer to be isolated in international studies. This is due to the multitude of products used in daily life which are derived from either polyethylene or polypropylene. Polyester and nylon fibres were the most common in fibres isolated. These were derived from waste material from textile factories in Asia and from population movement in Australia and North America.

Table 1.4: Summary microplastic particle sizes, locations, sampling methods and locations from freshwater microplastic research 2015 – 2019 (adapted from Triebskorn et al., 2019).

Matrix	Location	MP Type	Particle Size	Particle Number	Sampling Method
WWTP's Effluent	Scotland	Flakes, fibres, film, beads,	>65 µm	250-15700/m^3	Steel bucket and sieve
	Australia	Fragments, flakes, films, spheres	>25µm	280-1540/m^3	Fractionated filtering
	Los Angeles, USA	Irregularly shaped fragments	>20-400 µm	0.002/m^3	Sieving and filtering
	Oldenburg, Germany	Particles, fibres	>20 µm	10-9000/m^3	Custom pump and filtering
	Helsinki, Finland	Fragments, flakes, spheres	>20 µm	5-2000/m^3	Pump and filtering
	Amsterdam, Netherlands	Spheres, foils, fibres	>10 µm	9000/m^3	Effluent: grab sample
Surface Water	Laurentian Great Lakes	Microbeads	>355 µm	0.027/m^3	Manta trawl (333um)
	Lake Hovsgol, Mongolia	Fragments, foam, fibres, pellets, films	>355 µm	0.012/m^3	Manta trawl (333um)
	German Rivers	Fragments, foam, fibres, pellets, films	>300 µm	2.9-214/m^3	Manta trawl (300um)
	Tamar Estuary, UK	Floating plastic debris	>270 µm	0.028/m^3	Manta trawl (300um)
	Urban Lakes, Hanjiang and Wuhan Rivers, China	Coloured granule, films, pellets, fibres	>50 µm	1660-8925/m^3	Sieving
	North Sea coast, Netherlands	Pellets, films, fragments, fibres	>30 µm	27000/m^3	1 L glass jar
	Middle-Lower Yangtze River, China	Pellets, films, fragments, fibres	>20 µm	500-3100/m^3	Petersen sampler
	Amsterdam, Netherlands	Spheres, foils, fibres	>10 µm	48000-187000/m^3	2 L glass jar
Sediments	River Thames basin, UK	Fragments, fibres, films	>1 µm	0.66-0.100/m3	Scoop
	Middle-Lower Yangtze River, China	Pellets, films, fragments, fibres	>20 µm	15-1600/L^{-1}	Petersen sampler
	Quinghai Lake, China	Sheets, fibres, fragments, foam	>112 µm	500-12920/m^3	Shovelled
	Rivers in Shanghai, China	Fragments, spheres, fibres	>1 µm	410-1600/kg^{-1}	Shovelled

1.10. Microplastic research in New Zealand

Whilst freshwater microplastic research has gradually gained international interest, studies within New Zealand have been limited to three freshwater microplastic studies: These have been conducted in Auckland streams (Dikareva and Simon, 2019), surface waters of New Zealand urban rivers (Mora-Teddy and Matthaei, 2019) and Christchurch wastewater treatment plant study (Ruffell, 2020). The investigation of the impact of microplastics on urban streams in Auckland analysed both water and sediment samples while the wider New Zealand study only analysed surface water.

The Auckland investigations into microplastic contamination in stream sediment and surface waters agreed with previous international studies. Fragments (79%) and fibres (20%) were the most identified morphotype indicating the source of microplastic are derived from the breakdown of larger plastics (secondary microplastics). In the Mora-Teddy and Matthaei (2019) investigations, fibres (33%) were identified more frequently than fragments (15%). The lack of spherical morphotypes, such as microbeads, was likely in any of the urban streams tested as these beads are derived from industrial plastic production. While polyethylene (PE) was the most recognised polymer (Dikareva and Simon, 2015), the choice of sodium chloride solution used to recover microplastics may have underrepresented other polymer types such as polyethylene terephthalate (PET) and polyvinyl chloride (PVC) due to its lower density compared to the polymers detected (Rochman et al., 2015; Mani et al., 2019). No correlation was found in these studies relating to microplastic accumulation and population density.

Fibres were determined to be from wastewater or domestic sewage effluent where washing and other synthetic fibres from clothing will deposit into the stream (Ruffell, 2020). Other secondary microplastics were determined to have come from the dumping of rubbish, runoff from embankments and road runoff via stormwater networks (Dikareva and Simon, 2019). All three studies have shown local attributes such as land use were responsible for microplastic accumulation rather than from sources upstream or from other waterways as identified in the international literature.

1.11. Study location

This research was conducted along the 26 km long meandering Avon/Ōtākaro River; one of the two major river networks that flow within the city of Christchurch. Commencing from its spring-fed source in Avonhead, the Avon feeds through Central Christchurch and into the newly converted conservation land surrounding eastern Christchurch. The Avon discharges into the Te Wahapū Estuary in South New Brighton/Bexley (Lawrence and Whyte, 2017). The majority of the 380,000 residents who live within Christchurch reside within the 89 km^2 catchment (Figure 1.5). Residential land accounts for over 65% of the catchment with open space (20%) and commercial (15%) accounting for the remaining land.

The Avon/Ōtākaro River is fed by groundwater recharge from the Waimakariri River at 5-8 m^3/sec (Scott, 2000) and rainfall through seepage as evapotranspiration has limited impact annually (Scott and Thornley, 2009). The average baseflow is approximately 1.6 m^3/s but can reach 30 m^3/s from stormwater and land overflow discharges during heavy rainfall events. Several smaller tributaries also feed water flow towards the middle and lower reaches. These streams and other watercourses, (including 74 kms of stormwater drains) totalling 350 km, have the potential to be sources of microplastics.

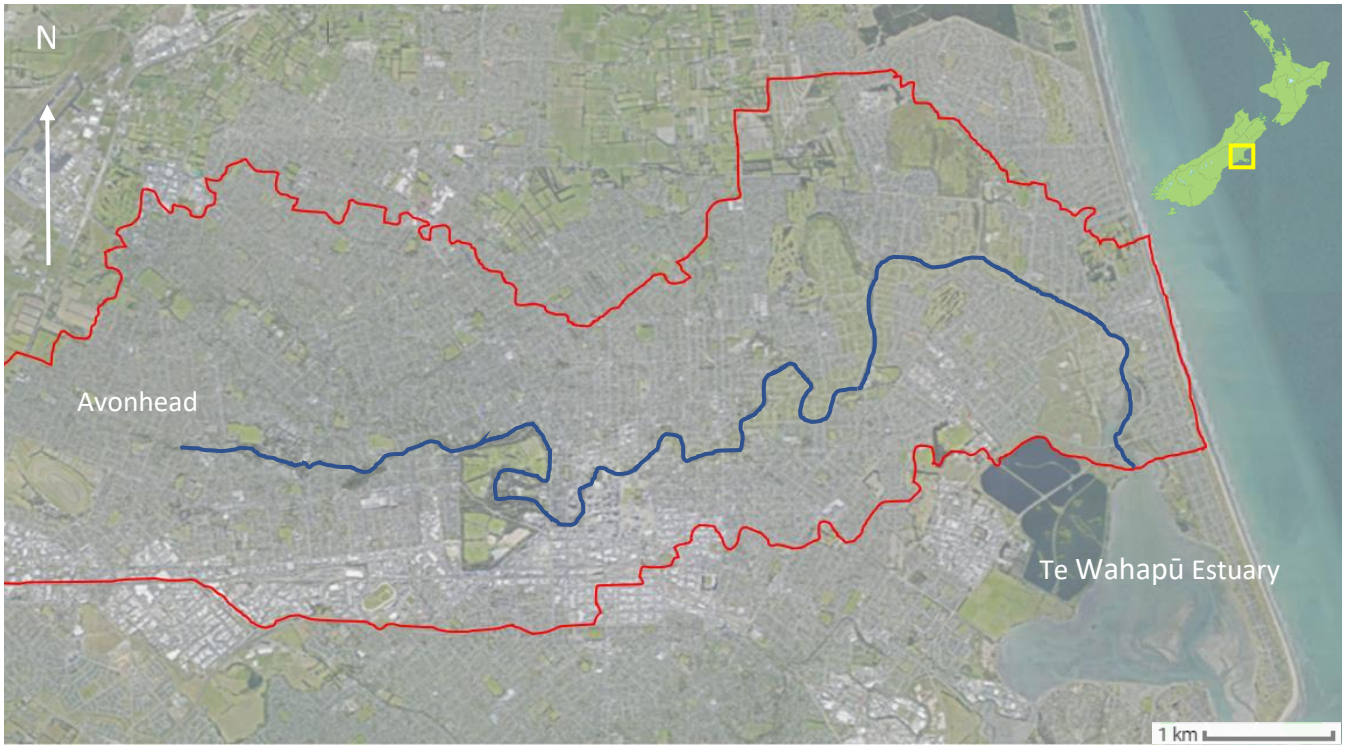

Figure 1.5. The location of Christchurch within the context of New Zealand (top right) and the catchment boundary of the Avon/Ōtākaro River within Christchurch City (red).

1.12. Thesis overview

Considerable research is being undertaken globally on the presence and implications of microplastics in freshwater networks. However, this research is not advanced for New Zealand. Research to date is restricted to the Auckland region (Dikareva and Simon, 2019) and urban areas of Wellington and Dunedin (Mora-Teddy and Matthaei, 2019). As Christchurch has limited textile industries and a relatively low population, microplastic volumes should be lower in concentrations compared to international studies. The hypothesis of this study is that polymer identification will be similar to international research but in lower concentrations. This thesis provides data on microplastics extracted from river sediments in the Avon/Ōtākaro River and sediments from stormwater drains adjacent to the Avon/Ōtākaro River. This data will be able to be used as a reference point for future research.

1.13. Aims

This research aims to:

1. Develop a suitable method for extracting microplastics from river and stormwater drain sediments.
2. Determine the concentration, type, morphotype, size, and colour of the microplastics in Avon/Ōtākaro River sediments.
3. Determine the concentration, type, morphotype, size, and colour of microplastics in stormwater drain sediments adjacent to the Avon/Ōtākaro River.
4. Identify whether microplastic accumulation zones occur through the use of GIS and to identify whether they are polymer dependant.

1.14. Thesis structure

Chapter 2 describes the river and stormwater drain sediment sample locations and the methodology used for collection, separation, and identification of plastics within the lab. Chapter 3 describes the validation of the methods used during this study. Chapter 4 contains the results and discussion attained from the Avon/Ōtākaro River sediment samples. Chapter 5 analyses the results and discusses the relationship of stormwater sediment microplastics to traffic, land use and river sediment microplastics. Chapter 6 summarises the conclusions of this study and identifies potential future research options.

2. Methods

2.1. Sample site selection

The sediment sampling sites were selected based on locations of stormwater outfall pipes, human movement, and confluence locations of tributaries and the Avon/Ōtākaro River (Appendix 1). Samples were collected from both river and stormwater drain sediments. Of the total samples collected (n=45), 30 river sediment samples were collected from 6 zones along the Avon/Ōtākaro River (Figure 2.1) whilst the remaining 15 stormwater drain samples were collected from land surfaces adjacent to the Avon/Ōtākaro River (Figure 2.2). The stormwater drain locations were identified using Environment Canterbury's Canterbury Map GIS platform (https://canterburymaps.govt.nz/). The sample sites were divided into 6 zones of the Avon/Ōtākaro River to allow for comparisons between the upper, middle and lower reaches, confluence locations, flow rates, total organic carbon, dredged and non-dredged locations, traffic volume, and alternative land uses.

A site name based on the physical location and surroundings was assigned to each sample where flow measurements were recorded (Table 2.1). Sample sites beginning at '1' were taken in Ilam (Figure 2.1a and 3.1a) and were numbered eastwards towards the lower reaches. Land in residential locations were sampled adjacent to intersections and open space land samples were collected adjacent to accessible locations (footpaths, bridges and carparks). Streams that fed into the Avon/Ōtākaro were sampled at the confluence location to gather potential input from other water bodies. All sites were sampled between the 15th – 19th July 2019 during dry weather conditions.

Table 2.1. Sampling zones, surface description and average flow velocities of the Avon/Ōtākaro River.

Sampling Site Name	Site Characteristics	Average Flow velocity (m/s^{-1})
Zone 1: Ilam	Residential / Special Purpose	0.43
Zone 2: Fendalton	Residential / Special Purpose	0.67
Zone 3: Hagley Park	Open Space	1.36
Zone 4: CBD	Commercial	1.78
Zone 5: Avonside	Residential / Open Space	1.27
Zone 6: New Brighton	Open Space / Residential	1.04

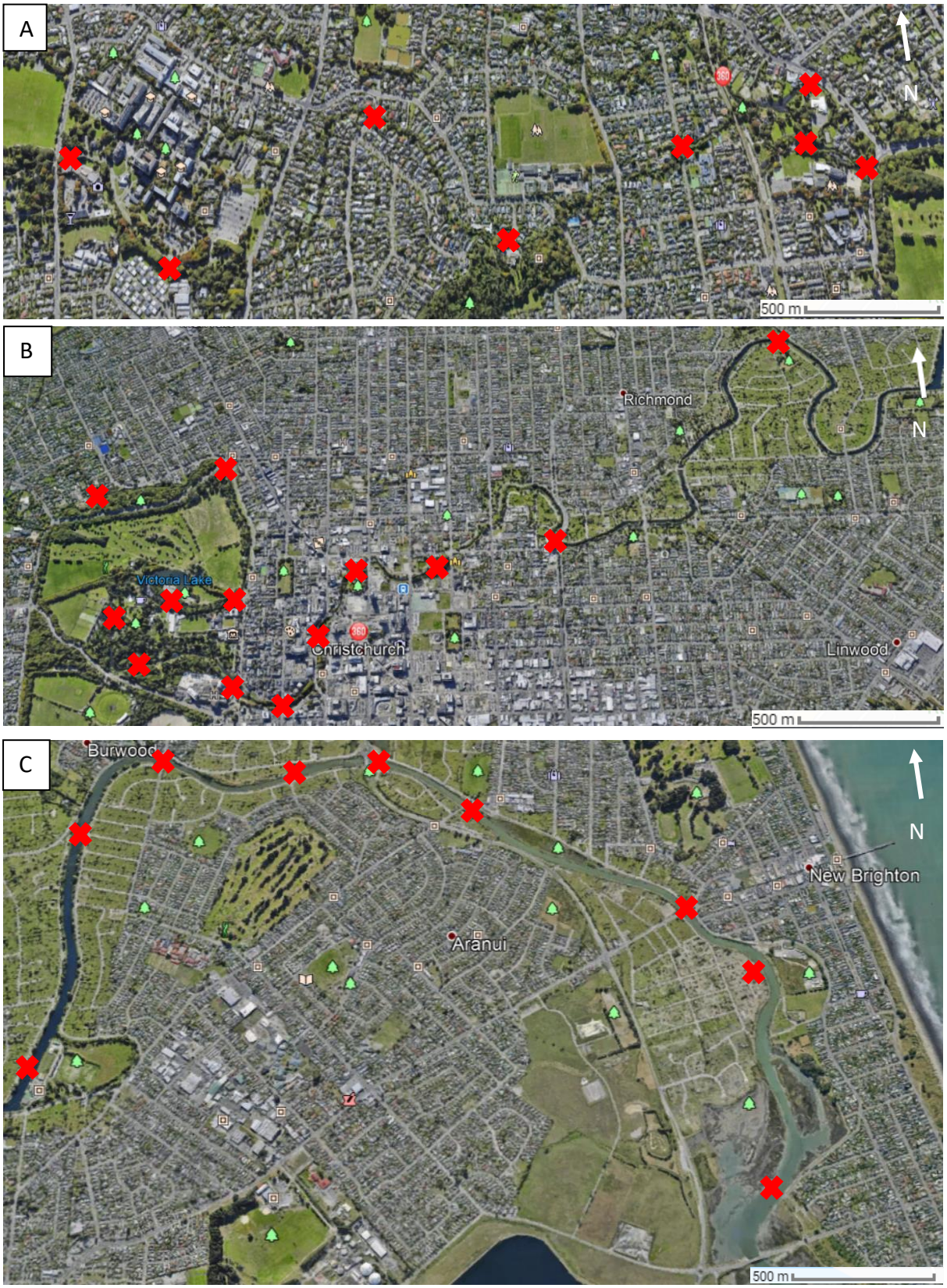

Figure 2.1: Locations of the river sediment sampling sites across the Avon/Ōtākaro River from (A) the upper reach (Western Christchurch) (B) the middle reach (Central Christchurch), and (C) the lower reach (Eastern Christchurch).

Legend: River sediment samples ✖

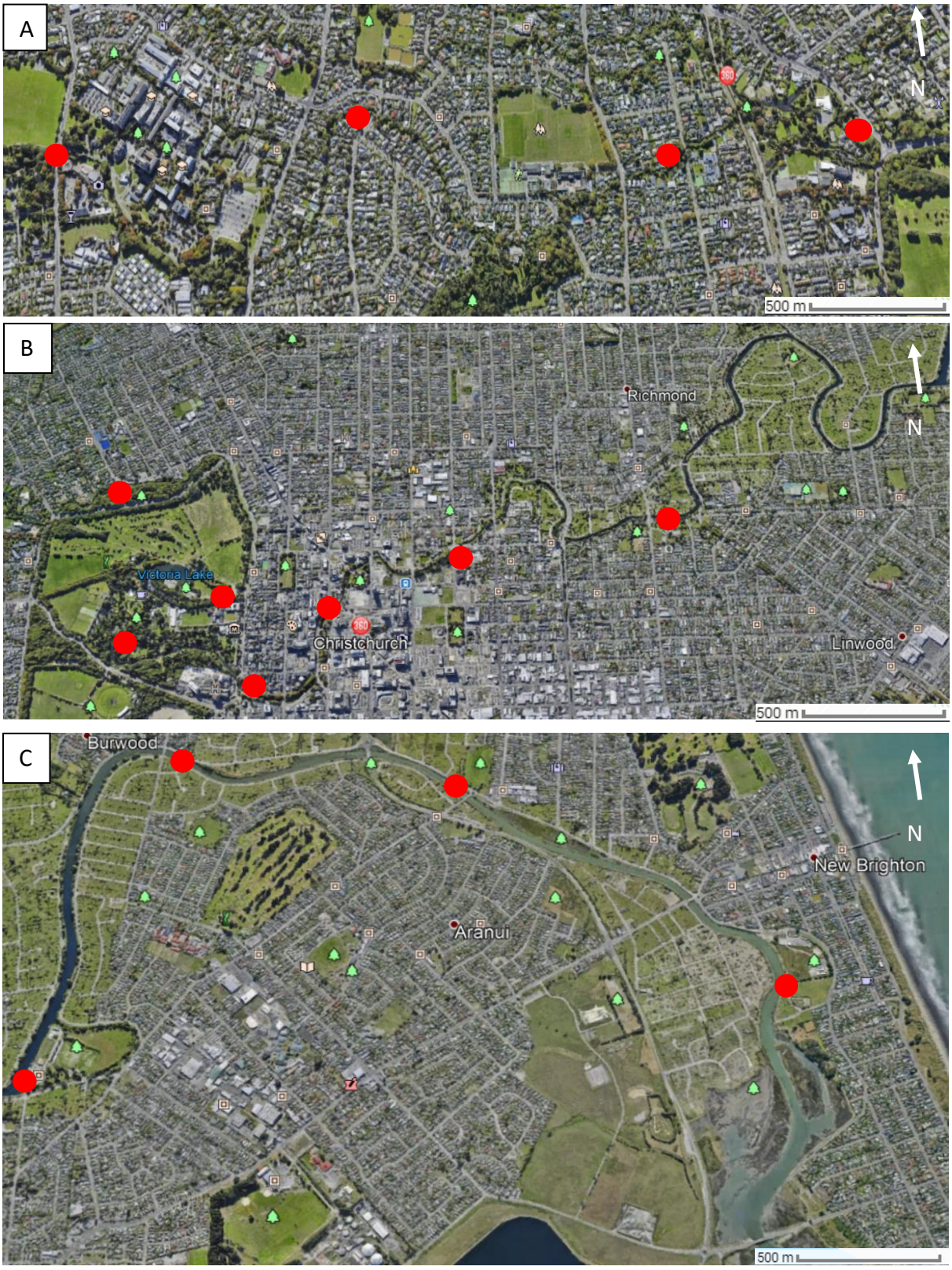

Figure 2.2: Locations of the stormwater drain sediment sampling sites across the Avon/Ōtākaro River from (A) the upper reach (Western Christchurch) (B) the middle reach (Central Christchurch), and (C) the lower reach (Eastern Christchurch).

Legend: Stormwater drain sediment samples

2.2. Sampling

2.2.1. River sediment sampling

River samples were collected using a 700 mL stainless-steel container connected to an extendable grab sampler. The container was placed into the riverbed sediments where the material was extracted from the river. The collected material was placed directly into a 500 mL glass mason jar where it was covered with aluminium foil and weighed before being stored in a 4°c chilly bin. The containers were rinsed downstream from the testing site with river water multiple times before reuse.

2.2.2. Stormwater drain sediment sampling

Stormwater drain samples were collected using a modified portable ZIP vacuum cleaner (Figure 2.3). A sheet of 20 µm stainless-steel sieve mesh was used to line the motor to protect from fine dust infiltration. Sampled dust was collected within a wooden quadrant measuring 1 m^2. The dust collected was brushed into a small stainless-steel funnel which was placed inside a 500 mL glass mason jar. The drain material was tapped into the mason jar. Compressed air was used to blow the remaining dust into the jar and for cleaning.

Figure 2.3 Portable vacuum cleaner used for sampling assembled (left) and lined with 20 µm sieving mesh (right) (adapted from Sampson, 2017).

2.2.3. Flow measurements

Once samples had been collected and water ripples had ceased, the sites flow velocity was recorded using the Sontek flow analyser. The velocity was taken from the centre of the river away from debris and organic material and measured 1 metre below the water surface to

avoid the influence of wind. The measuring height and angle were computed into the flow analyser before calculations were made. The analyser was held in the river for 30 seconds where an average flow rate was calculated based on the pre- and post-sampling times (Table 2.1).

2.3. Equipment

The equipment used for sampling and analysis are outlined in Table 2.2. Most equipment used during sampling and analysis were sourced from within the School of Physical and Chemical Science, however, the elutriation columns were made at an external fabrication company. The containers used for sediment collection were also collected from an external company. The reagents used during laboratory analysis were sodium Iodide (NaI) of 99% purity and isopropanol (C_3H_8O). Sodium iodide was used during sediment and stormwater drain sample extractions whilst isopropanol was used to clean identified microplastics during FT-IR analysis. The FT-IR suite and isopropanol supplied by the Institute of Environmental Science and Research (ESR) was used for chemical identification of microplastics. Geographic information systems (GIS) software was used to plot microplastic accumulation within river and land characteristics (dredged sediments, total organic carbon and land use). The GIS base layers for stormwater drains and land use were provided by Environment Canterbury's Canterbury Maps layer and adapted for this study through the University of Canterbury's ArcGIS system.

Table 2.2. Summary of the material used and suppliers during field sampling and laboratory analysis.

Materials & Equipment	Suppliers
Vacuum	Environmental Science, UC
Grabber	Environmental Science, UC
Sodium Iodide (NaI)	School of Physical and Chemical Science, UC
Stainless-steel containers	Southern Hospitality Ltd
Elutriation column	Taymac Stainless Ltd.
Spotlight 400 FT-IR Microscopy Suite	Institute of Environmental Science and Research
Isopropanol (C_3H_8O)	Institute of Environmental Science and Research
GIS map layers	Environment Canterbury

2.4. Elutriation column design

Microplastics were extracted from river sediments using a series of custom-made stainless-steel columns. The design used for the elutriation columns was modified from Hengstmann et al., (2018). To meet the specifications of this study, the waste chamber was increased to allow for silt material to be collected. The mesh size within the threaded collar was decreased to 20 µm due to the smaller sediment grain size in the Avon/Ōtākaro River (Figure 2.4).

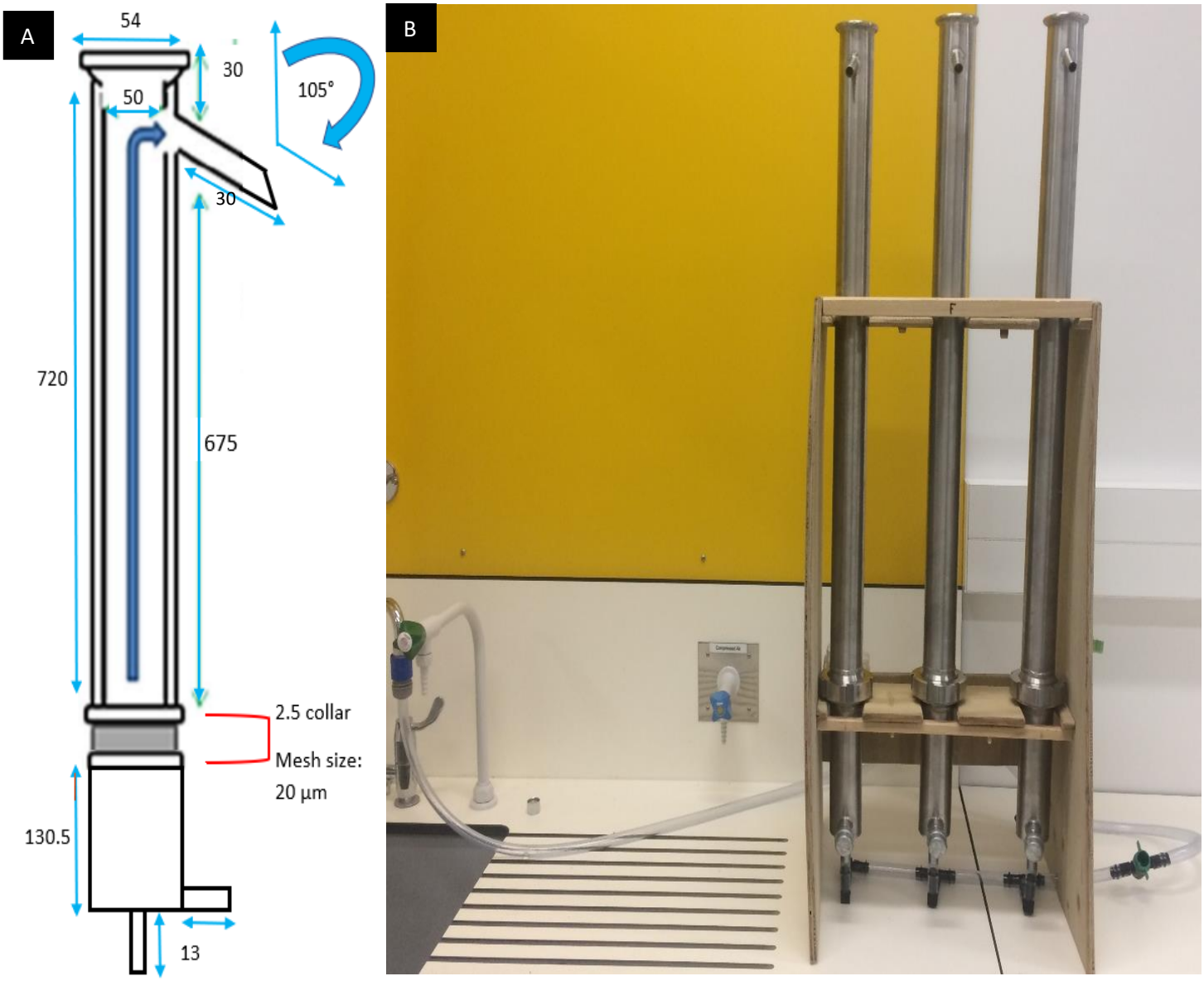

Figure 2.4. Elutriation column set up showing (A) schematic drawing of the elutriation column with measurements given in millimetres and (B) the three elutriation columns set in the laboratory contained within a wooden frame.

The elutriation columns used during analysis was comprised of three individual sections. The bottom section of the column was comprised of two spouts and a removable tank. The bottom spout allowed for the infiltration of water and the horizontal spout allowed for waste drainage and removal during cleaning. The columns were 720 mm tall with a spout welded 30 mm from the top for sediment extraction (Figure 2.4). The columns were built to this height to accommodate the total sediment output and space for the mixing process to take place. To allow for a free flow of plastics, the outlet spout was welded at a 105 ° angle so all the material collected could pour into the sieve without disrupting the mixing process.

A vinyl hose was attached to a tap where distilled water was pumped through the hose and into the tank through an input spout. The hose had flow control valves and a flow meter attached to regulate water flow. The remaining space was dedicated to a tank which held the water within the column before passing through the 20 µm mesh layer and to securely hold waste material before being removed from the column after use.

The middle section incorporated a 2.5 mm dairy-fitted threaded collar which was used to fuse the tank and column together whilst in operation. The collar also allowed for the quick and safe cleaning and replacement of the 20 µm mesh between sampling. A space of 2 mm was granted for the mesh to flex in case of excessive water force or an accumulation of sediment.

While the steel column allows for stronger inflow of water without any breakages occurring and eliminated the risk of plastic components infiltrating the sample, the ability to observe the separation process was limited. Previous methods using elutriation columns, as summarised in Table 2.3, outlined variations between sample size processed, water volume / speed, and the method of collection afterwards. For this study, a sample size of 100 g^{-1} was chosen based on the research of Claessens et al., (2013), Hengstmann et al., (2017, 2018), Kedzierski et al., (2018), and Pagter et al., (2018). As the sediments in this study are of silt origin, an alternative flow rate was required compared to other research that used sand and other heavier sediments.

Table 2.3. Summarised methodology for elutriation column studies from international literature.

Author	Sediment Size	Flow Rate	Collection Method
Zhu (2015)	500 g dry	250 L h^{-1} for 5 minutes	Sieves (300 µm)
Hengstmann et al., (2017)	50 g dry	200 L h^{-1} for 10 minutes	Sieves – unknown size
Hengstmann et al., (2018)	50 g dry	200 L h^{-1} for 10 minutes	Sieves – unknown size
Pagster et al., (2018)	App. 35 g dry	200 L h^{-1} for 10 minutes	Sieves (80 µm)
Dikareva & Simon (2019)	1 L^{-1} of wet sediment	250 L h^{-1} for 20 minutes	Sieves (60 µm)
This study	100 g dry	350 L $^{h-1}$ for 10 minutes	Sieves (20 µm)

2.5. River sediment separation

2.5.1. Elutriation set up

For each sample, 100 g^{-1} of dry sediment was weighed out and transferred using a stainless-steel micro spatula from the glass sample jar to a 500 mL glass funnel placed above the columns. Milli Q water in a squeeze bottle was used to remove sediment stuck to the micro spatula and glass funnel. This washing process was repeated three times. Once completed, a layer of aluminium foil was placed over the top of the columns to prevent airborne microplastics. Before the samples were run, a microscope slide with double-sided tape was placed behind the columns to capture any airborne microplastics within the laboratory.

2.5.2. Elutriation process

Distilled water flowing at 3.5 L/s^{-1} was pumped through a hose and into the base of the elutriation columns. The water passed up an internal spout and continued to flow until it rose above the 20 µm mesh lining separating the column from the waste tank. Once this occurred, the wet sediment began the mixing process. Once the water flow reached the external spout, the sediment samples began to fall into a 100 µm stainless steel sieve, clamped below the external spout. This process continued until 10 minutes surpassed.

2.5.3. Recovery of sediment samples

After 10 minutes, the water flow was stopped to allow for a collection of the external sieves. The material that passed from the elutriation spout and into the 100 µm sieves was carefully

removed with a micro spatula into a 50 mL polypropylene centrifuge tube. Milli-Q water was used to remove material stuck to the mesh sieve. Any water or additional material that had surpassed the sieves were collected in a büchner funnel below. This drained through a vinyl hose into three 5L glass beakers. The beakers stored the material until a visual inspection could be undertaken.

The columns were disconnected and unscrewed so a visual inspection could be undertaken. This was to identify any microplastics which became trapped during mixing. Residual material collected within the columns presented only sediment. As no microplastics were identified within the column or waste material, the columns were rinsed, and the waste material collected was disposed.

2.5.4. Cleaning

Before the columns were put back together, they were thoroughly rinsed three times with Milli-Q water. The 20 µm mesh lining between the column and waste tank was removed and cleaned of any coarse and organic material and replaced after every second sample. The aluminium foil covers used to mitigate airborne microplastics were replaced after every sample was processed. The microscope slide with double-sided tape was inspected after every sample. To maintain consistency, the tape was replaced after every elutriation run. The tape was stored inside a glass container to be inspected under a Nikon Sterio microscope. No microplastics were identified during any elutriation sample run.

2.5.5. Density separation

The material from the elutriation column then underwent a density separation. Along with the sediment, 50 mL of Sodium Iodide (Nal) made from 496 g of Nal with 1 L Milli-Q water (4.3 Mol) was added to each centrifuge tube. The sample was centrifuged at 3000 rpm (speed of 1643) for 10 minutes. The Hettich Rotina 420 centrifuge was used as it allowed for a faster suspension of microplastics when compared to previously reported overnight density separation methods (Carr et al., 2016a; Wolff et al., 2019). Material floating on top of the tubes were carefully poured onto a 47 mm glass microfibre filter paper and were left to dry under a vacuum for 5 minutes. Stainless steel tweezers were used to remove the filter paper from the beaker and funnel before being placed inside a petri dish.

The manipulation of the filter paper was conducted in a fumehood to limit the potential for particles to be blown away. A glass microscope slide with double-sided tape was place to the back of the fumehood to detect any airborne microplastics. This slide was replaced and inspected under a microscope. No contamination was observed. All sampled filter papers were weighed pre and post sample to ascertain the mass of the microplastic sample (Appendix 2).

2.6. Stormwater drain sediment separation

2.6.1. Extraction

Due to the highly variable sized particles within the stormwater drain sediments, the elutriation columns could not be utilised for stormwater drain sediment extraction. Using an analytical scale, 100 g^{-1} of stormwater drain sediment was measured. The measured stormwater drain sediments were removed from the glass mason jar, into a stainless-steel funnel. The funnel spout was placed inside a series of 60 mL centrifuge tubes for sediment deposition. To remove any remaining sediments from the jars, 20 mL of distilled water was poured into the jars and manually shaken for 2 minutes. A pressurised air canister was used to remove particles from the funnel. Large organic materials were removed from the sediment before the centrifugation process began. These particles were sprayed with pressurised air to remove any potential microplastics particles into the funnel before being discarded.

2.6.2. Density separation

The sediments were mixed with 50 mL of Sodium Iodide (Nal) using the same concentrations stated in section 2.5.5. The centrifuge tubes were centrifuged at 3000 rpm (speed of 1643) for 10 minutes. The material which floated to the top was carefully poured onto a 47 mm glass microfibre filter paper and left to dry under a vacuum for 5 minutes within a fumehood. Using stainless-steel tweezers, the filter paper was placed into a petri dish and secured with vinyl tape. A microscope slide with double-sided tape was placed inside the fumehood during vacuuming to identify potential airborne microplastics. No particles were identified on any of the 15 microscope slides during the vacuuming process.

2.7. Controls and mitigation

2.7.1. Field sampling

Contamination during sampling was minimised by using non-plastic materials. River sediments were collected using stainless steel containers and then placed inside glass jars. To avoid any aerosol paint and polyoxymethylene flaking from jar lids, the samples were covered with aluminium foil and sellotaped with vinyl tape for security. For protection when handling samples, nitrile gloves were used due to their stronger puncture resistance to reduce skin contact.

During stormwater drain sampling, the sediment was collected and transferred directly from the meshed lined vacuum cleaner to a stainless-steel funnel where the sample was stored inside a 120 mL glass jar. Between sampling, the vacuum cleaner filter was cleaned with a fine bristled brush and the 20 µm mesh netting was cleaned between every third sample.

2.7.2. Laboratory analysis

During laboratory work, all utensils were cleaned with detergent initially then subsequently cleaned three times with Milli Q water and once with acetone to remove any attached material from previous samples. All beakers, sieves and tubes were covered with aluminium foil. After every use, the aluminium foil was replaced before tears or breakages could occur. Samples within petri dishes or glass containers were secured with vinyl tape to prevent foreign material from entering. The petri dishes were opened in a secure fumehood and removed with stainless steel tweezers. Taped microscope slides were placed behind the elutriation columns and to the back of the fumehood to identify any airborne microplastics. Kimtech wipes dipped in isopropanol were used to clean the tweezers after every sample.

2.8. Fourier-transform infrared microscopy

The Spotlight 400 Fourier-transform infrared (FT-IR) microscopy was used to identify the microplastics identified during the density separation process. FT-IR allows for an identification of a variety of polymers from a wide database based on an infrared spectrum result. Before commencing with sample identifications, a test of the infrared spectrum was conducted using known standards. Polyester, PVC and PET microplastics were used to test the infrared accuracy (Figure 2.5).

2.8.1. FT-IR analysis

Once the extracted microplastics had been transferred to a secure petri dish, they were individually analysed under a Nikon Sterio microscope at 8x magnification. The petri dish lid was removed allowing a glass microscope slide to be placed on top of the filter paper to stop particles from moving. This also avoided the potential for airborne contamination. Identified particles of interest were carefully removed using fine tipped tweezers and placed on top of a diamond compression stage to secure the microplastic in place. A pexiglass cover was used to prevent contamination of dust during analysis.

Once the sample was placed on the diamond stage, an external joystick was used to find the centre of the diamond to complete a scan of the stage origin (Figure 2.5a). Markers were added to differentiate between the polymer and the external background. A total of 4 scans were completed on the polymer to ensure accurate identification. The completed scans produced a spectrum analysis for the individual particle (Figure 2.5b). The produced spectrum was compared to known spectrums within the FT-IR microplastic library database. A summary table of the top 4 similarly identified spectrums were produced from the library database (Figure 2.5c). To complete a successful identification, a similarity of greater than 90% was deemed acceptable based on literature research (Corami et al., 2020). As an identification of 97% was produced, there is confidence in the FT-IR identification.

2.8.2. Cleaning

As several polymers were initially identified by their external resin coating, two separate tests had to be conducted. The first scan involved no cleaning of the polymer so the external coating could be identified. Before a second run was conducted, the sample was washed in isopropanol (C_3H_8O) for 30 seconds. The polymer was then re-scanned for correct identification. The diamond stage compression was cleaned with Kimtech kimwipes and isopropanol before and after every sample to avoid cross-contamination. After use, the compression stage was washed with isopropanol and dried for 2 minutes before being stored.

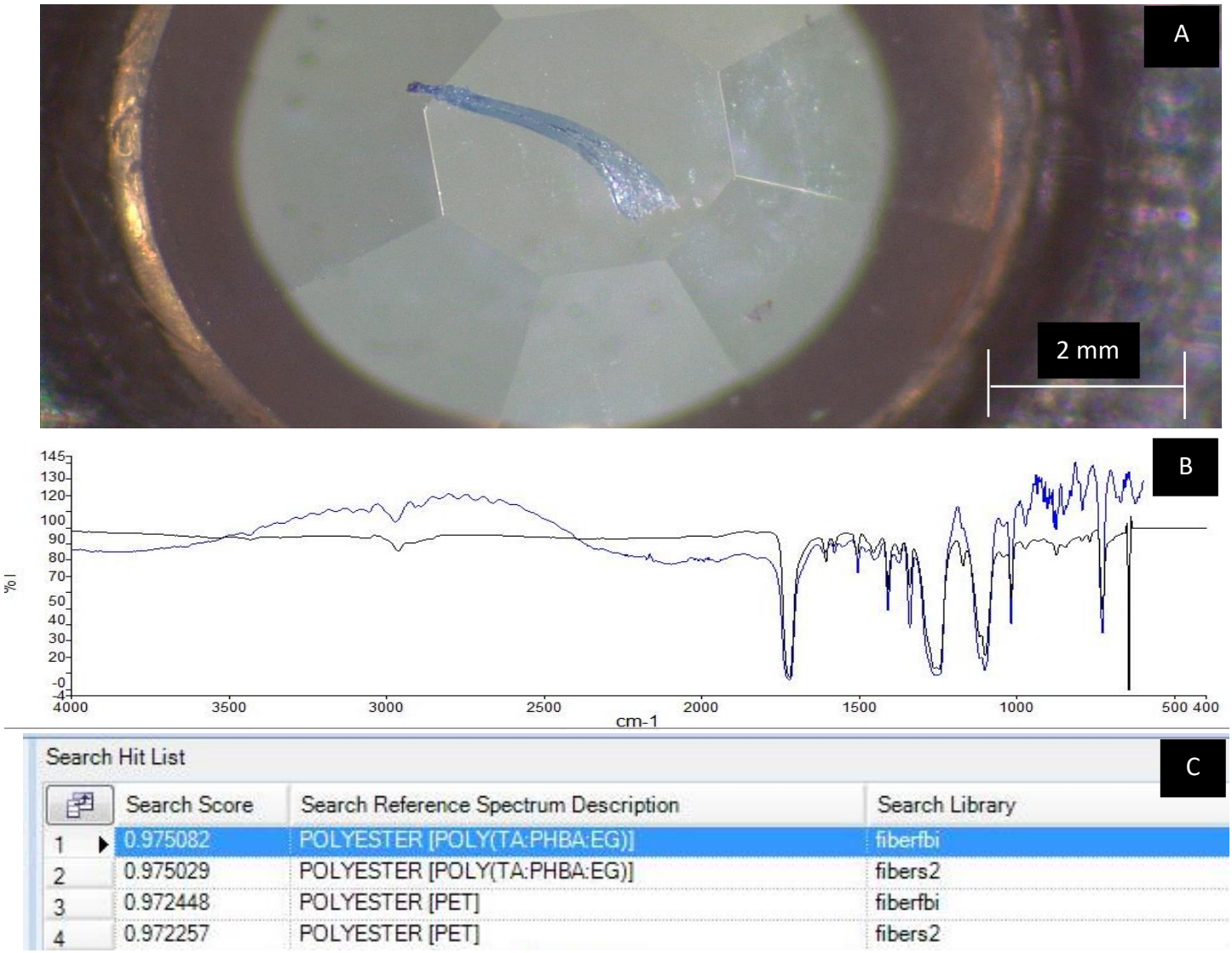

Figure 2.5: An example of (A) the blue polyester fibre at 16x magnification used to test the accuracy of the infrared spectrum with the (B) spectrum curve showing a (C) 97% match to polyester within the ESR Microplastic database.

2.9. Sediment characteristics

2.9.1 Total moisture content

A total moisture content test was conducted on the river sediments prior to using the elutriation column. Tinfoil dishes were individually weighed using an analytical scale before 10 to 15 g of the sediment sample was weighed into the foil dishes. The foil dishes were then placed in an oven to dry overnight at 103 °c. Once dried, the foil dishes were individually transferred using tweezers to a desiccator lined with 560 g of silica gel to cool. The dried samples were re-weighed to determine the moisture content lost (Appendix 3).

2.9.2. Particle size distribution

The dried river sediments from the total moisture content test were used to measure particle size. For each sample, 10 g^{-1} of dried river sediment was placed on the Microtrac Dry Image Analyser (DIA) which measured the total diameter and size of each sediment particle. The analyser reports median, highest, lowest and total grain size of each grain within a 100 µm category (Appendix 4).

2.9.3. Total organic carbon

River sediments that were analysed for total organic carbon (TOC) were conducted at Lincoln University under an Elementar Vario Max element analyser cube. Analysis was conducted under CN mode due to the use of sediments rather than water. Sediments analysed for TOC were first dried overnight at 110 °c. Using 1 g^{-1} of the dried river sediment, the sediments were placed into beakers made of silver foil. The sediment was acidified with 0.5 mL of hydrochloric acid (3%) until visible bubbles had dissipated. The sediment sample was left to dry in an oven at 130 °c for 4 hours before being placed into ceramic crucibles to be analysed. Total organic carbon was recorded as a percentage (%) of the total sediment weight (Appendix 5).

2.10. Health and safety

The primary concerns regarding health and safety was during the sampling process. As samples were collected from the Avon/Ōtākaro River, a University of Canterbury field participation form was required to be filled out, signed and submitted to a qualified and registered health and safety team member. A School of Physical and Chemical Science (SPCS) lab induction and safety inductions were completed into the use of equipment and behaviour in the field.

Samples were collected from the bank but when required to go into the river, full-length waders and a life jacket were worn. A second person was present in case of injury. Access to the river along private property boundaries was granted by owners and residents. During stormwater drain sampling, the drains were accessed by using a parked vehicle to provide protection from oncoming vehicles and Hi-Viz vests were worn whilst on the road. During laboratory analysis, appropriate training through the College of Science MPI course was conducted for the use and access of PC2 laboratories within the University of Canterbury.

3. Validation study for extraction of microplastics

3.1. Introduction

The validation of extraction and identification techniques are critical for understanding any potential limitations regarding microplastic identification or extraction in a sample (Budimir et al., 2018). To test the success of microplastic recovery from the elutriation columns, a validation study was undertaken involving the spiking of four microplastic polymers into a series of clean river sediment samples. The four microplastics utilised were derived from five different size clasts allowing for a test of successful recovery based on size and polymer composition. Whilst normally, recovery trials utilise a single microplastic clast and shape (Imhof et al., 2012; Dehaut et al., 2016), these trials have used digestion rather than elutriation separation as an isolation technique. Before commencing with the sediment analysis, the targeted recovery rate for spiked microplastics needed to be established. Based on previously published studies, a recovery rate of 85% for polymers 300 µm or larger was deemed successful (Thiele et al., 2019). Anything below 70% was classed as poor (Table 3.1). For this research, a recovery rate of 90% repeated for all four sample polymers from two morphotypes (fragments and spheres) was deemed optimal. As previous research into sediment spiking (Beiras et al., 2018; Budimir et al., 2018; Yu et al., 2019) was developed for sand or larger grained particles, a new method was required for silt-grained sediments. Different flow rates were also trialled to test if differing flow rates floated out or submerged specific polymers. The flow rates ranged between 3 L/s $^{-1}$ to 4.5 L/s^{-1} with extraction rates between 70% to 100% recovery rate of standards used. The most successful flow rate was 3.5 L/s$^{-1.}$ as it was the only rate which resulted in a 90% recovery rate for all polymer morphotypes (Figure 3.1).

The objectives of this study were to:

- To produce a validated method for microplastics within river sediments with a recovery rate of 90% or greater down to 30 µm for various polymers.
- To produce a validated method for microplastics from stormwater drain sediment with a recovery rate of 90% or greater down to 30 µm for various polymers.
- To produce a validated isolated method which can be applied to microplastics of different morphotypes.

Table 3.1. Previous sand-based methods and recovery rates for spiked microplastics.

Author	Methodology	Recovery rate (%)	Limitations
Budimir et al., (2018)	Mixture of PP and PE plastics (300 µm) – 10 MP's recovered	85% for clasts 300 µm to 1 mm	No limitations
Beiras et al., (2018)	25 g^{-1} of different microplastics – 15 MP's returned	100% recovery rate	No limitations
Yu et al., (2019)	7 microbeads of LDPE, PP, PET and PVC)	70% across all 10 samples tested	The 1:1 HNO_3:H_2O_2 mixture produced dense foam – heating caused high loss of microplastics

3.1.1. Polymer recovery from river sediments

The recovery of four size ranges of reference microplastics was determined. The sizes were 30 – 100 µm, 100 – 300 µm, 300 – 500 µm, 500 – 1000 µm, and 1000 – 5000 µm. Recovery microplastics below 100 µm were tested, however, this recovery was less significant as only microplastics above 100 µm were being investigated. A total of four different microplastics from five sample sizes were processed. As there were no reference PET microplastics below 300 µm, HDPE was used as a replacement for clast sizes between 30 – 300 µm. For each microplastic utilised, 20 particles were removed from a controlled glass vial and placed into 100 g^{-1} of wet sediment to mimic the samples collected from the riverbed.

A total of 60 microplastic particles were processed for each of the five size clasts, ranging from 30 to 5000 µm. For each size clast studied, three forms of reference microplastics were tested. A total of 300 reference microplastic polymers were processed across the wider validation study (Figure 3.1 – 3.4). Recoveries were determined for 10-minute elutriation periods. From the first elutriation trial of reference microplastics, there was a higher recovery for microplastics 500 µm or larger in size (Figure 3.1). A consistent 100% recovery of PVC polymers and 95 – 100% recovery of ABS were noted. Below 500 µm in size, microplastic recovery decreased below 95%. The recovery of reference microplastics within

the 30 – 100 µm clast range was only 92% on average. During this first trial, all reference microplastics used were fragments.

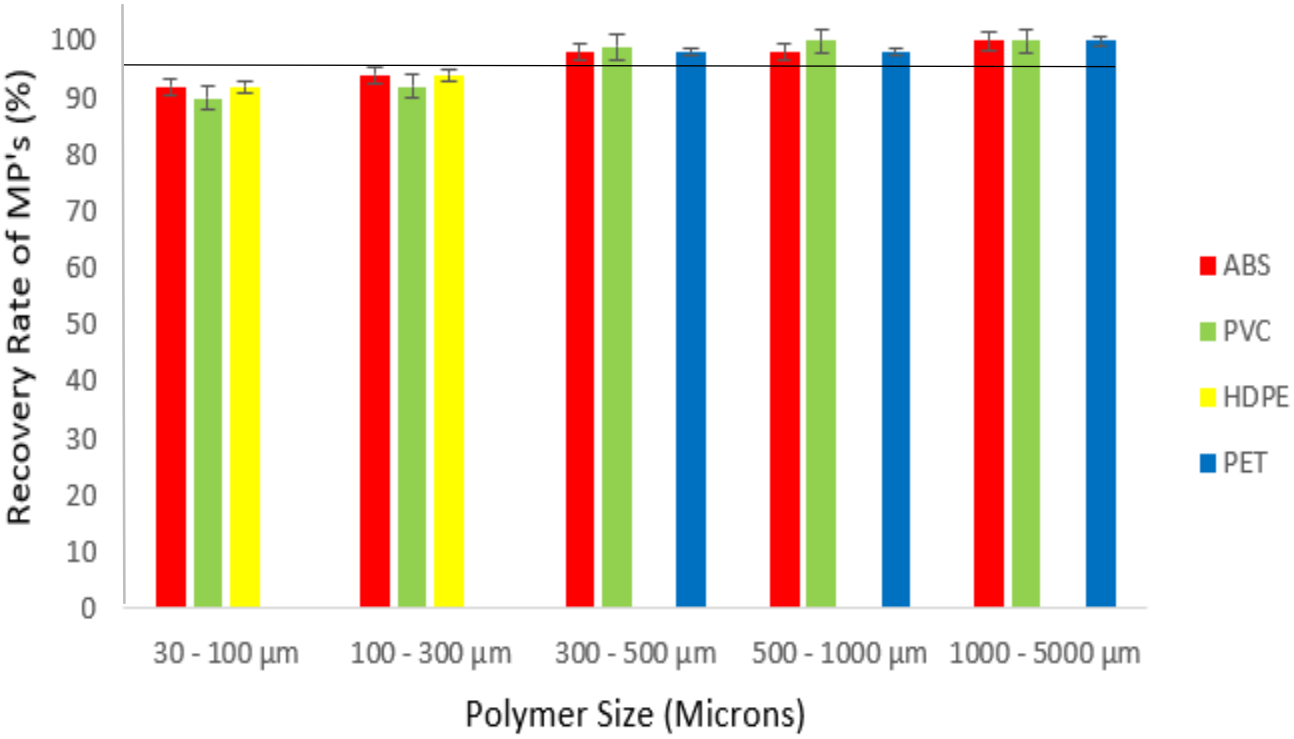

Figure 3.1: Recovery rate of reference microplastics during trial one with a 95% successful recovery rate line (Error bars denote data variability).

3.1.2. Alternative morphotype recovery trial

To test the ability for all morphotypes to be successfully isolated and identified, reference standards of pellets (PVC 300 – 500 µm) and spheres (PET 500 – 1000 µm) were spiked into the sediment samples. A second trial was conducted to determine reproducibility of recovery and to test whether the recovery rate for other morphotypes were comparable to fragments. Using a 3.5 L/s-^{1} flow rate and standards outlined above, 100 g^{-1} of silt was poured into the elutriation column followed by the reference microplastics before water was pumped through. During trial one, all polymers were fragments. The inclusion of the new morphotypes proved the elutriation columns can be utilised for all morphotypes (Figure 3.2).

When compared to trial one, there was no difference between the recovery of microplastics above 1000 µm but there was improvement in the recovery of microplastics in the lower size range (30 – 100 µm and 100 – 300 µm) (Figure 3.2). Recovery rates across all reference

microplastics 30 – 100 µm and 100 – 300 µm improved to above 90%. As 90% was the required success rate for microplastics above 300 µm, to produce an above 90% recovery rate for clasts below 300 µm is positive.

The recovery of formally PVC and PET reference microplastics, which were now pellet and spheres improved by 2% compared to the utilisation of PVC and PET fragments. Pellet-shaped microplastics had recovery rates of 96%. This rate was now consistent with other reference polymers of the same clast size. For the sphere-shaped PET microplastics there was no change in recovery when compared to fragment-shaped PET of the same clast size. The recovery rate was still 98%. The only microplastic to warrant a lower success rate in trail two when compared to trial one was 300 – 500 µm ABS polymers.

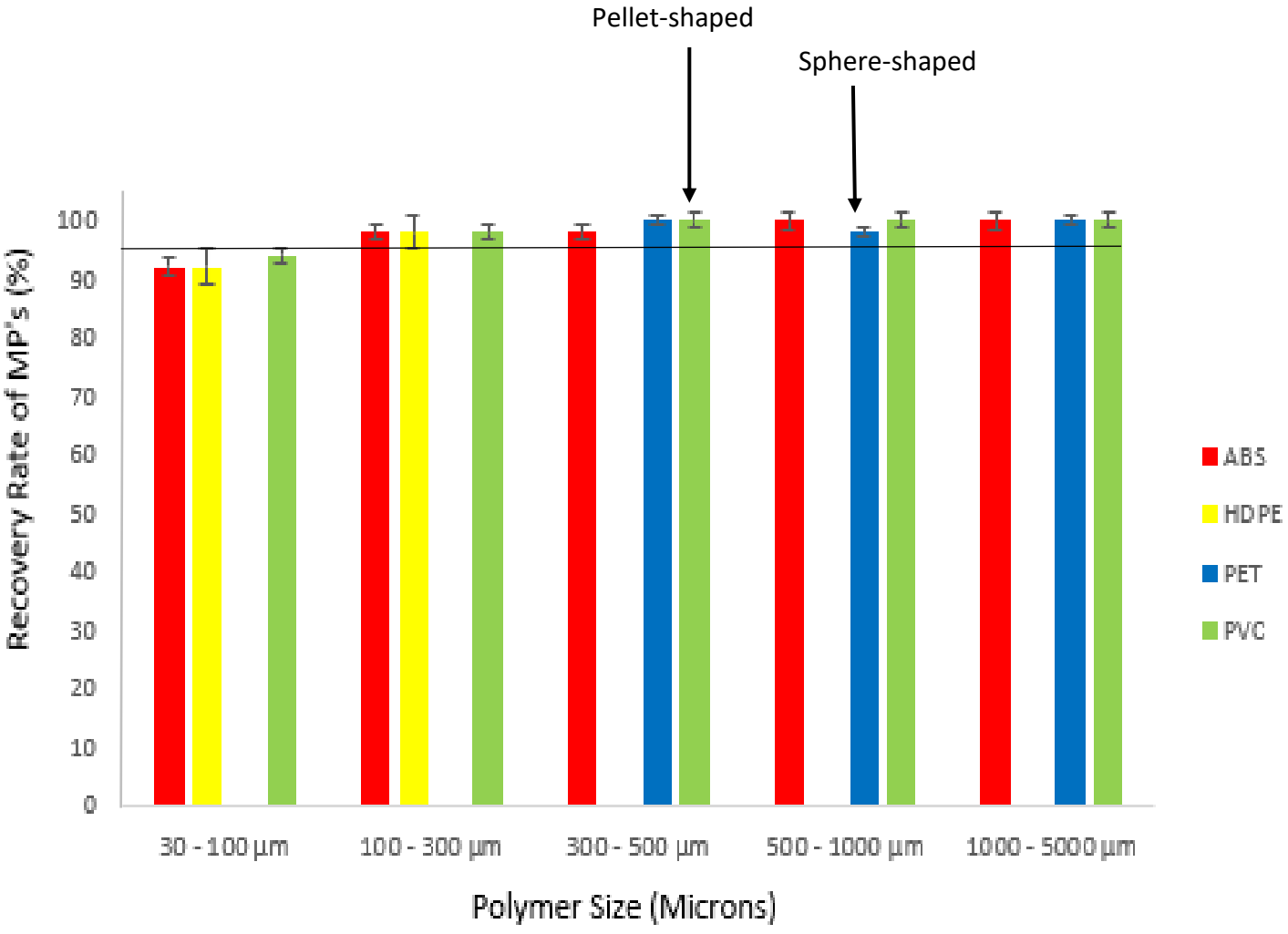

Figure 3.2. Recovery rate of reference microplastics during trial two with a 95% successful recovery rate line (Error bars denote data variability).

The introduction of pellet and sphere-shaped microplastics had no impact on the recovery rate of polymers. All recovery rates were either at or above 90% for the lowest clast size of 30 – 100 µm. The use of elutriation columns can be deemed a successful method for extracting microplastics from silt-based sediment. These recovery rates are 20% above the literature-based recommendation for lowest acceptable recovery rates (Yu et al., 2019).

3.2. Stormwater drain sediment

3.2.1. Isolation efficiency for stormwater drain sediments

An isolation efficiency was also determined for stormwater drain sediments. The sediment samples collected from stormwater drains were initially processed using the NOAA wet sieving methodology. The NOAA wet sieving method (Masura et al., 2015) was trialled as the stormwater drain samples contained several organic particles which could have damage the mesh layer within the elutriation columns. Previous NOAA methods utilised sand and water; not particles greater than 2 mm in diameter.

3.2.2. Microplastic recovery from stormwater drain sediments

The reference microplastics used in the sediment trials were also utilised for determining the recovery rate of stormwater drain sediment samples to maintain consistency. The standards used (PVC, PET, ABS and HDPE) were processed with 100 g^{-1} of stormwater drain sediment through a series of stainless-steel stacked sieves ranging from 30 µm to 2 mm in size. The standards were placed inside a covered glass beaker using stainless-steel tweezers and were manually shaken with 20 mL of distilled water for 2 minutes.

Once the reference microplastics had been decanted through the sieves, they were visually inspected to identify any standards which had been trapped within the mesh. When the standards were identified, the sieves were tipped upside-down over a funnel and rinsed with Milli-Q water into a 600 mL glass beaker. To maintain consistency, the sieves were rinsed three times with Milli-Q water starting from the bottom of the sieves then moving upwards. The Milli-Q water collected in the 600 mL glass beakers were transferred onto a glass microfibre filter and vacuum dried for 5 minutes before being inspected under a microscope. The recovery for reference microplastics ranged from 28% to 88% (Figure 3.3).

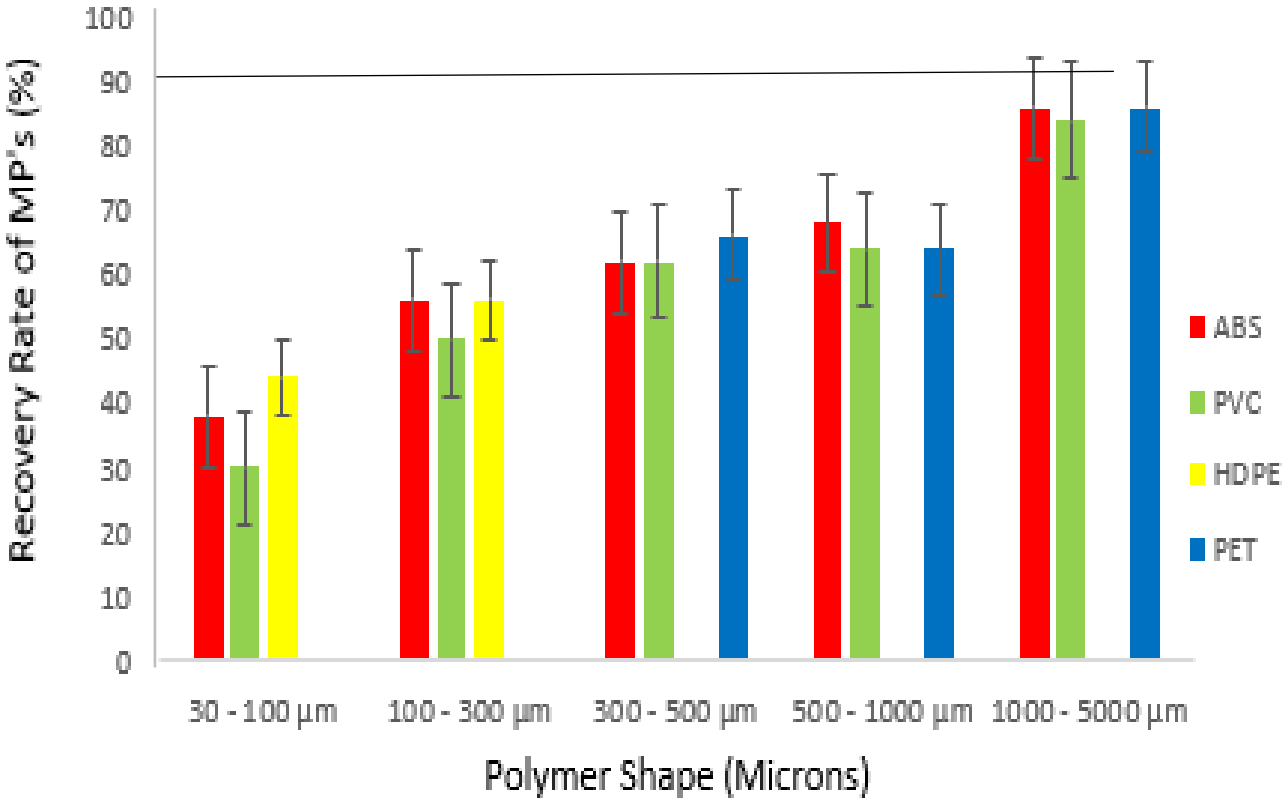

Figure 3.3: Recovery rate of reference microplastics using a series of decanting sieves with a 95% successful recovery rate line (Error bars denote data variability).

The recovery rate of the sieved samples was significantly lower than the targeted 90% recovery rate (Figure 3.3). This was due to the larger size of the sediments. Several reference microplastics processed either passed directly through all five sieves and were collected on the bottom tray or became stuck within larger-sized particles which blocked the pore space of the sieves. The results have indicated that the use of decanting sieves for these samples were not appropriate for the size of the stormwater particles.

3.2.3. Processing stormwater drain sediment through centrifugation

A second isolation method was trialled to improve the recovery of reference microplastics from stormwater drain samples. The 300 – 500 µm PVC and 500 – 1000 µm PET reference microplastics were replaced for pellet and sphere-shaped microplastics to be consistent with the river sediment trials conducted in section 3.1.2 (Figure 3.4). All other reference microplastics were fragments. The reference microplastics were placed inside a glass mason

jar with 20 mL of distilled water and 100 g^{-1} of sediment before being manually shaken for 2 minutes. The reference microplastics were transferred to a centrifuge tube with 50 mL of NaI and centrifuged at 3000 rpm (speed of 1643) for 10 minutes. The contents of the centrifuge tube were transferred onto a glass microfibre filter and vacuum dried for 5 minutes. Using the centrifugation extraction method produced higher recovery rates than using decanting sieves (Figure 3.4). All reference microplastics greater than 500 µm had a recovery rate of 100%. Whilst this recovery rate did decrease from 100% when clast sizes were less than 500 µm, the recovery rate of these reference microplastics were still at or above 90%. There was no difference in the recovery of microplastics based on morphotype.

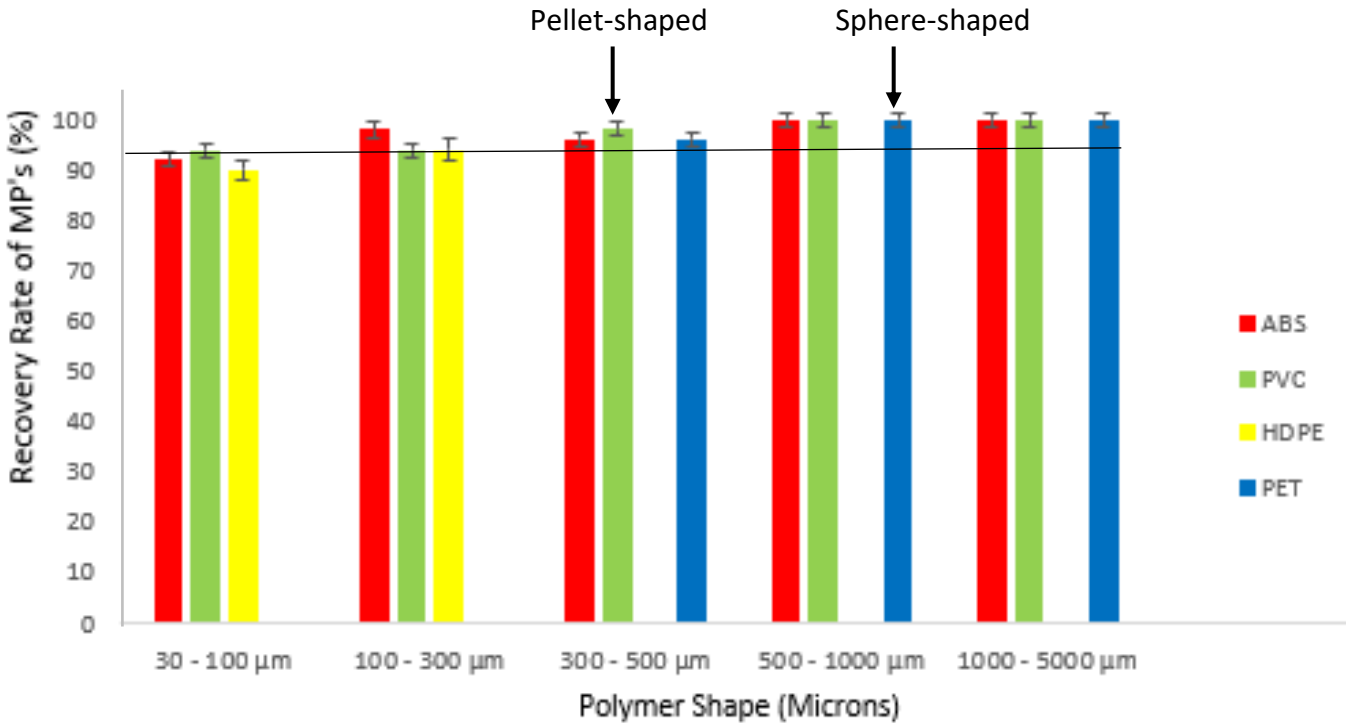

Figure 3.4. Recovery rate of reference microplastics using centrifugation with a 95% successful recovery rate line (Error bars denote data variability).

3.3. Fourier-transform infrared microscopy

3.3.1. Spectrum analysis on reference microplastics

To test the capability of the FT-IR to accurately identify the polymer type of microplastics isolated in this study, the reference microplastics were analysed by FT-IR and compared to previous polymer spectrums within the ESR microplastics library database to identify the closest match. Using ABS, HDPE, PET and PVC microplastics, the FT-IR scans were matched to the top six most similar-matched spectrums (Figures 3.5a to Figure 3.5d). An

identification of 90% similarity was deemed to be successful (Thiele et al., 2019). All four reference microplastics used were identified within the library database search. All search results except ABS returned a result above 95%. Whilst ABS returned a match below 95%, 90% is still within the acceptable range for microplastic identification (Corami et al., 2020).

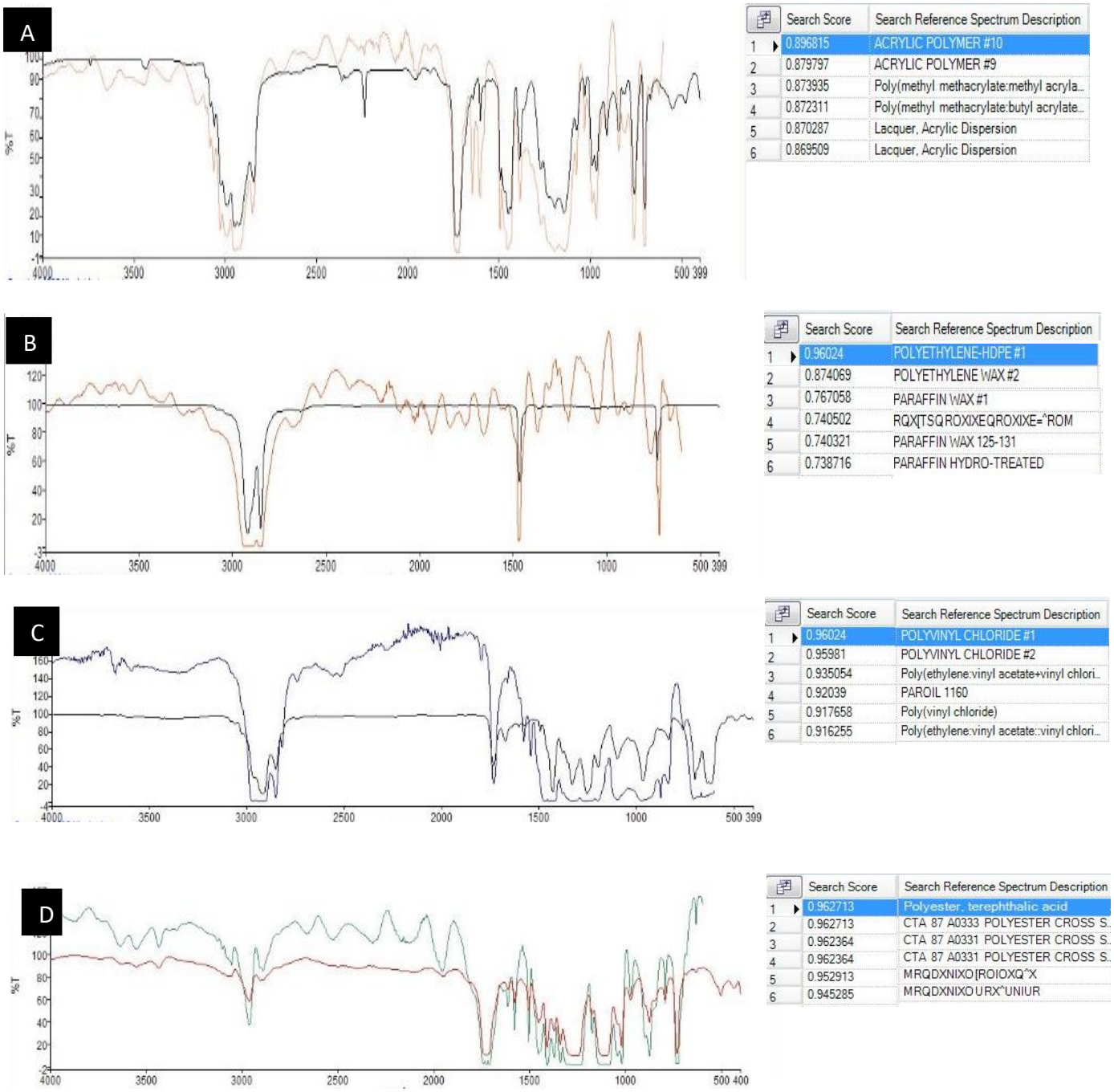

Figure 3.5. FT-IR spectrum analysis of (A) Acrylonitrile butadine styrene (ABS) reference microplastic with 90% match, (B) High-density polyethylene (HDPE) reference microplastic with 96% match, (C) Polyvinyl chloride (PVC) reference microplastic with 96% match, and (D) Polyethylene terephthalate (PET) reference microplastic with 96% match based on the search score on matches within the ESR Microplastic library database.

3.4. Conclusion

From the validation trials conducted, methods have been successfully developed for both river and stormwater drain sediments. The validation trials have proven successful for a rage of polymers, morphotypes and size ranges. The validation method for river sediment microplastics produced a recovery rate of 95% or greater for all morphotype polymers larger than 100 µm. Microplastics below 100 µm were still 90% successfully isolated. The validation method for stormwater drain sediments was successfully produced using centrifugation. The recovery results reported a consistent isolation at or above 90% down to 100 µm. The use of decanting sieves was not a viable option due to the poor recovery rate of microplastics within the stormwater sediments. The elutriation and centrifugation validation methods did not impact the recovery of different morphotypes. Pellet and sphere microplastics were recovered at the same or greater rate as fragment microplastics (Figure 3.2 and 3.4). The use of elutriation columns for river sediments was shown to be a suitable method for microplastic extraction. Centrifugation was identified as a successful method for the extraction of microplastics within stormwater drain sediments.

4. Microplastics in River Sediments

4.1. Introduction

Microplastics have been widely investigated as a major class of environmental pollution. The understanding of microplastics within Christchurch city's waterways is poor. The implications of how tributaries, flow rates, the presence of total organic carbon, and historical dredging impacts can aid in determining the accumulation zones and concentrations of microplastics within the Avon/Ōtākaro River. This can also identify if there are correlations spatially between sample locations.

4.1.1. Site selection

The sample locations within the Avon/Ōtākaro River were chosen based on land use from the Environment Canterbury 'Canterbury Maps' database (Table 4.1). These 30 sample sites were divided into 6 zones. In the upper reach of Ilam and Fendalton land use is predominantly residential. The middle reach of Hagley Park and the Central Business District are defined as open space and commercial while the eastern suburbs (lower reaches) of Avonside and New Brighton fall within the Residential Red Zone.

4.1.2. Study objectives

- To determine the concentration, polymer, morphotype, size, and colour of microplastics in Avon/Ōtākaro River sediment samples.

- To examine the relationship between concentrations of microplastics with tributaries, flow rates, total organic carbon and dredging.

Table 4.1. Site characteristics of the sample locations along the Avon/Ōtākaro River.

	Upper Reach	**Middle Reach**	**Lower Reach**
Suburbs	Ilam & Fendalton	Hagley Park & CBD	Avonside & New Brighton
Land Use	Residential / Open Space	Open Space / Commercial	Former residential land
Population (2006)	4,338 (Ilam) 2,901 (Fendalton	Between 200 – 3,850 (Central Christchurch)	3,240 (Avonside) 2,556 (New Brighton)
(2013)	4,857 (Ilam) 2, 874 (Fendalton)	Between 2400 – 5000 (Central Christchurch)	1,824 (Avonside) [1] 2,442 (New Brighton)

[1] Populations within Avonside and New Brighton decreased in 2013 due to the 2011 Canterbury earthquakes.

4.2. Results

4.2.1. Sediment concentration of microplastics

River sediment samples were collected from six zones along the Avon/Ōtākaro River. Five sediment samples were collected in each zone (Figure 4.1). Confirmed microplastic particles were collected from 27 of the 30 sediment samples. Overall, 156 microplastics were isolated. Across all sample sites, the mean abundance was 5.2 particles per 100 g^{-1} of sediment and ranged from 0 – 35 particles per 100 g^{-1} of sediment. The highest concentration of microplastics was in Avonside (10.4 particles per 100 g^{-1} of sediment) and the lowest in Ilam (2.2 particles per 100 g^{-1} of sediment). No microplastics were identified in one sediment sample from each of the Ilam, Hagley Park and CBD zones.

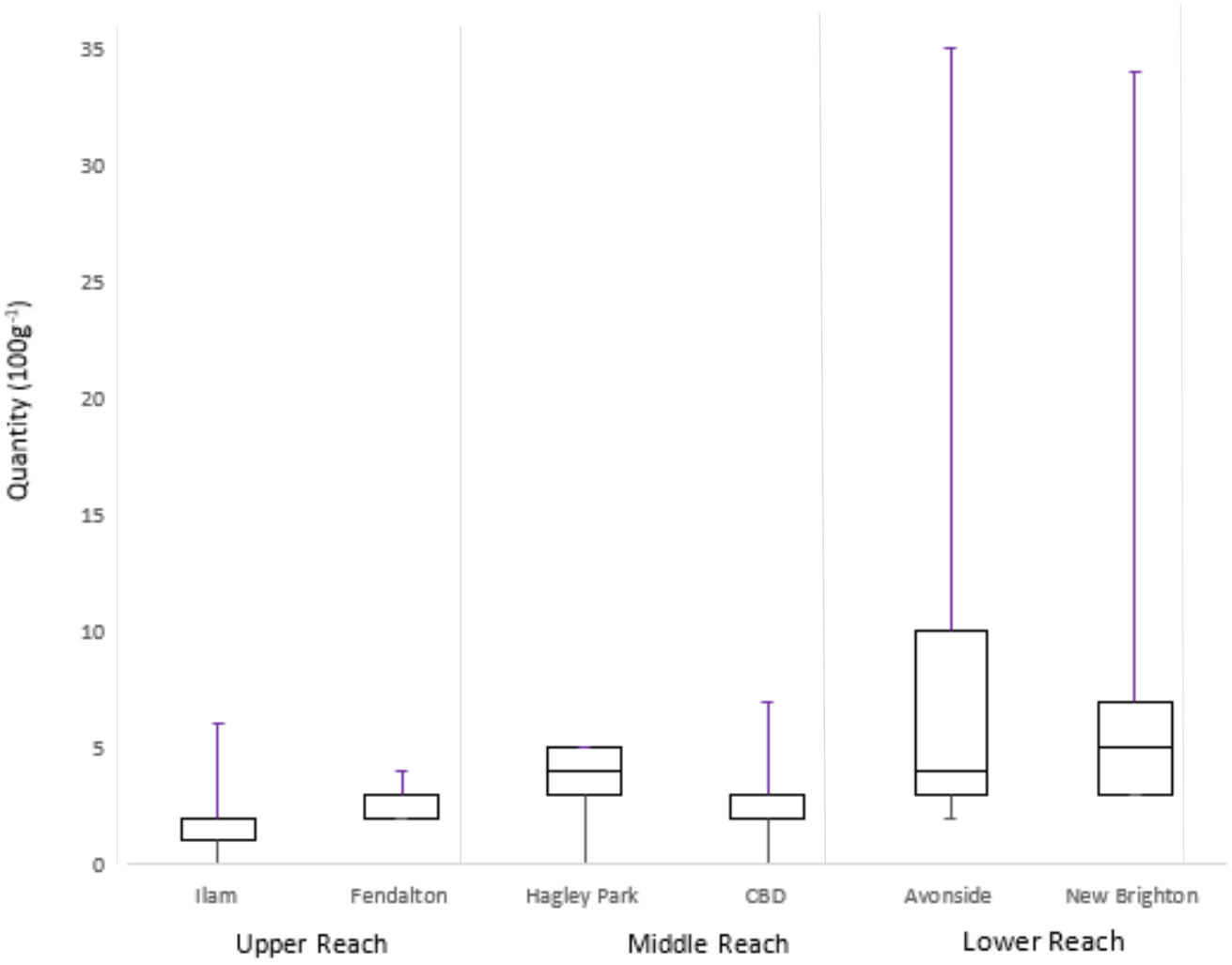

Figure 4.1. Microplastic polymers isolated from sediment within the Avon/Ōtākaro across the six zone locations in particles per 100 g^{-1} (n=156). Natural particles identified were excluded.*

There was a higher abundance of microplastics towards the mouth of the Avon/Ōtākaro River than at the source within the upper reaches. The concentration of microplastics generally increased from the upper reaches (17%), through the middle reach within Central Christchurch (19%) and towards the lower reaches in eastern Christchurch (64%). No significant difference in microplastic count was observed between the upper and middle reaches. The total count of microplastics from the middle to upper reach increased by 40%. Whilst this count decreases towards the mouth of the Avon/Ōtākaro River in New Brighton, the microplastics extracted still accounted for 30% of all total microplastics isolated.

4.2.2. Polymer type

FT-IR microscopy of all 197 particles isolated from the Avon/Ōtākaro River sediments identified fifteen polymer types (Table 4.2) (Appendix 6). Of the 197 particles isolated from river sediments, 156 of the 197 were confirmed as microplastic. Within this subset, 41 particles were confirmed as natural or, were not microplastic (20%) so were excluded from the final dataset. The most common polymers were PET (21%), PP (17%) and PS (16%) and were detected in sediment samples within the lower reaches. Other polymers such as polyester (7%) and nylon (7%) were detected in samples from the upper and middle reaches.

Of the twelve synthetic polymers identified across the Avon/Ōtākaro River, PET microplastics were most abundant (26%). There were five polymers identified which were below 2%. Within the upper reach, the most commonly isolated microplastics were polyester (7 particles) and nylon (6 particles). These accounted for almost half of the 27 microplastic particles isolated from within Ilam and Fendalton. Within the middle reach of Central Christchurch, polyester (5 particles) and nylon (6 particles) were most frequent. Due to the higher presence of other polymers, polyester and nylon only accounted for 38% of the microplastics within Hagley Park and the CBD. PET (5 particles), PS (4 particles) and PVC (4 particles) were also isolated from Central Christchurch. The lower reach of the Avon/Ōtākaro reported an abundance of PET (36 particles) PP (29 particles) and PS (24 particles). The majority of these microplastics were isolated from Avonside with PET being identified in a single location of New Brighton. These microplastics accounted for 80% of all microplastics in the lower reach. Compared to identifications undertaken upstream, polyester and nylon particles was identified as only 5% of all microplastics across Avonside.

Table 4.2. Summary of all particles isolated from the 6 zones along the Avon/Ōtākaro River. (Total microplastics = 156, total natural particles = 41).

Polymer[2]	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
PET	0	0	4	1	1	35	**41**
PP	2	1	0	1	25	4	**33**
PS	1	0	4	0	20	4	**29**
PE	0	5	1	2	0	2	**10**
PVC	0	1	0	4	0	1	**6**
PA	0	2	0	0	0	0	**2**
PPS	1	0	0	0	1	0	**2**
Rubber	0	0	0	0	2	0	**2**
PTFE	0	0	1	0	0	0	**1**
PU	1	0	0	0	0	0	**1**
Cellophane	0	0	1	0	0	0	**1**
Cotton	6	0	3	7	7	1	**24**
Rayon	4	0	5	4	0	3	**16**
Polyester	5	2	3	2	3	0	**15**
Nylon	1	5	4	2	0	2	**14**
Total	**21**	**16**	**26**	**23**	**59**	**52**	**197**

[2] Polyethylene Terephthalate (PET), Polypropylene (PP), Polystyrene (PS), Polyethylene (PE), Polyvinyl Chloride (PVC), Polyamide (PA), Polyphenylene sulfide (PPS), Polytetrafluoroethylene (PTFE), Polyurethane (PU).

4.2.3. Microplastic morphotype

Amongst the 156 microplastics identified by FT-IR analysis, four morphotypes were identified (Figure 4.2 to 4.3) (Appendix 7). The frequency of detection was fragments (47%) > fibres (35%) > foams (10%) > pellets (8%). Films were not detected in any sample. Fragments and fibres were isolated from all sample locations. Pellets were only identified in the lower reaches. Fragments (64%) and fibres (36%) were the most prominent morphotype in the upper reach. No film, foam or pellet microplastics were identified. Within the middle reach, fibres were most abundant (18 particles) followed by fragments (14 particles). These accounted for 88% of morphotypes identified across Central Christchurch. The introduction of foam microplastics (4 particles) accounted for the remaining 12% of morphotypes identified. No film or pellet microplastics were identified. Within the lower reach, a wider distribution of morphotypes were identified. Fragments (47 particles) and fibres (27 particles) were most abundant, accounting for 74% of all morphotypes. Pellets (13%) were identified firstly in the lower reach. Foam (12%) was identified again but only in Avonside.

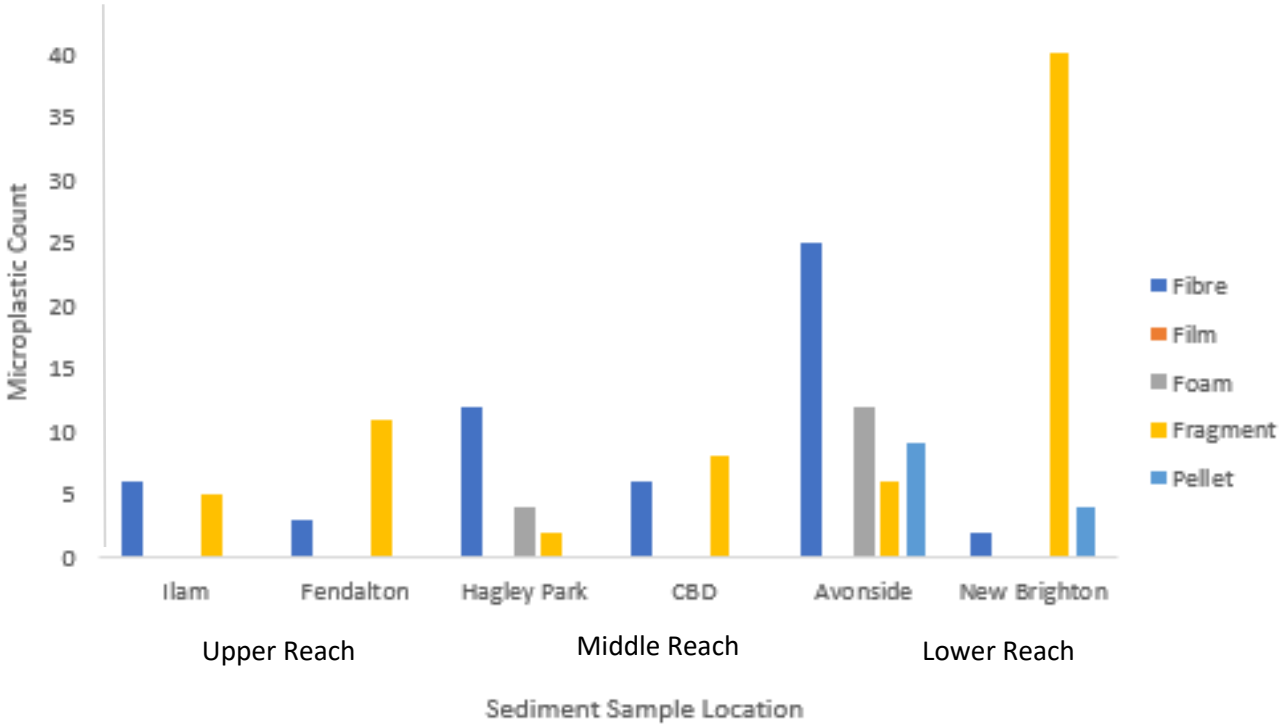

Figure 4.2. Microplastic morphotype distribution along the Avon/Ōtākaro River.

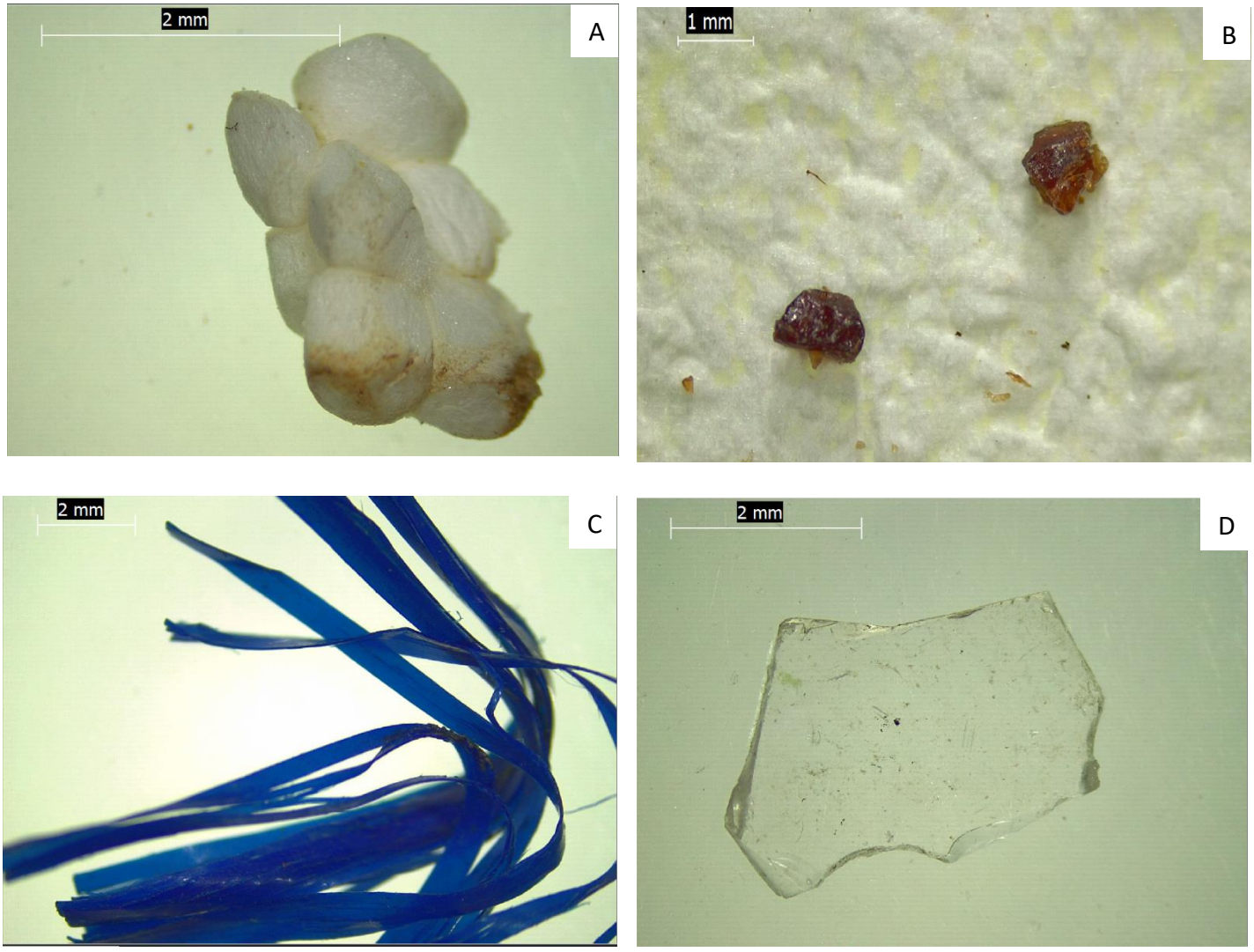

Figure 4.3. Examples of the four varieties of morphotypes isolated from sediment in the Avon/Ōtākaro River showing (A) white polystyrene foam from the lower reach, (B) brown polyethylene terephthalate pellet from the middle reach, (C) a collection of blue polypropylene fibres from fishing lures in the lower reach and (D) a white polyethylene fragment from the upper reach.

4.2.4. Microplastic size

Microplastics were detected down to 30 µm in size (Figure 4.4) (Appendix 7). The most abundant microplastic size range was 100 – 300 µm (43%), followed by 500 – 1000 µm (30%), 30 – 100 µm (18%), 300 – 500 µm (8%). Microplastics greater than 1000 µm only accounted for 2% of the total microplastics isolated. Microplastic sizes were highly variable within the upper reaches. Microplastics between 30 – 100 µm (9 particles) and 500 – 1000 µm (7 particles) were most common accounting for 64% of the microplastic size distribution. Other microplastic sizes were classed between 100 – 300 µm (4 particles) and 300 – 500 µm (3 particles). Only two microplastics were identified above 1000 µm. Only three size ranges were identified from microplastics in the middle reach. The most abundant size was between 500 – 1000 µm (16 particles) followed by 30 – 100 µm (10 particles) and 100 – 300 µm (6 particles). No microplastics isolated were within either 300 – 500 µm or above 1000 µm. The most abundant size range within the lower reach was 100 – 300 µm (57 particles). Only one particle above 500 µm was identified from the lower reach sample sediments.

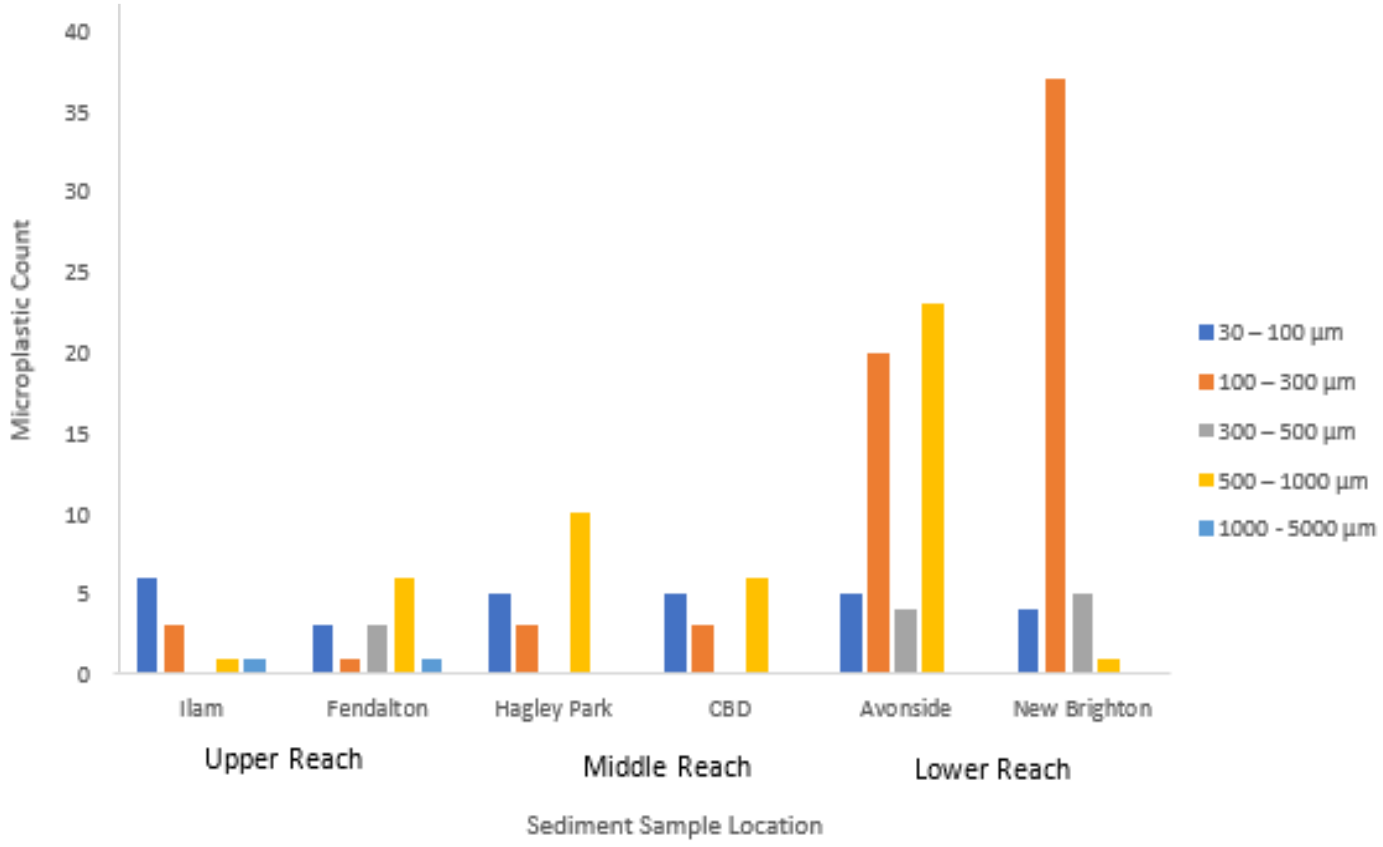

Figure 4.4. Microplastic size distribution along the Avon/Ōtākaro River.

4.2.5. Microplastic colour

Eight different colours of microplastics were isolated from sediments (Figure 4.5) (Appendix 7). The most abundant colours identified in all samples were red (27%) > Blue (26%) > white (21%) > black (19%) > orange (2%) > yellow (2%), green (2%) and multi-coloured (1%). In the upper reach, blue (7 particles), and blue and black (6 particles) were the most abundant accounting for 80% of microplastics. The remaining microplastics were of multi-coloured (2 particles), green, yellow or red colouring (1 particle each). No orange microplastics were identified. Black (12 particles) and white (9 particles) microplastics were most abundant in the middle reach of Central Christchurch. Other colours identified were blue (5 particles), red, yellow (2 particles), orange and green (1 particle). Red (39 particles) followed by blue (28 particles), white (16 particles), and black (12 particles) were most abundant in the lower reach. Orange (3 particles) and green (1 particle) were also isolated. No yellow or multi-coloured microplastics were identified.

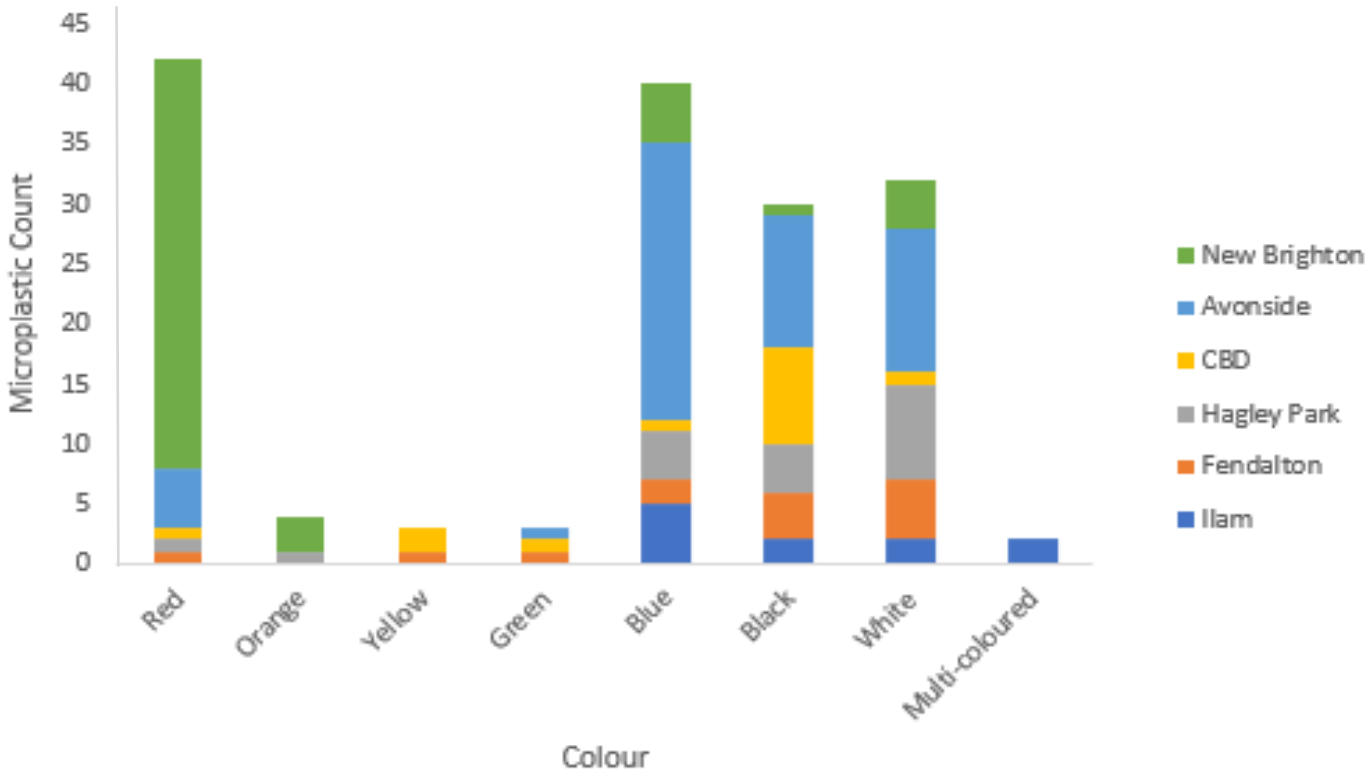

Figure 4.5. Microplastic colour distribution along the Avon/Ōtākaro River.

4.2.6. Influence of tributaries on microplastic concentrations in sediment

In Christchurch, 11 tributaries feed into the Avon/Ōtākaro River. These are a mixture of natural artesian springs and man-made drains. Sediment samples were collected from 9 of these tributaries at their confluence location with the Avon/Ōtākaro River. The mean abundance of microplastics in confluence locations was 94 particles per 100 g^{-1} of sediment compared to the main path of the Avon/Ōtākaro River of 46 particles per 100 g^{-1} of sediment (Figure 4.6). Of these 94 microplastics, 81 particles (86%) were identified in the lower reach (Table 4.3). The remaining 13 particles were identified in the middle reach (77%) and upper reach (23%).

Of the 12 polymers identified across the Avon/Ōtākaro River, 8 were isolated from confluence locations with tributaries. The frequency of detection was PET (39) > PP (27) > PS (18) > Polyester (5) > Nylon (3) > PVC (1) > PTFE (1). In the upper reach, three streams (Okeover, Waimairi and Wai-iti/Waimairi) identified 3 polymers in total (Figure 4.7). This was 7% of microplastics identified in the upper reach. The Riccarton Drain and Addington Brook within the middle reach contained 10 microplastics (Figure 4.8). These polymers accounted for 34% of middle reach microplastics. The five tributaries of the lower reach (Dudley Creek, Kerrs Reach, Horseshoe Lake, Corsers Stream and Travis Wetlands) identified 81 microplastic polymers. This was 83% of microplastics in the lower reach (Figure 4.9). PP and PET polymers were reported in abundance (greater than 20 particles) in these locations (Figure 4.10).

Morphotype detection within confluence locations identified fragments (43) > fibres (35) > foam (12) > pellets (4). Within confluence locations in the upper reach, fragments were most abundant (2 particles) followed by fibres (1 particle). Fragments and fibres in these locations accounted for 8 % and 11% in all upper reach morphotypes. Fibres (8 particles) and fragments (2 particles) were identified at Riccarton and Addington Drain confluences. This was 44% of fibres and 20% of fragments form the middle reach. Four morphotypes were identified from lower reach confluence locations. Fragments (39 particles) were most abundant, followed by fibres (26) > foam (12) > pellets (4). Over half of all morphotypes were identified in lower reach confluence locations. Of all morphotypes in the lower reach, 83% of fragments and 96% of fibres were identified in confluence locations. All foam and pellets identified in the lower reach were isolated in confluence locations.

Microplastics were detected in two sizes within confluence locations in the upper reach. Microplastics of 100 – 300 µm were most abundant (2 particles). A single 30 – 100 µm polymer was identified. These accounted for 50% of 100 – 300 µm and 11% of 30 – 100 µm polymers in the upper reach. Microplastics of 500 – 1000 µm (6 particles) were most abundant in middle reach confluence locations. Polymers of 100 – 300 µm (3 particles) and 30 – 100 µm (2 particles) were also identified. Of all middle reach microplastics, 38% of 500 – 1000 µm microplastics were identified at Riccarton and Addington Drains. These confluence locations also identified 50% of 100 – 300 µm and 20% of 30 – 100 µm. Four size classes were identified in the lower reach. The frequency of detection was 100 – 300 µm (49) > 500 – 1000 µm (24) > 300 – 500 µm (5) > 30 – 100 µm (3). No microplastics greater than 1000 µm were identified in any confluence samples. Microplastics of 100 – 300 µm size in confluence locations accounted for 86% of this size range in the lower reach. Confluence location also identified 33% of microplastics 30 – 100 µm and 55% of 300 – 500 µm polymers. All microplastics 500 – 1000 µm were identified in confluence locations.

Colours within upper reach confluence locations were identified as multi-coloured (2 particles) and black (1 particle). These accounted for all multi-coloured and 17% of black microplastics in the upper reach. Within the middle reach confluence locations, white (5 particles) were most commonly identified followed by black (2) > blue (2) > orange (1). These accounted for 55% of white, 17% of black, and 40% of blue microplastics in the middle reach. Six colours were identified in lower reach confluence locations. The detection of frequency was red (39) > white (16) > blue (23) > green (2) > orange (1). No yellow microplastics were identified in any confluence samples. All red, orange, white and green microplastics identified in the lower reach were identified in these confluence locations as well as 82% of blue microplastics.

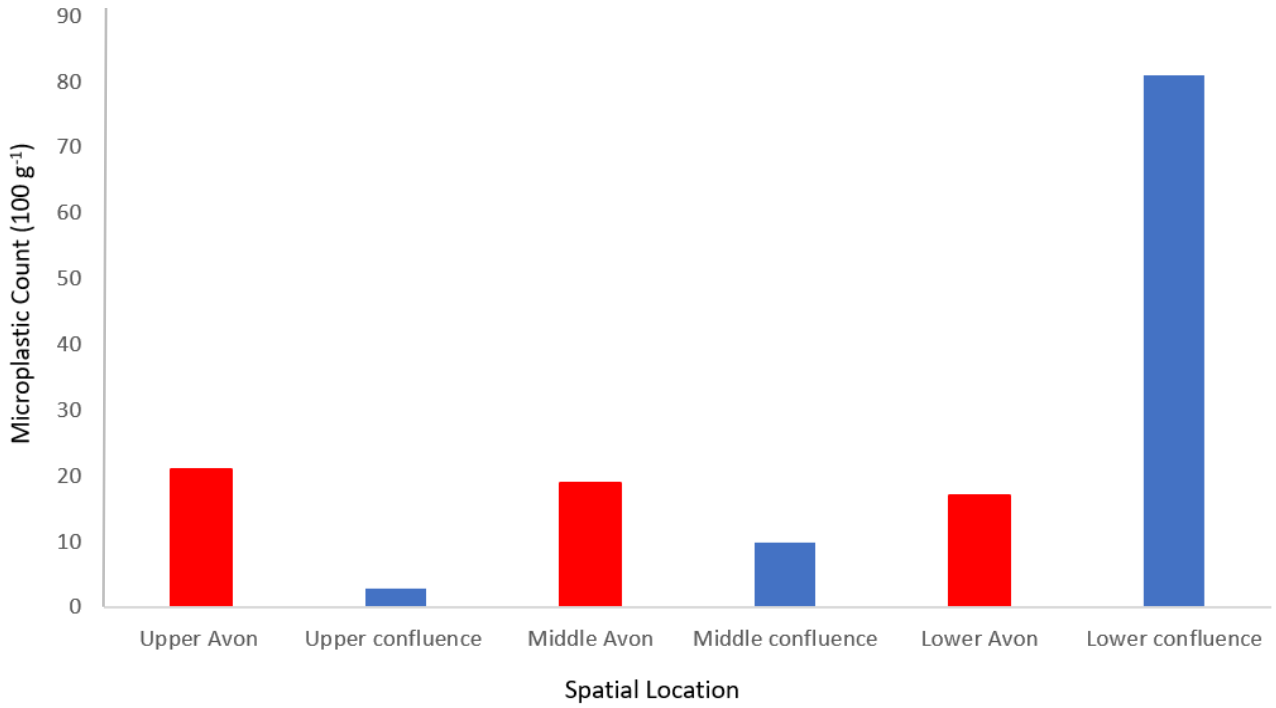

Figure 4.6. Spatial distribution of microplastics from confluence and main channel locations along the Avon/Ōtākaro River.

Table 4.3. Total count of microplastics from confluence locations along the Avon/Ōtākaro River.

Reach Location	Confluence Name	Microplastic Count / 100 g^{-1}
Upper	Okeover Stream	2
Upper	Waimairi Stream	1
Upper	Wai-iti/Wairarapa Stream	0
Middle	Riccarton Drain	5
Middle	Addington Brook	5
Lower	Dudley Creek	3
Lower	Kerrs Reach	35
Lower	Horseshoe Lake	4
Lower	Corsers Stream	34
Lower	Travis Wetlands	5

Figure 4.7. The Okeover, Waimairi, and Wai-iti /Wairarapa Streams (from west to east) that flow into the Avon/Ōtākaro River from Western Christchurch with microplastic count.

Figure 4.8. The Riccarton Drain and Addington Brook (from west to south) that flow into the Avon/Ōtākaro River from Central Christchurch with microplastic count.

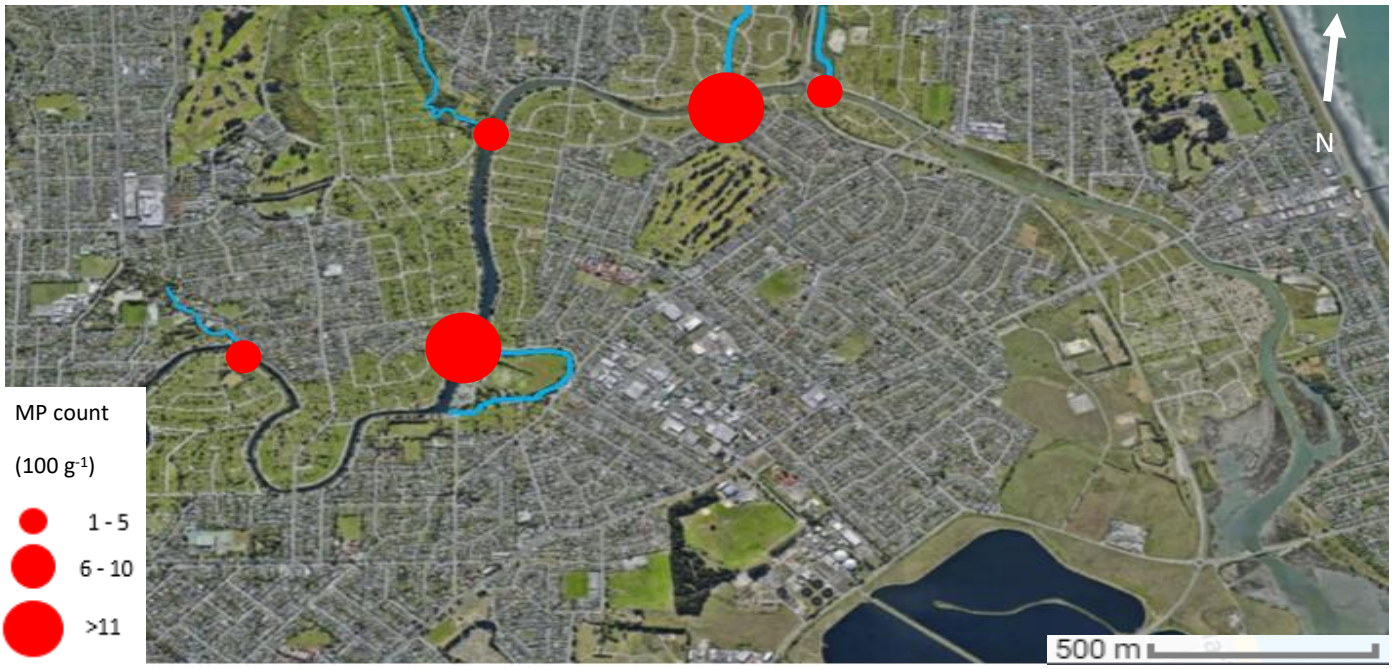

Figure 4.9. The Dudley Creek, Kerrs Reach, Horseshoe Lake, Corsers Stream ad Travis Wetlands (from west to east) that flow into the Avon/Ōtākaro River from Eastern Christchurch with microplastic count.

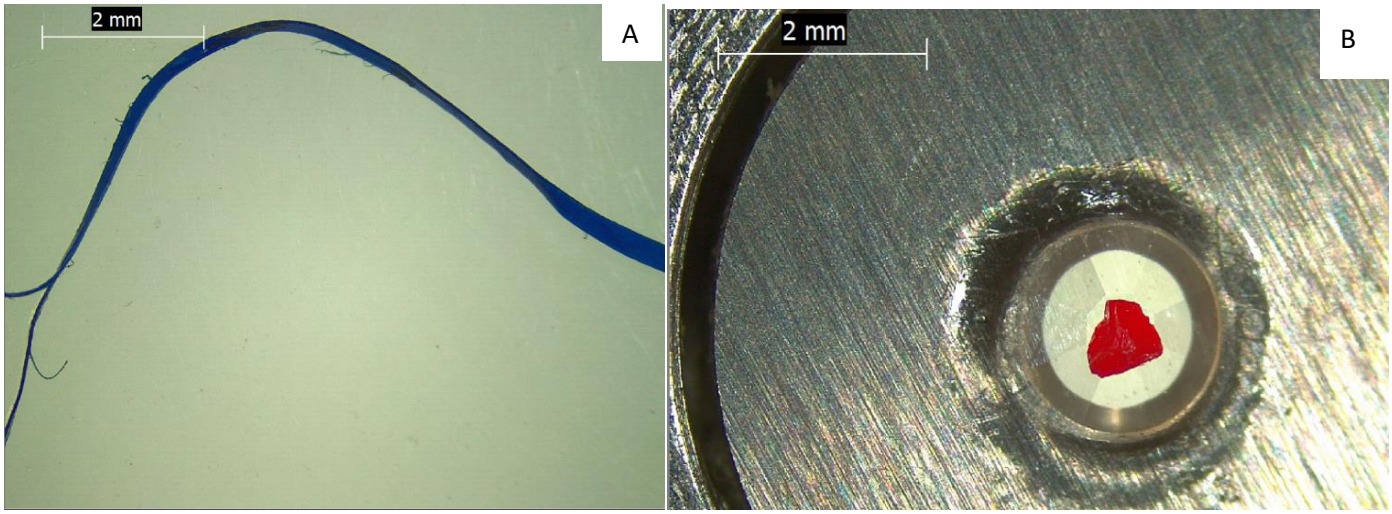

Figure 4.10. Microplastics from the lower reach of the Avon/Ōtākaro River showing (A) Kerrs Reach blue polypropylene (PP) fibre and (B) Corsers Stream red polyethylene terephthalate (PET) fragment at 8x magnification.

4.2.7. Influence of flow rate on microplastic concentrations in sediment

Flow rates of the Avon/Ōtākaro River generally increased towards the lower reach. This increase in flow rate also identified an increase in microplastic concentrations (Figure 4.11). Within the upper reach, flow rate varied between 0.14 m/s $^{-1}$ to 0.65 m/s^{-1}. Concentrations of microplastics isolated varied from 2 to 4 particles per 100 g^{-1} of sediment. Flow rates in the middle reach increased from 0.68 m/s^{-1} to 1.4 m/s $^{-1}$ downstream. Microplastic concentrations varied across the middle reach from 0 to 9 particles per 100 g^{-1} of sediment. In the upper reach of the Avon/Ōtākaro River, flow rates and microplastic concentrations remained consistent between 1.4 m/s^{-1} to 1.8 m/s^{-1} and microplastic concentrations of 7 to 10 particles per 100 g^{-1} of sediment.

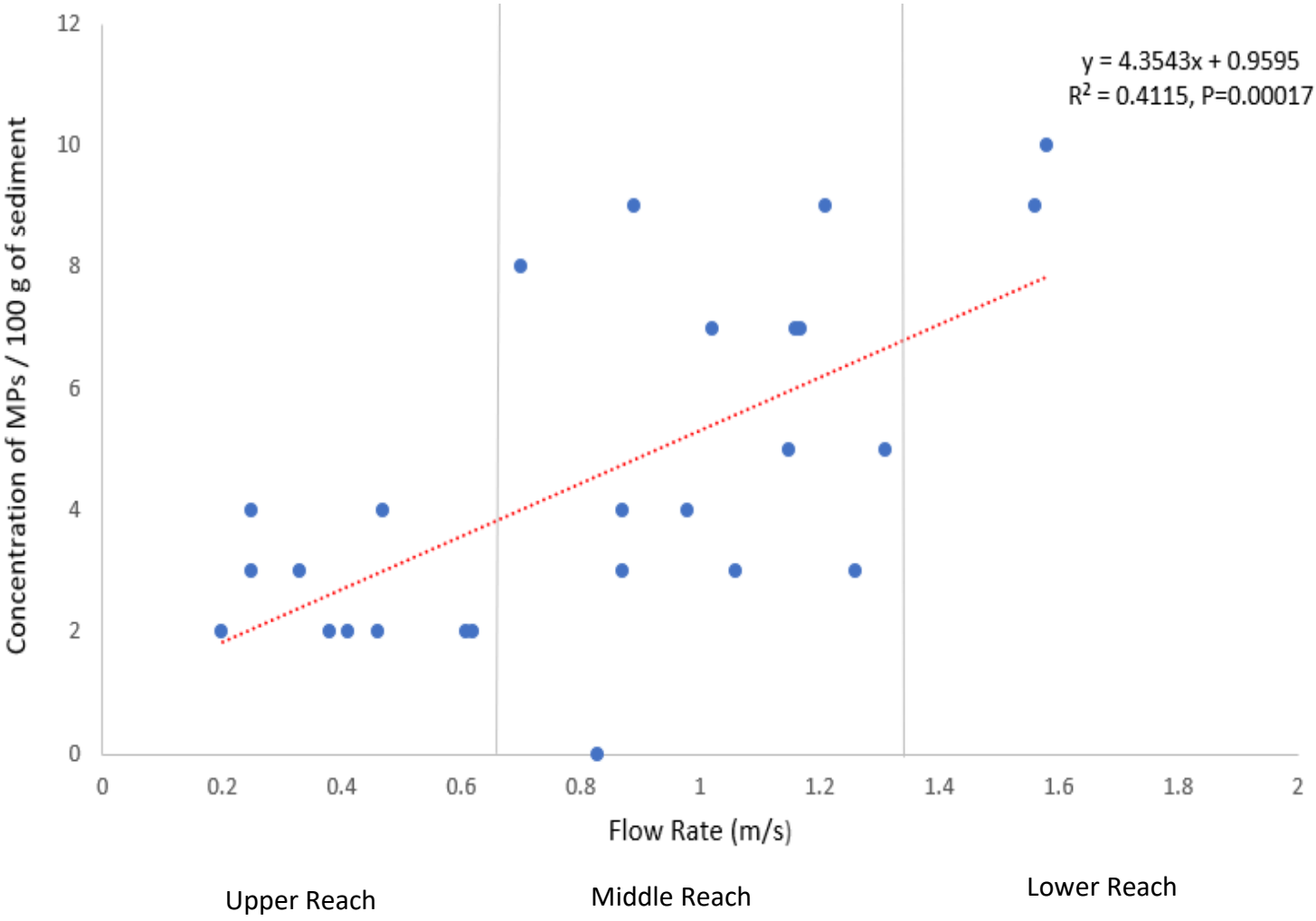

*Figure 4.11. Relationship between flow rate and microplastics within river sediment in the Avon/Ōtākaro River. *accumulations of PP and PET from Avonside/New Brighton have been removed as outliers.*

In the upper reach, polyester and nylon were the most common microplastics. As flow rates were below 0.65 m/s^{-1} and these polymers have a density greater than water (1 g/cm^3), these polymers would have been deposited into the sediment from their original pollution source (Table 4.4). A wider variety of microplastics were isolated from the middle reaches. All microplastics in this location were greater than 1 g/cm^3. Microplastics in the lower reach were abundant with polystyrene (0.98 g/cm^3) and polypropylene (0.91 g/cm^3). Based on the higher flow rates of above 1.4 m/s^{-1}, and the spatial distribution of PS and PP microplastics, it is likely these microplastics were transported from an upstream site.

Morphotypes in the upper reach identified fibre microplastics primarily deposited into the sediments and were not transported downstream. Fibre microplastics reported a similar pattern downstream in the middle reach. Fragments were also commonly identified in these locations. The introduction of foam did not show any influence on transportation. Fibres in the lower reach were not identified within two consecutive samples, hence did not transport. Fragments (PP) and foam (PS) were observed in consecutive samples highlighting some PP fragments and all PS foam microplastics were transported by flow rate.

Microplastic size within the upper reach was not influenced by flow rate. Only 1 microplastic polymer (nylon) was isolated from consecutive locations. Although the same size (30 – 100 µm), these were of different colouring. Within the middle reach, various sized microplastics were isolated from the same sediment sample. As these microplastics were multiple sizes, transportation was not possible. Within the lower reach, only polystyrene between 100 – 300 µm was identified within other sediment samples. Other microplastics whist identified in other sediment locations, were of varying sizes.

Multiple microplastic colours particles were identified in the same sediment sample location, identifying no correlation to flow rate. Only 1 polymer (nylon) of black colouring was identified within consecutive sediment sample sites. No colours were witnessed in consecutive upper reach sediment samples. No sediment samples provided consecutive colour identifications within the middle reach. In the lower reach, white polystyrene microplastics were isolated from four sediment samples. All white microplastics were derived from these polystyrene microplastics. No other colours were repeated in consecutive samples.

Table 4.4. Density of microplastics in relation to freshwater and their environmental fate when introduced to freshwater (Adapted from Estoppney et al., 2015).

Polymer	Density (g/cm^3)	Float or Sink
Polypropylene	0.90 – 0.92	Float
Polystyrene	0.96 – 1.05	Float or Sink
Cellophane	1.42	Sink
Polyurethane	1.20	Sink
Polyphenylene Sulfide	1.44	Sink
Polyethylene	0.85-0.98	Float
Polyamide	1.15	Sink
Polyvinyl Chloride	1.30 – 1.45	Sink
Polyethylene Terephthalate	1.38	Sink
Polytetrafluoroethylene	2.2	Sink
Rubber	1.52	Sink
Nylon	1.15	Sink
Polyester	1.40	Sink

4.2.8 Influence of total organic carbon on microplastic concentrations in sediment

Total organic carbon (TOC) concentrations in the Avon/Ōtākaro River sediments varied between 0.1% to 5.6% (Appendix 5). Total organic carbon and microplastic accumulations varied spatially along the Avon/Ōtākaro River but predominantly increased downstream. Overall, there was no relationship between total organic carbon and the concentration of microplastics across the three reaches. This is supported by the high P-value and low R^2 results (Figure 4.12). The highest TOC concentrations were isolated from confluence locations in the lower reach (Dudley Creek, Kerrs Reach, and Corsers Stream) which also identified some of the highest accumulation of microplastics along with some higher density population locations of Central Christchurch such as Victoria Square and Christ's College (Figure 4.13 and Table 4.5). Dredged locations had the lowest TOC concentrations within the sediment samples apart from Victoria Square. The only lower concentrations were identified in spring water locations from the upper reach. The lower TOC levels within dredged locations also correlated to lower microplastic concentrations (Table 4.6). There was no

effect on particle size distribution and total organic carbon due to the transportation effect of flow rate.

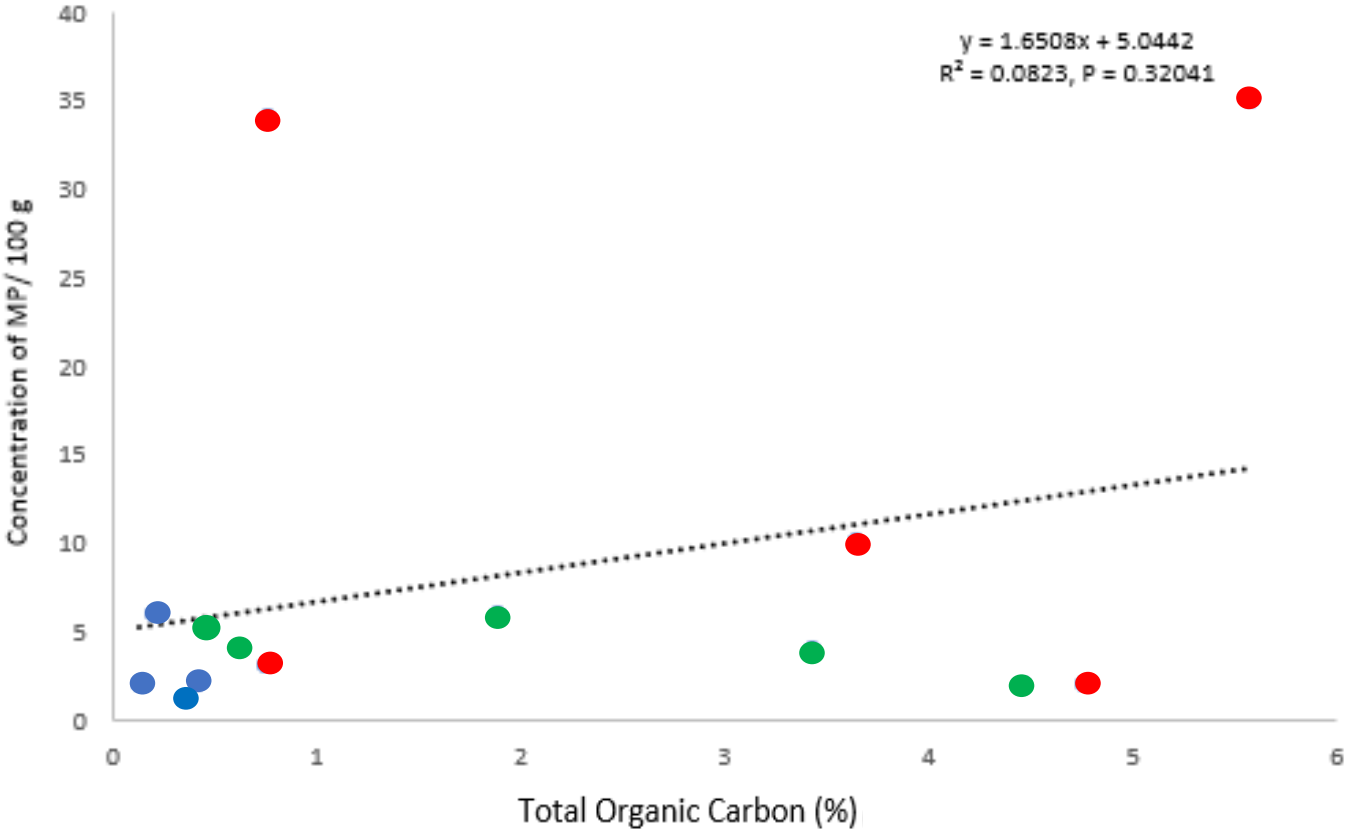

Figure 4.12. Relationship between microplastic concentration and total organic carbon within sediments of the Avon/Ōtākaro River showing (blue) upper reach, (green) middle reach, and (red) lower reach.

Table 4.5. The five locations with highest total organic carbon concentrations and their relative sediment microplastic concentrations along the Avon/Ōtākaro River.

Sample Location	Microplastics (100 g^{-1} sed)	Total Organic Carbon (%)
Kerrs Reach confluence	35	5.6
Victoria Square	2	4.8
Dudley Creek	2	4.5
Corsers Stream	10	3.7
Christ's College	4	3.4

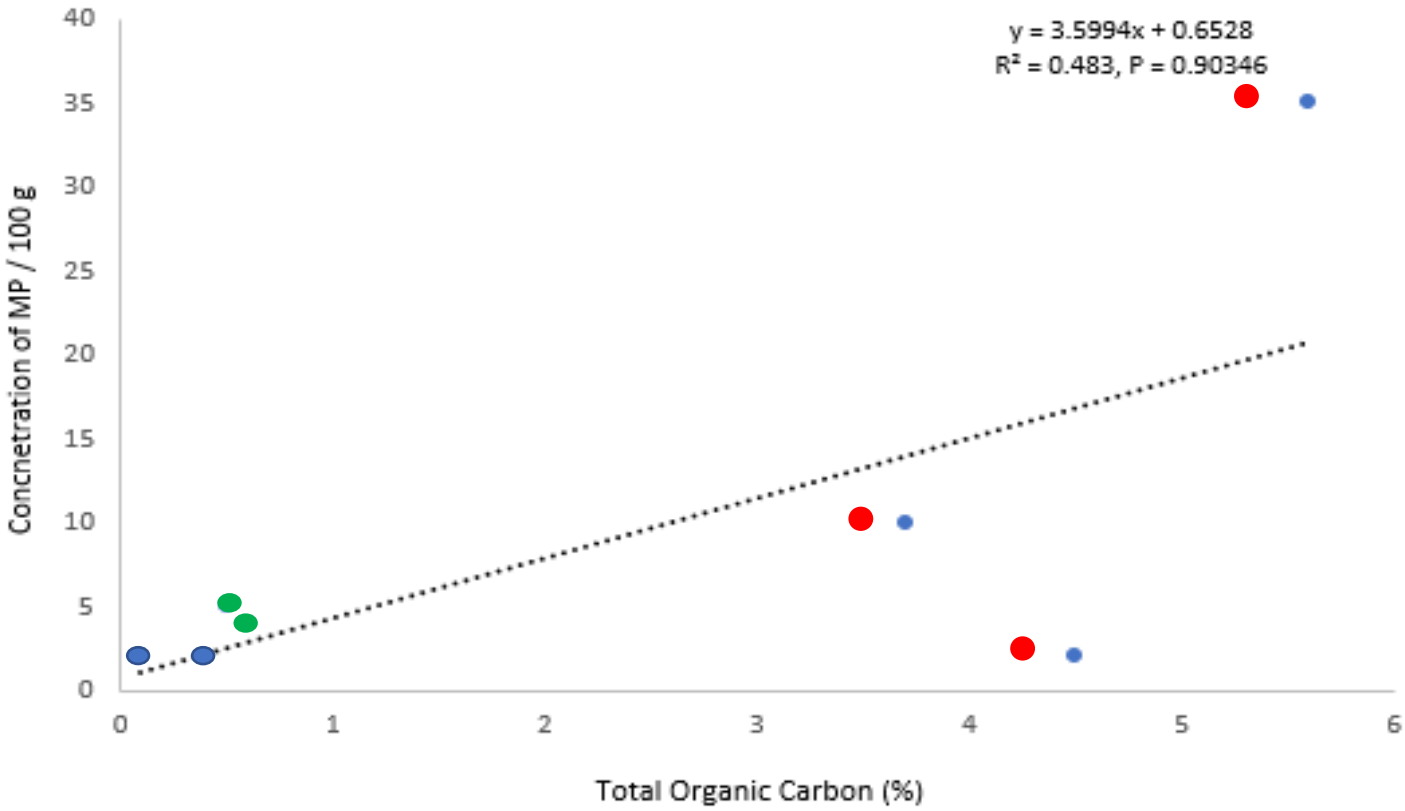

Figure 4.13. Relationship between microplastic concentration and total organic carbon within confluence location sediments of the Avon/Ōtākaro River showing (blue) upper reach, (green) middle reach, and (red) lower reach.

Table 4.6. The three locations with highest total organic carbon concentrations and their relative sediment microplastic concentrations compared to dredged sediment locations along the Avon/Ōtākaro River.

Sample Location	Microplastics (100 g^{-1} sediment)	Total Organic Carbon (%)
Kerrs Reach confluence	35	5.6
Victoria Square	2	4.8
Dudley Creek	2	4.5
Victoria Square	4.8	2
Avon Loop	1.9	0

4.2.9. Influence of dredged locations on microplastic concentrations in sediment

Sediment samples were taken from seven sites along the Avon/Ōtākaro River which have been dredged post-2011. (Figure 4.14). The Avon River Project (2014), Avon Loop (2013) and Kerrs Reach (2012) projects removed 23,600 m^3 of sediment after the Christchurch earthquakes in 2011. The summarised results from isolated dredged sediments identified 47 microplastics comprised of 7 polymers across the three dredged projects (Table 4.7). The Kerrs Reach project had the highest accumulation of microplastics (35 microplastics). These were comprised of all polymers outlined in Table 4.7. Within the Avon River Project, all polymers from Kerrs Reach except polystyrene were reported. The frequency of detection was Polyvinyl Chloride (4) > Nylon (2) > Polyester (2) > Polyethylene (2) > Polyethylene Terephthalate (1) > Polypropylene (1). No polymers were identified in the Avon Loop.

Within dredged locations, the mean abundance of microplastics was 6.7 particles per 100 g^{-1} of sediment. This abundance is exacerbated by accumulations in Kerrs Reach. If the 35 microplastics from Kerrs Reach were removed, the mean abundance decreased to 2 particles per 100 g^{-1}. Polypropylene was most abundant in dredged sediments accounting for 73% of all PP microplastics identified in river sediment samples. Dredged sediments also reported 66% of PVC from river sediment samples. Polyester (33%) and nylon (14%) were not as well identified in dredged sediments.

Fibres were the most abundant morphotypes (30 particles) followed by fragments (9 particles) and foam (8 particles). The fibres in dredged sediments accounted for 55% of all fibres within the Avon/Ōtākaro River. Half of all the foam microplastics in the Avon/ Ōtākaro River were from dredged locations. Size comparisons reported microplastics of 500 – 1000 µm were most abundant (30 particles) followed by 100 – 300 µm (10 particles) and 30 – 100 µm (7 particles). Dredged sediments contained 64% of all microplastics between 500 – 1000 µm. Microplastics between 100 – 300 µm were still most abundant in non-dredged sediments. Blue was most abundant colour in dredged sediments (24 particles) followed by white (9) > black (6) > red (4) > yellow (2) > green (2). No orange of multi-coloured polymers was identified.

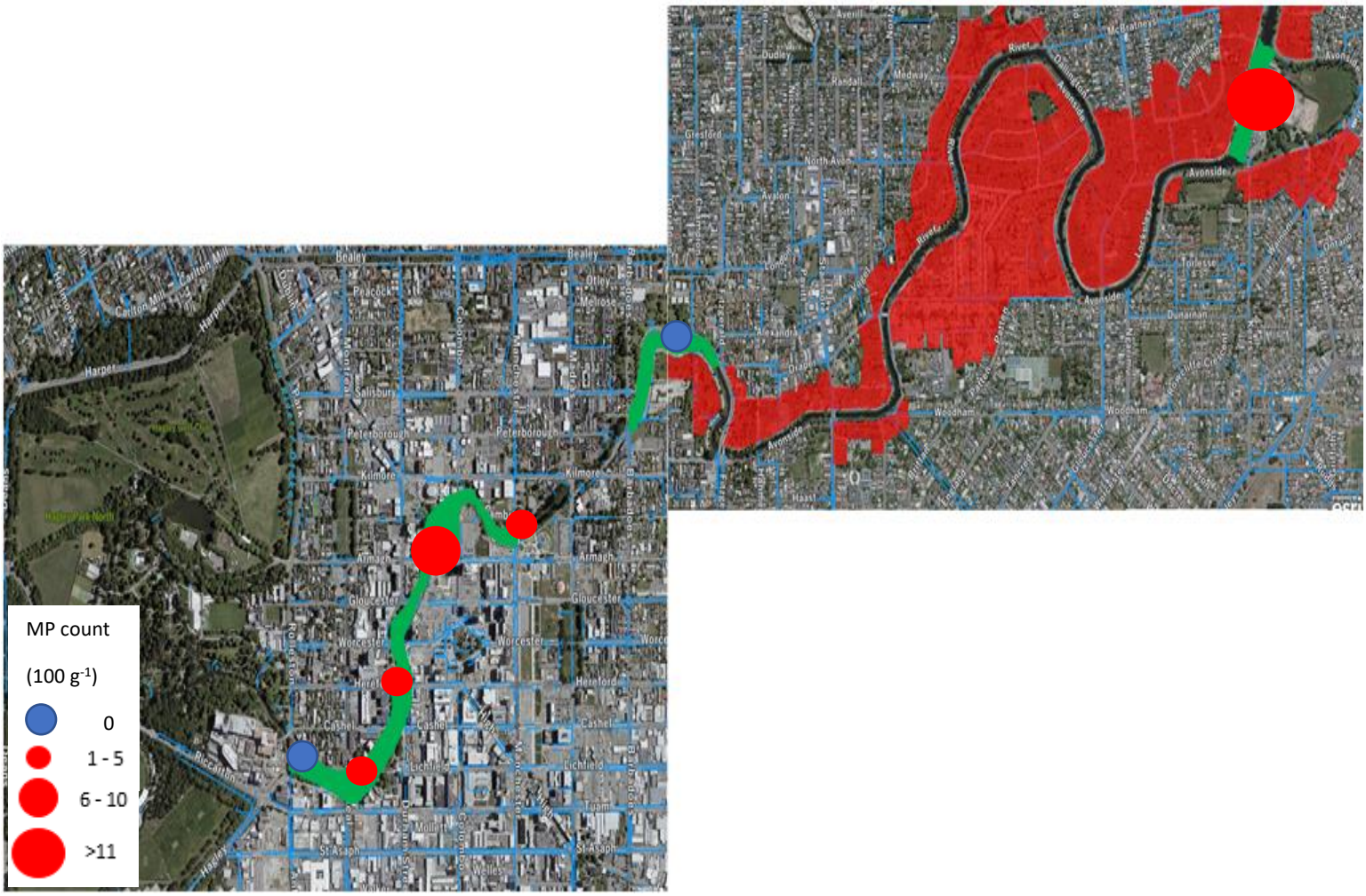

*Figure 4.14. Dredged locations of the Avon/Ōtākaro River showing the Avon River Project, Avon Loop and Kerrs Reach locations (from west to east) and the location of sediment sampling shown in triangles. *Blue lines show stormwater drains and red land shows the Residential Red Zone boundary.*

Table 4.7. Total count of isolated microplastics from sediment samples within dredged river projects along the Avon/Ōtākaro River.

Polymer	Avon River Project	Avon Loop	Kerrs Reach	Total
Nylon	2	0	0	2
Polyester	2	0	3	5
PE	2	0	0	2
PET	1	0	1	2
PP	1	0	23	24
PS	0	0	8	8
PVC	4	0	0	4
Total	12	0	35	47

4.3. Discussion

Total microplastic concentrations within the Avon/Ōtākaro River sediments ranged between 0 – 35 particles per 100 g^{-1}. Microplastic concentrations varied depending on the reach of the Avon/Ōtākaro River. Plastic polymers (PE, PP, PET and PS) were primarily identified within recreational locations such as Kerrs Reach and confluence locations while fibres were predominantly isolated from within the CBD. These values are significantly different to previous New Zealand studies. The average volume of microplastics reported within each sediment sample in Christchurch (6.5 particles per 100 g^{-1}) is equivalent to 65 particles per kg^{-1} with a variability of 0 – 350 particles per kg^{-1}. This concentration is comparable to reports conducted in streams within Auckland of 9 – 80 particles per kg^{-1} (Dikareva and Simon, 2019). There was no FT-IR identification conducted on polymers from these locations so microplastic identification could be lower. These results for the Avon/Ōtākaro River are consistent with concentrations measured in the Brisbane River (10 – 52 particles per kg^{-1}) (He et al., 2020) and the Qin River, China (10 – 97 particles per kg^{-1}) (Zhang et al., 2020). The results from the Avon/Ōtākaro River were considerably lower than microplastic concentrations identified in the Thames River, Canada (612 particles per kg^{-1}) (Corcoran et al., 2020), the Nakdong River, South Korea (1971 particles per kg^{-1}) (Eo et al., 2019), and the Milwaukee River, USA (32.9 – 4240 particles per kg^{-1}) (Lenaker et al., 2020). This could be a potential reflection on the lower population density within the Avon/ Ōtākaro river shed compared to watersheds of larger international city river basins.

Microplastics in the Avon/Ōtākaro River comprised of 5 main polymers (PET (26%), PP (21%), PS (19%), PE (6%), and PVC (4%)). Fibres were divided into polyester (10%), and nylon (9%). These results varied significantly to the Auckland and international research where ethylene ethyl/acrylate copolymer (EEAC) and PE were most commonly identified (Dikareva and Simon, 2019). The digestion method used within the Auckland study did limit the potential abundance of PET and PVC microplastics (Dikareva and Simon, 2019). New Zealand beach sediments show a similar correlation to this study as PET (30) > PE (21) > PP (6) were four of the most isolated microplastics (Bridson et al., 2020). Some international studies have reported PET as the main identifiable microplastic (e.g. Sakaret al., 2019; Lenaker et al., 2020) whilst other studies have reported PE as most abundant (Simon-Sanchez et al., 2019; Mani and Burkhardt-Holm, 2020). Fibres in river sediments also identified high

variability in composition internationally. Within the Seine River, Paris, PES and nylon (13%) were most common (Constant et al., 2020). Nylon and rayon were isolated within most urban sediment samples in Milwaukee (Lenaker et al., 2020).

Fragments (47%) and fibres (35%) were the most common morphotype identified in the Avon/Ōtākaro River sediments. Foams (10%) and pellets (8%) were also isolated. These results are consistent with the Auckland study where fragments (79%) were more common than fibres (20%) (Dikareva and Simon, 2019). No pellets were identified in Auckland study and foams only accounted for 1% of microplastics. Pellets were most common within wider New Zealand (Morra-Teddy and Matthaei, 2019) but these microplastics were isolated from surface water, not sediments. In New Zealand beach sediments, fibres (88%) were more abundant than fragments (8%) or films (4%) which show different results to freshwater sediments (Bridson et al., 2020). International studies report conflicting morphotype results for freshwater. Fragments (69%) were more common than fibres (30%) in South Korea (Eo et al., 2019) and the Tamsui River sediments, Taiwan (87%) (Wong et al., 2020). Fibres were more common in the Kelvin River, Scotland (83%) (Blair et al., 2019a), the Rhône and Têt rivers, France (98%) and (92%) (Constant et al., 2020), the Ganga River, India (91%) (Sakar et al., 2019) and Milwaukee River (29%) (Lenaker et al., 2020). Only one investigation reported foam as the highest microplastic morphotype (Lenaker et al., 2020). Microbeads were identified within some American and European rivers (Constant et al., 2020; Crew et al., 2020) but were not reported within any New Zealand study. This is most likely due to the 2018 ban on microbead use and production (Tilley, 2017).

Microplastic size varied across the Avon/Ōtākaro River, however, three microplastic size groups can be identified. Most microplastics were between small classes of 100 -300 µm (43%) and 30 – 100 µm (18%). There were a cluster of microplastics in the medium range 500 – 1000 µm (30%) but less than 2% were above 1000 µm. The trend of microplastic accumulation at or below 500 µm in Christchurch was also reported in Auckland (Dikareva and Simon, 2019) and wider New Zealand (Morra-Teddy and Matthaei, 2019). Both the Auckland and Avon/Ōtākaro River studies reported microplastic accumulation at or below 500 µm accounted for 50 – 60% of microplastics in sediments. The absence of microplastics above 1000 µm was consistent within the Auckland study as microplastics at or above 1000 µm were only reported to be 7% of all microplastics isolated. These results were

significantly different to New Zealand beach studies as microplastics less than 300 µm only accounted for 4% of all microplastics. The abundance of microplastics above 1000 µm (21%) was also conflicting to the Christchurch study. This result could be influenced by the different locations of river and ocean materials (Bridson et al., 2020). The prevalence of smaller microplastics (less than 500 µm) has been observed in other studies, regardless of morphotype (Ding et al., 2019; Simon-Sanchez et al., 2019; Eo et al., 2019). There was no correlation with microplastic colours.

Of the 156 microplastics isolated from the Avon/Ōtākaro River, 61% were derived from confluence locations of one of the Avon/Ōtākaro River's 11 tributaries. Microplastic polymers were deposited into the Avon/Ōtākaro River from suburban confluence zones. Central city tributaries transported fibres into the middle reach. The relationship of land use on microplastic accumulation in river sediments could not be completed due to the high percentage of microplastics which were transported from tributaries outside of the study location boundaries. Research into microplastics entering freshwater rivers from tributaries has not been previously reported within New Zealand. Overseas studies have identified tributaries as a considerable source of microplastics (Yonkos et al., 2014; Baldwin et al., 2016; Horton et al., 2017a) specifically, tributaries in urban watersheds. Within the UK, tributaries of the Tame and Thames River (Leach, Lambourn and the Cut) transport volumes exceeding 400 microplastics per kg^{-1} of sediment into the headwaters of the Thames before reaching urban settlements (Horton et al., 2017a; Tibbetts et al., 2018). The tributaries of the Great Lakes, USA, deposit 305 particles per kg^{-1} from urban sources. These microplastics were predominantly fibres (85%) and fragments (14%) (Baldwin et al., 2016). Similar results were reported in river to coastline tributaries of the USA. The tributaries that flow into Chesapeake Bay on average transport 347 particles per m^3 of water. This highlights the abundance of microplastics which originate from distant locations.

Flow rates and microplastic concentrations along the Avon/Ōtākaro River reported an apparent positive correlation. Fibres were primarily isolated from locations where flow rates were below 0.6 m/s^{-1}. Fragments and pellets were not witnessed until flow rates exceeded 0.8 m/s^{-1} in the middle reach. This apparent relationship may reflect the input of microplastics from tributaries within the lower reach of the Avon/Ōtākaro River and the increase in flow rate towards the lower reach. The lower land elevation within the lower

reach could also have impacted microplastic accumulation and flow rates (Appendix 8). While flow records were not collected in any previous New Zealand investigations, international research in Austria and Italy have reported that microplastics can become turbulent when river flows exceed 2.5 m/s^{-1} (Liederman et al., 2018; Campanale et al., 2020). This research is further supported by microplastic behaviour in storm flow conditions (above 4 m/s^{-1}) where microplastic polymers can travel between 1 – 23 kms from their source location before becoming benthic (Ockelford et al., 2020). As the Avon/Ōtākaro River can reach flow rates of 30 m/s^{-1} in heavy rainfall conditions, the distribution of microplastics from high flow conditions is highly probable for all locations sampled.

Total organic carbon concentrations from sediments in the Avon/Ōtākaro River ranged between 0.1 to 5.6%. The mean concentration across the Avon/Ōtākaro River was 2%. The mean concentration at confluence locations increased to 2.2%. Microplastic concentrations did not correlate to total organic carbon due to influence from tributaries and the resuspension and distribution of microplastics. Of the five highest TOC results, three were from confluence locations of lower reach tributaries. Whilst no other microplastic investigations have considered total organic carbon as an explanatory factor, previous research of the Avon/Ōtākaro by Gadd and Sykes (2014) reported TOC was considerably lower in dredged locations than other sediment samples (0.8% compared to 1.9% in this study). TOC results were higher in this study compared to TOC results from the 2014 study. This is due to the difference in sampling years and the accumulation of organic material within the river sediments over time. International studies have shown an exponential relationship between fine sediments with high TOC concentration and volume of microplastics (Li et al., 2020).

Dredged locations reported some of the lowest microplastic accumulation zones across the Avon/Ōtākaro River, with the exception of Kerrs Reach. Of the three locations sampled that produced no microplastics, two were in recently dredged locations. National research did not investigate sediment samples from dredged locations, but international research has reported the impact dredging has on temporal accumulation of material within sediments (Maes et al., 2017). The international research identifying dredged sediments as low accumulation zones of material would support the findings seen in this study (Maes et al., 2017).

4.4. Conclusions

Microplastic concentrations within the Avon/Ōtākaro River were variable between sample locations, ranging from 0 – 35 particles per 100 g^{-1} of sediment. PET was the most commonly isolated polymer in these sediments (26%). The polymer frequency within this study is similar to polymer frequency identification in international settings (Estoppey et al., 2015). Fragments were the most common morphotype (47%). Microplastics were generally smaller than 1000 µm (98%). These results are consistent with some of the national and international studies but as the Avon/Ōtākaro has a different catchment and tributary system to other rivers, some of the results regarding prominent polymer, morphotype and size differed. Flow rates reported an apparent relationship with microplastic accumulation, but this was influenced by an increase in flow rate within the lower reach. Microplastic input from tributaries and the lower land elevation in the lower reaches could have also influenced these results. This was evident in the accumulation of PS and PP microplastics in the lower reach. There was no relationship between microplastic accumulation and sediment TOC. Microplastic accumulation was lowest within dredged locations except for Kerrs Reach which is a recreational zone for boating and fishing.

5. Microplastics in Stormwater Drain Sediments

5.1. Introduction

The concentration and spatial distribution of microplastics within the Avon/Ōtākaro River was presented in Chapter 4. To assess the relationship between sediment and land-based microplastics, this chapter presents the results of microplastics collected from stormwater drain sediments adjacent to the Avon/Ōtākaro River. Stormwater drains are widely regarded as a major transportation route for terrestrial microplastics based on land to enter the freshwater environment (Kole et al., 2017; Sutton et al., 2019). Within urban environments, stormwater drains can be accumulation zones for microplastics which have fractured from everyday objects such as vehicles, buildings and roads (Kreider et al., 2010). The route of these stormwater drains determines microplastic accumulation. During high rainfall events, microplastics collected within stormwater drains can be transported directly into the freshwater networks (Liu et al., 2019), into stormwater retention basins (Ziajahromi et al., 2020) or into rivers (Piñon-Colin et al., 2020).

Within Christchurch, there is a lack of knowledge regarding stormwater as a source of microplastics. No investigations have been undertaken on the concentration of microplastics within stormwater drains or where the highest accumulation of land-based microplastics are. Knowledge is also lacking regarding the sources of land-based microplastics and the percentage of microplastics within stormwater drains which flow into freshwater systems.

5.1.1. Study objectives

- To determine the polymer, morphotype, size and colour of microplastics within Christchurch stormwater drain samples.
- To determine the composition and location of polymers and fibres with exterior coatings.
- To examine the relationship between microplastics and traffic density, land use and river sediment microplastic concentrations.

5.2. Results

5.2.1. Sediment concentrations of microplastics

Stormwater drain sediment samples were collected from the same six zones utilised for river sediment samples. Stormwater drain sediment samples were collected from every second river sediment sample location (Figure 2.2) (Appendix 9). Confirmed microplastic particles were collected from 14 of the 15 sample sites. Overall, 45 microplastics were isolated. Across all sample sites, the mean abundance was 3.3 particles per 100 g^{-1} of sediment and ranged from 0 – 8 particles per 100 g^{-1} of sediment. The highest concentration of microplastics was in Hagley Park (5.3 particles per 100 g^{-1} of sediment). The lowest concentration was in New Brighton (1 particle per 100 g^{-1} of sediment) (Figure 5.1). No microplastics were identified in one of the sediment samples from the New Brighton zone.

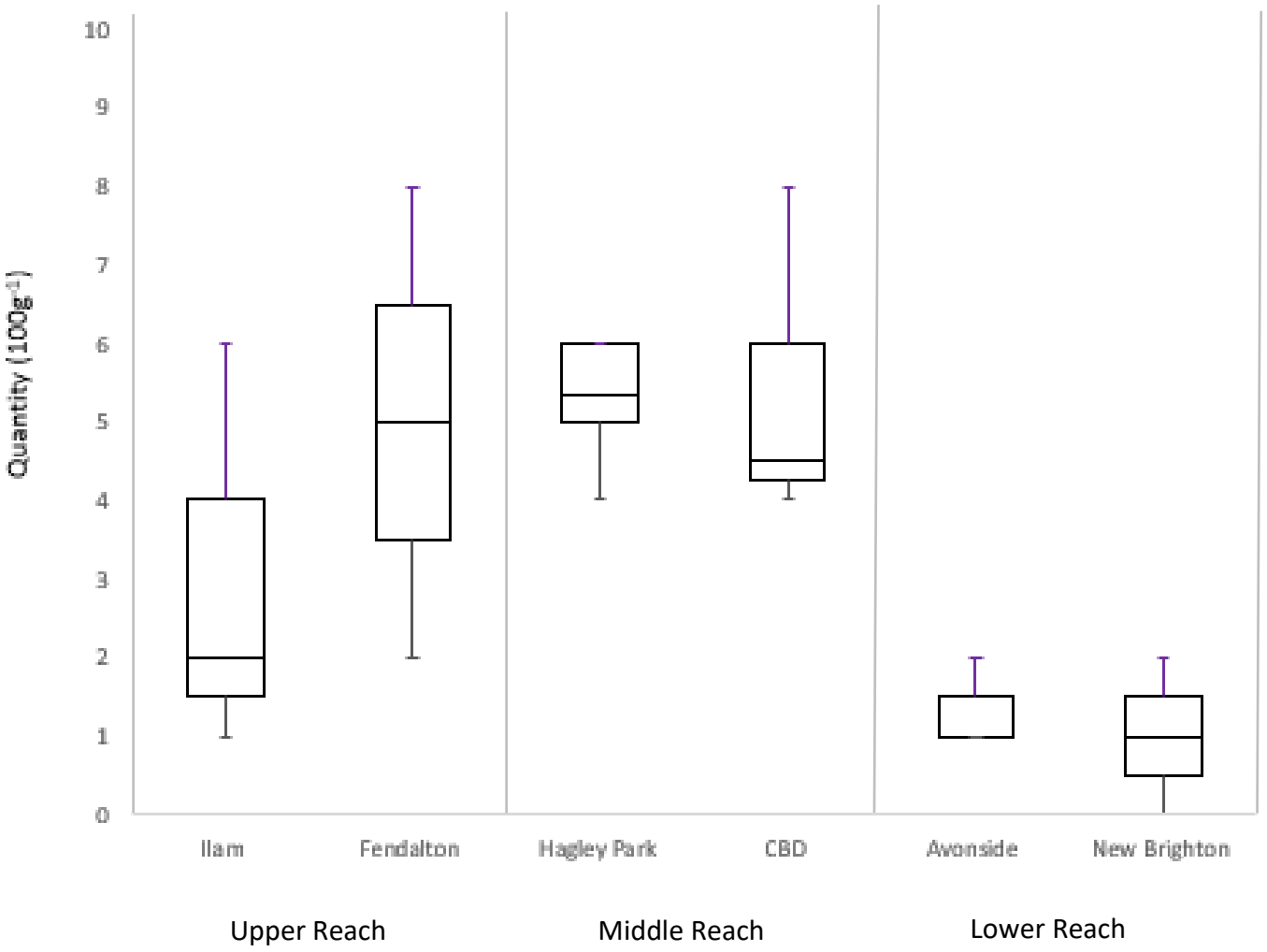

Figure 5.1. Microplastic polymers isolated from stormwater sediments along the Avon/Ōtākaro across the six zone locations in particles per 100 g^{-1} (n=45).

The highest abundance of microplastics isolated from stormwater drain sediments were in the upper and middle reaches of the Avon/Ōtākaro River compared to the lower reach. The concentrations of microplastics in the upper reach (3.8 particles per 100 g^{-1} of sediment) increased into the middle reach (5 particles per 100 g^{-1} of sediment) before concentrations decreased to 1.2 particles per 100 g^{-1} of sediment in the lower reach. Within the sample zones, microplastics within the middle reach accounted for half of all stormwater drain microplastics isolated. The remaining microplastics were isolated from the upper reach (38%) and lower reach (12%).

5.2.2. Polymer type

FT-IR microscopy of all 99 particles isolated from the stormwater sediments identified seventeen polymer types (Table 5.1). Of the 99 particles isolated from stormwater sediments, 45 of the 99 were confirmed as microplastic. Nylon, rubber and polypropylene were most abundant (Figure 5.2). Nylon (8 particles), rubber and PP (6 particles) were most commonly detected within upper and middle reaches. Acrylonitrile butadiene styrene plastic was identified for the first time in this study (Figure 5.3). Within this subset, 54 particles were identified as natural or, were not microplastic (54%) so were excluded from the final dataset. Of the twelve synthetic microplastic polymers isolated from stormwater drain sediments, nylon was most abundant (16%) followed by rubber and PP (12%), PE, (10%), PC (8%), polyester (6%), PET, ABS and PVC (4%). The remaining polymers only accounted for 2% of microplastics identified. Within the upper reach, microplastic polymers were identified in 8 forms. The most abundant was PP, PE (4 particles) and nylon (3 particles). These three polymers accounted for 73% of all microplastics within the upper reach. Rubber (5), silicone (5) and nylon (5) were the most commonly isolated microplastics from within the middle reach, accounting for 60% of microplastics in the middle reach. Other common polymers were Polycarbonates (4) and polyester (2). The remaining four polymers (PP, Polybutyl, PET and PVC) were only identified as a single item. Only six polymers were identified in the lower reach. The most abundant was ASB (2). The remaining five (PP, PE, polybutyl, PET and nylon) were all identified as a single item.

Table 5.1. Summary of all particles isolated from stormwater sediments within the 6 zones along the Avon/Ōtākaro River. (Total microplastics = 45, total natural particles = 54).

Polymer[3]	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
Rubber	0	1	3	2	0	0	**6**
PP	2	2	0	1	0	1	**6**
Silicone	0	0	5	0	0	0	**5**
Lignin	0	5	0	0	0	0	**5**
PE	3	1	0	0	0	1	**5**
PC	0	0	2	2	0	0	**4**
Cellophane	0	2	1	0	0	0	**3**
Polybutyl	1	0	0	1	1	0	**3**
PET	0	0	1	0	0	1	**2**
ABS	0	0	0	0	2	0	**2**
PVC	1	0	1	0	0	0	**2**
Sandpaper	0	0	0	1	0	0	**1**
Rayon	0	0	0	22	0	0	**22**
Cotton	0	1	2	14	0	2	**19**
Polyester	1	0	0	2	0	0	**3**
Nylon	3	0	2	2	1	0	**8**
Wool	0	0	3	0	0	0	**3**
Total	**11**	**12**	**20**	**47**	**4**	**5**	**99**

[3] Polycarbonate (PC), Acrylonitrile butadiene styrene (ABS).

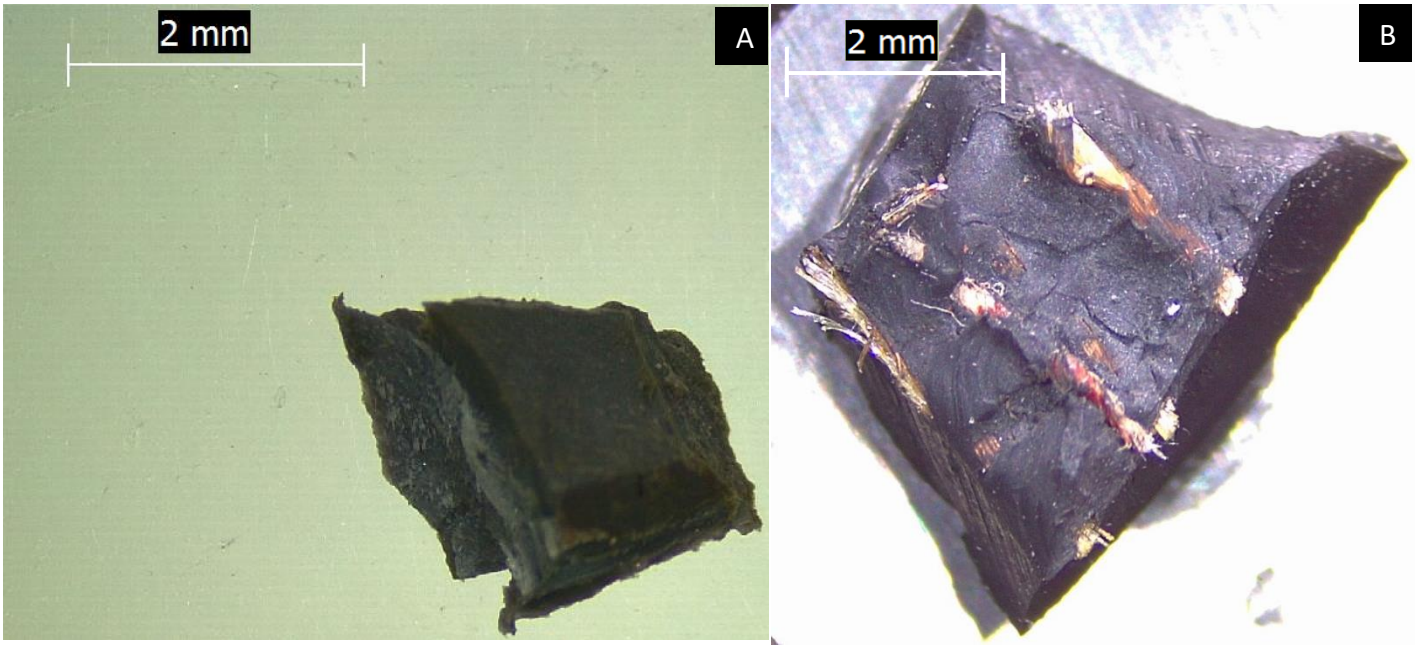

Figure 5.2. Rubber fragments found within Central Christchurch showing (A) section of road hump from Park Terrace and (B) bicycle tyre embedded white nylon fibres.

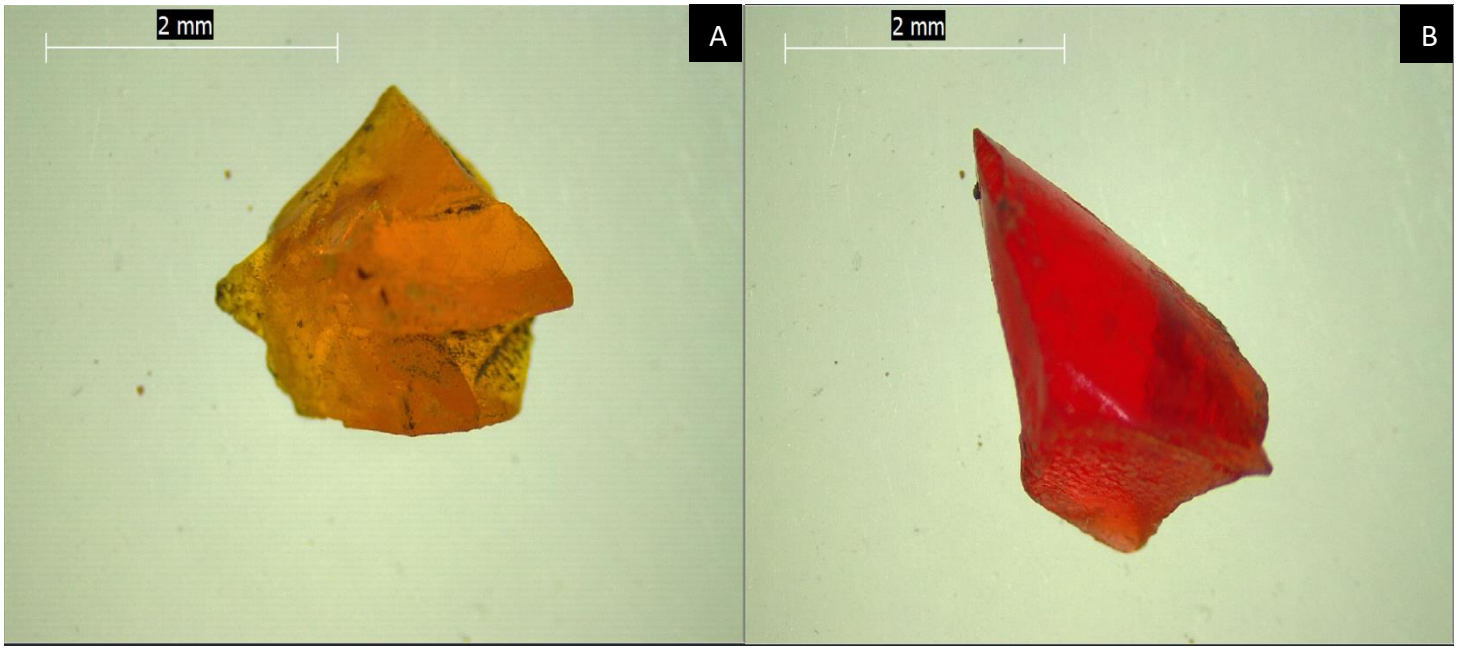

Figure 5.3. A variety of brake lights isolated from stormwater drain sediment within Avonside showing (A) yellow and (B) red acrylonitrile butadiene styrene microplastics.

5.2.3. Microplastic morphotype

Amongst the 45 microplastics isolated from FT-IR analysis, three morphotypes were identified (Figure 5.4) (Appendix 10). The frequency of detection was fragments (66%) > fibres (26%) > pellets (8%). Foams and films were not detected in any sample. Fragments were isolated across all sample locations. Fibres were identified across all sample zones except New Brighton. Within the upper reach, fragments accounted for 63% of microplastics isolated and fibres accounted for 26%. Pellet morphotypes only accounted for 11% as they were not identified in Fendalton. Within the middle reach, fragment morphotypes (68%) were most abundant. Fibres were identified as 28% of all morphotypes in the middle reach. A single pellet was identified in the CBD, accounting for 4% of all middle reach morphotypes. Within the lower reach, only a single fibre and pellet microplastic was identified. No fibres were identified in Avonside and no pellets were identified within New Brighton. Of the six microplastics identified, four were fragment morphotypes (66%).

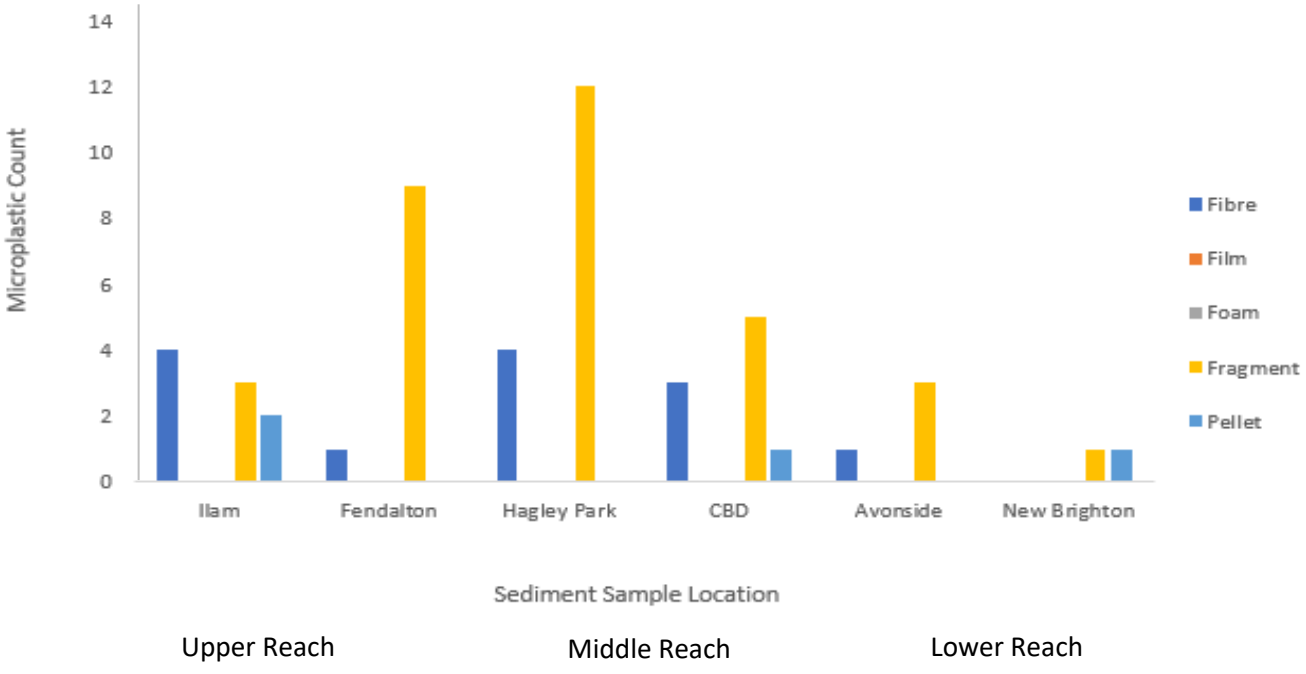

Figure 5.4. Microplastic morphotypes from stormwater drain samples along the Avon/Ōtākaro River.

5.2.4. Microplastic size

Microplastics were detected down to 30 µm in size (Figure 5.5) (Appendix 10). The most abundant microplastic size range was 500 – 1000 µm (34%), followed by 30 – 100 µm (24%), 1000 – 5000 µm (18%), 300 – 500 µm (14%) and 100 – 300 µm (10%). Microplastics between 500 - 1000 µm were isolated from stormwater drain sediments in all locations except New Brighton. Four size classes were identified in the upper reach. Microplastics between 500 – 1000 µm (8) were most abundant. Five microplastic particles were identified in both 30 – 100 µm and 300 – 500 µm size ranges. Only a single 100 – 300 µm microplastic was identified. No microplastics were identified above 1000 µm. All size ranges were identified in the middle reach. The most abundant microplastics were in the size range 500 – 1000 µm (32%). Size ranges of 1000 – 5000 µm and, 30 – 100 µm accounted for 52% of microplastics. Microplastics between 100 – 300 µm and 300 – 500 µm were identified twice. Microplastics in the lower reach identified two particles of 100 – 300 µm and 1000 – 5000 µm range. Only a single item was identified between 30 – 100 µm and 500 – 1000 µm. No microplastics were reported between 300 – 500 µm.

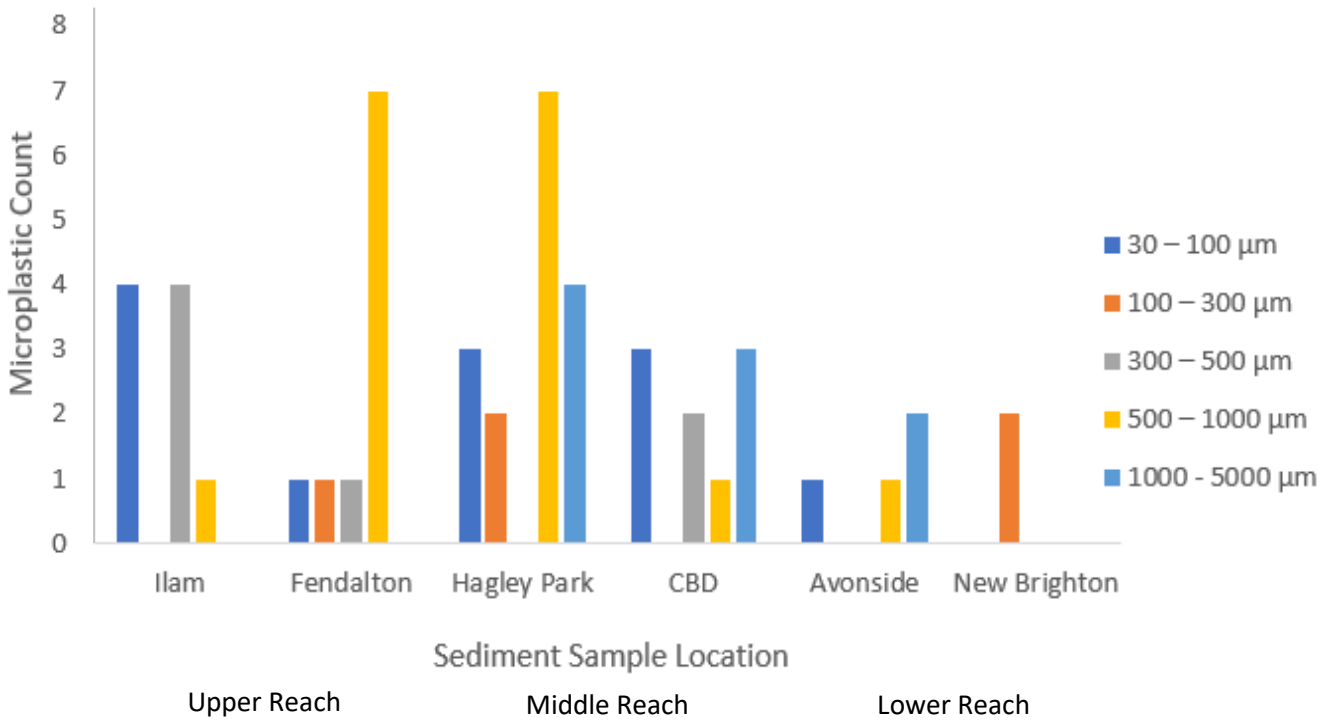

Figure 5.5. Microplastic size distribution within stormwater drains sediments along the Avon/Ōtākaro River.

5.2.5. Microplastic colour

From all stormwater drain sediments, 8 colours were identified (Figure 5.6) (Appendix 10). The most abundant colours were white (20%) > green (18%) > black (14%) > blue (12%) > yellow (12%) > red (10%) > orange (10%). Multi-coloured microplastics were only identified twice (4%) and only within the upper reach stormwater drain sediment samples. In the upper reach, green (7 particles) and yellow (6 particles) were most abundant accounting for 68% of microplastics. The remaining microplastics were derived from multi-coloured and blue (2 particles), orange and red (1 particle each). No black or white microplastics were reported. White (9 particles) and black (6 particles) were most abundant in the middle reach of Central Christchurch. Other colours identified were blue and orange (3 particles), red and green (2 particles). No multi-coloured or yellow microplastics were identified. Red microplastics (2 particles) were not common in the lower reach. A single microplastic each of orange, blue, black and white were also identified. No yellow, green or multi-coloured microplastics were identified in the stormwater drain sediments.

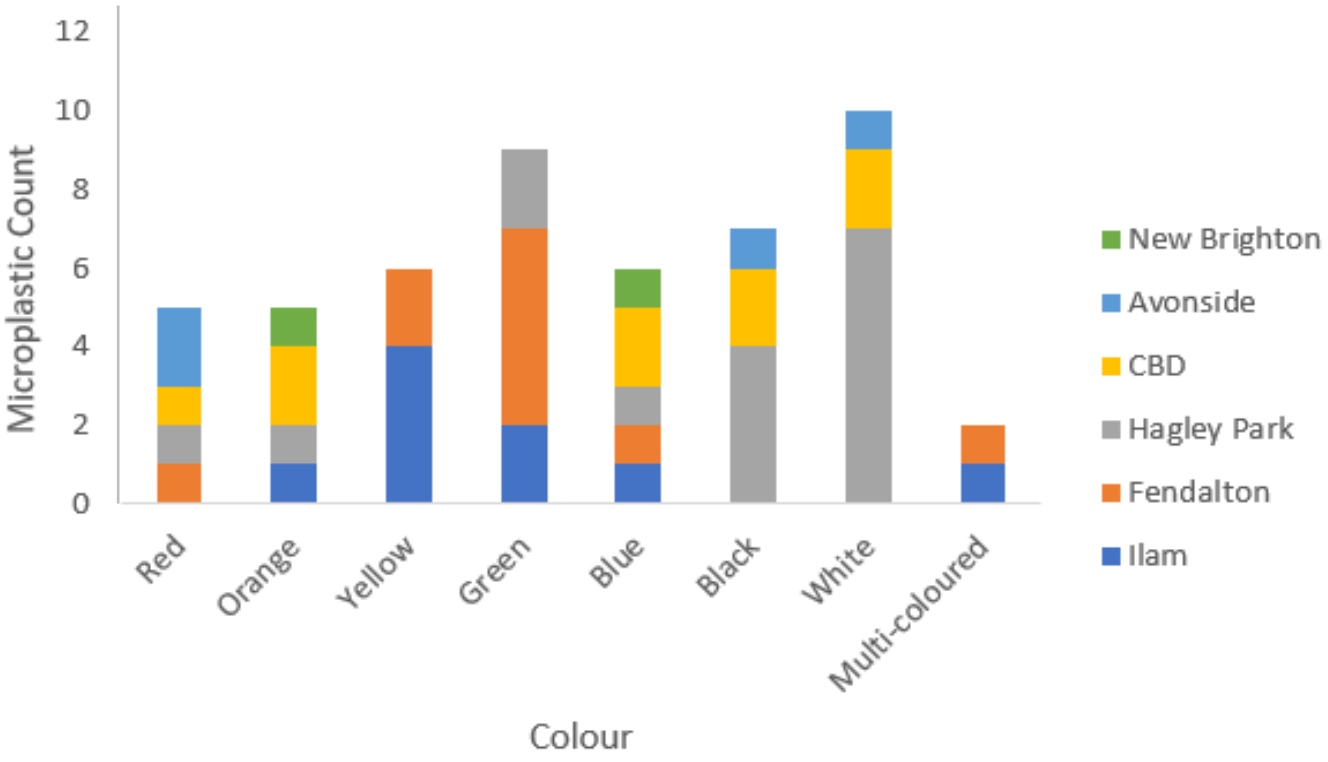

Figure 5.6. Microplastic colour identified in stormwater drains sediments along the Avon/Ōtākaro River.

5.2.6. External coatings on stormwater drain microplastics

Microplastics from stormwater drain sediments were analysed by FT-IR to identify external coatings (Table 5.2 to 5.4). Of the 45 microplastics isolated, 21 microplastics still contained an external coating (46%). Of the 21 microplastics with coatings, 10 were identified in the upper reach (48%), 7 microplastics were isolated in the middle reach (33%), and 4 microplastics were from the lower reach (19%).The external coatings reported correlations to surrounding land use. Coatings from paints and building materials were derived from residential land use in the upper reach. Open space land locations of Hagley Park reported coatings from transportation such as bicycle rubber and vehicle panelling.

In the upper reach, 10 of the 19 microplastics identified had an external coating (53%). The frequency of inner plastics with an external coating was reported as PE (4) > PP (3) > PVC (1) > Nylon (1) > Polyester (1). These coatings were predominantly identified on fragments (7 particles), fibres (2 particles) and pellet (1 particle). Of the 10 external coatings identified, 6 were identified as an additive (paints, outdoor materials, manufacturing). The remaining 4 coatings were identified to be from localised activities (cooking, food packaging, ammunition). Only 1 coating was from transportation (bicycles).

Only 7 of the 25 microplastics isolated from stormwater drain sediments identified an external coating in the middle reach (28%). The frequency of external coatings on microplastics was reported as PVC (2) > Silicone (2) > PE (1) > PET (1) > Nylon (1). Most coatings were identified on fragments (6 particles) and fibres (1 item). Of the 7 external coatings identified, 3 were related to transportation activities (bicycles and cars), 3 were related to outdoor preservation (paints and coatings) and 1 coating was from human activities (cigarette smoking).

Of the 6 microplastics identified across the lower reach, 4 particles had an external coating (66%). The frequency of detection was reported as Silicone (1) > PP (1) > PE (1) > nylon (1). External coating applications varied across microplastics identified in the lower reach compared to other sample locations. Microplastics were identified to have been derived from different building materials (fire-resistant plastic, electrical components, plexiglass) and mechanical uses (degreasers). These coatings were derived from locations where residential housing once stood.

Table 5.2. External coatings on microplastics from stormwater drain sediments within the upper reach.

External Coating	Location	Polymer	Application
Bisphenol A	Ilam	Polyethylene	Plastic chemical
CTG 076	Ilam	Polypropylene	Paint & sealants
CTA 87 C0479	Ilam	Polypropylene	Outdoor building materials
Durocryl 720	Ilam	Polyvinyl Chloride	Production of glue
Ethylene glycol	Ilam	Polyester	Manufacturing polyester
Spensol L-51	Ilam	Polypropylene	Water-based coating
Acrylamide	Fendalton	Polyethylene	Cooking of starch foods
Alox 2138F	Fendalton	Polyethylene	Ammunition lubrication
CTA 87 A0461	Fendalton	Nylon	Bicycle rubber
Vinyl alcohol	Fendalton	Polyethylene	Food packaging

Table 5.3. External coatings on microplastics within stormwater drain sediments in the middle reach.

External Coating	Location	Polymer	Application
CTA 87 A0442	Hagley Park	Nylon	Bicycle rubber
Gilsonite	Hagley Park	Polyethylene Terephthalate	Asphalt coating
Latex	Hagley Park	Polyethylene	Coating materials (paint)
Polyvinyl Acetate	Hagley Park	Polyvinyl Chloride	Latex paint
Acrylonitrile butadiene styrene	CBD		Automobile body parts
Cellulose Acetate	CBD	Polyvinyl Chloride	Cigarette filters
Polycarbonate	CBD	Silicone	Vehicle brake lights

Table 5.4. External coatings on microplastics within stormwater drain sediments in the lower reach.

External Coating	Location	Polymer	Application
Acrylonitrile butadiene styrene (FR)	Avonside	Silicone	Fire-resistant materials
DuPont R-940	New Brighton	Polypropylene	Electrical components
Poly(methacrylate)	New Brighton	Polyethylene	Acrylic/Plexiglass
Whitconate	New Brighton	Nylon	Degreaser and wetting agent

5.2.7. Influence of traffic settings on stormwater drain microplastics

Microplastics from stormwater drain sediments were isolated from various traffic settings. These sample locations were divided into three traffic settings: carparks, intersections and roadsides (Table 5.5). The statistics of traffic movement from 2015 were utilised to compare microplastic accumulation to traffic volume (Table 5.6 to Table 5.8) (Christchurch City Council, 2015). The highest microplastic accumulation zones across all stormwater drain sediment samples were roadsides and carparks (76%). The remaining 24% were derived from intersections. Within the upper reach, the highest quantity of microplastics were derived from roadsides (53%) and intersections (47%) and no microplastics were identified for carpark settings. Within the middle reach, microplastics were most abundant in carpark sediments (64%) followed by roadsides (36%). No microplastics were reported for intersection settings. In the lower reach, microplastics were evenly distributed between carparks and intersections (50%) with none for roadsides.

Table 5. 5. Traffic settings along the Avon/Ōtākaro River where stormwater drain microplastics were isolated.

Location	Carpark	Intersection	Road	Total
Upper reach	0	9	10	19
Middle Reach	16	0	9	25
Lower Reach	3	3	0	6
Total	**19**	**12**	**19**	**50**

Table 5. 6. Traffic volume at stormwater drain sample locations in the upper reach.

Sample Location	Traffic Volume (Per hour) (2015)
University Drive	383
Puriri / Kotare Street	No data
Harakeke Street	170
Harper Ave / Fendalton Rd	561
Carlton Mill / Hewetts Rd	404

Table 5. 7. Traffic volume at stormwater drain sample locations in the middle reach.

Sample Location	Traffic Volume (Per Hour) (2015)
Park Tce / Armagh St	577
Te Kura Tennis Club	No data
Cambridge Tce / Cashel St	322
Durham St South / Cashel St	284
Manchester St / Oxford Tce	295

Table 5.8. Traffic volume at stormwater drain sample locations in the lower reach.

Sample Location	Traffic Volume (Per Hour) (2015)
River Rd / Stanmore Rd	148
Hockey Ln / Avonside Dr	No data
Avondale / New Brighton Rd	89
New Brighton Rd/ ANZAC Dr	71
Owles Tce / Hardy St	No data

Traffic volume correlated with microplastic accumulation (Figure 5.7). This data, however, is of 2015 origin as no recent data was available. Concentrations of microplastics greater than 4 particles per 100 g^{-1} of sediment were identified in the middle reach and major intersections of the upper reach where traffic volumes exceeded 250 vehicles per hour. The higher volume of traffic and microplastics within the middle reach highlighted the impact of vehicles on microplastic accumulation through the identification of polycarbonate brake lights (Table 5.6) and bicycle tyre rubber (Table 5.5).

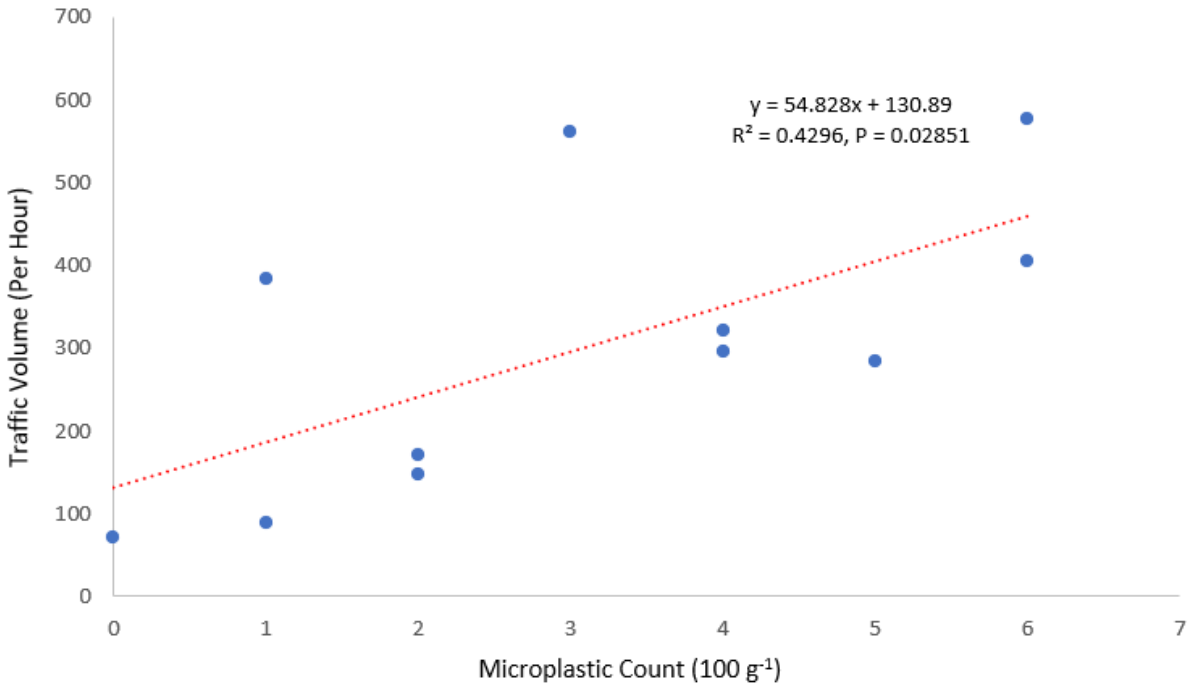

Figure 5.7. Relationship of stormwater drain microplastics and traffic volumes from along the upper, middle and lower reaches of the Avon/Ōtākaro River.

5.2.8. Influence of land use on microplastic accumulation

Microplastics isolated from stormwater drain sediment samples were divided into four land zone categories: Open space (areas of green space such as Riccarton Bush, Mona Vale, and Hagley Park), Residential, Special purpose (University of Canterbury, Christchurch Girls' High School, Christs' College, Christchurch Hospital) and Commercial/Business (Figure 5.8 to 5.9). Land within the lower reach was classed as 'Residential Red Zone' as the land is no longer suited for residential purpose and has not been assigned 'open space' zoning (Figure 5.10). The highest abundance of microplastics was reported within Open space (34%) and Special purpose (30%) land zones (Table 5.9). No microplastics were identified in commercial land zones in the upper reach or residential land in the middle reach as these areas did not exist within the sample network.

Microplastics were identified from three land zones in the upper reach. Special purpose land zones contained the highest abundance (47%) followed by residential (37%) and Open space (16%). Fragments were identified in all land zones. Fibres were identified only in Special purpose and Residential land zones. Fibres in special purpose zones (nylon) were identified from high school locations. The polymer and colour of which were derived from school uniforms. Microplastic accumulation in stormwater drain sediments from the middle reach was predominantly within Open space land zones (48%) followed by Commercial (36%) and Special purpose (16%). As reported in the upper reach, fragments were identified in all land zones. Fibres were not isolated in special purpose zone. Microplastics in the lower reach are not comparable due to the single land use within this location.

Table 5.9. Land use allocation of stormwater drain sediment microplastics along the Avon/Ōtākaro River.

Location	Open Space	Residential	Special Purpose	Commercial	Total
Upper Reach	3	7	9	0	19
Middle Reach	12	0	4	9	25
Lower Reach	0	0	0	0	0
Total	15	7	13	9	44

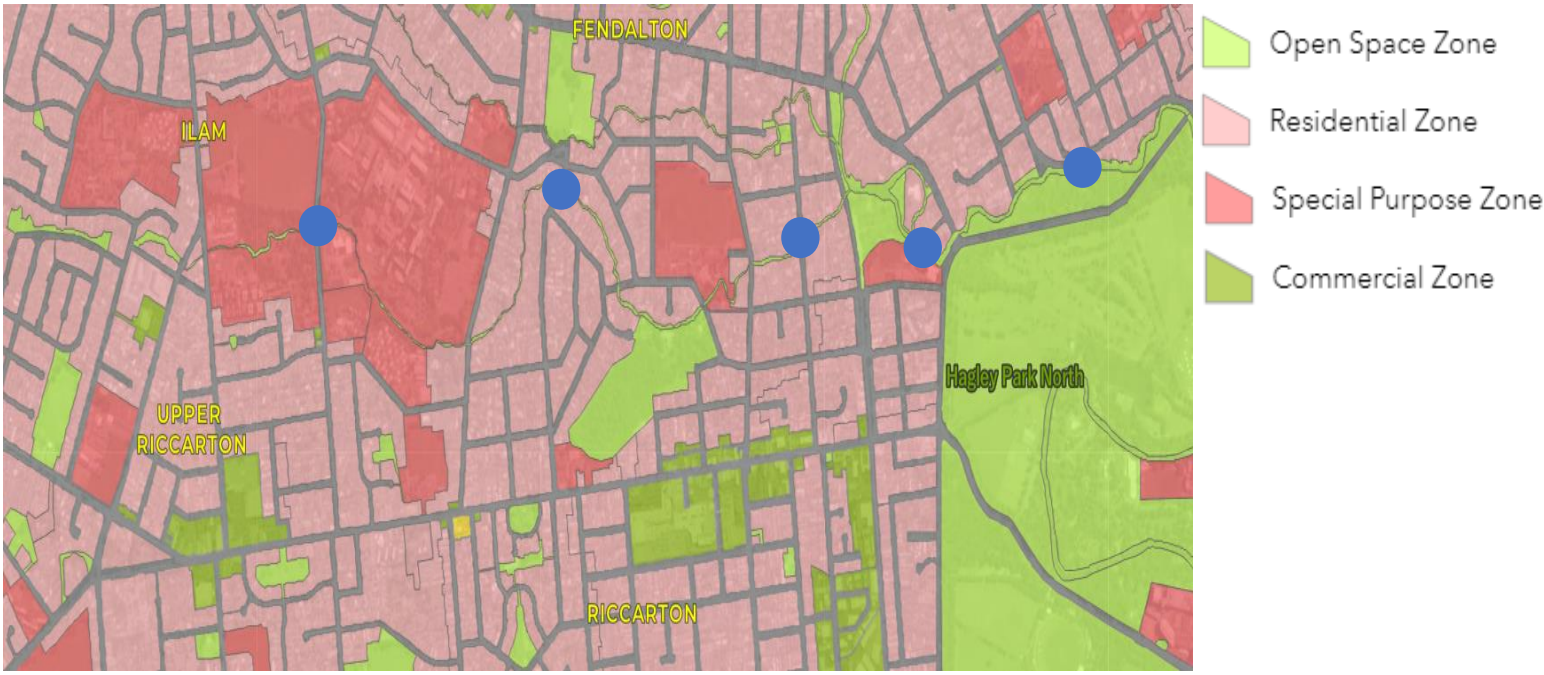

Figure 5.8. Land zones of stormwater drain sediment sample locations within the upper reach of the Avon/Ōtākaro River (adapted from Environment Canterbury, 2020).

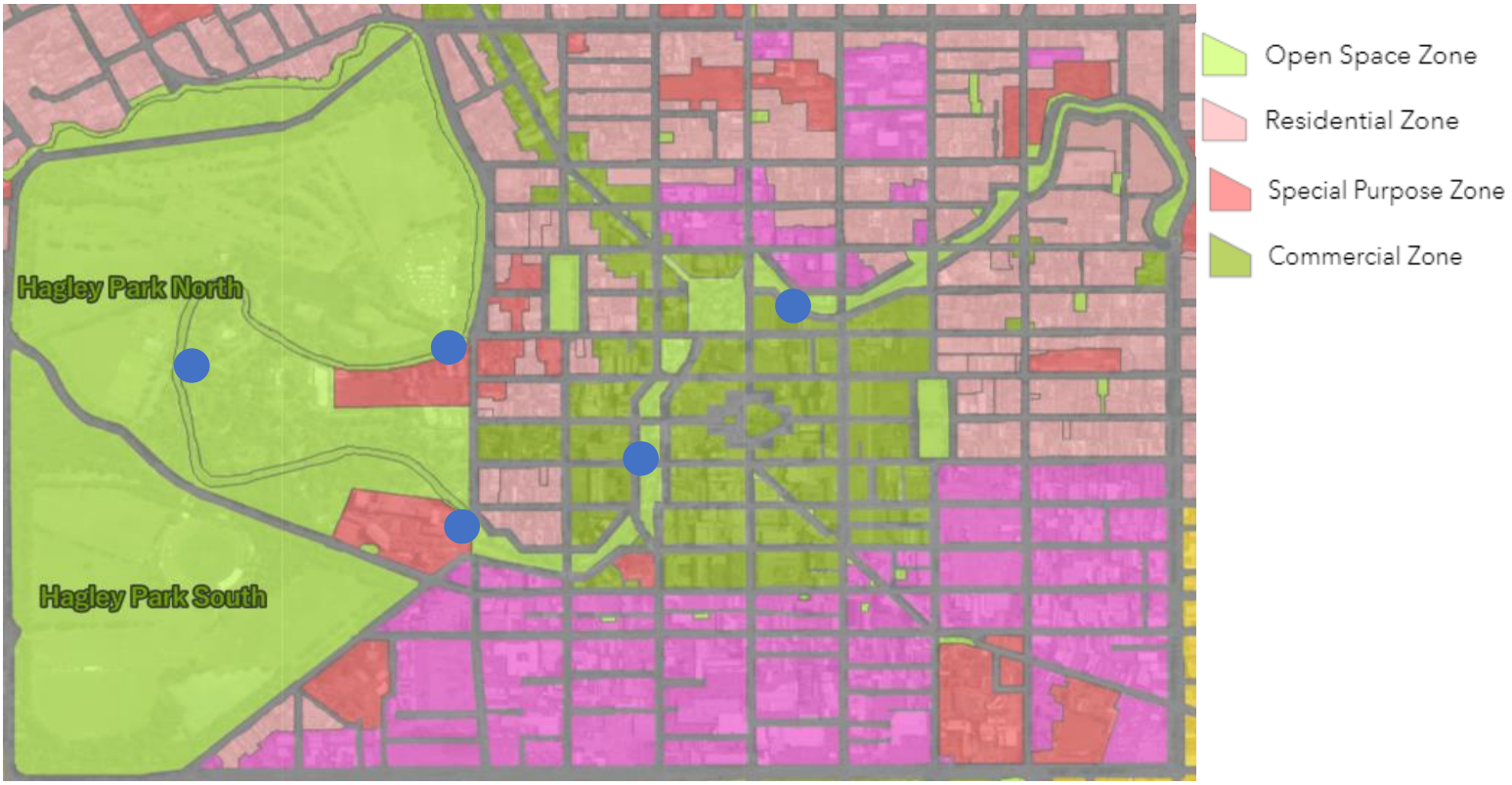

Figure 5.9. Land zones of stormwater drain sediment sample locations within the middle reach of the Avon/Ōtākaro River (adapted from Environment Canterbury, 2020).

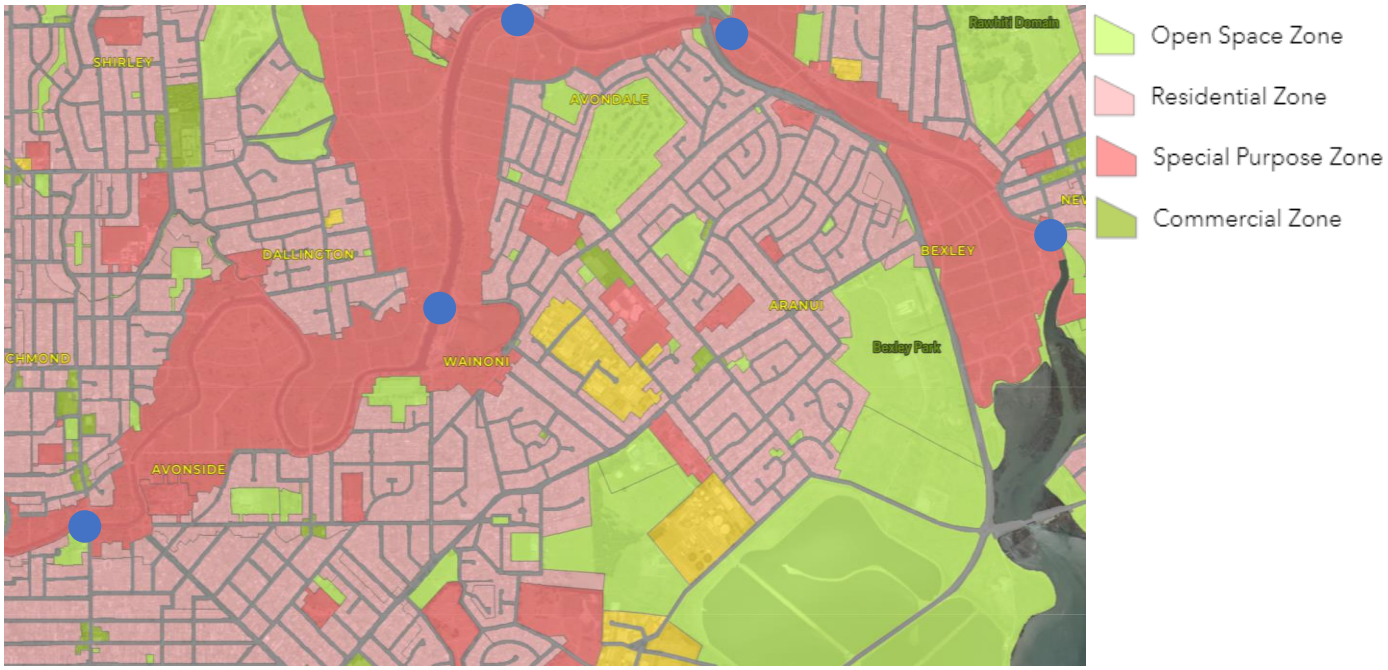

Figure 5.10. Land zones of stormwater drain sediment sample locations within the lower reach of the Avon/Ōtākaro River (adapted from Environment Canterbury, 2020).

5.2.9. Spatial relationship between river and stormwater sediment microplastics

There was no relationship present between stormwater drain sediment microplastics and river sediment microplastics (Figure 5.11). The highest accumulation zones of stormwater drain microplastics were in the upper and middle reaches whilst the highest accumulation zone of river sediment microplastics were in the lower reach. Polymers were also variable between the two sediment sources. Within stormwater drain sediments, nylon (8) was most abundant followed by rubber and PE (6). This is a significant contrast to river microplastics where PET (41) > PS (29) > PE (10) were most abundant. PP was identified in both sediment classes as the second-most abundant polymer (33 particles in river sediments and 6 in stormwater drains). Rubber and nylon, whilst identified as 31% of all stormwater drain microplastics, only accounted for 10% of river microplastics isolated.

Microplastics from stormwater drains and river sediments reported similarities across all samples. Fragments were most abundant in both sediment types (66% in stormwater drains and 47% in river sediments) followed by fibres (26% and 35%), and pellets (8% in both sediment types). Both sediment types also identified no film morphotypes. The only difference between the sediment types was the identification of foam morphotypes.

Microplastic sizes varied significantly between stormwater drain and river sediments. Most stormwater drain microplastics were between 500 – 1000 µm (34%) whilst microplastics in river sediments were most abundant in 100 – 300 µm ranges (43%). Microplastics greater than 1000 µm accounted for 18% of stormwater drain microplastics but only 2% of river sediment microplastics. The higher size range in stormwater drain microplastics is most likely due to the lack of fracturing and breakdown due to a shorter time within the environment compared to river microplastics.

Eight colours were identified within stormwater drain sediments. White was the most commonly identified colour (20%). Red was most abundant in river sediments (27%). Blue (12% and 26%), black (14% and 19%) and white (20 % and 21%) were commonly identified in both sediment types. Orange, yellow and green colours only accounted for 6% of river sediment colours. This identification increased to 40% in stormwater drain sediments. The greater concentration of microplastics within these colour ranges were also correlated to external coatings highlighting the use of these primary colouring in the external resin coating production.

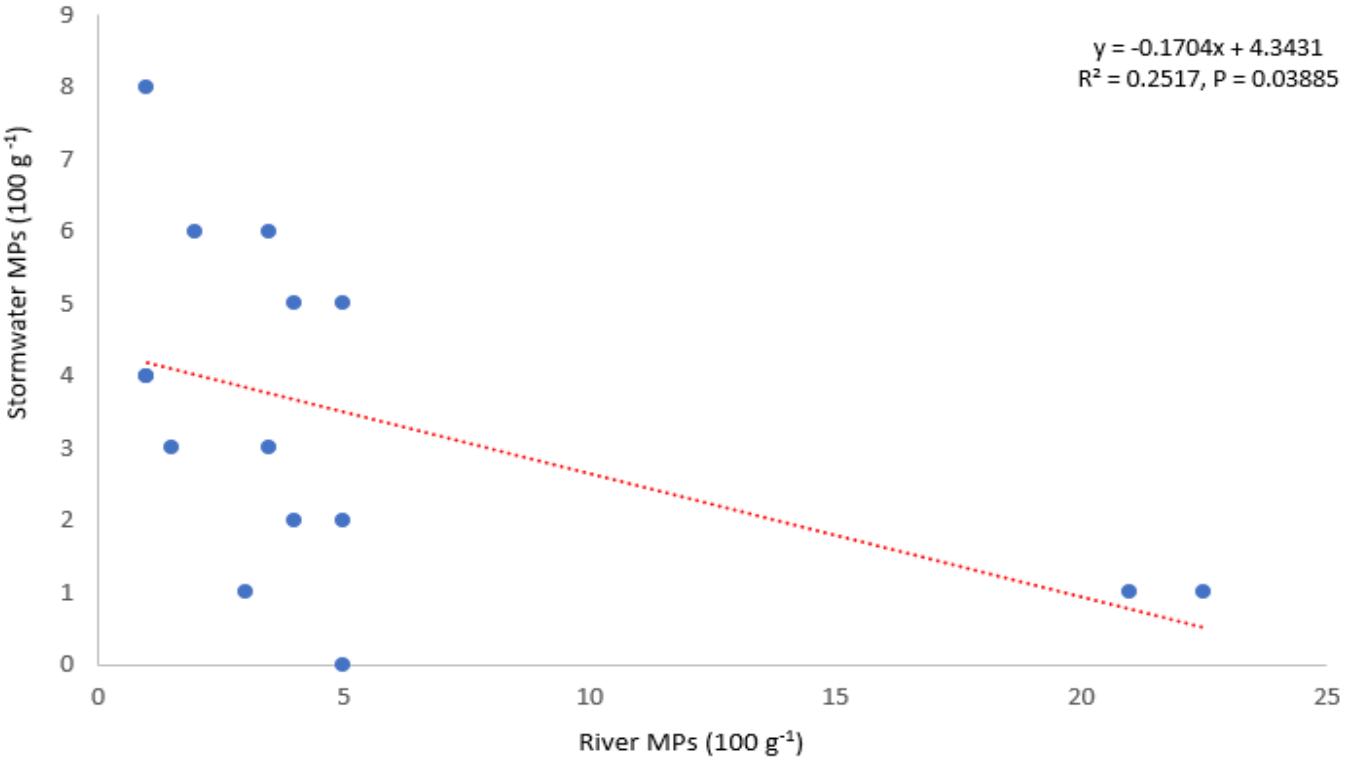

Figure 5.11. Relationship between stormwater drain-based microplastics and microplastics from the Avon/Ōtākaro River sediments.

5.3. Discussion

Total microplastic concentrations in stormwater drain sediments along the Avon/Ōtākaro River ranged between 0 – 8 particles per 100 g^{-1}. The concentrations of microplastics varied depending on the sample location. The highest mean concentration of microplastics was within the middle reach (5.3 particles per 100 g^{-1} of sediment). The lowest concentration of microplastics was within the lower reach (1 item per 100 g^{-1} of sediment). There is no New Zealand data available for comparison. The concentration of microplastics within Christchurch are comparable to other studies internationally investigating microplastics within city centres of Beshehr, Iran (Abbasi et al., 2017), Tehran, Iran (Dehghani et al., 2017) and Japan, Nepal and Vietnam (Yukioka et al., 2020) where concentrations varied between 0 – 12 particles per 100 g^{-1}. The results of these studies, however, become less comparable within suburban Christchurch (upper and lower reaches) as these investigations focused primarily in commercial locations and limited suburban environments.

Nylon (8 particles) was the most abundant polymer across the stormwater drain sediment results. Rubber and PP microplastics accounted for 6 particles each. Spatially, nylon and PP microplastics were only identified in four of the six sampling zones. Rubber was reported in only three sampling zones. Internationally, PE, PET and PP microplastics are most abundant in land-based sediments throughout Asia (Yukioka, et al., 2020), Africa, (Matsuguma et al., 2017) and Europe (Rodrigues et al., 2018). While the identification of rubber as one of the most abundant forms of microplastics is in agreement with international research, higher concentrations have been reported overseas (Kole et al., 2017; Polukarova et al., 2020). The accumulation of tyre wear particles is prevalent in higher volumes internationally (greater than Christchurch 400 vehicles per hour traffic volume) (Sieber et al., 2020). The lack of rubber microplastics compared to international research could be due to the variety of transport options available to residents of Christchurch or that street sweeping removed the tyre rubber from drain systems (NIVA, 2018; Halle et al., 2020; Polukarova et al., 2020).

Only three morphotypes were identified from the 45 microplastics isolated. Fragments (66%) were the most abundant morphotype followed by fibres (26%) and pellets (8%). No film or foam microplastics were identified. The fragments identified were derived from several uses and applications. The fibres identified outside of schools within Special purpose land zones were derived from the school uniforms. The colours of nylon fibres identified

outside Christchurch Girls' High School and Christ's College correlate to the composition of their uniforms. Fibres outside of these land areas were in zones of high population movement. The abundance of fragments is consistent with results from previous studies (Kole et al., 2017; Rodrigues et al., 2018; Yukioka et al., 2020). The identification of fragmented vehicle plastics such as ABS and PC agreed with international studies on vehicle derived microplastics (NIVA, 2018). Some research within Asia, however, has noted that other morphotypes such as microbeads were of higher quantities, exceeding 80% of all microplastic morphotypes, regardless of suburban or city land zoning (Chen et al., 2020). This is most likely due to poor waste management practices in place within Asia and other developing locations (Steinmetz et al., 2016). The dispersal of fibres was a feature noted within this study, whilst the cause of fibre dispersal was not investigated in this study, fibre distribution in other studies has been reported to be from aeolian processes or high population concentration in a given location, such as a city centre or tourist attraction (Enyoh et al., 2019; Liu et al., 2019; Wright et al., 2020).

Size ranges varied across the stormwater drain sediment samples. The most common size range was between 500 – 1000 µm (34%). Microplastics in this range were predominantly isolated in the upper and middle reaches. All fibres identified were in the 30 -100 µm category (24%). The remaining size ranges were 1000 – 5000 µm (18%), 300 – 500 µm (14%) and 100 – 300 µm (10%). Only 10% of microplastics were classed above 1000 µm. Within international literature, various size ranges were identified due to population density and waste management processes (Lui et al., 2019). The size distribution of rubber from tyres in this study were between 1000 – 5000 µm which contradicts results from Norway where tyre particles were only identified below 350 µm (NIVA, 2018). The presence of microplastics within 30 – 100 µm was uncommon in international studies (Enyoh et al., 2019; Lui et al., 2019; Yukioka et al., 2020). This is due to the higher abundance of fragmented plastics which are significantly larger in size than fibres and other smaller microplastics. The limited identification of microplastics between 300 – 500 µm was also uncommon internationally. As stated by Yuikoka et al., (2020), the identification of large microplastics is due to the poor waste management practices within Asia hence, smaller plastics are limited. Unlike microplastics in Christchurch, most microplastics in Asia are the result of littering rather than the releasing of fibres or from vehicle materials.

External coatings on polymers within Christchurch were derived from a variety of uses such as recreation, additives and external paints and fillers. International studies have reported the impact of synthetic coatings from paints which flake from ship hulls into the ocean (Song et al., 2014; Van Wezel et al., 2016). Some coatings have been proven to withstand saturation in water such as PLA and PBSA coatings (De Falco et al., 2019). Other investigations have reported that when microplastics are submerged in water, the external coating can begin to dissolve from anywhere between 2 days to 3 weeks (Kokalj et al., 2019).

The identification of microplastics within different traffic settings in Christchurch reported carparks and roadsides as the most abundant accumulation zones (76%) followed by intersections (24%). Microplastic accumulation was also correlated to traffic volume ($P = 0.02$). Research from Norway and the UK identified that microplastics which accumulate in stormwater drains from traffic movement will either deposit into the stormwater system (49%) or, will accumulate within carparks, intersections and along roadsides (51%) (Amundsen and Roseth, 2004; Unice et al., 2019). As the total volume of microplastics that entered freshwater systems from stormwater drains cannot be confirmed, no correlations can be drawn between international research and this study regarding traffic-based microplastics.

Land use was a distinct characteristic in microplastic concentrations. The highest accumulation of microplastics was identified within Open space (34%) and Special purpose (30%) land zones. Due to the rezoning of land within the lower reach of the Avon/Ōtākaro River, correlations between microplastic accumulation and land use were not conducted. Research into microplastics on different land zones is not available in New Zealand although, the impact of plastics on land animals has been conducted (Harper and Fowler, 1987). The lack of microplastics in residential land zones of Christchurch is supported by studies in Melbourne, Australia which reported a decrease in microplastics within residential zones and an increase in microplastics within industrial and commercial zones (Townsend et al., 2019). Microplastic concentrations from different land uses is also supported in literature reviews which identify higher population movement zones such as industrial, commercial and school zones to have higher quantities of microplastics than residential zones (Bondelind et al., 2018; Fahrenfeld et al., 2019a). The differing land surface types have also

been investigated in several land zones (Peters and Bratton, 2016) which identified paved surfaces as a higher accumulation zone of microplastics over grass by 75%. Some research has reported there is no comparable difference between microplastics in residential and commercial land zones (Alam et al., 2019), however, these areas of comparison are slum lands where building regulations regarding building proximity to the river boundary is not enforced.

The comparison between stormwater and river microplastics reported significant differences between concentration rates, polymer identification, size and colour ranges. Only fragment and fibre morphotypes were comparable between the two sediment types. Nylon (8), rubber and PP microplastics (6) were most abundant in stormwater drain sediments. This was a significantly lower volume to the highest polymer accumulations in river sediments where PET (41) > PP (33) PS (29) > PE (10) were most abundant. Fragments (66% in stormwater drains and 47% in river sediments) were the most abundant morphotype within both sediment types. Fibres were the second-most abundant (26% and 35%). No films were noted in either sediment study. Microplastics sizes were predominantly larger in stormwater drain sediments than river microplastics. Stormwater drain microplastics were primarily identified between 500 – 1000 µm (34%). River sediment microplastics were most abundant in 100 – 300 µm range (43%). Microplastics greater than 1000 µm were seldomly identified. This could be the result of fracturing and breakdown within the environment. Colours were highly variable within both sediment types. White (20%) was most abundant in stormwater drains and red (27%) was most abundant in river sediments. Multi-coloured microplastics were only identified five times across both sediment types. The higher identification of orange, yellow and green microplastics in stormwater drain sediments (40%) provided layers which were identified within external coating identifications. This identiifed that external coating identification could be more likely processed through primary coloured microplastics than black and white microplastics.

5.4. Conclusions

Microplastic concentrations within stormwater drain sediments had a low variability between sample locations of 0 – 8 particles per 100 g^{-1} of sediment. Nylon was the most commonly isolated polymer from these sediments (8 particles) followed by rubber and PP (6 particles). Fragments were the most abundant microplastic morphotype accounting for 66% of microplastics isolated. Microplastics between 500 – 1000 µm were most common (34%). Only 18% of microplastics were above 1000 µm in size. White (20%) was the most common polymer colour. Multi-coloured particles were the lowest (4%). External polymer coatings related directly to applications applied on land use. Additives and paint could be identified from homes in suburban locations. Coatings from rubber production and vehicle panelling were identified in high traffic volume locations within Central Christchurch. The results of this study are consistent with some international studies, however, the total count of microplastics is much lower than what has been identified overseas. This is most likely due to the better waste management practices in operation throughout Christchurch. Microplastic accumulation in different land settings reported more microplastics in open space or commercial areas than residential. This is consistent with international studies which identify the driver of accumulation in these land zones are higher population movement and industrial operations.

6. Summary

6.1. Conclusions

This thesis has presented the results of microplastic accumulation in river and stormwater drain sediments. Concentrations of microplastics within river sediments had a higher mean abundance (5.2 particles per 100 g^{-1}). The highest concentration of river microplastics were in the lower reach zone of Avonside (10.4 particles per 100 g^{-1}). The results presented from this study are comparable to previous research in Auckland, Brisbane, Melbourne, and Shanxi (China). Concentrations were lower than reported in South Korea, Canada and the USA. The mean abundance of stormwater drain sediments was 3.3 particles per 100 g^{-1}. The highest accumulation zone of microplastics in stormwater drains sediment was in Hagley Park (5.3 particles per 100 g^{-1}). These results were comparable to research in Iran, Japan and Vietnam but only for central location. These results became less comparable in suburban locations due to the Residential Red Zone land allocation in eastern Christchurch.

Microplastics of polyethylene Terephthalate, polypropylene, and polystyrene from the lower reach were either transported from tributaries into confluence locations (PET and PS) or were from recreational activities such as fishing on the Avon/ Ōtākaro River (PP). The identification of nylon and polyester in both river and stormwater drain sediment primarily occurred within the middle reach, an area of high foot traffic and population movement. Polymers such as rubber were identified in the upper and middle reach stormwater drain sediments but were only isolated in lower reach river sediments. This would suggest that some microplastics could be transported by river flow from their original sources.

Fragments (47% in river and 66% in stormwater drains) were the most abundant morphotype in both sediment studies. Fibres (38% in river and 26% in stormwater drains) and pellets (8% for both studies) were also identified. No films were detected in either sediment type. Microplastics in both sediments were likely to be smaller than 1000 µm. Only 2% of microplastics in river and 18% in stormwater drain studies were greater than 1000 µm. The average microplastic size in river sediments was 300 – 500 µm (43%) increasing to 500 – 1000 µm in stormwater drain sediments (34%). There was no relationship between microplastic abundance and colour composition. Red (27%) was most prevalent in river sediments whereas white (20%) was most abundant in stormwater

sediments. The only correlation reported between sediment colours was the lack of multi-coloured microplastics identified (1% in river and 4% in stormwater drains).

Sixty percent of microplastics were isolated from confluence locations with tributaries of the Avon/Ōtākaro. This was evident in the lower reach where polystyrene isolated from Dudley Creek was continuously isolated towards the Avondale Bridge (5 kms). The influence of flow rate on microplastic accumulation stated a positive relationship which was exacerbated by the influence of tributaries as flow rates increased downstream. The difference of land elevation could have also influenced microplastic accumulations as the zones of highest accumulation in the lower reach of the Avon/Ōtākaro river are 3 metres lower than in the middle reach.

Traffic settings in stormwater drain sediments identified roadsides (53%) and intersections (47%) as major sources of microplastics. The highest accumulation of microplastics within these roadside and intersection locations were areas where traffic volume exceeded 250 vehicles per hour. This directly related to land use as the locations of major intersections were identified in areas of open space use. The main microplastics identified in open space land were rubber from bicycle tyres and fragments of vehicle panelling. Open space land use accounted for 47% of stormwater drain microplastics and special purpose land accounted for 37%. Whilst there were several polymers within these land zones, nylon fibres identified in special purpose land zones were derived from the uniforms of schools within these land locations. This would support international studies which have reported microplastics in stormwater drains do not travel far from their original source (Sutton et al., 2016; Kole et al., 2017: Sommer et al., 2018). The accumulation of microplastics in river sediments of the Avon/Ōtākaro River are derived from tributary influences. These microplastics, based on spatial tends, show no relationship to stormwater drain microplastics. This study would suggest that microplastics in rivers are not derived solely from stormwater drains. Microplastics that feed into the Avon/Ōtākaro River have originated from other sources besides for stormwater drains such as land overflow, road runoff or aeolian processes.

This thesis investigated two aspects of microplastic concentrations within the Avon/ Ōtākaro River catchment in Christchurch. The hypothesis of this study is that polymer identification will be similar to international research but in lower concentrations. The following conclusions were drawn:

- The average microplastic concentration within river sediments was 5.2 particles per 100 g^{-1} with a range of 0 - 35 particles per 100 g^{-1}. These results are comparable to previous research in Auckland and Brisbane. Concentrations were significantly lower than reported in South Korea, Canada, and the USA. These concentration values are believed to be influenced by population density.
- PET (41 particles), PP (33 particles) and PS (29 particles) were most abundant.
- The greatest proportion of microplastics in river sediments were fragments (47%) and 100 – 300 µm (43%). Less than 2% of microplastics were greater than 1000 µm.
- There was no relationship between microplastic accumulation and sediment total organic carbon.
- The average microplastic concentration within stormwater drain sediments was 3.3 particles per 100 g^{-1} with a range of 0 - 8 particles per 100 g^{-1}. These results are comparable to studies internationally but only for microplastics within Central Christchurch. No comparisons could be made with suburban Christchurch.
- Nylon (8 particles) was the most abundant polymer in stormwater drain sediments followed by rubber and PP (6 particles).
- The greatest proportion of microplastics in stormwater drain sediments were fragments (66%) and 500 – 1000 µm (34%). Both morphotype and size distributions were conflicting to international research due to the polymers identified.
- External coatings collected from stormwater drain microplastics identified sources of plastics including a range of recreational, building and outdoor uses.
- There was a correlation to microplastic abundance in stormwater drain sediments and traffic volume.
- There was no relationship between concentrations of river and stormwater microplastics. This was due to the spatial distribution between the two sample types. The highest accumulation of river microplastics was in the lower reach while highest accumulation of stormwater drain microplastics was in the upper reach.
- Stormwater drains were not the only source of microplastic accumulation in rivers. Land overflows, road runoff, and aeolian processes all impacted accumulation rates.

6.2. Recommendations

As microplastics have been reported as detrimental to river environments, solutions are required to mitigate or to eliminate the impact microplastics have on aquatic life and water quality. While the methods for reducing microplastics into the environment are still a new topic, some potential solutions are described below:

6.2.1. Legislation and policy amendments

Whilst several countries, including New Zealand, have banned the use of single-use plastic bags, primary microplastics such as microbeads are still commonly permitted (McDevitt et al., 2017). Many countries still classify plastics as harmless solid waste so can be discharged into sewers or tributaries (Rochman et al., 2013; Lechner and Ramler, 2015).

The definition of 'plastic' can be disputed and varies between countries and subsequently, laws and legislation. Several legislative policies are either vague or have loopholes which can be exploited. Within the United States of America, plastic is defined as 'a synthetic material that can be extruded at high heat' (Illinois Bill SB2727). This definition does not cover plastics which have a low melting temperature nor, does it define what a high temperature is (McDevitt et al., 2017). This legislation would also not apply to by-products of plastic production such as glitter (Yurtsever, 2019). Many other legislations do not deem cosmetic microbeads as a form of plastic so only apply microbead rules to 'rinse-off' products such as car cleaners.

Most legislation globally does not cover microfibres such as polyester and nylon. While it would be impossible to restrict the selling of man-made fibre clothing and materials, the encouragement of purchasing natural fibre clothing through the introduction of taxes on man-made fibre apparel is being trialled in Singapore (Lam et al., 2018).

The introduction of new and tightening of existing microplastic legislation is a fundamental step into improving our dependency on plastic materials. There still needs to be a certain degree of accountability from governments, an introduction of penalties for individuals and companies who do not adhere to these new regulations and an alternative option for consumers. A blanket ban on all plastic is not feasible if no other alternative measure has been introduced to the public.

6.2.2. Control of secondary microplastics

Once in the aquatic environment, microplastics are almost impossible to halt. The controls of macroplastics before they reach secondary form is a crucial step to reducing potential microplastics (Fu and Wang, 2019). The introduction of natural or alternative packaging is a newly emerging option for consumers but for distributors and businesses, the alternative materials for packaging would need to be as cost-effective and beneficial to them as to the environment (Petrescu-Mag et al., 2019).

One control measure could be to increase the cost of using polystyrene pellets, bubble wrap and other forms of non-recyclable packaging. This would only prove beneficial if several situations were to occur simultaneously. Firstly, if the removal of non-recyclable plastics was to occur overtime. Secondly, if the cost of biodegradable plastics were cheaper, or at least equal to other packaging and thirdly, if public awareness is made of the danger plastic has to the environment (Deng et al., 2018). Controls of microplastics within rivers is difficult to achieve as netting or technology to remove the progress of microplastics downstream would impact on aquatic invertebrates. Therefore, methods need to be implemented to reduce the volume of plastics before reaching waterways.

6.2.3. Stormwater baskets

Stormwater drains are a crucial transportation route of micro and macroplastics from land to rivers (Hwang et al., 2016; Liu et al., 2019). To control what can be passed through a stormwater grate, a device that could capture all particles whilst allowing water to freely flow through the device would be most suitable for microplastic collection. Other systems such as stormwater trains have been trialled (Saberi and Nikraz, 2008) but would be highly expensive in production and maintenance.

A current project in Berlin, Germany, the '*Reifen Abriebin der Umwelt*' (Tyre wear in the environment) is designed to capture material only during rainfall events but could be utilised as a permanent form of filtration. Currently, the cost of producing a basket would be a one-off cost (depending on how many baskets councils would wish to purchase). The cost of maintenance and cleaning of the mesh sieves, however, would be unaffordable especially if they are required to be cleaned after every rain event. If cost measures were introduced to limit cleaning or maintenance, it would be a highly desirable system for all settlements.

6.2.4. Discussion

To control the introduction of microplastics into the aquatic environment, a mixture of legislative and scientific controls will be necessary. To say that one method would be suitable over another would not be functional. Legislative change is required to stop the production of primary microplastics and plastic polymers which are readily destined for landfills. The controls of secondary microplastics are important as they will allow for public notification of the negative impacts microplastics or, plastics that can become secondary have on the environment and aquatic life. Stormwater baskets and other devices to trap microplastics in the environment should be considered a last resort. If legislative purposes work, then the reliance on these devices should become obsolete.

6.3. Future work

This research provides a baseline data on microplastic concentration within river sediments and in stormwater sediments adjacent to the Avon/Ōtākaro River. There is, however, research that can be conducted to strengthen our knowledge on microplastics within Christchurch city. A temporal dataset could be completed by sampling over longer periods. This would allow for a comparison between summer, winter, storm and low flow profiles.

Testing of microplastics within other waterbodies of Christchurch would allow for more in-depth understanding of how microplastics behave once within river systems. This could be achieved by sampling tributaries and other major Christchurch river systems such as the Heathcote/Ōpāwaho or the Styx/Pūharakekenui. Testing of river sediments outside of the Christchurch city boundary such as the Kaiapoi River could be used as a comparison between microplastic volumes of urban and semi-urban locations. Sampling could also be conducted within the Avon/Ōtākaro River's groundwater-fed sources from the Waimakariri River to identify if microplastics are prevalent before becoming surface-born.

Airborne microplastics research, especially within the CBD, would provide further detail into how microfibres move when airborne. This research has established there are high quantities of microfibres within anthropogenic-dominated environments such as schools and shopping locations. Understanding where these fibres are derived from will give valuable insight into how microfibres behave, a topic that has not been researched within New Zealand to date.

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Appendices

Appendix 1: Sample locations along the Avon/Ōtākaro River

Sample Zone	River sediment sample location	Stormwater drain sample location
Ilam	Ilam Road/ University Drive	University Drive
Ilam	Kirkwood campus, University of Canterbury	
Ilam	Puiri Street bridge	Puriri/ Kotare Street
Ilam	Riccarton Bush	
Ilam	Harakeke Street	No. 73 Harakeke Street
Fendalton	Mona Vale	
Fendalton	Christchurch Girls' High School	Harper Avenue/ Fendalton Road
Fendalton	Deans Avenue / Harper Avenue	
Fendalton	Rossall Street	Carlton Mill Road / Hewetts Road
Fendalton	Carlton Mill Bridge	
Hagley Park	Christ's College	Park Terrace / Armagh Street
Hagley Park	Botanic Garden footbridge	
Hagley Park	Te Kura Tennis Club	Te Kura Tennis club
Hagley Park	Hagley Oval	
Hagley Park	Antigua Boat Sheds	Cambridge Terrace / Cashel Street
CBD	National Earthquake Memorial	
CBD	Bridge of Remembrance	Durham St South / Cashel Street
CBD	Victoria Square	
CBD	Margaret Mahy Playground	Manchester Street / Oxford Terrace
CBD	Barbadoes Street Bridge	
Avonside	Stanmore Road Bridge	River Road / Stanmore Road
Avonside	Dudley Creek Confluence	
Avonside	Kerrs Reach	Hockey Lane / Avonside Drive
Avonside	Horseshoe Lake Pump Station	
Avonside	Avondale Bridge	Avondale Road / New Brighton Road
New Brighton	Corsers Stream Confluence	
New Brighton	ANZAC Bridge	New Brighton Road / ANZAC Drive
New Brighton	Pages Road Bridge	
New Brighton	Owles Terrace Pontoon	Owles Terrace / Hardy Street
New Brighton	Te Wahapū estuary	

Appendix 2: Filter paper weight before and after the addition of microplastics from Avon/Ōtākaro River sediments.

Sample site	Filter paper weight	Filter paper weight with sample	Difference
Ilam Road/ University Drive	64 g	68 g	4 g
Kirkwood Campus	64 g	73 g	9 g
Puiri Street Bridge	64 g	72 g	8 g
Riccarton Bush	64 g	89 g	25 g
Harakeke Street	64 g	75 g	11 g
Mona Vale	64 g	78 g	14 g
Christchurch Girls' High School	64 g	83 g	19 g
Deans Avenue / Harper Avenue	64 g	91 g	27 g
Rossall Street	64 g	66 g	2 g
Carlton Mill Bridge	64 g	76 g	12 g
Christ's College	64 g	69 g	5 g
Botanic Gardens	64 g	72 g	8 g
Te Kura Tennis Club	64 g	75 g	11 g
Hagley Oval	64 g	78 g	14 g
Antigua Boat Sheds	64 g	84 g	20 g
National Earthquake Memorial	64 g	94 g	30 g
Bridge of Remembrance	64 g	68 g	4 g
Victoria Square	64 g	68 g	4 g
Margaret Mahy Playground	64 g	73 g	9 g
Barbadoes Street Bridge	64 g	64 g	0 g
Stanmore Road Bridge	64 g	68 g	4 g
Dudley Creek Confluence	64 g	84 g	20 g
Kerrs Reach	64 g	102 g	38 g
Horseshoe Lake Pump Station	64 g	93 g	29 g
Avondale Bridge	64 g	87 g	23 g
Corsers Stream Confluence	64 g	85 g	21 g
ANZAC Bridge	64 g	68 g	4 g
Pages Road Bridge	64 g	92 g	28 g
Owles Terrace Pontoon	64 g	82 g	18 g
Te Wahapū estuary	64 g	67 g	3 g

Appendix 2: Filter paper weight before and after the addition of microplastic particles from stormwater drains.

Sample site	Filter paper weight	Filter paper weight with sample	Difference
University Drive	64 g	69 g	5 g
Puriri / Kotare Street	64 g	67 g	3 g
No. 73 Harakeke Street	64 g	74 g	10 g
Harper Avenue / Fendalton Road	64 g	72 g	8 g
Carlton Mill Road / Hewetts Road	64 g	72g	8 g
Park Terrace / Armagh Street	64 g	75 g	11 g
Te Kura Tennis Club	64 g	69 g	5 g
Cambridge Terrace / Cashel Street	64 g	66 g	2 g
Durham St South / Cashel Street	64 g	68 g	4 g
Manchester Street / Oxford Terrace	64 g	73 g	9 g
River Road / Stanmore Road	64 g	74 g	10 g
Hockey Lane / Avonside Drive	64 g	72 g	8 g
Avondale Road / New Brighton Road	64 g	74 g	10 g
New Brighton Road / ANZAC Drive	64 g	66 g	2 g
Owles Terrace / Hardy Street	64 g	78 g	14 g

Appendix 3: Pre and post weight of sediment to ascertain total moisture content.

Sample Location	Foil Dish Weight (G)	Wet Sediment Weight (G)	Dry Sediment Weight (G)	Difference (Wet – Dry)
Ilam Road/ University Drive	0.73	9.21	2.43	6.78
Kirkwood Campus	0.72	9.72	6.12	3.60
Puiri Street Bridge	0.72	5.75	3.77	1.98
Riccarton Bush	0.74	7.25	5.18	2.07
Harakeke Street	0.71	12.57	9.19	3.38
Mona Vale	0.73	6.56	4.78	1.78
Christchurch Girls' High School	0.72	6.65	2.73	3.9
Deans Avenue / Harper Avenue	0.75	6.37	2.48	3.89
Rossall Street	0.74	6.29	3.75	2.54
Carlton Mill Bridge	0.73	8.45	6.54	1.91
Christ's College	0.72	11.17	6.85	4.32
Botanic Gardens	0.76	12.68	7.65	5.03
Te Kura Tennis Club	0.73	10.67	4.06	6.61
Hagley Oval	0.78	16.04	9.20	6.84
Antigua Boat Sheds	0.74	10.49	2.65	7.84
National Earthquake Memorial	0.72	5.85	2.07	3.78
Bridge of Remembrance	0.75	9.62	2.01	7.61
Victoria Square	0.74	6.35	2.52	3.83
Margaret Mahy Playground	0.74	5.99	2.38	3.61
Barbadoes Street Bridge	0.73	5.79	2.00	3.79
Stanmore Road Bridge	0.72	7.58	1.13	6.45
Dudley Creek Confluence	0.74	6.27	0.92	5.35
Kerrs Reach	0.79	6.82	1.09	5.73
Horseshoe Lake Pump Station	0.78	6.02	2.62	3.40
Avondale Bridge	0.76	6.89	4.30	2.59
Corsers Stream Confluence	0.77	10.15	3.01	7.14
ANZAC Bridge	0.76	7.49	3.47	4.02
Pages Road Bridge	0.71	11.51	5.18	6.33
Owles Terrace Pontoon	0.78	7.22	5.63	1.59
Te Wahapū estuary	0.75	30.09	19.98	10.11

Mean	4.59
Minimum	1.59
1st Quartile	3.39
2nd Quartile	3.91
3rd Quartile	6.42
Maximum	10.11
Standard Deviation	2.10

Appendix 4: The percentage of sediment particle size distribution along the different reaches of the Avon/Ōtākaro River.

Particle Size (µm)	Upper Reach	Middle Reach	Lower Reach
0 - 299 µm	13.18	4.45	2.23
300 - 399 µm	5.16	5.45	1.5
400 - 499 µm	0.5	18.21	2.41
500 - 599 µm	4.53	10.52	3.59
600 - 699 µm	5.05	9.5	4.24
700 - 799 µm	4.32	10.22	4.71
800 - 899 µm	6.77	5.94	5.82
900- 999 µm	6.97	5.31	7.72
> 1000 µm	46.48	30.4	67.78
Mean	10.33	11.11	11.12
Minimum	0.5	4.45	1.5
1st Quartile	4.43	5.38	2.32
2nd Quartile	5.16	9.5	4.24
3rd Quartile	10.08	14.37	6.77
Maximum	46.48	30.4	67.78
Standard Deviation	13.17	7.91	20.12

Appendix 5: Total organic carbon (TOC) results from Avon/Ōtākaro River

Sample Location	TOC (ppm)	Microplastic concentration (100 g sediment)
University of Canterbury	0.2	6
Okeover Stream	0.4	2
Harakeke Street	0.4	1
Wai-Iti/Waimairi Stream	0.1	2
Christ's College	3.4	4
Riccarton Drain	0.6	4
Addington Brook	0.5	5
Victoria Square	4.8	2
Avon Loop (Barbadoes Street to Stanmore Road)	1.9	6
Dudley Creek	4.5	2
Kerrs Reach	5.6	35
Corsers Stream	3.7	10
Pages Road	0.8	5
Te Wahapū estuary	0.8	3
Mean	2.0	6.2
Minimum	0.1	1
Maximum	5.6	35
Standard Deviation	1.9	8.3

Appendix 6: Particles identified from river sediments within the Avon/Ōtākaro River

Site	Polymer	Morphotype	Size (microns)	Colour
1	Propylene glycol monestrate	Fragment	100 - 300	White
	Polyester x 5	Fibre	30 - 100	Blue
	Cotton x 2	Fibre	30 - 100	White
2	Rayon x 4	Fibre	30 - 100	Black
3	Polystyrene	Fragment	100 - 300	Purple/Pink
	Cotton	Fibre	30 - 100	White
	Nylon	Fibre	30 - 100	Black
4	Polyurethane	Fragment	1000 - 5000	White
	Cotton x 2	Fibre	30 - 100	Red & Black
	Polyphenylene Sulfide	Fragment	500 - 1000	White
5	Cotton	Fibre	30 - 100	Red
	Polypropylene	Fragment	100 - 300	White/Red
6	Polyethylene	Fragment	1000 - 5000	Black
	Polyamide	Fragment	500 - 1000	Blue
7	Polyamide	Fragment	300 - 500	Black
	Nylon	Fibre	30 - 100	Black
8	Polyethylene	Fragment	500 - 1000	Blue
	Nylon	Fibre	30 - 100	Black
9	Polypropylene	Fragment	500 - 1000	Green
	Polyester	Fibre	30 - 100	Yellow
	Polyester	Fibre	30 - 100	Red
10	Polyethylene x 3	Fragment	300 - 500	White
	Polyvinyl	Fragment	100 - 300	White
11	Polyester	Fibre	30 - 100	Blue
	Nylon	Fibre	30 - 100	Black
	Nylon	Fibre	30 - 100	Red
12	Polystyrene x 4	Foam	500 - 1000	White
13	Polyethylene Terephthalate x 4	Fibre	500 - 1000	White
	PTFE	Fragment	500 - 1000	White
14	Polyester x 2	Fibre	30 - 100	Blue
	Nylon x 2	Fibre	100 - 300	Black

	Polyethylene (High-Density)	Fragment	500 - 1000	Orange
15	Rayon x 5	Fibre	30 -100	Black
	Cotton x 3	Fibre	30 -100	Green
	Cellophane	Fragment	100 - 300	Red
16	Cotton x 7	Fibre	30 - 100	Blue
	Polyethylene	Fragment	100 - 300	Red
	Polyethylene	Fragment	100 - 300	Green
17	Polypropylene	Fragment	500 - 1000	White
	PET	Fragment	500 - 1000	Blue
18	Nylon x 2	Fibre	30 - 100	Black
	Rayon x 3	Fibre	30 - 100	Black
	Polyvinyl Chloride x 4	Fragment	500 - 1000	Black
19	Polyester x 2	Fibre	30 - 100	Yellow
20	No Plastics found			
21	Cotton x 5	Fibre	30 - 100	Red
	Rubber x 2	Fragment	1000 - 5000	Black
22	Polystyrene x 3	Foam	100 - 300	White
23	Polystyrene x 8	Foam	100 - 300	White
	Polyester x 3	Fibre	30 - 100	Red
	Polyethylene Terephthalate	Fragment	500 - 1000	Green
	Polypropylene x 23	Fibres	500 - 1000	Blue
24	Cotton x 3	Fibre	30 - 100	Black
	Polypropylene x 2	Fragment	300 - 500	Red
	Polypropylene	Fragment	300 - 500	Green
	Polystyrene	Foam	300 - 500	White
25	Polystyrene x 9	Pellet	100 - 300	White
	polyphenylene	Fibre	30 - 100	Black
26	Cotton	Fibre	30 -100	Black
	Polyethylene Terephthalate x 34	Fragment	100 - 300	Red
27	Polystyrene x 4	Pellet	100 - 300	White
	Polyvinyl Chloride	Fragment	500 -1000	Orange
	Rayon x 2	Fibre	30 - 100	Green
28	Polypropylene x 3	Fragment	300 - 500	Blue

29	Polypropylene	Fragment	300 - 500	Blue
	Polyethylene x 2	Fragment	300 - 500	Orange
30	Nylon x 4	Fibre	30 - 100	Red
	Polyethylene Terephthalate	Fragment	100 - 300	Black

Appendix 7: Summary of microplastic characterisations from Avon/Ōtākaro River sediments.

Table 4.3. Summary of microplastic morphotype distribution across the Avon/Ōtākaro River.

Morphotype	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
Fibre	6	3	12	6	25	2	**54**
Film	0	0	0	0	0	0	**0**
Foam	0	0	4	0	12	0	**16**
Fragment	5	11	2	8	6	41	**73**
Pellet	0	0	0	0	9	4	**13**
Total	**11**	**14**	**18**	**14**	**52**	**47**	**156**

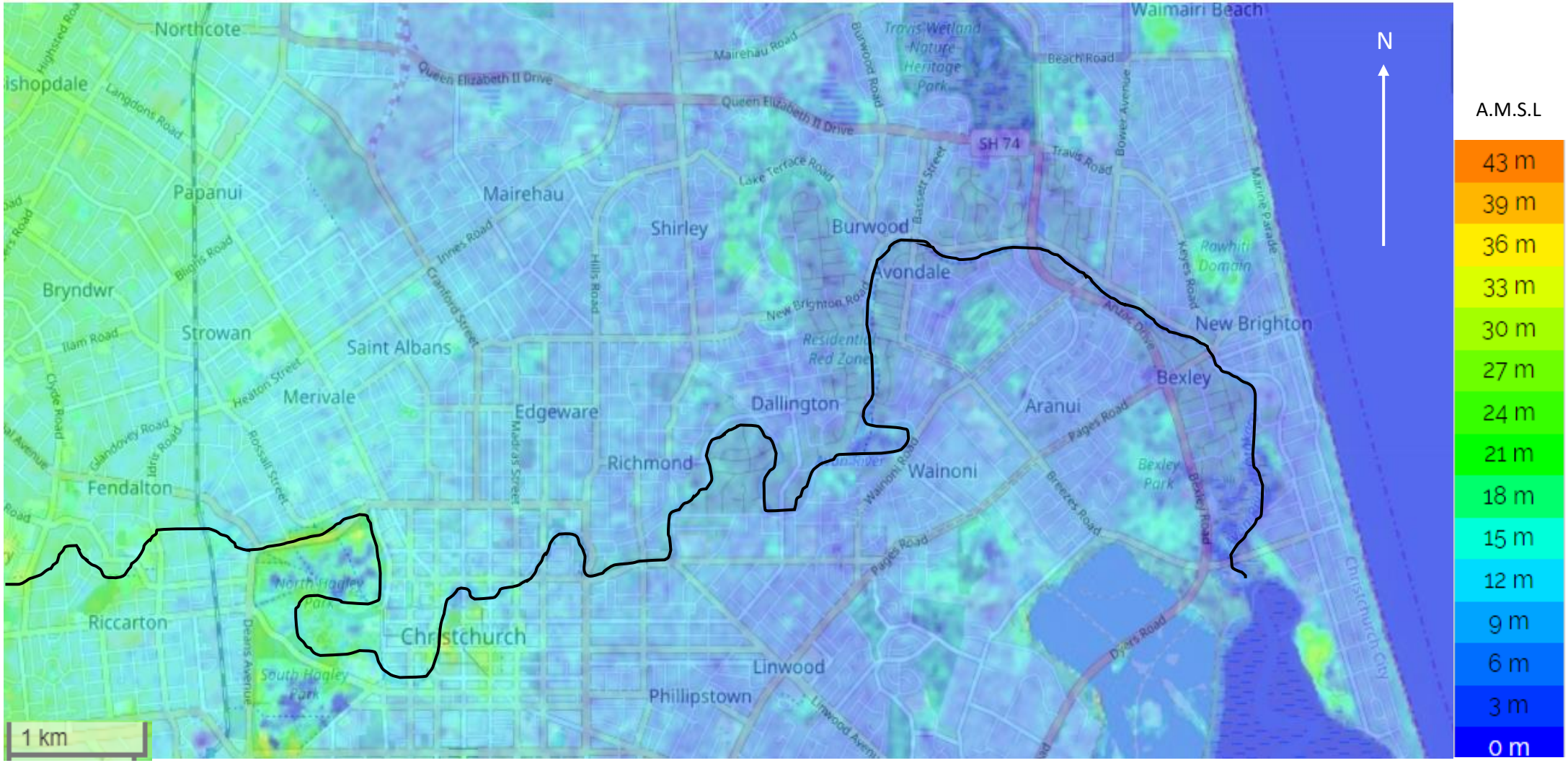
Table 4.4. Summary of sizes distribution across the six cluster locations of the Avon/Ōtākaro River

Size (µm)	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
30 – 100	6	3	5	5	5	4	**28**
100 – 300	3	1	3	3	20	37	**67**
300 – 500	0	3	0	0	4	5	**12**
500 – 1000	1	6	10	6	23	1	**47**
1000 - 5000	1	1	0	0	0	0	**2**
Total	**11**	**14**	**18**	**14**	**52**	**47**	**156**

Table 4.5. Microplastic colour identification from sediments within the Avon/Ōtākaro River.

Colour	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
Red	0	1	1	1	5	34	42
Orange	0	0	1	0	0	3	4
Yellow	0	1	0	2	0	0	3
Green	0	1	0	1	1	1	4
Blue	5	2	4	1	23	5	40
Black	2	4	4	8	11	1	30
White	2	5	8	1	12	4	32
Multi-coloured	2	0	0	0	0	0	2
Total	**11**	**14**	**18**	**14**	**52**	**47**	**156**

Appendix 8: Digital elevation model of Christchurch city showing land height above mean sea level (A.M.S.L) from the upper reach (west) to the lower reach (east) post 2011 earthquake (adapted from LINZ, 2020).



Appendix 9: Particles identified from sediments from Stormwater drains adjacent to the Avon/Ōtākaro River

Site	Polymer	Morphotype	Size (microns)	Colour
1	Polyethylene	Pellet	300 - 500	Yellow
2	Polyethylene	Fragment	300 - 500	Orange
	Polyethylene	Fragment	300 - 500	Blue
	Polypropylene	Fragment	500 - 1000	Green
	Polyvinyl Chloride	Pellet	300 - 500	Yellow
	Polyester	Fibre	30 - 100	Green
	Nylon	Fibre	30 -100	Yellow
3	Nylon	Fibre	30 - 100	Yellow/White
	Nylon	Fibre	30 - 100	Yellow
4	Lignin Sulfonate x 5	Fragment	500 - 1000	Green
	Polypropylene	Fragment	500 - 1000	Yellow
	Polypropylene	Fragment	300 - 500	Blue
	Cellulostic Ester acetate	Fibre	30 - 100	Red
	Polyethylene	Fibre	30 - 100	Yellow
5	Aluminium Silicate	Pellet	30 - 100	White
	Cotton	Fibre	30 - 100	Black
	Rubber (speed hump)	Fragment	500 - 100	Black/Yellow
	Cellophane	Fragment	100 - 300	Red
	Polyethylene	Fragment	100 - 300	Red
6	Nylon	Fibre	30 - 100	Green
	Cotton	Fibre	30 - 100	White
	Vehicle brake lights (polycarbonate)	Fragment	1000 - 5000	Orange
	Vehicle brake lights (polycarbonate)	Fragment	1000 - 5000	White
	Rubber (tyre)	Fragment	1000 - 5000	Black
	Silicone	Fragment	1000 - 5000	Green
	Polyvinyl Chloride	Fragment	500 - 1000	Red
	Cotton	Fibre	30 - 100	Black
7	Nylon	Fibre	30 - 100	Blue
	Wool x 3	Fibre	30 -100	Black
	Rubber x 2 (tyre)	Fragment	500 - 100	Black
	Nylon	Fibre	30 - 100	White
	Polyethylene Terephthalate	Fragment	100 - 300	Black
8	Cellophane	Fragment	100 - 300	White
	Silicone x 4	Fragment	500 - 1000	White
	Cellophane	Fragment	100 - 300	Red

9	Rayon x 21	Fibre	30 - 100	Black
	Poly(butyl) methacrylate	Fragment	500 - 1000	Orange
	Polycarbonate	Fragment	1000 - 5000	White
	Polycarbonate	Fragment	1000 - 5000	Orange
	Polypropylene	Pellet	300 - 500	White
	Sandpaper	Fragment	300 - 500	Blue
10	Rayon	Fibre	30 - 100	Black
	Cotton x 14	Fibre	30 -100	White
	Rubber (tyre)	Fragment	1000 - 5000	Black
	Polyester	Fibre	30 -100	Blue
	Polyester	Fibre	30 -100	Black
	Nylon	Fibre	30 - 100	Red
11	Acrylonitrile Butafirne Styrene	Fragment	1000 - 5000	Red
	Acrylonitrile Butafirne Styrene	Fragment	1000 - 5000	White
12	Poly(butyl) methacrylate	Fragment	500 - 1000	Red
13	Nylon	Fibre	30 -100	Black
14	Cotton	Fibre	30 -100	Blue
15	Polyethylene	Fragment	100 - 300	Orange
	Polyethylene Terephthalate	Pellet	100 - 300	Blue
	Cotton	Fibre	30 -100	Red

Appendix 10: Summary of microplastic characterisations from stormwater drain sediments.

Table 5.2. Summary of microplastic morphotype from stormwater drain samples along the Avon/Ōtākaro River.

Morphotype	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
Fibre	4	1	4	3	1	0	**13**
Film	0	0	0	0	0	0	**0**
Foam	0	0	0	0	0	0	**0**
Fragment	3	9	12	5	3	1	**33**
Pellet	2	0	0	1	0	1	**4**
Total	**9**	**10**	**16**	**9**	**4**	**2**	**50**

Table 5.3. Summary of microplastic size distribution within stormwater drains sediments along the Avon/Ōtākaro River.

Size (µm)	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
30 – 100	4	1	3	3	1	0	**12**
100 – 300	0	1	2	0	0	2	**5**
300 – 500	4	1	0	2	0	0	**7**
500 – 1000	1	7	7	1	1	0	**17**
1000 - 5000	0	0	4	3	2	0	**9**
Total	**9**	**10**	**16**	**9**	**4**	**2**	**50**

Table 5.4. Microplastic colour identification from stormwater drain sediments.

Colour	Ilam	Fendalton	Hagley Park	CBD	Avonside	New Brighton	Total
Red	0	1	1	1	2	0	5
Orange	1	0	1	2	0	1	5
Yellow	4	2	0	0	0	0	6
Green	2	5	2	0	0	0	9
Blue	1	1	1	2	0	1	6
Black	0	0	4	2	1	0	7
White	0	0	7	2	1	0	10
Multi-coloured	1	1	0	0	0	0	2
Total	**9**	**10**	**16**	**9**	**4**	**2**	**50**