Plastic Characterization in Sediment Along the Bow River near Calgary, Alberta

by

Liam Murphy

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Royal Roads University
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Supervisor: Dr. Rick Kool
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COMMITTEE APPROVAL

The members of Liam Murphy's Thesis Committee certify that they have read the thesis titled Plastic Characterization in Sediment Along the Bow River near Calgary, Alberta and recommend that it be accepted as fulfilling the thesis requirements for the Degree of Master of Science Environment and Management:

Dr. Richard Kool [signature on file]

Dr. Rick Robinson [signature on file]

Final approval and acceptance of this thesis is contingent upon submission of the final copy of the thesis to Royal Roads University. The thesis supervisor confirms to have read this thesis and recommends that it be accepted as fulfilling the thesis requirements:

Dr. Richard Kool [signature on file]

Abstract

Macro- and microplastics are a pervasive anthropogenic pollutant in aquatic environments. My exploratory research quantified and characterized macro- and microplastic contamination in shoreline sediments along the Bow River in Calgary, Alberta, and identified trends in their abundance and distribution. Sediment samples were collected from eight sites along banks of the Bow River. At each site, 1 kg of wet sediment was obtained then filtered, dried, and mixed into 100 g dry-weight subsamples. Natural debris were removed using hydrogen peroxide, and microplastics were extracted from sediment using NaCl density flotation. Macroplastics were collected along a 10 m transect deployed parallel to the shoreline at the same eight sites.

Microplastics were detected in all samples. A total of 93 microplastics were detected, with an average of 11.6 pieces/100 g sample. The most frequent microplastic shape (type) was fiber (n=52), followed by fragment (n=26), foam (n=14), and round (n=1). A two-tailed t-test (p-value <0.01) and linear regression (r^2 value of 0.68) showed a statistically significant difference in microplastic concentrations downstream compared to upstream sites. For macroplastics, a total of 826 individual pieces of macroplastic debris were collected from all sites. Most pieces were identified as single-use plastics. A two-tailed t-test (p-value of 0.24) and linear regression (r^2 value of 0.32) suggested a weak relationship between macroplastic concentrations downstream compared to upstream.

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Chapter 1: Introduction

Canada is a unique country with an abundance of available freshwater; approximately 20% of global surface feshwater is located in Canada. Despite this, data on plastic pollution in Canadian freshwater ecosystems, particularly riverine environments, is severely limited (J. C. Anderson et al., 2016). Most studies on plastic pollution in the scientific literature focus on the marine environment. The marine environment serves as a sink for macroplastic[1] and microplastic pollution[2], with rivers being a major land-based source; however, not all plastics particles transported from rivers to oceans. Recent studies demonstrate rivers and lakes are just as contaminated as marine environments (Eriksen et al., 2013; Lechner et al., 2014; Luo et al., 2019), highlighting a widespread issue. For freshwater ecosystems, there is an emphasis on microplastic pollution research whereas less focus is paid to macroplastics which are also omnipresent in riverine systems, particularly near densely populated areas. More research on macroplastics is needed to form a comprehensive understanding of plastic pollution in freshwater environments (Blettler et al., 2018).

In Canada, high levels of freshwater microplastic contamination have been reported in the Great Lakes (Eriksen et al., 2013), Lake Winnipeg (P. J. Anderson et al., 2017) the St. Lawrence River (Castañeda et al., 2014; Crew et al., 2020), the Ottawa River (Vermaire et al., 2017), Muskoka Headwater Lakes (Welsh et al., 2022), the North Saskatchewan River (Bujaczek et al., 2021), and the Bow River (Ross et al., 2023). These studies suggest freshwater pervasive microplastic contamination across Canada but because of the limited number of published studies the full extent of contamation across the country remains relatively unknown. In addition,

[1] Macroplastics are defined as plastic particles with a diameter of ≥ 5 mm.
[2] Microplastics are defined as plastic particles with a diameter of < 5 mm.

most studies in Canada have focused on instream or suspended microplatics extracted from water samples; there is a noticable gap in the scientific literature with respect to macro- and microplastic contamiation in littoral (shoreline) sediments[3] of major rivers and tributaries. I investigated shoreline sediments as opposed to instream (suspended) microplastics or river bottom sediments because the shoreline is readily accessible. To capture instream sediments across the Bow River's wetted width would require equiment I did not have access to such as a motor boat, a permit to operate the boat in city limits, and specialized sampling equipment (e.g., Plankton net, Ekman grab sampler). In addition, the shoreline of a waterbody serves as an interface between land and water where people congregrate and enjoy recreational activities. Seeing plastic pollution along any shoreline elicits a visceral response in people that I felt warranted additional investigation. This includes the shoreline of the Bow River near Calgary, Alberta, which serves as the location for this thesis.

The Bow River originates in the Rocky Mountains and runs through reservoirs, towns, rural communities, the City of Calgary, and agricultural land until it eventually reaches its confluence with the Oldman River. From there, the Oldman River and Bow River coalesce to form the South Saskatchewan River, one of the longest rivers in Canada, and that ultimately drains into Hudson Bay. The Bow River is an important source of social and cultural identity, irrigation and drinking water, and an economic, environmental, and societal benefit to all Calgarians. As a previous resident of Calgary and someone who spent recreational time on the Bow River, the health of this important freshwater ecosystem is of personal and professional interest.

[3] For the purposes of this thesis the terms "littoral zone" and "shoreline" are used interchangeably to define the part of the shore where water meets the land, including the adjacent beach area extending inland from the shoreline.

There are no published studies of freshwater macro- or microplastic contamination in shoreline sediments of the Bow River near Calgary, Alberta. There is, however, a recently published study on microplastic contamination in Calgary's stormwater system during base flow and rain fall events (Ross et al., 2023). The study by Ross et al. (2023) focused on suspended microplastics in stormwater runoff from upland areas, not shoreline contimination. The absence of any macro- and microplastic data on shoreline sediments presents a unique opportunity for me to be one of the first to investigate and document plastic contamination along the Bow River in Calgary, Alberta.

In my study, the primary focus is on microplastic pollution because of personal curiosity and interest; however, to provide a comprehensive understanding of plastic pollution in the Bow River, macroplastic data was also collected and analyzed. Although macroplastics are not the main subject of this study, the breakdown and fragmentation of macroplastics are the main source of microplastics; therefore, they are an integral component of the overall assessment of plastic pollution and important to the context of microplastic pollution. By including both microplastic and macroplastic data, this research aims to present a holistic view of the state of plastic pollution in the Bow River.

The question this exploratory research will attempt to answer is: what are the concentrations, types, and relative abundances of macro- and microplastics in shoreline sediments of the Bow River near Calgary? The study's objectives are to quantify, enumerate, and characterize microplastic pollution within littoral (shoreline) sediments and identify trends in their distribution along the Bow River. The quantitative and qualitative data obtained in this baseline study will enhance our understanding of plastic contamination within a heavily urbanized Canadian river, which can inform future monitoring programs and scientific research.

Chapter 2: Literature Review

Plastics as a Persistent Pollutant

Since their development in the 1930s, plastics have become ubiquitous and used around the world. They continue to provide significant contributions to societal well-being (Wagner et al., 2014). A world without plastics is unimageable today, yet large-scale industrial production of plastics occurred only in recent history starting in the 1940's and 1950's.

Since its inception, plastics development and manufacturing has experienced exponential growth. Annual global plastic production increased from 2 million metric tonnes (Mt) in 1950 to 380 million Mt in 2015 (Geyer et al., 2017). Forecasts of future plastic production vary, but it is estimated that plastic production is expected to again double within the next 20 years (Lebreton et al., 2017; Lebreton & Andrady, 2019). Plastic use has now permeated all aspects of our daily lives and is used in a plethora of products such as clothing, food, buildings, appliances, machines, communication devices, transportation, and medicine, to name a few (Worm et al., 2017). The demand for plastics is considerable and new applications of plastic are being developed every year. This demand, combined with the present reality that no industrial-scale substitution exists, means plastic use will exist long into the future.

Plastics have several characteristics that make them suitable for the manufacturing of a wide variety of products: they are lightweight, strong, durable, and cheap to produce (Derraik & Derriak, 2002). With a high strength-to-weight ratio, stiffness, toughness, corrosion resistance, high thermal/electrical insolation, and a low cost compared with competing materials, plastics are very resource-efficient. However, the difficulty of recycling plastic polymers, combined with the pervasive production and usage, has resulted in accelerated pollution.

None of the commonly used synthetic plastics are biodegradable and consequently they accumulate in landfills or the natural environment rather than decomposing (Geyer et al., 2017) . According to Geyer et al. (2017), as of 2015, approximately 6300 million Mt of cumulative plastic waste had been generated, around 9% of which had been recycled, 12% was incinerated, and 79% was accumulating in landfills or the natural environment. Geyer et al. (2017) also estimate that if current production and waste management trends continue, roughly 12,000 million Mt of plastic waste will be in landfills or in the natural environment by 2050. These figures emphasize the reality that most discarded plastics are not recycled; therefore, discarded plastics have become a widespread anthropogenic pollutant and a growing contaminant of concern (Ziajahromi et al., 2017).

Terrestrial and aquatic life are in a constant state of exposure to plastic debris. It can be found in virtually every habitat on earth, from Himalayan peaks to deep oceanic trenches (Lebreton & Andrady, 2019; K. Zhang et al., 2016). In fact, some scholars suggest that plastic stratification in terrestrial and marine sediments are a key geological indicator of the Anthropocene, the conceptual geological epoch which is defined by the disruption of Earth's systems by human activity (Zalasiewicz et al., 2016).

Because of its ubiquity, durability, and ease of mobilization, plastics are now pervasive in the environment, and the negative influence of plastic accumulation in the environment has garnered significant attention in recent years (Rocha-Santos & Duarte, 2015; Wagner et al., 2014). The seriousness of plastic pollution in the environment has become prevalent in scientific literature, mainstream media, and other popular media sources (Catarino et al., 2021; MacLeod et al., 2021; Wagner et al., 2014).

Types of Plastic and Their Additives

Plastics are predominately synthetic organic materials derived from petrochemicals such as crude oil, natural gas, and other derivatives of hydrocarbon resources. To form synthetic plastics, petrochemical molecules are bonded together to form long, repeating chains of chemical compounds known as polymers through a process of polymerization (Worm et al., 2017). Depending on their expected use, plastic polymers can be designed to leverage preferable traits such as ductility, elasticity, impact resistance, and pliability. This makes plastic polymers incredibly versatile.

Different types of plastic vary in their global distribution, chemical composition, and environmental impact (Worm et al., 2017). The most widely manufactured and used plastic polymers, including some examples of their application, are outlined in Table 1. The most widely used types of plastics include polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET), and polystyrene (PS). The majority of these plastics are used in one-time, disposable packing materials or products (Hanvey et al., 2017). There are promising developments in the field of biodegradable plastics and substitute products; however, none of the commonly used hydrocarbon-derived plastic polymers are fully biodegradable. Consequently, the most widely manufactured polymers are also the most dominant plastic pollution found in the environment because rather than decomposing they accumulate in landfills or the natural environment (Geyer et al., 2017).

Table 1

A list of the most common types of plastics, usage, and density (Hanvey et al., 2017)

Type of Plastic Polymer	Abbreviation	Applications[4]	Density (g/cm^3)
Polyetheylene terephthalate	PET	Disposable beverage bottles, textiles/clothing, tape, food packaging, thermal insolation	1.37 - 1.38
High-density polyethylene	HDPE	Plastic bags, fuel tanks, bottle caps, milk crates	0.93 - 0.97
Low-density polyethylene	LDPE	Plastic bags, six-pack rings, packaging foam, snap-on lids	0.91 - 0.92
Polyvinyl chloride	PVC	Plumbing pipes, door frames, hoses, inflatable products	1.10 - 1.47
Polypropylene	PP	Bottle caps, rope, carpet	0.89 - 0.92
Polystyrene	PS	Styrofoam©, disposable cutlery, dinnerware	0.28 - 1.04
Other (polycarbonate, nylon, acrylic)	Other	Rope, synthetic fiber, electrical wire insulation, Plexiglas©	1.15 - 1.22

Plastics are rarely manufactured without resin and additive compounds used as colorants, stabilizers, and plasticizers (J. Li, Liu, Chen, et al., 2018). These compounds are added during the plastic production process to improve or change physical and chemical properties such as color, flame resistance, and hardness, to name a few. The most common additives are plasticizers which are used to improve the plasticity (i.e., malleability) or viscosity of the plastic material. Plasticizer additives incorporated into plastics can include endocrine-disrupting chemicals, heavy metals, phthalates, styrene, bisphenol A, and polybrominated diphenyl ethers (Hammer et al.,

[4] This column highlights examples of plastic polymer applications. It should not be considered an exhaustive list.

2012; Yu et al., 2018). Thus, plastic pollution may provide a mechanism to transport concentrated contaminants to organisms across multiple trophic levels (Browne et al., 2013).

Degradation of Plastics in the Environment

One of the reasons for the versatility and persistence of plastic polymers in the environment is their high resistance to degradation (Wagner & Lambert, 2018). Plastics take a very long time to degrade leading to an accumulation of plastic litter in the environment (Andrady & Neal, 2009; Klein et al., 2015). According to Wagner and Lambert (2018), degradation of plastic polymers in the environment can generally be classified into several processes:

- Physical/Mechanical degradation (abrasive forces, heating/cooling, freezing/thawing);
- Photodegradation (usually by UV-B radiation);
- Chemical degradation (oxidation or hydrolysis); and
- Biodegradation by organisms (bacteria, fungi, algae).

These processes weaken plastic polymer chains, which decreases the strength of the polymer bonds, causing embrittlement (Wagner & Lambert, 2018; Weinstein et al., 2016). This embrittlement helps mechanical forces, such as friction, sheering, and compression, to further weaken plastic polymers causing fragmentation and disintegration of large plastic pieces into smaller particles (Hanvey et al., 2017; Weinstein et al., 2016). Over time, the repeated process of large plastic pieces being fragmented into progressively smaller pieces leads to the occurrence of microscopic plastic particles known as microplastics.

Microplastics

Microplastics are defined as plastic particles with an upper size limit of 5 mm and a lower size limit of 1 μm (micrometer), and are as widely distributed as larger plastic debris

(macroplastics) in marine, freshwater, and terrestrial environments (Wagner & Lambert, 2018). The upper size limit of 5 mm is a generally accepted definition of microplastics and has been adopted by United Nations Environment Programme (UNEP) (Cheshire et al., 2009) and the U.S. National Oceanic and Atmospheric Administration (NOAA) (Lippiatt et al., 2013). More recently, researchers now recognize a new term, nanoplastic, that defines plastic particles smaller than 1 μm (Da Costa et al., 2016; Gigault et al., 2018). Nanoplastics have very different environmental fate and biological effects when compared to larger plastic fragments, but the difficulties in detection, separation, visualization, and chemical identification of nanoplastics in the environment have limited widespread investigation (W. Zhang et al., 2022). For the purposes of this thesis, the term microplastics will be defined as any plastic debris with an upper size limit of 5 mm.

There are two types of microplastics. *Primary microplastics* are deliberately manufactured to microscopic dimensions and can be found in textiles, medicines, and personal care products such as facial cleansers and body scrubs (J. Li, Liu, & Paul Chen, 2018; Wright et al., 2013). They are also produced as microplastic pellets used as a base or feeder material for plastic production. In the last few years, as scientific research on microplastic pollution intensified, many governments around the world, typically in wealthier countries such as Canada, USA, and the UK, began implementing bans of primary microplastics used in personal care products (Mitrano & Wohlleben, 2020). Multinational companies are also taking action to reduce microbeads in some of their products (C. M. Rochman et al., 2015). Regulation of primary microplastic production and use may be a step in the right direction, but primary microplastics represent a small portion of the estimated volume of microplastics in the natural

environment. A more significant amount of microplastics in the environment are considered secondary microplastics.

Secondary microplastics are derived from fragmentation or breakdown of larger plastic debris from processes such as ultraviolet-degradation, abrasion, and other physical, chemical, and biological forces (J. Li, Liu, Chen, et al., 2018). The majority of the microplastic particles in terrestrial, marine and freshwater environments are secondary microplastics (Horton et al., 2017; Mitrano & Wohlleben, 2020). Secondary microplastics are pervasive in the environment and occur through accidental release, indiscriminate discarding of plastic waste, and point (i.e., wastewater treatment plants (WWTP), stormwater system outfalls) and non-point (e.g., surface water runoff, windblown debris, spills, etc.) source pollution (J. Li, Liu, Chen, et al., 2018; C. Rochman et al., 2017; Wright et al., 2013). Reducing the occurrence of secondary microplastics would be incredibly difficult and would involve major action during all steps of plastic production, usage, and waste management.

Effects of Macroplastic Pollution on Biota

Macroplastic debris has been shown to have harmful consequences to terrestrial, marine, and freshwater biota and overall ecosystem health (Rocha-Santos & Duarte, 2015; Wright et al., 2013). Most studies on the effects of plastic debris on biota have focused on marine organisms because plastic debris is listed among the major perceived threats to marine biodiversity (Gall & Thompson, 2015; Gray, 1997) and receives significant media attention. It is apparent in the scientific literature that a wide range of species are affected by ingestion of and entanglement in plastic debris (Gall & Thompson, 2015; Kühn et al., 2015). Gall and Thompson (2015) provide evidence that macroplastic pollution has affected at least 693 marine vertebrate species. Many species of seabirds, fish, turtles, and marine mammals suffer from entanglement of marine plastic

debris, which often leads to mortality (Gall & Thompson, 2015; Worm et al., 2017). Sessile organisms such as sponges and corals, who play fundamental roles in the health of marine ecosystems, are affected through tissue abrasion when contacted or entangled by plastic debris, causing partial or total mortality (Gall & Thompson, 2015). In addition, tissue abrasion and stressors from contact and entanglement of corals can provide pathway for invasion from pathogens, increasing the risk of disease for these organisms (Lamb et al., 2018).

The ingestion of plastic by marine organisms is detrimental to their health and survival. Smaller plastic items, such as bottle caps, cigarette lighters, and fragmented plastics, may be ingested by a wide variety of organisms, leading to obstruction of the gut, reduced food consumption due to satiation (malnutrition or even starvation), or intestinal blockage leading to death (Kühn et al., 2015). Ingestion can occur intentionally, when a species assumes the plastic debris is a food source, or accidentally, through filtration or indiscriminate grazing. Up to 44% of seabird, 100% of marine turtle, 59% of whales, and 36% of seal species have been observed to ingest large and small plastic debris (Kühn et al., 2015). Ingestion also risks causing uptake of chemicals from plastic into tissue and organs (Eerkes-Medrano et al., 2015). The main route by which chemicals transfer into an organism is ingestion via the gut, rather than sorption through the body wall (Browne et al., 2013). C. M. Rochman (2019) summarized both a wide range of possible direct toxic effects from plastics via adsorption and transfer to organisms and found that toxicological studies in the laboratory demonstrate adverse effects in fish and lugworms.

Effects of Microplastic Pollution on Biota

One of the primary environmental risks associated with microplastics is their bioavailability for organisms (Lambert & Wagner, 2018). Microplastics are similar in size to fine sediments; therefore, they are bioavailable to a large number of organisms, including sediment-

feeding benthic invertebrates (Browne et al., 2013; Eerkes-Medrano et al., 2015; C. Rochman et al., 2017). Microplastics in polychaetes (marine worms), mussels, crabs, fish, and seabirds, have been shown to cause "negative effects including reduced feeding and reproductive success, reduced survival, cellular-level toxicity, changes in immune function, changes in enzyme function, and gene expression" (Worm et al., 2017). A species' feeding type is important when considering the injestion of microplastics, because feeding types are commonly used to classify and group biota (Wagner & Lambert, 2018). For instance, suspension feeders obtain nutrients from particles suspended in water, deposit feeders forage for food in sediments, and suction feeders ingest their prey together with the surrounding water. All of these feeding types are prone to injest microplastics present in the environment (Derraik & Derriak, 2002; Wagner & Lambert, 2018). Similar to the ingestion of macroplastics, ingested microplastics can have toxic effects via the adsorption and uptake of chemical and the transfer of these chemicals to organisms that ingest them (C. M. Rochman, 2019). The scientific understanding of hazardous chemicals transferring from plastic pollution to animals is still growing and further research is needed to understand the ecological consequences.

Effects of Microplastic Pollution for Human Health

The ubiquity of microplastics have raised concerns regarding how ingestion of microplastics may influence human health and wellbeing (J. Li et al., 2015; C. Rochman et al., 2017). Like other organisms, there are several routes of microplastic exposure to the human body including ingestion, inhalation, and dermal contact. Ingestion is considered the main route of human exposure and microplastics have been reported in tap water, bottled water, sugar, salt, beer, honey, and seafood (Cox et al., 2019; Prata et al., 2020; Rist et al., 2018). According to Cox et al. (2019), the estimated intake of microplastics via ingestion is 39,000-52,000 particles

per person annually, although these estimates increase to 74,000-121,000 particles when inhalation is considered. The inhalation route is gaining traction as a notable source of microplastic intake because microfibers from synthetic materials, such as clothing, are easily mobilized into the air (Prata et al., 2020; Rist et al., 2018). The scientific literature demonstrates that microplastics are present in humans, but the question remains: what are the consequences for human health? Although it is alarming that plastic is found to be ingested regularly, to date, little information seems to be available on the potential toxicity of microplastics to humans; however, a recent clinical study suggests microplastic and nanoplastics are emerging as a potential risk factor for cardiovascular disease, although direct evidence is still lacking (Marfella et al., 2024).

Rivers as Sources of Plastic into Marine Environments

Despite widespread acknowledgement that plastics are detrimental to ecosystems and pervasive in the environment, most contemporary research focuses on marine plastic pollution (Lambert & Wagner, 2018). This is because marine environments are sinks for plastic pollution and their persistence and effect on marine environments has been studied for decades (Jambeck et al., 2015). Regardless, freshwater environments are subject to extensive pollution by plastics and should receive more scientific attention (Blettler et al., 2018). One major reason why continued studies on freshwater environments is paramount is because land-based sources, such as rivers and estuaries, are one of the biggest sources of plastic pollution into the marine environment (Jambeck et al., 2015; Lebreton et al., 2017; Strokal et al., 2023).

A significant portion of terrestrial plastic waste is transported from land via river systems and ends up in the ocean (Teuten et al., 2009), with an estimated 70% to 80% of plastic debris found in marine environments being emitted to the ocean via rivers (Wagner et al., 2014). Lebreton et al. (2017), using a model based on global waste management practices, population

densities, and hydrological information, estimated that between 1.15 and 2.41 million Mt of plastic flows from rivers into oceans every year (Lebreton et al., 2017). Other studies estimate much higher plastic flows, such as 4.8 to 12.7 million Mt (Jambeck et al., 2015). An accurate estimate of plastic debris flowing from river systems into marine environments is nearly impossible because of a lack of data, inconsistent methodologies, and unaccounted for activities (e.g., illegal dumping).

There is a growing disparity between developing countries and industrialized, wealthy countries when it comes to plastic pollution. Developing countries – and particularly, China – are responsible for most of our global primary plastic production. These regions also tend to have high population densities, rapid urbanization, and ineffective waste management systems, which generally leads to significantly higher amounts of mismanaged plastic waste and plastic pollution (Blettler et al., 2018; Chen, 2015; Jambeck et al., 2015). For example, the global model study by Lebreton et al. (2017) concluded that the top 20 polluting rivers are predominately located in southern Asia – a fast-developing middle income region – and accounted for more than two thirds (>67%) of the global total plastic pollution into marine environments (Lebreton et al., 2017). Additionally, the top 122 polluting rivers accounted for >90% of the plastic input into the ocean; 103 rivers were in Asia, 8 in Africa, and 8 in South and Central America. Of those 122 rivers, only one was in a highly developed region (Europe). Plastic production and pollution will continue to grow with increased population and increased per capita consumption associated with economic growth, particularly in urban areas and developing nations (Jambeck et al., 2015); therefore, improving waste management infrastructure in developing countries is of prime importance and will be a major step towards reducing plastic waste.

Plastics in Freshwater Environments

Knowledge about the spatial distribution of microplastics in freshwater systems is lacking. According to Lambert & Wagner (2018), who reviewed 1,228 papers containing the term 'microplastic', just 45 publications (3.6%) were associated with freshwater environments, and Akdogan & Guven (2019) reported that 54% of publications on microplastic pollution between 2006 and 2018 were related to marine environments, with only 11% focused on riverine environments. Although significantly less data are available compared to marine systems, studies of freshwater systems show comparable microplastics concentrations and pervasiveness (Wagner & Lambert, 2018).

Urban areas, and in particular large cities, have been recognized as considerable sources of microplastic pollution (Lambert & Wagner, 2018; Mani et al., 2015; Peng et al., 2017). The concentration of manufacturing industries, WWTPs, stormwater systems, and high population densities, are documented as being a significant contributor of plastic pollution into adjacent freshwater environments (Wagner & Lambert, 2018). The concentration and abundance of microplastics in freshwater systems is influenced by several factors, such as proximity to WWTPs, proximity to storm water outfalls, adjacent land use, flow regimes, geomorphological characteristics, and seasonal and weather events; therefore, riverine plastic pollution is highly complex (Mani et al., 2015). Realistic estimates of plastics flowing from rivers to oceans are very important in helping to raise the awareness of plastic debris sources and, ultimately, encourage measures to reduce plastic pollution (Lechner et al., 2014).

Chapter 3: Methodology

There are no standardized methods for macroplastic and microplastic sampling and analysis in freshwater environments and many studies use an array of different methods (Hidalgo-Ruz et al., 2012; Lu et al., 2021); however, across the scientific literature, there are several commonly used methodologies for site selection, sampling procedures, organic debris digestion, density separation, and quantifying microplastics. The primary sources for methods used in my thesis were adapted from technical papers published by the NOAA Marine Debris Program (Lippiatt et al., 2013; Masura et al., 2015; Opfer et al., 2012). Although these publications focus on sediments in marine environments, such as beaches and shorelines, the sampling and laboratory methods can be transferred to freshwater systems. My thesis used the NOAA methods where possible and any deviations from common methods are identified as needed.

Sampling Site Selection

A desktop review was completed to determine the suitability of sampling sites before commencing a field investigation. The desktop review reduced the time and effort required to find suitable sampling sites in the field. Information reviewed included aerial maps and topographic imagery, shapefiles, land-use maps, wastewater treatment plants, and stormwater infrastructure.

Specific sampling sites along the Bow River and Nose Creek were selected based on several criteria which had been adopted from (Lippiatt et al., 2013). including:

- The sites must be accessible;
- Sandy beach or pebble shorelines (i.e., depositional zones); and
- No regular cleanup activities.

These criteria were met where possible, but can be modified based on site-specific conditions (Cheshire et al., 2009; Lippiatt et al., 2013). There are annual City of Calgary and privately sponsored clean-up programs along the Bow River, where registered volunteers help remove litter along the city's pathways and riverbanks. I could not find data on clean-up programs or results of clean-up activities along the Bow River, although it can be reasonably assumed that some sites selected, particularly sites that were easily accessible, may have had cleanup activities at some point in the past.

A total of seven sites along the banks of the Bow River and one site in Nose Creek, a tributary to the Bow River, were sampled between April 14 and April 25, 2019 (Figure 1 and Figure 2). Starting upstream and working downstream, the sites included Bowness Park (Site Number: S1), Prince's Island Park (S2), Nose Creek (S3), Harvie Passage (S4), Ogden Road Bridge (S5), Glenmore Trail (S6), Fish Creek Provincial Park (S7), and Policeman's Flats (S8). The Glenmore Trail (S6) was selected as the dividing line that considers S1-S5 as "upstream sites" and S6-S8 as "downstream sites". A brief description of each site is as follows:

- The Bowness Park site (S1) is approximately 1 km downstream of Bearspaw Dam, which marks the western administrative boundary of the City of Calgary. Surrounding land use includes a municipal park (Bowness Park), a golf course, walking trails and pathways, and the suburban community of Valley Ridge.
- Prince's Island Park (S2) is in Calgary's downtown core and therefore the surrounding land is heavily urbanized. The site was situated in a depositional zone comprised of fine sediments along the mainstem of the Bow River. This site was selected because sediment-based shorelines were limited in this reach of the river due to heavily modified

riparian areas, with flood protection (e.g., riprap armouring) and channel modifications prevalent along this reach of the Bow River.

- The sample site at the tributary of Nose Creek (S3) was upstream from its confluence with the Bow River. The Nose Creek watershed is influenced by increasing residential and commercial developments, industrial activities (including plastic manufacturers), stormwater discharge, agricultural activity, and stream modification and channelization. This makes Nose Creek an interesting tributary to investigate to see if land use plays a significant role in the formation of plastic pollution in littoral sediment. The sampling site selected in Nose Creek is adjacent to Deerfoot Highway, a CN Railway bridge, and an off-leash dog park.
- The Harvie Passage site (S4) is located downstream of both the Elbow River and Nose Creek's confluence with the Bow River. The site was located immediately upstream of a riprap groin that provided a back eddy depositional zone with a sandy shoreline.
- The Ogden Road Bridge site (S5) is located immediately downstream of a stormwater outfall (outfall identifier B5) that drains a large, predominately industrial area. Surrounding land use is dominated by a rail yard, industrial manufacturing, including several plastics manufacturers, and is located immediately upstream of the Bonnybrook WWTP. The riparian area at S5 included several transient camp sites and large volumes of debris and litter scattered throughout the wooded riparian zone.
- The Glenmore Trail site (S6) is located downstream of Glenmore Trail Bridge along a river island which divides the Bow River between its mainstem and a permanent side channel. The sampling site was located along the east side of the island, which faces the

mainstem of the Bow River and not the smaller side channel. Surrounding land use at S6 consists of commercial properties, parkland, and residential housing.

- The Fish Creek site (S7) is adjacent to Fish Creek Provincial Park, a golf course, and new residential developments, and is upstream of the Fish Creek WWTP.
- Finally, the Policeman's Flats site (S8) is located outside of the City of Calgary's eastern municipal boundary. Surrounding land use is predominately agricultural and forested lands with sparse residential housing. The site is located upstream of a popular boat launch to access reaches of the Bow River downstream of the City of Calgary.

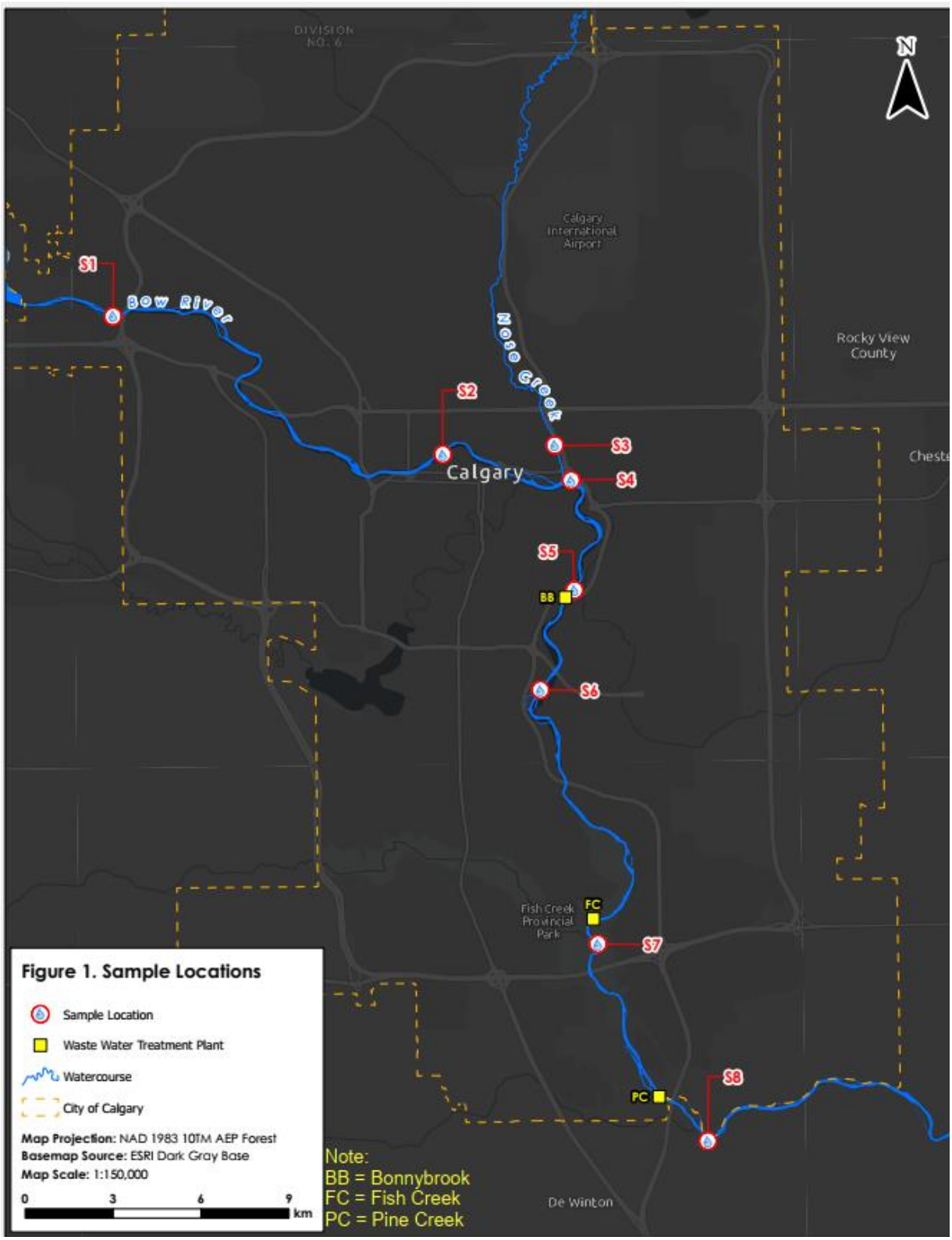

DIVISION NO. 6
N
Calgary International Airport
S1
Bow River
Nose Creek
Rocky View County
S2
S3
Calgary
S4
S5
BB
S6
Fish Creek Provincial Park
FC
S7
PC
S8
De Winton
Figure 1. Sample Locations
Sample Location
Waste Water Treatment Plant
Watercourse
City of Calgary
Map Projection: NAD 1983 10TM AEP Forest
Basemap Source: ESRI Dark Gray Base
Map Scale: 1:150,000
0 3 6 9 km
Note:
BB = Bonnybrook
FC = Fish Creek
PC = Pine Creek

Figure 2

View Looking Upstream at Four Sampling Locations Along the Bow River at A) Bowness Park (S1), B) Prince's Island Park (S2), C) Ogden Road (S5), and D) Glenmore Trail (S6), Showing the Variability of Shoreline Bank Characteristics at Sampling Locations.

Macroplastic Shoreline Sampling Methodology

To sample each site for macroplastic debris, a 10 m long transect was deployed parallel along the littoral (shoreline) zone. When possible, the transect was situated along a visible strandline. The strandline was identified by a high concentration of conifer needles, leaves, and other organic debris under the assumption that floating macro- and microplastics would also accumulate there (R. Robinson, personal communication, September 28, 2018; Claessens et al., 2011). A 10 m transect length was selected for sampling. This sampling methodology was

adapted from the NOAA Marine Debris Program recommendations (Lippiatt et al., 2013) but did not follow them directly because it was difficult to find long stretches of suitable shoreline (i.e., >10 m) that could be repeated throughout all sampling sites. After the 10 m transect was established, a 1x1 m quadrat was deployed on each side of the 10 m transect for a total of 20 sampling quadrats (a 20 m^2 area). Once deployed, each quadrat area was photographed to document pre-sampling conditions (see Figure 3).

Figure 3

View of 1 x 1 m Quadrat Deployed at Ogden Road (S5) for Sampling Macroplastic Debris Along the Transect.

Larger organic debris such as logs, sticks, and vegetation were inspected for plastic debris and discarded. All manufactured debris (plastics, metals, glass) within each quadrat was counted, classified, photographed (if applicable), collected, and later discarded at an approved waste management facility. Plastic debris that could not be visually classified because of excessive weathering or fragmentation were photographed, their longest length measured, qualitatively described, and stored in bags or jars.

Microplastic Shoreline Sampling Methodology

To investigate microplastic contamination, two shoreline sediment samples were obtained from each site. Using a random number generator, a number was selected between 1 and 10 which related to a location along the 10 m transect, measuring upstream to downstream, where the sediment sample would be obtained. For example, if the number 5 were randomly generated then sediment samples would be obtained at the 5 m mark along the transect. Once a location was selected using the random number generator, a 25 x 25 cm quadrat was deployed along the suspected strandline (see Figure 4). Within the quadrat, stones and non-plastic objects >10 mm in size were removed (Klein et al., 2015) and the top 2-3 cm of sediment was extracted from the quadrat using a metal hand trowel (Lippiatt et al., 2013; J. Wang et al., 2017). Samples were placed in 1 L sterilized glass jars, sealed, labeled with site-specific information such as date, time, and location, and placed in dry storage until laboratory analysis. The quadrat was rinsed between sampling at each site and deployed at the second locaiton along the transect. The two samples were then mixed together to form one composite sample from each location. A total of 8 composite sediment samples – one sample from each site – were obtained.

Figure 4

View of 25 x 25 cm Quadrat Deployed at Sampling Locations, showing Pre-Sampling (A) and Post-Sampling (B).

Sediment Sample Laboratory Processing

Sample Preparation

Sample preparation for the isolation of microplastics followed methods outlined by Masura et al. (2015) with deviations to the sieving process. First, each bulk sample volume was filtered through a 5 mm sieve to remove debris larger than 5 mm (i.e., rocks, wood, metal, plastics). Any plastic debris larger than 5 mm were counted and classified as macroplastics and removed. From each bulk sample, 500 g of wet sediment was weighed, covered, and subsequently dried in an oven at 90°C overnight until the sediment was dry (Masura et al., 2015). After drying, each bulk sample was weighed again to determine the dry sample weight. To ensure a consistent volume of sediment for processing, 100 g of sediment was extracted from the dry bulk sample. The 100 g was obtained by shaking each dry sediment sample inside their closed container, collecting ~20 g of dry sediment and placing it in a separate covered glass jar,

and repeating this process until the desired mass of 100 g dry sediment was reached. All samples remained covered as much as possible to reduce the potential for airborne contamination.

Removal of Organic Debris (Sample Digestion)

Organic debris and biofilms need to be removed from each sediment sample to facilitate easier identification of microplastics during analysis (Mai et al., 2018; Masura et al., 2015). To dissolve organic debris, a solution of 30% hydrogen peroxide (H_2O_2) was used (Masura et al., 2015) . First, the 100 g dry sediment sample was transferred to a 400 ml beaker. Depending on the volume of organic debris in the sample, 20 ml of H_2O_2 combined with 20 ml of deionized water was added to the beaker sample. The sample was covered with tin foil, placed on a hotplate magnetic stirrer under a fume hood, and vigorously stirred for 5 minutes at room temperature. Once finished, the sample was left to sit for 5 minutes. The sample was then simultaneously stirred and heated to 75°C until gas bubbles were observed on the surface, after which the sample was removed from the hotplate and remained in the fume hood until the boiling subsided. If the reaction appeared to have potential to overflow the beaker, distilled water was added to slow the reaction (Masura et al., 2015). Once the boiling subsided the sample was again heated to 75°C and stirred for 30 minutes. If organic debris were still visible, an additional 10-20 ml of H_2O_2 was added to the sample and the process was repeated until no (or negligible) organic debris was visible. The samples were now prepared for density separation.

Density Separation

The most common and economical method used to separate microplastics from sediment is by using a saturated NaCl solution with a density of 1.2 kg/L (J. Li, Liu, & Paul Chen, 2018). For each 20 ml of sample volume, ~6 grams of salt (NaCl) was added to increase the density of the sample solution (Masura et al., 2015). The solution was stirred and heated to 75°C until the

salt dissolved. The solution was then covered with tinfoil and left to settle for at least 24 hours at room temperature (Klein et al., 2015). The settling process allows low-density particles such as microplastics to float to the upper layer of water (i.e., the supernatant) and high-density particles, such as clay and suspended sediments, to sink to the bottom. All the supernatant, including any visible particles floating on the surface of the supernatant, was withdrawn using a glass pipette (J. Li, Liu, & Paul Chen, 2018). During withdrawal, the sides of the beaker were occasionally rinsed with a wash bottle containing distilled water to dislodge any particles adhering to the side of the beaker to ensure they were collected by the pipette. An additional 100 ml of deionized water was added to the sediment sample and the density separation procedure was repeated. The second procedure was used to extract any additional microplastics remaining in the sediment sample that may not have been captured during the first round of separation.

Filtration

The pipetted solution was decanted onto Whatman® 25 μm pore size, 55 mm white non-grided mixed cellulose filter paper connected to a vacuum filter apparatus (Mathalon & Hill, 2014). After filtration was completed, each filter paper was placed in its own sterilized Petri dish. The Petri dishes were covered and left to dry at room temperature for at least 48 hours prior to microscopic analysis (Hidalgo-Ruz et al., 2012; Mathalon & Hill, 2014).

Laboratory Analysis

Once the samples were dry, visual examination could proceed. Visual examination of the filtered sample is a necessary step in nearly all reviewed studies (Hidalgo-Ruz et al., 2012; Lu et al., 2021; Mai et al., 2018). Careful visual inspection of the processed samples is necessary to separate potential microplastics from other materials, such as organic debris or non-plastic items (i.e., glass, salt, cotton, sand), and to determine the physical characteristics of potential

microplastics. Visual inspection was completed using a stereoscope (also known as a "dissecting microscope") at 40X magnification. During examination, the filtered paper was left in the petri dish with the lid off and placed in a stationary position on the microscope's mechanical stage. The filter paper was then examined starting from the top of the paper. From there, the mechanical stage was slowly moved in a back-and-forth (i.e., left to right) pattern, continuing this sequence while moving down the filter paper until the entire filter paper was examined.

If a suspected piece of microplastic was discovered, but I was not confident it was a plastic polymer, it was carefully removed using metal tweezers and placed onto a glass side for further inspection using an Olympus CH30RF100 Compound Microscope at 4X to 100X magnification. A compound microscope was used because the resolution is higher and, particularly at 100X magnification, biological structures are more readily visible. Individual pieces inspected under the compound microscope were subjected to several tests to increase the confidence of its origins, which were adapted from the Marine & Environmental Research Institute's *Guide to Microplastic Identification* (2016). Several rules were followed including:

- No cellular or organic structures are visible, except in the unlikely case where biofouling or a small piece of organic debris (which should have been removed during sample digestion) overlaying the plastic was visible and easily distinguished from the polymer itself.
- Fibers should be equally thick throughout their entire length, except for potentially frayed or split ends.
- Particles should exhibit clear and homogenous colour throughout.

- Tweezers or a needle-tip is used to prod the object. When prodded, salt crystals, sand, and organic debris will break apart. Microplastics will, in most cases, not break or bend and retain its shape/structure.

If needed, a test was used which involved prodding the object using a hot needle. The hot needle test is based on De Witte et al. (2014), as outlined in the *Guide to Microplastic Identification* (2016). Although it is not a fool-proof method for plastic identification, it does provide greater confidence in the assessor's judgement as they try to distinguish between plastic debris and organic matter (Beckingham et al., 2023). To perform the test, a metal needle was heated under an open flame until the metal tip was glowing red. Once sufficiently hot, the needle was carefully situated near the suspected plastic piece. The reaction of the object was observed through the stereoscope. If the suspected particle is plastic, the hot needle will cause the plastic piece to melt, bubble, and/or curl. If the piece is organic in nature, such as cotton or wool fibers, there will be a burning reaction. Sand grains and salt crystals will not respond to the heat of the needle and will not melt, curl, or burn. The hot needle test, used in conjunction with the rules mentioned above, increased the confidence of my visual examination results.

Microplastic particles were classified by shape, which is roughly divided into four category types: fiber, round (pellet), foam, and fragment (Hidalgo-Ruz et al., 2012; Peng et al., 2017).

Limitations

Several limitations were encountered during this study that need to be acknowledged. The first limitation is using a stereomicroscope for identification of microplastics, which is necessary but time consuming and laborious. Visual identification is economical, given the cost of advanced analytical instruments, but it can lead to erroneous characterization of microplastics

because when looking at small pieces of microplastics, it is difficult to differentiate them from other organic and inorganic particles of similar size and shape (J. Li, Liu, Chen, et al., 2018). For example, assuming a cotton (organic) fiber is a polymer-based fiber (or vice versa); assuming a black particle is sediment or undigested organic debris; a transparent particle is a piece of weathered glass; or the possibility of missing small or transparent plastic particles entirely. For more accurate results, it is often necessary to couple visual enumeration with other instruments such as Fourier-transform infrared spectroscopy (FT-IR) or Raman spectroscopy (J. Li, Liu, Chen, et al., 2018; Mai et al., 2018). FT-IR spectroscopy subjects samples to infrared radiation with a defined range and the resulting molecular vibrations are compared to a known database of plastics polymers with distinct band patterns to categorize the polymer of the microplastic (e.g., polystyrene, polypropylene, polyester, etc.) (J. Li, Liu, Chen, et al., 2018). Unfortunately, access to this equipment was unavailable for my study. According to the scientific literature, microscopic visual identification alone may not be accurate enough to determine the abundance of microplastics in a sample, yet it is an integral part of most studies on microplastics (Blettler et al., 2018; Eriksen et al., 2013; Hidalgo-Ruz et al., 2012). Consequently, visual identification alone does not diminish the validity of my research, but this limitation should be overcome in any future research by prioritizing access to specialized analytical equipment.

Another limitation my use of NaCl for density separation. Although concentrated NaCl is widely used for similar studies, the density of NaCl in water (1.2 kg/L) can result in underestimation of microplastics because some plastic polymers have higher densities (e.g., polyvinyl chloride, polyurethane). These high-density plastics will sink to the bottom of the sample and will not be suspended and recovered from the supernatant. Higher-density solutions such as sodium iodide, sodium zinc chloride, and polytungstate solutions have all been used in

other studies to recover these high-density microplastics; however, compared to NaCl, these solutions are expensive, hazardous to work with, and not environmentally friendly. Considering NaCl is still widely used for separating microplastics from a water or sediment matrix, this limitation does not compromise the validity of my research.

Another limitation was the timing of the coronavirus disease 19 (COVID-19) pandemic, which made sourcing laboratory space difficult because universities and other institutions were experiencing periods of lockdowns and social distancing during that period. This delayed sample processing timelines and made sourcing of space and equipment challenging, but otherwise did not influence the field sampling program.

Quality Assurance and Quality Control

For accurate results it is necessary to minimize sources of contamination during sampling and laboratory analysis (Hidalgo-Ruz et al., 2012); therefore, I wore cotton clothes during sample collection, extraction, processing, and analysis. All glassware and laboratory utensils were washed thoroughly with deionized water prior to use.

Blank samples were taken during the field program at one site and during one full round of laboratory analysis to monitor potential contamination from the ambient environment. Field blanks consisted of a 1 L glass jar filled with 1 L of distilled, deionized water. The lid was removed, and the glass jar was placed near the assessor during the sample collection process. Once field sampling was completed, the glass jar was sealed with a metal lid and stored until laboratory analysis. Laboratory blanks consisted of an open Petri dish containing a filter paper. The Petri dish was left open and placed adjacent to me during all steps in the laboratory analysis process. Once laboratory analysis was completed the Petri dish was sealed. Two microplastic fibers were found in the field blank and no microplastics were found in the laboratory blank.

Chapter 4: Results

The results of macro- and microplastic sampling and analysis of shoreline sediment samples are presented below. A summary of my key findings are arranged starting with results of microplastic shoreline sediment sampling, followed by macroplastic data. My findings are presented via text and non-textual elements (i.e., tables and figures). The results provide data to help achieve the objectives of this research: quantifying, enumerating, and characterizing macroplastic and microplastic pollution within the littoral (shoreline) sediments of the Bow River, and discerning trends in their distribution.

Microplastics in Bow River Shoreline Sediments

Each 100 g dry weight sediment sample analyzed (n=8) contained microplastics (Table 2). A cumulative total of 93 microplastic particles were isolated from the 100 g sediment samples[5]. As shown in Table 2 and Figures 5-7, the most frequent microplastic shape (or type) was fiber (n=52), followed by fragment (n=26), foam (n=14), and finally beads/spheres (n=1). The number of microplastic pieces enumerated at each location ranged from 4 to 20 pieces/100 g of sediment. The location with the lowest number of microplastics per 100 g sample was Harvie Passage (S3; n=4). The location with the highest number of microplastics was Fish Creek (S7; n=20). The most identified colour of microplastic particles found in the sediment samples was blue (n=44) followed by white/transparent (n=25), red (n=10), black (n=9), other (n=3), and green (n=2).

An independent two-tailed t-test was conducted to assess whether the concentration of microplastics in shoreline sediments along the Bow River increases downstream. For this test, the demarcation point that distinguished upstream and downstream sites was the Glenmore Trail

[5] All references to sediment should be considered dry weight.

(S6). Meaning, S1-S5 were considered upstream, where the most heavily urbanized and populated regions of Calgary are located, and S6-S8 were considered downstream locations where microplastics may accumulate, potentially leading to higher concentrations of microplastics in shoreline sediment. The t-test yielded a p-value of <0.01, indicating a statistical significance between upstream and downstream microplastic concentrations. Furthermore, linear regression was used to investigate the relationship between microplastic concentrations (dependent variable) and distance in kilometers downstream, starting from S1 as 0 km (independent variable; Figure 8). The analysis yielded an r^2 value of 0.68. This suggests a relatively strong relationship between shoreline microplastic concentration and distance downstream.

Table 2

Categories of Microplastic Debris Collected from all 100 g Dry Weight Sediment Samples from the Shoreline of the Bow River

Category	Bowness Park (S1)	Prince's Island Park (S2)	Nose Creek (S3)	Harvie Passage (S4)	Ogden Road (S5)	Glenmore Trail (S6)	Fish Creek Park (S7)	Policeman's Flats (S8)	Total
Fiber	6	2	7	3	7	7	10	10	52
Fragment	2	5	3	1	5	1	6	3	26
Foam	0	0	0	0	0	5	3	6	14
Round	0	0	0	0	0	0	1	0	1
Total	**8**	**7**	**10**	**4**	**12**	**13**	**20**	**19**	**93**

Figure 5

Total Number of Microplastics and Category Type per Sample Location

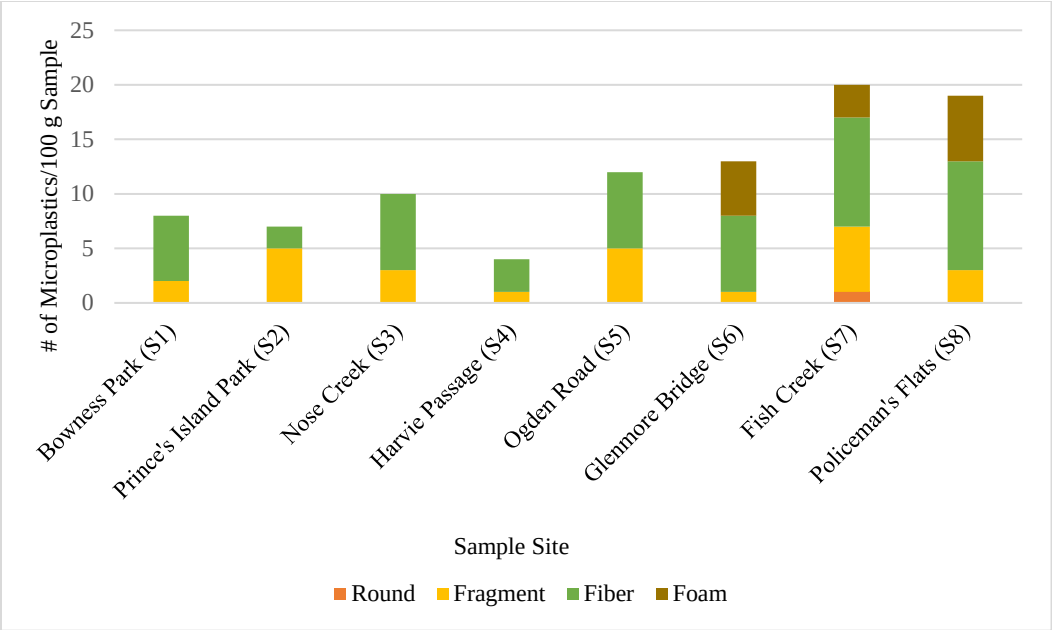


Figure 6

Relative Percentage of Microplastic Categories from Each Sample Location

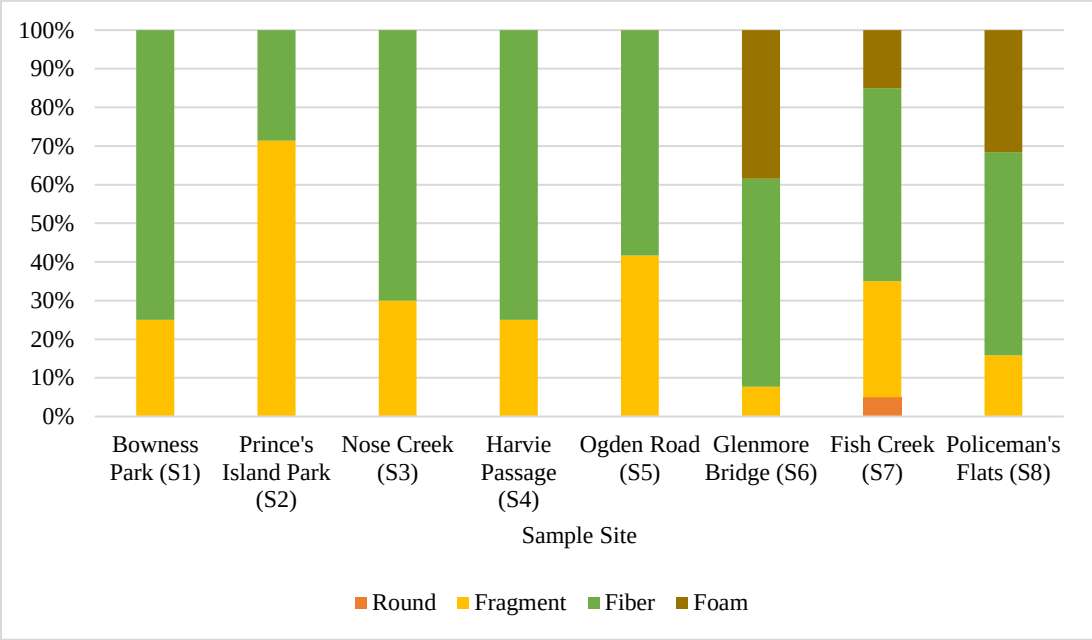

100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Bowness Park (S1)
Prince's Island Park (S2)
Nose Creek (S3)
Harvie Passage (S4)
Ogden Road (S5)
Glenmore Bridge (S6)
Fish Creek (S7)
Policeman's Flats (S8)
Sample Site
Round
Fragment
Fiber
Foam

Figure 7

Percentage of Microplastic Shape Categories Across all Samples of Bow River Sediment

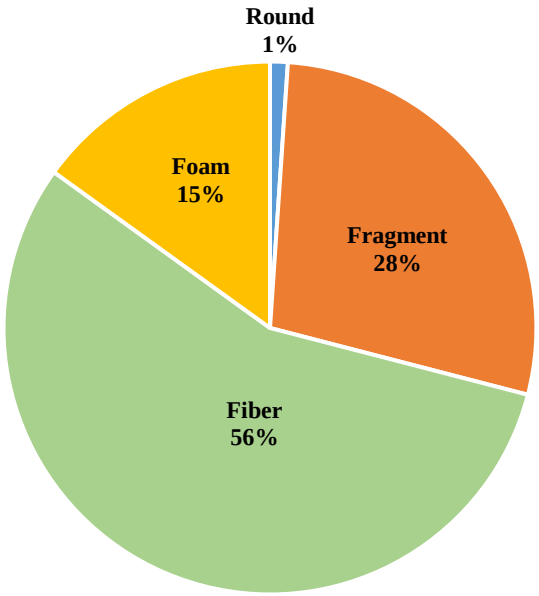
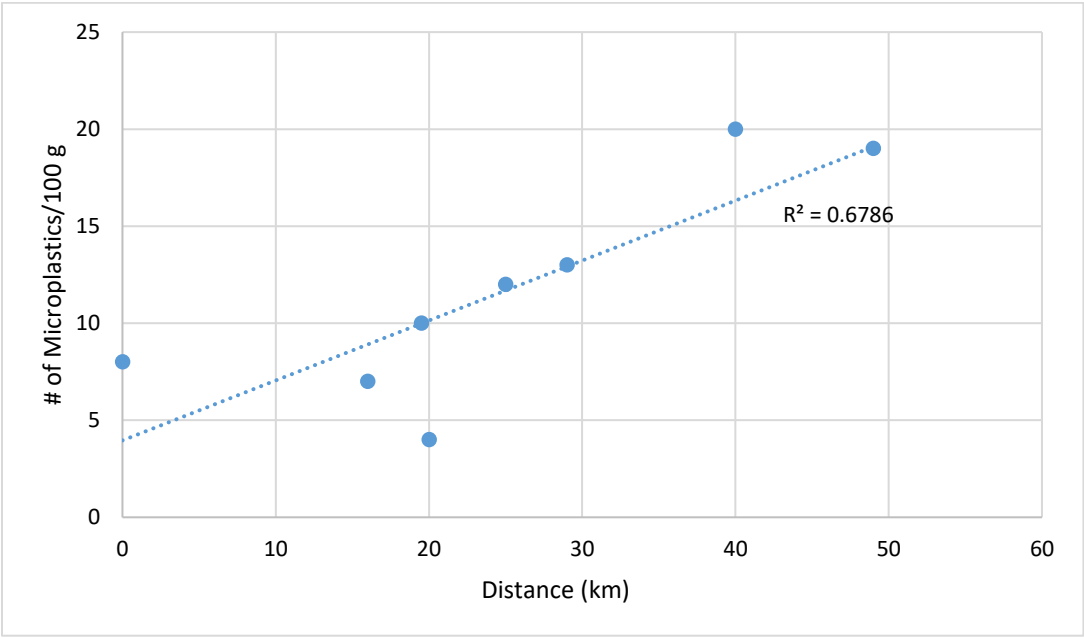


Figure 8

Regression Analysis of Microplastic Count vs Distance in the Bow River



Macroplastics in Bow River Shoreline Sediments

Macroplastic debris was found at every sample location (Table 3; Figures 9-10). A total of 826 individual pieces of macroplastic debris were collected from all sample locations. The most abundant type of plastic debris identified across all sites was small polystyrene (Styrofoam) pieces (n=230). The percentage of macroplastic per category is shown in Figure 9. The number of macroplastic debris enumerated at each location ranged from a low of 6 pieces at Bowness Park (S1) to a high of 294 pieces at Ogden Road (S5). In addition to macroplastic debris, non-plastic debris was also enumerated (Table 4). A total of 31 pieces of non-plastic debris (i.e., metal, glass, paper) were identified across all sampling sites.

An independent two-tailed t-test was conducted to assess whether the concentration of macroplastics in shoreline sediments along the Bow River increases downstream. The analysis followed the same methodology as the microplastic t-test. The macroplastic t-test yielded a p-value of 0.24, suggesting the difference between upstream and downstream macroplastic concentrations is not statistically significant. Linear regression was used to investigate the relationship between macroplastic concentrations and distance in kilometers downstream, starting from S1 as 0 km (Figure 11). The analysis yielded an r^2 value of 0.32. This suggests a weak relationship between macroplastic concentration and downstream distance.

To determine if there was a relationship between macro- and microplastic concentrations, a Pearson correlation coefficient was calculated. The Pearson correlation coefficient between macroplastic and microplastic concentrations across the eight sites was 0.54. This value indicates a moderate positive correlation, but this result should be interpreted with caution because of the small sample size. See Chapter 5: Discussion for a detailed analysis.

Table 3

Categories of Macroplastic Debris Collected from Sampling Locations along the Bow River

Plastic Debris Category	S1	S2	S3	S4	S5	S6	S7	S8	Total Pieces
Small Foam	0	4	1	12	**141**	**16**	33	23	230
Food Packaging	1	4	**14**	**16**	36	3	24	37	135
Small Plastic Pieces	1	2	7	5	26	2	**37**	**44**	124
Cigarettes/Cigars	0	**14**	4	6	38	11	19	18	110
Bottle Caps/Lids	0	2	2	3	3	1	23	20	54
Straws/Stirrers	0	0	2	10	18	1	15	6	52
Films	0	4	11	0	3	3	8	1	30
Recreational Items	**3**	1	1	0	8	6	5	3	27
Plastic Bags	0	1	1	7	8	3	0	0	20
Rubber	0	0	2	0	5	2	1	4	14
Plastic Utensils	0	1	1	1	5	0	0	4	12
Fishing Gear	0	0	1	5	0	0	6	0	12
Personal Hygiene	1	0	0	1	1	0	1	0	4
Plastic Bottle	0	0	0	0	1	0	1	0	2
Total	6	33	47	66	293	48	173	160	826

Note: Bolded numbers for debris categories indicate the most abundant type of plastics debris found at each sample site.

Figure 9

Percentage of Total Macroplastic Categories Identified Across all Sample Locations

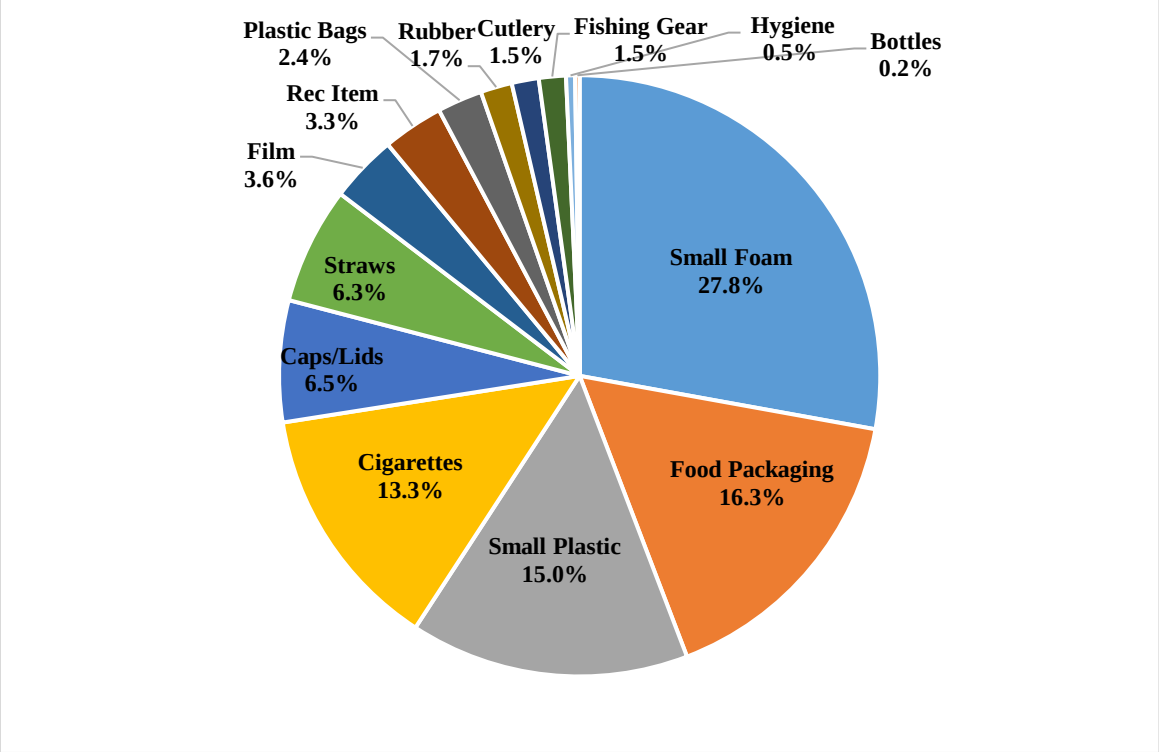

Plastic Bags 2.4%
Rubber 1.7%
Cutlery 1.5%
Fishing Gear 1.5%
Hygiene 0.5%
Bottles 0.2%
Rec Item 3.3%
Film 3.6%
Straws 6.3%
Caps/Lids 6.5%
Cigarettes 13.3%
Small Foam 27.8%
Food Packaging 16.3%
Small Plastic 15.0%

Figure 10

Number of Macroplastic Debris Collected at Each Sample Location along the Bow River

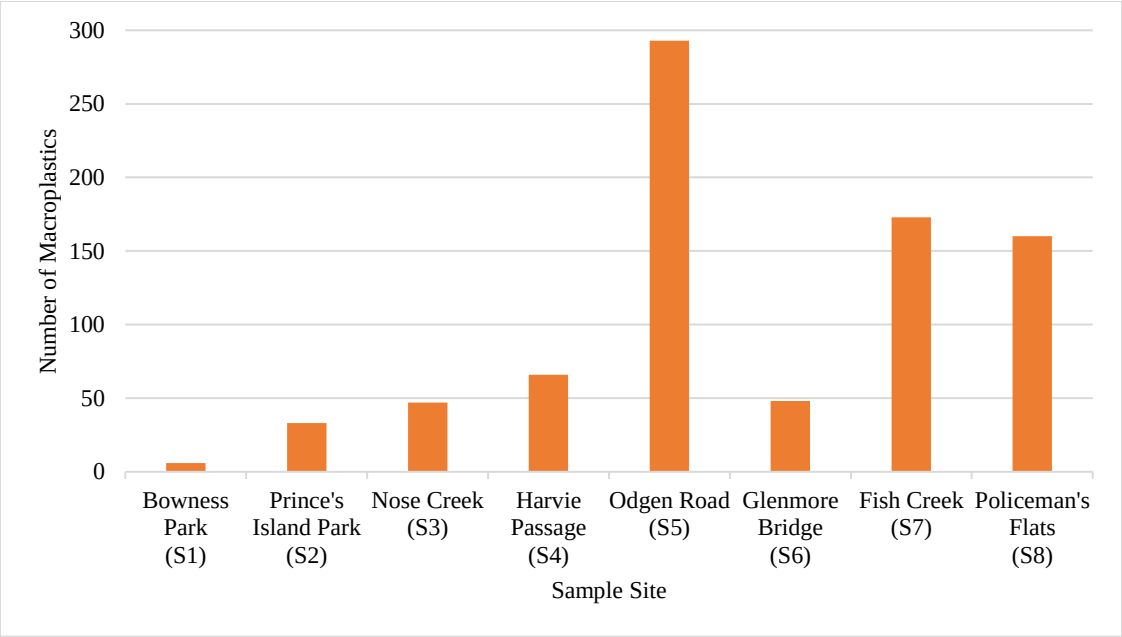


Figure 11

Regression Analysis of Macroplastic Count vs Distance in the Bow River

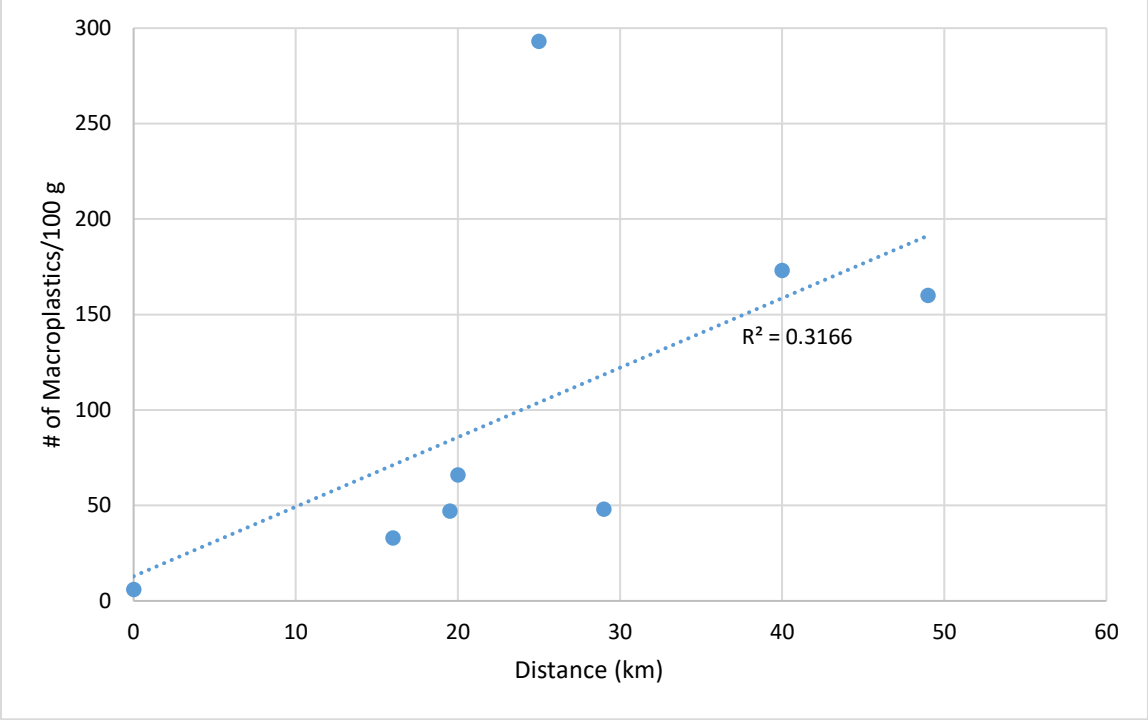


Table 4

Categories of Non-Plastic Debris Collected from Sampling Locations along the Bow River

Non-Plastic Debris Category	Bowness Park (S1)	Prince's Island Park (S2)	Nose Creek (S3)	Harvie Passage (S4)	Ogden Road (S5)	Glenmore Trail (S6)	Fish Creek Park (S7)	Policeman's Flats (S8)	Total Plastic Pieces
Glass	0	0	1	7	2	0	0	1	11
Metal	0	0	5	2	2	2	0	0	11
Paper	1	0	2	0	4	2	0	0	9
Total	**1**	**0**	**8**	**9**	**8**	**4**	**0**	**1**	**31**

Chapter 5: Discussion

The results indicate that shoreline sediment along the Bow River is polluted with macro and microplastics; however, there was nothing unexpected about this because plastics are found in nearly every environment investigated. In this chapter, I explain and evaluate the results of this study and attempt to show how it relates to my original research objectives, how the results compare to similar studies in the scientific literature, and present arguments to support the conclusions and recommendations provided in Chapter 6.

Abundance of Microplastics in Bow River Sediments

I anticipated that the abundance of microplastics in sediment along the Bow River would be lower at upstream sites S1-S5 when compared to downstream sites S6-S8. This is because downstream sample sites would have an increasing number of terrestrial inputs – such as more stormwater outfalls, WWTP outfalls, and urban runoff – and additional tributaries (e.g., Elbow River, Nose Creek, Fish Creek) joining the Bow River as the river navigates through the City of Calgary. These inputs, combined with increasing population densities and human activities in the urban environment, would result in more prevalent microplastic concentrations in downstream sediments.

As anticipated, the highest concentration of microplastics was recorded at downstream sites S6-S8. Furthermore, the results from the independent two-tailed t-test (p-value of <0.01) and the regression analysis (r^2 value of 0.68), when comparing upstream to downstream sites, both indicate statistical significance. This provides evidence that the City of Calgary is likely acting as a general point source for microplastic pollution into the Bow River, with increasing concentrations downstream of the city. This trend of higher concentrations downstream of urban environments is consistent with similar studies (Baldwin et al., 2016; Mani et al., 2015;

McCormick et al., 2014); however, this trend is not universally observed across all research because microplastic transportation and deposition in dynamic river systems is highly complex and variable (He et al., 2020; Zbyszewski et al., 2014).

There was an unexpected drop in microplastics at Harvie Passage (S4; n=4). There are several reasons why this drop may have occurred. For instance, the boundary between the wetted shoreline and the high-water mark (i.e., where riparian vegetation is established) was relatively narrow compared to other sampling locations; approximately 1.25 m wide from wetted shoreline to riparian vegetation, with a moderately steep gradient and evidence of undercut banks. Another factor that could have caused the drop at S4 may be the large groyne (a large deposit of rock [i.e., riprap] running perpendicular to the shoreline) immediately downstream of the sampling location at Harvie Passage. The groyne may cause an eddy that affects this sampling location by causing increased erosion of the bank and mobilization of shoreline sediments. Based on these site features at S4, it is reasonable to assume that the sampling transect was deployed in an area that is frequently submerged during periods of moderate to high discharge in the Bow River. The unexpected drop in microplastic concentration at S4 provides further evidence that channel morphology and flow regimes play an important role for depositional patterns of macro- and microplastics, which is documented in the scientific literature (He et al., 2020; Zbyszewski et al., 2014).

Microplastic Composition in Bow River Sediments

The most abundant category of microplastic debris in shoreline sediments was fibers. This aligns with similar studies of microplastic composition in rivers and estuaries (Bujaczek et al., 2021; Crew et al., 2020; Ding et al., 2019; Huntington et al., 2020; Mathalon & Hill, 2014; Ross et al., 2023; Vermaire et al., 2017). According to Mathalon and Hill (2014), studies have

shown that a single synthetic clothing garment can release >1900 microplastic fibers per wash. Another study estimated over 700,000 fibers could be released from an average 6 kg wash load (Napper & Thompson, 2016). The data indicates fibers released by washing clothes are an important source of microplastic pollution to aquatic habitats, and these fibers enter the aquatic environment via effluent discharge from WWTP. Calgary has 3 WWTPs: Bonnybrook, Pine Creek, and Fish Creek. These facilities have different levels of effluent treatment, with Pine Creek having the most advanced treatment methods. Nonetheless, even with treatment, complete removal of microplastic particles from WWTP discharge is not feasible (Ziajahromi et al., 2017); therefore, it is reasonable to assume that Calgary's WWTPs are a substantial source of microplastic pollution discharged daily to the Bow River. Although, they are not the only point source. Another point source is Calgary's stormwater system. The stormwater system is entirely separate from the WWTP system (i.e., separate pipe systems) and, unlike the WWTP system, there is no treatment of stormwater runoff. This means that urban runoff from Calgary goes directly to the Bow River watershed via ~450 outfall locations along the Elbow and Bow Rivers (R. Robinson, personal communication; (Ross et al., 2023).

The study by Ross et al. (2023) investigated concentrations and characteristics of suspended microplastics in Calgary's stormwater runoff under baseflow conditions and following individual rain events. Microplastics were identified in all samples, with higher concentrations occurring after rain events. Other factors such as catchment size, percentage of impervious surface areas, and runoff flow rate were all positively correlated to microplastic abundance. Like my study, fibers were the most prevalent category of microplastics, followed closely by fragments. Although, this result was very site- and time-specific (i.e., more fibers and larger particles under baseflow; more fragments and smaller particles during rain events). The

findings by Ross et al. (2023) suggest Calgary's stormwater system contributes a significant amount of microplastics to the Bow River, highlighting the need for effective mitigation solutions to reduce plastic pollution from this point source.

The results of my study found the highest number of fibers at the two furthest downstream sites (S7 and S8), which are located downstream of the three WWTPs servicing the City of Calgary and downstream of most of the City's stormwater outfalls; however, there was microfiber contamination across all sites sampled. Considering fibers were found at each sample site, including S1, there are likely other sources of contamination, such as the towns of Cochrane and Banff, which are upstream of Calgary. In addition, airborne contamination from atmospheric fallout is recognized as a sizable contributor of fibers (and other microplastics) into the aquatic environments (Vermaire et al., 2017; Wong et al., 2020).

Although the most-identified microplastic category in my study was fiber, it is possible this may be an overestimation because this study lacked FT-IR (or equivalent) analysis, which would help determine if misidentification of non-polymer fibers occurred. Misidentification of microfibers has been discussed in the literature (Blair et al., 2019; J. Wang et al., 2017) and visual characterization alone may lead to overestimation; however, the majority of studies using FT-IR analysis have also identified fibers as their most prevalent microplastic category (Peng et al., 2017; Simon-Sánchez et al., 2019; Vermaire et al., 2017; Yang et al., 2021).

An interesting result from the microplastic data was the presence of white foam microplastics at Glenmore Trail (S6), Fish Creek (S7), and Policeman's Flat (S8), and the absence of these foam microplastics upstream at S1-S5. When examining the macroplastic data, a pattern emerges that may explain this result. For macroplastics, the most abundant plastic category was larger (i.e., >5 mm in diameter) white foam pieces (230 pieces total; 28% of the

total enumerated macroplastics) and the largest concentration of these foam pieces was observed at S5 (141 pieces; or 61% of all foam pieces identified). No obvious source of these larger foam fragments was found at S5, but it is possible foam debris was blown in from a transient camp in the adjacent riparian area or discharged from the stormwater outfall (Outfall B5) immediately upstream. Nonetheless, it is plausible that the foam microplastics found exclusively at S6-S8 are derived from the physical breakdown, fracturing, and mobilization of the larger foam pieces found at S5. Although this pattern is inferred and lacks direct confirmation, it underscores the concept of secondary microplastic pollution and the downstream mobilization of plastic debris in riverine environments.

Macroplastic Debris Along Bow River Shorelines

Macroplastic debris was present at all sampling locations and, when compared with non-plastic debris (e.g., metal, glass, etc.), accounted for ~96% of all litter found along the Bow River. A total sampling area of 160 m^2 (20 m^2 x 8 sites) of shoreline contained ~826 pieces of macroplastic, sampled during a period of low flow. The most dominant category of macroplastics were foam pieces (n=230), food packaging (n=135), small plastic pieces (fragments; n=124), and cigarettes/cigars (n=110). These four categories can be considered single-use plastics and accounted for ~72% of the total macroplastics enumerated.

In general, sites located further downstream (S6-S8) had higher volumes of macroplastic debris compared to upstream sites; however, S5, which is in the middle reaches of the Bow River through Calgary, had the highest abundance of macroplastic debris (n=293). One plausible reason for this is the presence of several transient encampments and associated debris in the riparian floodplain immediately adjacent to S5. Debris from this area accumulates along the banks of the Bow River at S5. Another point source may be the stormwater outfall (Outfall B5),

which drains a large catchment area, located immediately upstream of S5. These both serve as examples of a discrete point source for macroplastic debris into the aquatic environment.

Excluding the S5 data, a noticeable trend emerges in the total concentration of macroplastic debris from upstream to downstream sites. This trend resembles what is observed for microplastic data, although the r^2 value of 0.32 is much lower for macroplastics. The data from S5 may disproportionately influence the results of regression analysis. If we consider S5 to be an outlier and remove this dataset from the analysis the r^2 value increases to 0.83, indicating a much stronger statistical significance. However, I opted not to remove S5 from the findings because, although it is an extreme value, it remains a valid data point and highlights the complexity of plastic pollution deposition in riverine systems.

To determine if there is a relationship between macroplastic and microplastic concentrations, a Pearson correlation coefficient was calculated, which yielded a value of 0.54 across the eight sites. This value indicates a moderate positive correlation, suggesting when the quantity of macroplastics increases there is a corresponding increase in microplastics. This relationship does not mean that the physical breakdown of shoreline macroplastics are the source of microplastics in *local* sediments; polymer type, shape, and density (e.g., PE, PET, PS, PVC) behave differently in rivers, so macro- and microplastic transport and deposition mechanisms are distinct. Rather, the correlation suggests depositional areas should be prioritized for any clean-up activities as more debris across different size fractions is likely to accumulate.

Comparison of Bow River Microplastic Composition with Other Studies

I attempted to find similar studies of microplastic contamination along riverbank areas to compare against my study along the Bow River. There is a noticeable lack of studies of microplastic pollution in riverine environments globally because researchers have tended to

focus primarily on marine and beach environments (J. C. Anderson et al., 2016; Jenkins et al., 2022; J. Wang et al., 2017). Furthermore, studies that do investigate microplastic contamination in riverine "sediments" tend to focus on river substrate (bottom) rather than riverbanks. In Canada, there are a limited number of studies of microplastic pollution in Canadian rivers in general, and I am unaware of any similar studies of microplastic contamination in riverine littoral sediments in Canada. Most microplastic contamination studies in Canada focus on substrate environments and/or floating microplastics, so I needed to broaden my search criteria to include global published data. For the studies that are available, a comparison with my results can provide an indication of the level of contamination in the Bow River relative to other watercourses.

I attempted to find studies that followed similar methods for sample collection, processing, digestion, analytical equipment, and analysis; however, it is important to acknowledge that no standardized methodology exists for sampling microplastics in riverine littoral environments. As such, it is necessary to highlight uncertainties when comparing microplastic contamination in sediments across studies that use different sampling and methodological procedures.

The results of my study were presented as "items per 100 g of sediment". Although there is no standardize unit of measurement, many studies in the scientific literature appear to use "items per m^2" or "items per kilogram (kg) sediment" (Mai et al., 2018). I used 100 g of dry weight for my study because it is less labour intensive and more economical to process 100 g of sediment compared to 1 kg. To align my results with other studies that reported data in items per kg, I converted my findings to 1 kg of dry weight by multiplying them by a factor of 10.

Table 5 displays the results of my study compared to other studies that sampled riverine shoreline sediments. The results provide evidence that Bow River littoral sediments are not as contaminated compared to other regions, although a variety of factors influence this conclusion. A more detailed interpretation of data presented in Table 5 is provided below.

Table 5

Comparison of Bow River Sediment Microplastic Concentrations with Similar Studies

Study Area	Matrix	# of Sample Sites	Mean (items/kg)	Range (items/kg)	Dominant Category	Reference
Bow River, Calgary, Alberta	Sediment	8	116	40 – 200	Fibers	this study
Rhine River, Germany	Sediment	8	861	228 – 3763	Fragment	Klein et al., 2015
Main River, Germany	Sediment	2	1077	786 – 1368	Fragment	Klein et al., 2015
River Kelvin, Scotland	Sediment	1	278	161 – 432	Fibers	Blair et al., 2019
Tijuana River Basin, Mexico	Sediment	11	187	57 – 469	Fibers	Piñon-Colin et al., 2024
River Ganga, India	Sediment	7	-	99– 410	Fibers	Sarkar et al., 2019
Beijiang River, China	Sediment	8	312	178 – 544	N/A (Polyethylene)	J. Wang et al., 2017
Yangtze River and tributaries, China	Sediment	54	717 (Yangtze River) 753 (tributaries)	35 – 51968 (Yangtze River) 52 – 1463 (tributaries)	Flake	W. Li et al., 2022

Microplastic Composition Compared to Other Studies

The results of my study indicated fiber was the primary microplastic category found in Bow River sediments. This result aligns with other, similar studies of microplastic contamination in freshwater environments (Blair et al., 2019; Ding et al., 2019; Piñon-Colin et al., 2024;

McCormick et al., 2014; Sarkar et al., 2019). In addition, fibers have been identified as the most dominant microplastic category in beach and river delta sediments (Luo et al., 2019; Simon-Sánchez et al., 2019; Zhao et al., 2014); however, fibers are not always the most dominant category identified in the scientific literature.

Klien et al. (2015) investigated shoreline sediment samples in the Rhine and Main Rivers in Germany. They aligned their results based on the size fraction of sediment particles, which I did not do. For the smallest size fraction (63 – 200 µm), spheres were the most abundant category. For the larger size fractions (200 – 630 µm; 630 – 5000 µm) fragments were the most abundant. In their study, Klien et al. (2015) acknowledged that the low abundance of fibers in sediment samples was surprising to them, as they had expected higher numbers. No explanations or assumptions for this anomaly were provided in their study.

The study by W. Li et al. (2022), along the Yangtze River and its tributaries, determined microplastic flakes were the most abundant category. Their definition of flake particles originates from discarded woven plastic bags, derived from PP, which degrade in the environment. Most of the literature reviewed did not have flake as a standalone category. In contrast, Turner et al. (2022) determined microplastic flakes could be one of the most abundant types of microplastic in the ocean; however, their definition of ‘flake’ is weathered particles derived from paint, which does not align with the ‘flake’ definition from W. Li et al. (2022). The different definitions of ‘flake’ highlight the need for standardized categories of microplastics.

Microplastic Concentration Compared to Other Studies

Referring to Table 5, the concentration of microplastic contamination in Bow River sediments was lower compared to other studies. There are several possible reasons for this result including: differences in methodological approaches, such as the lower limit of detection;

sediment size fractioning; use of other solutions for density separation (e.g., $ZnCl_2$, NaI, etc.); and/or using advanced analytical steps (e.g., FT-IR, Raman Spectroscopy) to confirm polymer particles, which was beyond the scope of this thesis. Ultimately, without a standardized method of sample collection, sample processing, and laboratory analysis, comparing the relative abundance of microplastics across other studies can only be expressed qualitatively.

Another likely reason the Bow River has a lower concentration of microplastic pollution is the City's population. Calgary has a population of 1.4 million people, which is far lower when compared to population densities and the number of urban centers along the Rhine, Main, Ganga, Yangtze, and Beijiang Rivers. It is reasonable to assume that plastic pollution is more prevalent in these rivers because the prevalence of microplastics in freshwater environments is closely linked to areas with high population densities and proximity to urban centers (Lebreton et al., 2017; Wong et al., 2020). In contrast, the River Kelvin flows through a portion of Glasgow, which has a comparatively lower population than Calgary, but has a higher abundance of microplastics. Interpreting this result at face value could be misleading because the study used only one sample location along an inside meander bend (depositional area) which does not provide a robust basis for comparison.

These studies can also be used to identify trends in microplastic distribution in riverine shoreline sediments. Similar to my study, the results from the Main and Rhine Rivers (Klien et. al, 2015) and the Tijuana River (Piñon-Colin et al., 2024) indicate a pattern of increased microplastic concentration further downstream. Conversely, no discernable pattern of increased microplastic concentration in sediments further downstream in the Beijiang River (J. Wang et al., 2017) and Ganga River (Sarkar et al., 2019). These studies suggest microplastic concentrations in littoral sediments may be largely based on localized anthropogenic activity, land use, sediment

grain size, and depositional patterns in riverine systems. In contrast, for the Yangtze River and its tributaries (W. Li et al., 2022), microplastic abundance was higher in the upper reaches (average 418 particles/kg) compared to the lower reaches (average 252 particles/kg), with shape playing a significant role. Interestingly, more foam particles were present downstream compared to upstream, consistent with my findings. The variability observed in these data underscores the need for further research into the distribution of microplastics along rivers.

Based on my data, Bow River littoral sediments are contaminated with microplastic, but not to the extent recorded in other rivers in more densely populated areas. As previously mentioned, there are limitations when comparing data to other studies. Nonetheless, despite challenges and limitations, the comparative data remains valuable for understanding the presence of microplastics in Bow River sediments relative to other regions.

Chapter 6: Conclusion and Recommendations

My study begins to reveal the state of macro- and microplastic contamination in shoreline sediments of the Bow River in Calgary. Microplastics were extracted from all sediment samples, leading to the conclusion that microplastic contamination in Bow River littoral sediments is pervasive throughout the City of Calgary. The dominant type of microplastics contamination in this study was fiber, coming from clothes washing and discharge from WWTPs, stormwater outfalls, and most likely atmospheric fallout, which aligns with the current consensus in the scientific literature on the sources of microfibers in aquatic environments. Macroplastics were also found at all sites, with single-use plastics (e.g., Styrofoam, food wrappers, cigarettes, caps, straws) being the most observed categories.

Despite the small sample size, my study revealed a statistically significant difference in both macro- and microplastic concentrations between downstream and upstream sites. This finding aligns with the results in similar studies in the scientific literature; however, the concentration of plastics along a shoreline is heavily influenced by variables such as flow conditions, riverine morphology, seasonality, polymer type and shape; therefore, while there is a general tendency for plastic concentrations to increase downstream in rivers, it is not universally observed.

I strongly encourage further research on macro- and microplastics in freshwater environments. Rivers and freshwater sources remain fundamental to our collective economic, social, and environmental wellbeing. Most people on Earth live close to water and are directly dependent on access to freshwater sources for survival (Kummu et al., 2011). In fact, it is estimated that rivers and their floodplains are home to 2.7 billion people (Best, 2019). Our freshwater sources are increasingly being stressed from a variety of factors, plastic pollution

being one of the them. Compared to macroplastics, more research is needed on microplastics because despite their ubiquitous presence we still know little of the effect and harm microplastics are doing to humans, wildlife, and our collective watersheds.

There are several recommendations I would make for future research on microplastic pollution in shoreline sediments.

We need to develop reliable and repeatable methods for sampling including a standardized method of extracting (isolating) microplastics from sediments. Different sampling regimes, extraction procedures, and analytical techniques will continue to lead to inconsistencies in microplastic concentrations and characterizations. For example, in a recent systematic review, Weiss et al. (2021) determined there were overestimates (by orders of magnitude) in the flux of microplastics from rivers to the ocean. They identified inappropriate mixing of data collected using different sampling techniques as one reason for the discrepancy. We need to ensure sampling, laboratory, and analytical methods are standardized so comparative studies can help us form a fully integrated picture of microplastic pollution across different spatial and temporal scales.

To help close the gaps in our collective understanding of microplastic pollution, further research is also needed to:

- Qualify and quantify microplastics in freshwater environments including investigations into point and non-point microplastic pollution, microplastic distribution in watersheds, and temporal patterns of microplastic deposition in sediments.
- Evaluate the effects of microplastic ingestion, particularly at smaller size fractions, on freshwater biota and human health, including chronic low-dose exposure, to assess the potential long-term consequences.

- The interactions and transfer pathways for microplastics between organisms, potential biomagnification effects, and the leaching of toxins from microplastics into the environment. These pathways are not well understood.

By addressing these research recommendations we can gain a deeper understanding of microplastic pollution, its ecological and human health impacts, and begin to develop effective mitigation strategies. Ultimately, there is no simple, across-the-board solution to abate plastic and microplastic pollution entirely and we cannot simply ban all plastics from production. The applications for plastic materials are too valuable and entrenched in our society and no scalable replacement exists that would allow society to maintain the *status quo*.

There are ways for the harmful effects of plastic pollution to be managed or mitigated, to some extent, with no additional research. For example, improving the quality of terrestrial waste management will encourage large-scale reductions in the loss of plastic to the environment during their life cycle (Wong et al., 2020). Also, improving wastewater treatment systems, including the ability of those systems to capture microplastics before they enter treatment facilities, will reduce the volume of plastics discharged to rivers via effluent. For stormwater systems, strategic engineering and infrastructure solutions such as rain gardens, bioretention ponds, and engineered wetlands, have been shown to be effective at removing macro- and microplastics (Stang et al., 2022). Any effective, large-scale solution to abate plastic pollution requiring infrastructure changes or engineered methods will require political will and substantial capital inputs, which may not be feasible for some countries.

Another way to reduce plastic pollution may be in the emerging field of biodegradable polymers and substitute products, although at present, most biodegradable plastics do not degrade quickly enough and therefore are not a perfect solution. Plastic polymer substitutes are

still in development and have yet to scale to the global demand for plastic products so at present, they are neither economically nor practically feasible. This field of research, however, is still in its infancy and the potential for biodegradable polymers is promising, particularly as a substitute for single-use plastics.

Arguably, the most effective solutions for reducing environmental plastic pollution are curbing production, demand, and usage, which can be achieved by using policy and regulatory instruments. The global plastic recycling environment was upended in 2017, when China implemented a ban on the import of certain types of waste products, including certain types of plastic, to their recycling facilities (W. Wang et al., 2019). These facilities handled nearly half the world's recyclable waste for the previous 25 years (W. Wang et al., 2019). There is now added urgency to reduce and manage plastic waste in developing and developed countries, including Canada. Prior to the ban, Canada exported nearly 40% of its plastic waste to China (Huang et al., 2020). Since the ban has been enforced, Canada has been scrambling to deal with increasing volumes of plastic waste with limited financial and governmental resources to manage the situation (Diggle & Walker, 2020). This causes a burden on municipalities and taxpayers and increases the volume of plastic debris entering landfills and leaking into the environment. Starting in 2018, Canada shifted the export of plastic waste from China toward other regions such as Malaysia, Vietnam, and Thailand; however, these regions have, or are in the process of, implementing similar bans on imported waste products (C. Wang et al., 2020). The onus is on Canada to reduce its plastic waste problem.

One regulatory pathway for Canada to reduce its waste is the extended producer responsibility (EPR) strategy. An EPR strategy is intended for producers and manufacturers to consider the end-of-life financial responsibility for their products, such as single use plastics and

packaging, to incentivize those producers to create more sustainable packaging, reduce packaging, or shift to reuse/refill. In doing so, they generate less waste. Canada has had a national EPR strategy since 2009 (Diggle & Walker, 2020), although by 2024, it has been put into practice in only five provinces: British Columbia, Saskatchewan, Manitoba, Ontario and Quebec. Recently, the Canadian federal government committed to move towards zero plastic waste using several strategies including an implemented, harmonized EPR policy across Canada. Internationally, negotiations on plastic pollution are ongoing through the United Nations Intergovernmental Negotiating Committee, which aims to develop a legally binding instrument to manage plastic pollution. Any regulatory pathways aimed at reducing plastic waste will need to be enforceable and measurable to be effective. Reducing macroplastic pollution at the source will undoubtably lead to a reduction in microplastic pollution.

In closing, this thesis provided a baseline for future monitoring of macro- and microplastic contamination in the Bow River. There are many opportunities for new and much-needed research on macro- and microplastic contamination not only in the Bow River but globally. The entrenchment of plastics in our society and the forecast of much higher volumes of plastic production in the future means that plastic contamination will only increase; therefore, more research efforts are needed to expand our understanding of this issue as we strive towards a world devoid of plastic waste.

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