# A Big Tiny Problem: Flows of Primary Microplastics in Canada

by

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# Author’s Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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# Abstract

Microplastics are ubiquitous across ecosystems, posing chemical and physical risks to wildlife and human health. To effectively monitor and manage the microplastics problem, a baseline of the mass of primary microplastics produced, used, and discarded is measured. The literature lacks a comprehensive assessment of the different flows of microplastics, especially in North America. This study quantifies the mass of seven microplastic types based on an analysis of the plastics during production and use stages and their flows to six final compartments, including surface water, soil, agricultural soil, roadside, landfill, and incineration for Canada.

A material flow analysis (MFA) was conducted for Canada in 2016, using data gathered from academic literature, government reports, and industry reports. The results showcase that in Canada, 60,100 tonnes of microplastics were unintentionally and intentionally released into Canada's environment in 2016. The top three sources of microplastics were tire wear particles (TWP), releasing a total of 51,300 tonnes; paint fragments releasing 8,000 tonnes and microfibres, releasing 913 tonnes. The flows responsible for the most microplastic emissions were direct release to roadsides, contributing 41,400 tonnes; direct release to soils, contributing 3,470 tonnes; and direct release to surface waters releasing 5,090 tonnes. Roadsides and surface waters received the most microplastics, totalling to 46,000 and 9,120 tonnes, respectively. Regarding the polymer composition of microplastics released, rubber and poly (methyl methacrylate), found in TWPs and paints, respectively, are estimated to be deposited the most commonly in the environment.

This work is the first to map the flows of primary microplastics in Canada and distinguish between polymer types. The findings of the MFA allow stakeholders to identify significant points of microplastic leakage and inform legislative, infrastructural, or technological solutions to reduce emissions upstream and downstream of the plastics lifecycle. In addition, the results of this study inform actions concerning the Canada-wide Strategy on Zero Plastic Waste and Action Plan, suggest the need for product design change, and provide insight into waste diversion and recovery. The results of this study are a foundational piece for environmental fate models to be conducted. This investigation provides a baseline to compare preventive scenarios to reduce microplastic generation in Canada.

Keywords: Microplastics, Material Flow Analysis, Waste Management, Circular Economy

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# List of Abbreviations

CaPSA: Canada Plastics Science Agenda

CCME: Canadian Council of Ministers of the Environment

CEPA: Canadian Environmental Protection Act

CIAC: Chemistry Industry Association of Canada

CPP: Canada Plastics Pact

CSO: Combined Sewage Overflow

CSS: Combined Sewage System

DIY: Do-it-yourself

DP: Domestic Production

ECCC: Environment Canada Climate Change

EPS: Expanded Polystyrene

GHG: Greenhouse Gas

GESAMP: Joint Group of Experts on the Scientific Aspects of Marine Environmental Pollution

HCBD: Hexabromocyclododecane

LLDPE: Linear low-density polyethylene

MFA: Material Flow Analysis

MP: Microplastics

NI: Net Import

PCCP: Personal Care Cosmetic Products

PE: Polyethylene

PET: Polyethylene terephthalate

PP: Polypropylene

PMMA: Polymethyl methacrylate

SDW: Stormwater

SSS: Separate Sewage System

SW: Surface Water

TWP: Tire Wear Particle

WWTP: Wastewater Treatment Plant

XPS: Extruded polystyrene

# Chapter 1: Introduction

## 1.1 The Plastic Problem

Plastic pollution is recognized as a concerning anthropogenic issue across aquatic and terrestrial environments globally (Jambeck et al., 2015; Koelmans et al., 2017; MacLeod et al., 2021). Recent studies have indicated that if the current trajectory of plastic waste generation, inadequate waste management and lack of clean-up initiatives are continued, there could be up to 90 million metric tonnes of plastic waste entering aquatic systems by 2030 (Borrelle et al., 2020; Lau et al., 2020; MacLeod et al., 2021). Plastics have various applications and have been integrated into many aspects of society. However, the lack of foresight regarding plastic waste management has caused it to be one of the leading drivers of global contamination, from reaching remote mountain lakes to the deep abyss of the ocean (Jaibachi et al., 2018; Woodall et al., 2014; Wright & Kelly, 2017).

Although plastics pose a pressing environmental challenge, they have also provided many benefits, such as reducing food waste, improved sanitation, and lowered carbon footprints compared to alternative materials such as glass, wood, and aluminum (Klemeš et al., 2021). The increase in plastic use within society is partly due to its favourable characteristics: low cost, lightweight, versatility, and durability (Geyer et al., 2017). Since the creation of plastic polymers ~100 years ago, they have been modified and manufactured to ensure their stability and longevity in crucial sectors such as automotive, textile, electronic industries, and construction (Millican & Agarwal, 2021). Nevertheless, the increase in production and applications is also why plastics are found across various ecosystems. When plastics leak into the environment, they do not chemically degrade but instead break down into small plastic particles called microplastics (Thompson et al., 2004; Cole et al., 2011; Andrary et al., 2011).

Microplastics are classified into two categories: primary and secondary microplastics (Barnes et al., 2009; GESAMP, 2015). Primary refers to microplastics that are intentionally produced for commercial use (pre-production pellets, microbeads) or through the wear and tear of products (textiles, tires). In contrast, secondary microplastics are created unintentionally through the breakdown of larger pieces of plastic due to biological, chemical, or physical processes (GESAMP, 2015). Due to microplastics' ability to stem from diverse product types and travel through many different flows, it has many source points, such as stormwater drains, washing machines, and wastewater treatment plants (WWTP) (Horton et al., 2017; Al-Jaibachi et al., 2018; Conley et al., 2019). Given their small size and ubiquity, there is widespread ingestion of microplastics by animals globally, including humans (Wright et al., 2013; Gall & Thompson, 2014; Wright & Kelly, 2015; Guzzetti et al., 2018). The consequences of microplastic ingestion to organisms can be grave, such as internal/external injuries, pseudo satiety sensation (starvation) and physiological stress, reduction in fertility and lower survival rate of offspring (Cole et al., 2013; Wright et al., 2013; Caron et al., 2016). However, in humans, there is still significant uncertainty regarding their potential physical and chemical consequences (Wright & Kelly, 2017; Smith et al., 2018; Chen et al., 2020).

There is growing concern among society about the risks microplastics pose, as they have been found in drinking water, food and within the human body (De-la-Torre, 2020; Koelmans et al., 2019; Prata et al., 2020; Rainieri & Barranco 2019). Microplastics can act as vectors for other chemical contaminants and have been shown to adsorb these from the environment and release them to biota (Fred-Ahmadu et al., 2020; Hartmann et al., 2017). In addition, researchers have found that there is potential for bioaccumulation and magnification of microplastics across food chains, although more research is necessary to understand which organisms are at the highest risk and the ecological implications this may have (Miller et al., 2020). Due to the various unknown health and environmental impacts of microplastic pollution, many local and global actions have ensued (global treaty other nation's initiatives on microplastics). Moreso, solving the microplastics issue is complex and relatively less straightforward than the issue of macroplastics (plastics larger than 5 cm). Microplastics signify economic inefficiency, as there is currently no technology to recycle or upcycle them, thereby further contributing to the problematic linear plastics economy (Calero et al., 2021; Chen et al., 2022).

Canada (along with the United States) has the highest per capita consumption of plastics in North America (Deloitte & ChemInfo Services, 2019; PlasticsEurope, 2019). Canada's plastics industry is linked to generating 87% of plastic waste, where the majority ends up in landfills and a small portion leaks into the environment, fueling a mainly linear plastics economy (Deloitte & ChemInfo Services, 2019). To address this issue, the government announced a ban on manufacturing, importing, and selling toiletries used to exfoliate or cleanse that contain plastic microbeads (Government of Canada, 2017a). Following was a single-use plastics ban which came into effect in 2022 – banning six single-use plastic products deemed harmful (Government of Canada 2017b). While efforts to prevent plastic waste and microplastics are being made, it is important to measure the stocks and flows of microplastics to target and manage plastic waste disposal and leakage. This is especially true for North America, as researchers have estimated that more microplastics are released into the environment than macroplastics (Boucher & Friot, 2017).

Over the past decade, there has been an exponential increase in microplastic research within the literature (Zhou et al., 2021). However, most of the research is focused on identifying the potential sources, fate, and effects of microplastics and their associated chemicals across ecosystems (Thompson et al., 2004; Cole et al., 2011; Jambeck et al., 2015; Geyer et al., 2017; Horton et al., 2017; Horton & Dixon, 2018; Akdogan & Guven, 2019). In contrast, relatively few studies have quantified the flows of microplastics from the point of production to their release into different economies and subsequent environments (Jambeck et al., 2015; Geyer et al., 2017; Boucher & Friot, 2017; Ryberg et al., 2019; Kawecki & Nowack, 2019; Sieber et al., 2020; Liu & Nowack, 2022). To effectively tackle the plastic pollution issue, research regarding the baseline amounts of produced, consumed, and end-of-life microplastics should be measured to effectively monitor and manage the progress in reducing plastic waste. A methodology to do so is known as stock and flow accounting. It aims to quantify the mass of materials across key stages of a material's life cycle in a defined system (Brunner & Rechberger, 2004). With stock and flow accounting, patterns of material consumption, model waste generation, and evaluation of recycling potential can be measured (Brunner & Rechberger, 2004; Wang et al., 2021). The primary method for stock and flow accounting is known as material flow analysis (MFA). It is applied widely to examine anthropogenic stocks and flows of various materials, most commonly metals (Brunner & Rechberger, 2004).

### 1.2 Rationale

The foundation for understanding the release of plastics into the environment is through gathering information on the production, use and disposal of the various distinct products and their associated polymers (Ryberg et al., 2019). Thus far, there has been increasing literature regarding the flows and stocks of macroplastics; however, literature regarding microplastic flows from production to their potential flows into different environmental compartments remains limited (Boucher & Friot, 2017; Ryberg et al., 2019; Kawecki & Nowack, 2019). By assessing the flows of microplastics, key entry points can be identified and targeted mitigation strategies can be implemented and measured over time. As microplastics are present in much higher count than macroplastics within the environment due to their small size, it is important to understand their various flows across Canada's economic and environmental systems to advise on prevention methods and recommend preventative and end-of-pipe solutions (Lebreton et al., 2018)

### 1.3 Research Aims & Objectives

This study aims to quantify the current flows of seven microplastic types (pre-production pellets, microfibres, microbeads, tire wear particles, paint fragments, film (from agricultural plastic mulch), and foam (from construction)). The flows of each microplastic type are then modelled to six final outputs, which are defined as final compartments for this study. Final compartments are divided into two categories: environmental and technological. Environmental compartments include surface waters (oceans and freshwaters) and terrestrial systems (soils, agricultural soils), while technological compartments include landfill, incinerators, and roadsides. The results inform relevant stakeholders of the current source points and processes microplastics are potentially entering and exiting the system boundary of Canada. Through a material flow analysis (MFA), emissions of microplastics across their production and use stages are quantified. The polymer types associated with the seven types of microplastics above were considered. Different sources of microplastics in the environment are quantified and compared. This research aims to answer the following:

What are the flows of primary microplastics in Canada?

To answer this question, this study addresses the following objectives:

1. Quantify the flows of primary microplastics in Canada.
2. Identify the hotspot sources, flows, and sinks of microplastics in Canada.
3. Provide recommendations for microplastic mitigation strategies and how they can be managed in a circular economy.

### 1.4 Thesis Structure

Following this section, Chapter 2 includes a literature review which will outline the current literature investigating microplastics, their sources and receiving environmental compartments, the few pieces of literature which have investigated their stocks and flows thus far and lastly, policies put in place in Canada to manage the plastics and microplastics problem. Chapter 3 will introduce the methodology used to quantify and track emissions of microplastics: MFA. Background on this methodology will be provided, along with variations of this method used by various researchers to quantify the stocks and flows of microplastics. The methods section will also outline how the data was calculated to quantify the stocks and flows of microplastics and the approach used to model the microplastics across Canada. Next is Chapter 4, the results section. Here the findings of this study are presented as Sankey diagrams for each microplastic type. The dominant polymer types will also be shown in each environmental compartment. Lastly, a qualitative discussion regarding the uncertainty associated with the results is discussed to provide transparency and disclose the assumptions made to calculate the microplastic flows. Lastly, Chapter 5 discusses the study results critically analyzed and compared to other MFA studies conducted in different countries and regions globally. In addition, the implications of the plastics policies already implemented within Canada will be addressed and what it means to achieve a circular economy.

# Chapter 2: Literature Review

Microplastics have become a pressing global contaminant in the Anthropocene, posing a threat to humans and organisms across terrestrial, freshwater, and marine systems (de Souza Machado et al., 2018; Guzzetti et al., 2018; Chen et al., 2020; Earn et al., 2021). There is limited literature regarding the mass or count of microplastics which flow through global and regional economic systems and, thereby, how much is released to the subsequent environmental compartments (Boucher & Friot, 2017; Ryberg et al., 2019; Wang et al., 2021; Cholewinski et al., 2022). In Canada, an MFA of macroplastics was conducted by Environment Canada Climate Change (ECCC), but there remains a lack of investigation for microplastics (Deloitte & ChemInfo Services, 2019). Therefore, in efforts to effectively address the research objectives and provide further insight into Canada's problematic plastics economy, this literature review will cover the following: benefits and consequences of our increasing use of plastics, an overview of macro- and microplastics, the ecological impacts, potential sources, the current focus of the literature on microplastics, the different categories of microplastics and their origins, current Canadian plastics policy and integration of plastics circular, investigation of plastics utilizing material flow analysis, and need for a material flow analysis for microplastics.

## 2.1 Background on Plastics and the 'Throw-away' Culture

Plastics are integrated into our everyday lives, so much so that this era has been coined the "Age of Plastics" (Avio et al., 2017). Plastics are synthetic or semi-synthetic materials derived from petroleum or produced from renewable raw materials (Alauddin et al., 1995; Rennie, 1999). Currently, the market is still largely comprised of plastics from petroleum, while those derived from biobased monomers remain to be around 2% of the market (Chinthapalli et al., 2019). There are three main classifications of plastics, thermoplastics, thermosets and elastomers. Thermoplastics are linear polymer chains with no crosslinks, making a recyclable polymeric material (Garcia & Robertson, 2017; Scheutz et al., 2019). Given the lack of chemical bonding, this type of plastic can be heated many times and reshaped without risking the integrity of the material (Garcia & Robertson, 2017; Scheutz et al., 2019).

On the other hand, thermoset plastics are highly crossed-linked covalent network polymers and provide extensive mechanical properties, chemical and heat resistance, and dimensional stability

(Morales, 2018; Morici & Dintcheva, 2022). However, due to their chemical bonding, they cannot be reprocessed or recycled further (Jin et al., 2019; Rennie, 1999). Elastomers are chemically wide-meshed crosslinked plastics that are elastic and are rubber-like polymers from lower temperatures up to a set decomposition temperature (Eyerer, 2010; Kutz, 2015). The most common polymers in Canada are polyethylene, polyvinyl chloride, and polypropene, which are used for food packaging, product packaging, buildings, vehicles, electronics, and elastomers, as shown in Table 1 (Deloitte & ChemInfo Services, 2019; Figure 1). The creation of various plastic polymers has provided immense opportunities for plastics to be utilized in any industry.

Table 1: Plastic classifications, polymer types, and examples of some main applications and uses within society (Choong et al., 2021; Deloitte & ChemInfo Services, 2019).

| **Classification of plastics** | **Plastic polymer types** | **Major application** |
|---|---|---|
| Thermoplastics | Polyethylene | Packaging |
| | Polypropylene | Automotive parts, Textiles, Packaging, Electronics |
| | Polystyrene | Packaging |
| | Polyester | Textiles |
| | Polyamide/Nylon | Textiles, Automotive parts |
| | Polyethylene terephthalate | Packaging |
| | Polyvinyl chloride | Building and Construction |
| | Acrylonitrile butadiene styrene | Electronics |
| | Ethylene-vinyl acetate | Others |
| | Polyurethane | Building & Construction |
| | Polymethyl methacrylate | Paints |
| Thermoset | Epoxy | Paints, Building & Construction, Automotive parts |
| | Phenolic | Building & Construction, Automotive parts |

| | | |
|---|---|---|
| Elastomer | Styrene butadiene rubber | Tires |
| | Butadiene rubber | |
| | Isoprene rubber | |
| | Natural rubber | |

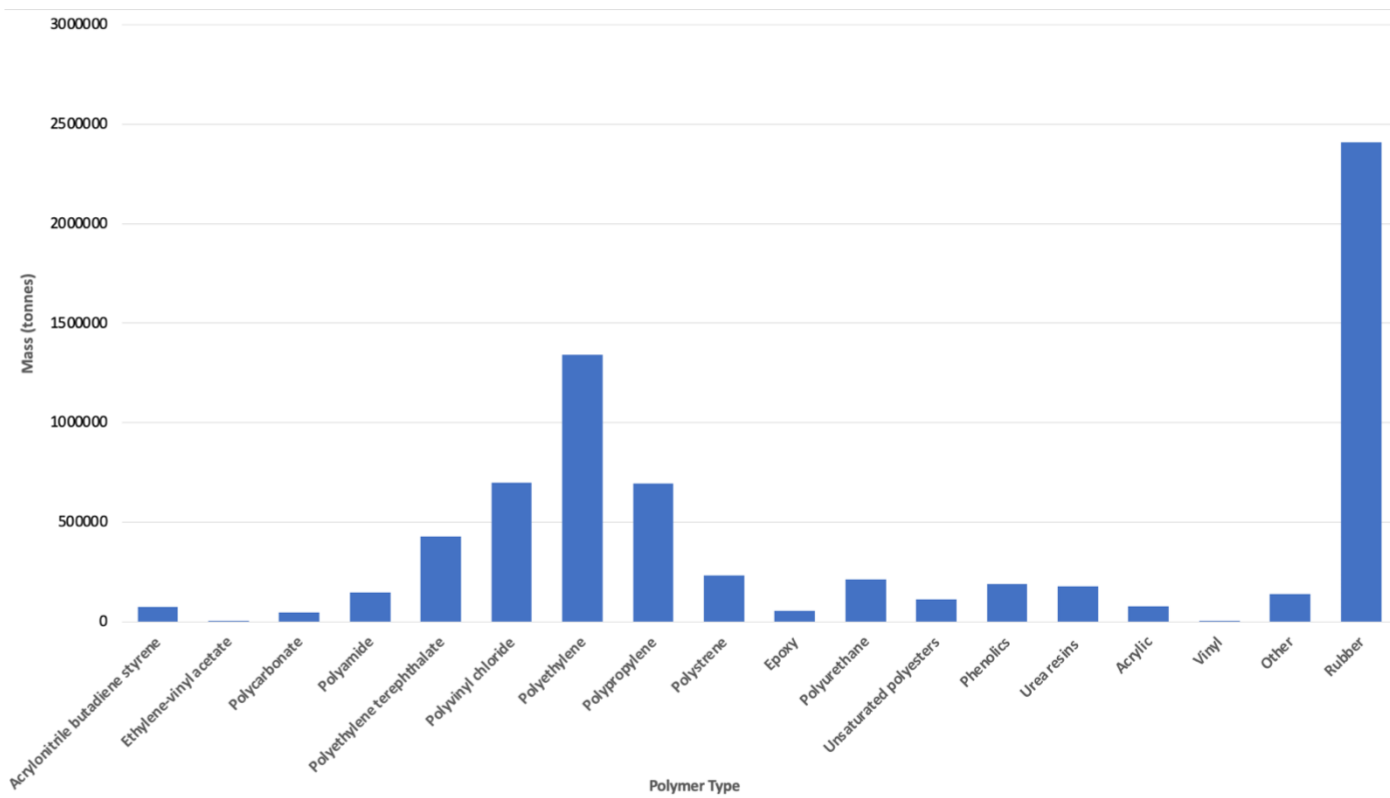


Figure 1: Canadian plastic consumption in 2016 (Deloitte & ChemInfo Services, 2019; Dillon Consulting & Oakdene Hollins, 2019).

Plastic production and waste have grown exponentially due to plastics' versatility, low cost, lightweight, and durability, making it a $600 billion global industry. (Jambeck et al., 2015; Geyer et al., 2017; Klemeš et al., 2021). These characteristics have made plastics a significant commodity in most industries, and they have become irreplaceable in some sectors, such as the healthcare sector (Hervé

Millet et al., 2019; Klemeš et al., 2021). Plastics can result in a decrease in greenhouse gas (GHG) emissions in comparison to other common materials like glass in applications like packaging (Miller et al., 2020; Boucher et al., 2020). For example, the entire lifecycle of a PET bottle generates 41% fewer GHG emissions than a glass bottle (Cornstar, 2010). In addition, researchers have found that glass has a higher marine plastic footprint than plastic packaging due to the weight of the materials during transportation. The weight of glass causes more tire wear particles to be generated than when plastic packaging is the material being transported (Boucher et al., 2020).

While it is clear plastics have provided many benefits, these benefits have also caused society to become dependent on plastics in almost all contexts. The dependence on plastics has manifested with single-use packaging as consumers demand longer product shelf life, freshness, and convenience (Mcdermott, 2016; Hervé Millet et al., 2019; Miller, 2020). This created a "throw-away culture" and a collective low-value regard for these plastic products. The result of this culture has overwhelmed the waste management systems of developed and developing countries as there is a lack of facilities and technology to deal with the heterogeneous types of plastic products (Mcdermott, 2016; Browning et al., 2021). Developed countries are dependent on developing countries to take 100,000s tonnes of plastic waste per year as it is much more cost-effective (Browning et al., 2021). This causes significant economic and social harm in developing countries with little to no infrastructure or waste management system in place, where they often illegally dump plastic waste into the environment or burn it (Barnes et al., 2009; Jambeck et al., 2015; Borrelle et al., 2020; Browning et al., 2021).

## 2.2 Characterization of Plastics: Macro vs Micro

The first significant discovery of plastic pollution was the Great Pacific Garbage Patch by Captain Charles Moore in the mid-1990s, where he and his crew spotted millions of plastic pieces and products floating in the North Pacific Subtropical Gyre (Moore et al., 2001). At the time, it was described as an island of plastic, filled with abandoned fishing gear, plastic bottles, bags, sheets of film, packaging, and many miscellaneous single-use plastics (Moore et al., 2001). These macroplastics were thought to be the most common form of plastic pollution floating around in the ocean and harming wildlife (Sigler, 2014). The reason plastics can infiltrate a wide range of ecosystems, and what makes up 94% of the 1.8 trillion pieces of plastic in the "garbage patch," is because of small particles called microplastics (Moore et al., 2001; Thompson et al., 2004). Microplastics are small synthetic polymer particles which form an important component of micro-sized litter. Due to their small size, microplastics are widespread in aquatic and terrestrial systems and organisms (Barnes et al., 2009; Cole et al., 2011; Eriksen et al., 2014).

Considering the size, there are a total of four categorizations of plastic waste found in the environment as defined by the Joint Group of Experts on the Scientific Aspects of Marine Environmental Pollution (GESAMP) (Table 2) (GESAMP, 2016). The two categories related to microplastics are described in Table 2; however, the definition of microplastics remains debated (Frias & Nash, 2019; Kooi & Koelmans, 2019). To date, there is still no international agreement on a single definition which is all-inclusive of its various properties and criteria that describe what a microplastic is (Frias & Nash, 2019; Hartmann et al., 2019). However, most of the literature and researchers have agreed that microplastics are plastic particles smaller than 5mm in size (Thompson et al., 2004; Arthur et al., 2009; Cole et al., 2011; GESAMP, 2015, 2019). Frias and Nash (2019) proposed the definition of microplastics as any synthetic solid particle or polymeric matrix ranging from 1um to 5mm with primary or secondary manufacturing origin and is insoluble in water. In contrast, Hartmann et al. (2019) proposed a framework to define and categorize what plastic debris entails. Hartmann et al. (2019) suggested including criteria such as polymer composition, solid state, and solubility, thereby allowing sources such as rubber and paint to be concluded in the scope of (micro) plastics' research. Therefore, this current study confers with their definition of plastic debris and, thus, what constituents as a microplastic.

Table 2: Summary of size definitions of plastic litter (GESAMP, 2016).

| **Size categories of plastic litter** | **Micro <5mm** | **Meso** | **Macro** | **Mega** |
|---|---|---|---|---|
| **Diameter** | <5mm | <2.5cm | <1m | >1m |
| **Source** | Primary and secondary microplastics (fibres, tires, pre-production pellets, personal care products, | Direct or indirect inputs of plastic waste, including fragmentation of larger plastic items | Lost items from maritime activities or rivers | Abandoned gear, or catastrophic events |

| | | | | |
|---|---|---|---|---|
| | degradation of meso-, macro- or megaplastics) | | | |

The two categories of microplastics help determine their potential source and where to implement policy or technological interventions (McDevitt et al., 2017). Primary microplastics are produced intentionally for commercial use, such as in personal care products, pre-production pellets or industrial abrasives or created unintentionally from the wear and tear of products like clothing or tires (Table 3) (Duis & Coors, 2016; GESAMP, 2019). Conversely, secondary microplastics are produced unintentionally through the break-up of larger plastics like meso, macro or mega categories into smaller fragments via biological, physical, or chemical processes such as UV radiation, high temperatures or wave activity in marine environments (Rochman et al., 2018 GESAMP, 2019). These environmental factors cause chemical changes in plastic products, making them more brittle and more suspectable to fragmentation (Andrary, 2011; Duis & Coors, 2016; Horton et al., 2017; GESAMP, 2019). However, microplastic pollution is often unintentional, such as when microbeads in personal care products are washed down drains after use. (Horton et al., 2018). Most loss from primary and secondary microplastics remains to be unintentional (Boucher & Friot, 2017).

Due to their small size and varying densities (depending on the type of polymer/material the microplastic is comprised of, i.e., polyethylene, polypropylene, polystyrene, rubber etc.), microplastics can be transported to various parts of the environment and have been found in remote locations such as polar regions and remote mountain lakes (Hidalgo-Ruz et al., 2012; Obbard et al., 2014; Rocha-Santos & Duarte, 2015; Woodall et al., 2014). Given their ubiquity, managing microplastics becomes much more complex than larger plastics (Eriksen et al., 2018). However, through standardized characterization, microplastics found in the environment can be tied back to the original source or intermediate sources, thereby revealing the potential flows (Horton et al., 2017; Boucher & Friot, 2017).

## 2.3 Sustainability Impacts

The increased production of plastics has caused many negative ecological implications across marine, freshwater, and terrestrial ecosystems (Eriksen et al., 2013; Lusher, 2015; de Souza Machado et al., 2018). Microplastics are highly resistant to biodegradation and can persist in the environment for hundreds of years (Yoshida et al., 2016). Their size allows them to be bioavailable to organisms throughout the food web and present at all depths of aquatic systems and, more recently, discovered across terrestrial terrain (de Souza Machado et al., 2018; He et al., 2019). Microplastics have been documented in zooplankton, phytoplankton fishes, turtles, birds, and mammals – either through direct ingestion or indirect ingestion through mechanisms like bioaccumulation (Wright et al., 2013; Caron et al., 2016; Cole et al., 2013; Munno et al., 2021; Sherlock et al., 2022). Once ingested, microplastics can cause many adverse effects to organisms, such as internal/external injuries, blockages of the gut track which result in pseudo satiety sensation and physiological stress, altered feeding and retarding of growth, reduction in fertility, fecundity, and survival rate of progeny (Bucci et al., 2020; Guzzetti et al., 2018; Rochman et al., 2013; Wright et al., 2013).

Microplastics are not just a material but a diverse contaminant suite (Rochman et al., 2019). During plastic production, chemical additives such as plasticizers, colourants reinforcements, fillers, flame retardants, and stabilizers are added to the polymers (Lithner et al., 2011; Rochman et al., 2019; Padervand et al., 2020). When microplastics are released into the environment, they can be exposed to a cocktail of chemicals, such as persistent organic pollutants and trace metals and can act as a vector of contaminants to aquatic and terrestrial organisms when ingested (Rochman, 2015; Rochman et al., 2019).

However, the impact of microplastics is not limited to the environment, as their effects have social and economic impacts (Chaudhry & Sachdeva, 2021; Fuschi et al., 2022). Examples can include negative consequences to the fishing industry, which represented the second-largest food export in Canada in 2015 (DFO, 2020). Studies have shown plastic waste, and microplastics can cause a loss in tourism revenues and a risk to human health through inhalation and consumption (Seltenrich, 2015; Van Cauwenberghe et al., 2015; Barboza et al., 2018; Chen et al., 2020; DFO, 2020). There is a lack of research regarding the potential health effects, but there is a call for more research regarding this area (Wright & Kelly, 2017; Campanale et al., 2020). The release of microplastics in the Global South may pose a larger risk due to the lack of wastewater treatment plants (WWTP) and water filtration (Eerkes-Medrano et al., 2019; Iyare et al., 2020). In addition, its effect on public spheres has much to do with the media, which amplifies the perception of suggested risks with microplastics using research that has unverifiable results due to inconsistent methodology and unreliable data (Kramm et al., 2018). The media also takes away from the larger concern of microplastics, which is the persistent organic pollutants and leachates associated with them which cause chemical or physical harm to human health, as it has already been shown to do so in aquatic and terrestrial organisms (de Sá et al., 2018; Earn et al., 2020; Rahman et al., 2021).

## 2.4 Prevalent Sources and Sinks of Microplastics

The sources of microplastics are diverse, but 80% are suspected to originate from land-based sources (Jambeck et al., 2015; Wang et al., 2016). Over the past decade, researchers have focused on seven types of microplastics: synthetic fibres, city dust (includes losses from synthetic soles of footwear, cooking utensils, household dust, city dust, artificial turfs, harbours and marinas, building coatings, and blasting from abrasives), tire dust, marine coating, road marking paint, personal care products, and plastic pre-production pellets (Boucher & Friot, 2017; Ryberg et al., 2019). Their potential flows to the environment, their chemical and physical implications on organisms and interactions with other emerging contaminants have been investigated globally, nationally, and for specific watersheds (Wright et al., 2013; Boucher & Friot, 2017; Bucci et al., 2019; Vieira et al., 2021; Paruta et al., 2022).

The scope of this current study includes the following primary microplastics: pre-production pellets, microfibres, microbeads, tire wear particles, paint fragments, film (agricultural plastic film), and foam (construction). Each is discussed below. These types of microplastics were investigated due to their prevalence in environmental sampling and the rate of their generation given their sources. Here, the rate of generation refers to primary microplastics created through the wear and tear of a product (i.e., tires, clothing, construction foam, agricultural plastic mulch, paints on roads and buildings).

For example, clothing today is composed mainly of synthetic fibres derived from a select set of polymers (Belzagui et al., 2020; Gavigan et al., 2020). Paint is used for many applications and sectors such as architectural, marine, road marking, general industrial and automotive. Pre-production pellets are used to mould and create thermoplastic products (Karlsson et al., 2018). Each of these microplastics is prevalent in day-to-day operations.

Microplastics generated from construction foam and agricultural plastic film have not received as much attention as other microplastic types. Recently in literature, these sources have been identified as significant, where films from agricultural plastic use are assumed to accumulate in agricultural soils for decades and are thus a direct source of microplastics (Tian et al., 2022). Construction foam (specifically

EPS and XPS) is increasingly receiving attention due to the chemical associated with the material, and as construction continues, especially in urban settings, its prevalence is increasing in aquatic and terrestrial environments (Gao et al., 2023). Microplastic sources that were not included in this current study but have been identified as significant contributors to the environment are marine paints, industry abrasives, cleaning products, waste processing facilities, artificial turf, municipal solid waste, dryer vents, and secondary microplastics (Kawecki & Nowack, 2018; Suzuki et al.,2022; Paruta et al., 2022; Shi et al., 2023). Their omission is due to a lack of data regarding their flows, rate of generation, or lack of Canadian input data, which is further discussed in section 5.4.

### 2.4.1. Pre-production Pellets

Colloquially known as nurdles, pre-production pellets are the raw materials for thermoplastic products and are, on average, smaller than 5mm in size (Boucher & Friot, 2017; Rochman et al., 2018; Peano et al.., 2020). Pellets are considered a primary microplastic and can also be produced as flakes or powders (OSPAR, 2018). As pellets are the basis for many commonly used plastic products, they are available in various polymer types and consequently have been found in aquatic and terrestrial environments (Karlsson et al., 2018).

Pellets are produced in many colours, such as translucent, grey, white, yellowish white to amber, black, blue, and red dependent on the product they will be transformed into (OSPAR, 2018). Pellets are usually regular in shape; however, fine particulate powders have more irregular shapes and sizes (Karlsson et al., 2018). Plastic production in Canada primarily occurs within a handful of major petrochemical industries. Canadian plastic production facilities produce plastic polymers using natural gas. These plastic producers are mainly Dow Chemical Company, NOVA Chemicals Corporation, DuPont, Imperial Oil Limited and BASF Canada, which carry out the primary production of plastic pellets (Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). The points of pellet loss can occur during production, although a low percentage (~0.4%), and during transportation and storage (Kawecki & Nowack, 2018). Once the pellets are produced, they are stored in large silos until the time for transportation (OSPAR, 2018; Policy Manager Chemistry Industry Association Canada, personal communication August 9th, 2022). Next, they are sold to converters or processers, which are facilities that transform pellets by remelting, moulding, or extruding pellets into plastic products like water bottles, food packaging, construction materials etc., or they are first transported to distribution centres, then to the converter or processor facilities (OSPAR, 2018). Within Canada, there are approximately 2,555 plastic manufacturing facilities, and 45.2% of these facilities are in Ontario (Government of Canada, 2019).

Pellets are transported through a few different avenues from the manufacturers to the processors and converters within Canada (Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). They can be transported by train, which is most common, and each cart on the train has taps where the pellets are then released from the cart to the processor or converter facilities (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). They are transferred from the cart to the processing facility through a long hose, which connects directly from the train cart to the silo which is on site of the plastic converter/processor (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). Once the pellets have entered the silo, they have entered the manufacturing process of being transformed into a new plastic product and therefore have little to no chance of spilling (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022).

Pellets can be transported from manufacturers or distribution centres to the processors in trucks, where the pellets are in large plastic bags or octane boxes (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). Pellets are then unloaded again through pipes which connect directly from the truck to the silo. Pellet spills mainly occur during the loading, unloading and transport phase from the manufacturer, distribution centre and process/converter centres (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). Spills are most often accidental due as once pellets are spilt, they can no longer be used to manufacture plastic products and therefore cannot be sold, so manufacturers and processors tend to maximize their pellet transfer efficiency (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022).

Plastic producers across Canada are beginning to include their pellet spills within their sustainability reports to be transparent and address public scrutiny (NOVA Chemicals, 2021; Dupont, 2022; Dow Chemical, 2022). In addition, this is a requirement to be a part of the Operation Clean Sweep (OCS) Initiative, an international prevention-focused program for environmental stewardship designed to help every plastic resin manufacturing and handling operation implement good housekeeping and resin containment practices (CIAC, 2023). Members of the OSC must commit to the responsible management of plastic resin throughout all aspects of their company's business (CIAC, 2023). Once the pellets have been split, their flows to the environment are often directly to land, into stormwater drains and eventually to surface waters (Ballent et al., 2016; OSPAR, 2018).

### 2.4.2 Microbeads

There are two main sources of microbeads: personal care cosmetic products (PCCP) and abrasive cleaning agents. Microbeads from personal care and cosmetic products include products like face scrubs, toothpaste, nail polish, shower gels and other 'rinse-off' products (Government of Canada, 2015; Bashir et al., 2021). The most common ingredients used to create cosmetic microbeads are PE, PET, PP, and PMMA (Gouin et al., 2015; An et al., 2020). Microbeads are added to aid in skin exfoliation and cleansing and provide a smooth and silky feeling (UNEP, 2015). The term rinse-off insinuates that these cosmetic products have microbeads intended to be rinsed down the drain after use (Government of Canada, 2015). Manufacturers have chosen microbeads over other natural products due to their smoother exfoliation, low cost, lower density prevention of clogging of drains, compatibility with other products in the cosmetic product and no damage to the container they are packaged in (Sherrington et al., 2016; Anagnosti et al., 2021). Once washed down into household drains, they end up in WWTPs or directly into watersheds depending on the type of sewage system in place or if there is a sewage system at all (Rochman et al., 2015; Government of Canada, 2015). If the wastewater is directed to a WWTP, a small fraction of microbeads stays in the effluent and gets back into aquatic systems (Rochman et al., 2018). In contrast, the remaining portion ends up in sludge produced by the WWTPs, which can be applied to land and agricultural soils and/or sent to landfill or incinerated (Government of Canada 2015; Xanthos & Walker, 2017).

For the past decade, there have been worldwide bans and phase-out efforts for microbeads (Anagnosti et al., 2021). Thus far, South Korea, Sweden, Taiwan, France, Ireland, Italy, New Zealand, the United Kingdom, the United States and Canada have banned the manufacturing and importing of products with microbeads (Xanthos & Walker, 2017; Anagnosti et al., 2021).

The second and less investigated source of microbeads is abrasive cleaning agents (Verschoor et al., 2016). These products include included in laundry detergents, dishwasher detergents, bathroom cleaners, bleaching cleaners, and surface cleaners (Sherrington, 2016). Microbeads are often an effective component in these types of cleaning agents as they are softer than other materials used, such as minerals and therefore can be used to clean delicate surfaces that consumers do not want to be scratched or have a removing factor to paint or other materials but can still clean (Wu et al., 2017). For this study, only the emissions of microbeads from PCCP are focused on due to the lack of data regarding the amount of microbeads in abrasive products in Canada.

### 2.4.3 Microfibres

Microfibres are one of the best-studied sources of microplastics within the literature and have been identified in nearly every study conducted, including those in remote locations (Boucher & Friot, 2017; Carr, 2017; Gavigan et al., 2020; Athey & Erdle, 2022; Adams et al., 2021; Padha et al., 2022). They are tiny threads (<5mm) of most commonly polyester, nylon acrylic and other synthetic textiles that are released from the process of production, laundering and wear (Henry et al., 2018). Most microfibres found in aquatic systems have been reported from the wear and tear of textiles from indoor and outdoor laundering, washing of clothing, textile manufacturing, or direct dumping of waste garments into rivers (Almroth et al., 2018; Figure 4).

Specifically, the mechanical abrasion or physical stress applied at different stages of the textiles, i.e., usage day to day, wear and tear, and laundry, are the main cause of microfibre shedding (Boucher & Friot, 2017; Ryberg et al., 2019; Belzagui et al., 2019). In addition, the deposition of microfibres to the environment has been attributed to atmospheric transport, where urban centres tend to act as the primary source due to the shedding and wear and tear of clothing (Österlund et al., 2023). Microfibres can be transported at great distances via long-range transport (Bergmann et al., 2019; Zhang et al., 2020).

However, given the lack of investigation and standardization of methodology to measure the atmospheric transport of microfibres and their emissions to indoor and outdoor air, this pathway has been excluded from this current study (Österlund et al., 2023).

Clothing in Canada is mostly imported; $12 billion of clothing were imported into Canada, while only $2 million was domestically produced in 2016 (Deloitte & ChemInfo Services, 2019). During the manufacturing stage of textiles which is less of a concern in Canada, the higher physical agitation, mechanical stress, and chemical treatments provided in the wet processing stages, like bleaching, dyeing, and finishing, release the most microfibres to wastewater (Ramasamy & Subramanian, 2021). Although there is awareness that the shedding of microfibres occurs during manufacturing, few studies investigate the rate at which this occurs (Sherrington et al., 2016). The Nature Conservancy calculated that 0.19% of pre-consumer microfibres are released and directly enter waterways surrounding manufacturing processes, while other studies have looked at microfibre emissions from entire textile parks that exist in China (Zhou et al., 2020; Belzagui et al., 2020; The Nature Conservancy, 2021). Estimates of microfibre emissions from one textile mill in these textile parks can be up to 430 billion particles per day (Belzagui et al., 2020). However, textile mills across different regions may have different production and water quality assurance rates; therefore, more work is necessary to estimate microfibre emissions from production (Chan et al., 2021; Kamble & Behera, 2021).

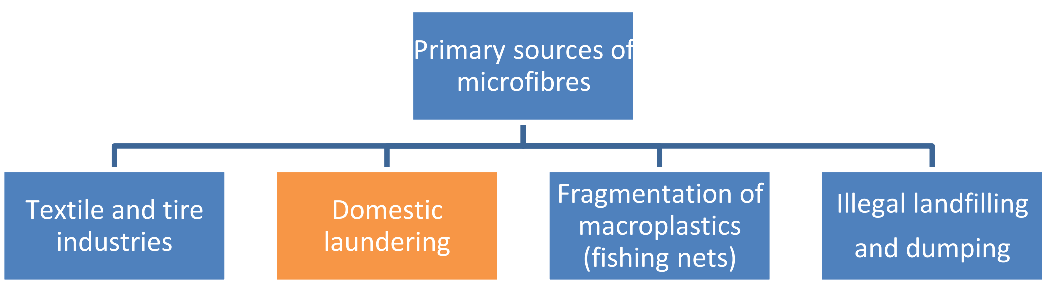


Figure 2: Primary sources of microfibres to the environment adapted from Carr (2017).

The composition of clothing imported, produced, and therefore purchased by Canadians is mainly plastic based. It is estimated that 63% of clothing available to Canadians is made of synthetic fibres (Cheminfo Services, 2022). Polyester is the most popular polymer used, followed by polyamide, polypropylene, acrylic and elastane (Cheminfo Services, 2022; Figure

5). The expansion of fashion, specifically fast fashion, would not have been possible without synthetic fibres, as PE is low-cost and readily available (Mishra et al., 2019).

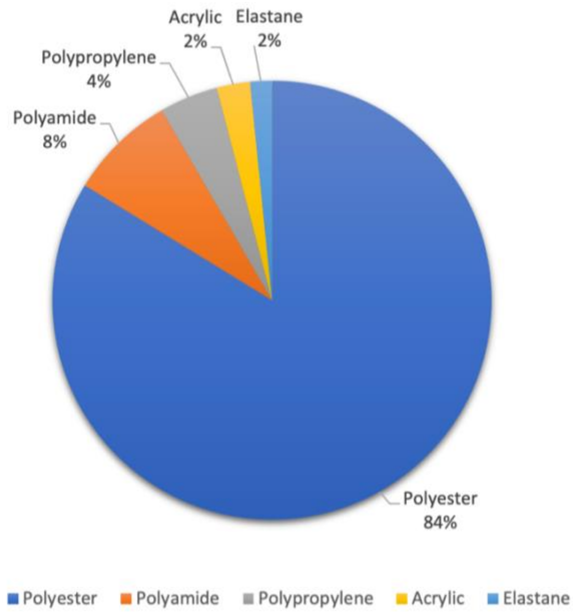


Figure 3: Types of synthetic polymers used in apparel consumed and worn by Canadians (Cheminfo, 2022).

Various studies have investigated microfibre release from washing machines. Researchers have investigated dryers and the production of microfibres within a lint catcher and their release indoors when cleaning the lint tray, and, more significantly, their release to outdoor air through the vent which connects to outside the home (De Falco et al., 2019; O'Brien et al., 2020; Kapp & Miller, 2020; Yousef et al., 2021). This is particularly interesting to the Canadian context as 81% of Canadians own washing and dryer machines (Statistics of Canada, 2009). Studies have found that microfibre release is promoted by longer drying times; however, more investigation is needed to assess the detachment rate of different dryer machines, like what has been done for the different types of washing machines (Belzagui et al., 2022).

The release of microfibres from washing machines into the environment is straightforward and follows the same pathway as microbeads. In Canada, most washing machines are connected to the wastewater system (separate or combined sewage system). This wastewater system then leads to a WWTP, and depending on the level of treatment at that facility, the effluent or sludge will have different concentrations of microfibres (Athey & Erdle, 2022; Gavigan et al., 2019). Compared to the microplastics described below, the leakage points of microfibres from washing machines are simple, and strategies for mitigation are clear (Athey & Erdle, 2022).

### 2.4.4 Paints

Paint consists of binders, fillers, pigments, solvents, and water (Verschoor et al., 2016). Within the binder, resin polymers are utilized and combined with one or more additives and therefore share compositional similarities with microplastics (Gaylarde et al., 2021). Paint products contain intentionally added microplastics either as dispersed polymer particles in water-based paint for film formation or as microbeads/microfibres for enhancing paint traits in water and solvent-based paints, where paints can be composed of up to 35% of plastics (Hann et al., 2018; Faber et al., 2021).

Microplastics from paints have been investigated for architectural, marine, road marking, general industrial, automotive, and industrial wood sectors (Paruta et al., 2022; Gaylarde et al., 2021). Paint fragments tend to originate from the removal or deterioration of surface applications of paint. Paint can contain a multitude of different polymers, such as acrylonitrile butadiene styrene, polyethylene, polyvinyl chloride, polyethylene terephthalate, polystyrene, acrylic, alkyd, polyurethane and epoxy (Turner, 2021; Faber et al., 2021). For this study, the sources of paints previously identified are residential buildings within the architectural sector and road marking paints, as they have been presented as the most significant source of paint in the literature and have the most data regarding their emissions (Paruta et al., 2022; Turner, 2021).

Road marking paints are another prevalent source of paint fragments; however, their release and degradation to the environment depend on several factors such as polymer composition, location on the road, traffic levels and climate (Lassen et al., 2015). Road marking application in Canada is controlled by municipalities and provincial governments, where each province has its specified regulations regarding road paint application for environmental health and safety (LaFrentz, 2023; ECCC, 2023). Paint fragments from road markings are assumed to have a similar fate to TWPs, therefore, have identical flows, such as direct entry to surface waters, soils, roadsides or stormwater drains (Verschoor et al., 2016).

Building paints are used for decorative purposes but also to prevent corrosion and fouling, thereby allowing construction materials to have a longer lifespan (Gaylarde et al., 2021). Building paint fragments occur due to removal processes and through degradation from UV-irradiation (Gaylarde et al., 2021; Paruta et al., 2022; Haave & Henriksen, 2022). Paints in residential buildings are applied on both interior and exterior surfaces. Studies state that between 71-73% of paints sold are for interior purposes, while 27-29% are sold for exterior purposes (Verschoor et al., 2016; Paruta et al., 2022). This is due to different regions' regulatory requirements for health and safety to repaint interior spaces and for decorative purposes. However, exterior surfaces are more likely to deteriorate quickly due to exposure to environmental factors, while emissions from interior paint are assumed to be contained in the building (Verschoor et al., 2016; Paruta et al., 2022). Any paint particles from interior paint that ends up in wastewater and, therefore, possibly in terrestrial and aquatic environments are due to improper disposal methods of paint by washing the paintbrush in the sink or dumping the remaining paint down the sink (Verschoor et al., 2016; Paruta et al., 2022).

### 2.4.5 Agricultural Plastic Mulch Film

Agricultural plastic mulch film (which will now be referred to as plastic film) is vital for agricultural practices globally (Wang et al., 2018; Gao et al., 2019). It is used in large quantities to improve crop yield as applying the film across the fields allows for water conservation, increased soil temperature, protection against soil erosion, weed prevention, and pathogen and pesticides (Kapanen et al., 2008 Steinmetz et al., 2016; Li et al., 2023). However, it has also been identified as the primary source of microplastic accumulation in agroecosystems (Huang et al., 2020). In Canada, LLDPE film is used on the fields and often is predrilled with holes at specific intervals for plants (Clean Farms, 2019). It is primarily used for vegetable and strawberry production and left for one to three seasons before disposal (Clean Farms, 2019). When it is time for disposal, it is difficult for farmers to collect the film as they are thin (between 0.01-0.05mm) and, therefore, often just left on the fields intentionally or unintentionally (Kasirajan & Ngouajio, 2012; Astner et al., 2019). The frequency of this occurring in Canada has yet to be studied, and therefore, it is assumed that the farmers can collect much of the plastic film, and only a portion that remains on the agricultural soil is used for the next season (Deloitte & ChemInfo Services, 2019).

Research regarding microplastics from agricultural plastic film is mainly from regions in China, as they account for 90% of the world's production and use of plastic film (Steinmetz et al., 2016; Huang et al., 2020). However, these investigations have identified the significance of this source and how it may contribute to a greater accumulation of microplastics in soils than in oceans (Liu et al., 2014; De Falco et al., 2019; Huang et al., 2020). Within Canada, crop production occurs mostly in Western Canada (Saskatchewan, Alberta and Manitoba) (Hein, 2020). Given its application nationwide, plastic film may significantly contribute to direct microplastic pollution in Canadian agricultural fields (Agriculture and Agri-Food Canada, 2023).

### 2.4.6 Construction Foam

Within the construction industry, expanded polystyrene (EPS) foam and extruded polystyrene (XPS) foam are used extensively as they are light, inexpensive and provide high thermal insulating properties for roofs, walls, and floors (Yucel et al., 2003; Charbonnet et al., 2020). During the installation and demolition processes with buildings, EPS and XPS foam are often mismanaged and lost directly into the environment through activities such as shaving to ensure the insulation foam board is even and improve adhesive bonding to other materials or through general demolition processes (Gao et al., 2023). However, there needs to be more investigation into EPS and XPS flows throughout their lifecycle (Minet et al., 2021; Figure 4). There are many uninvestigated flows that may contribute to the release of EPS and XPS throughout their lifecycle to the environment. Due to the lack of understanding, in this study, only the direct inputs of EPS and XPS foam to the environment through production, installation and demolition processes are considered.

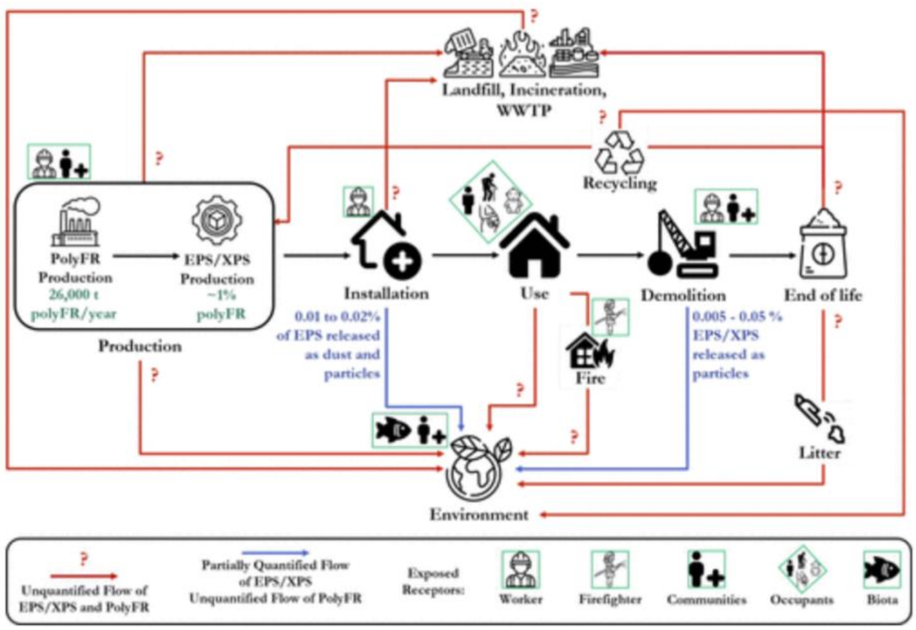


Figure 4: Figure created by Minet et al. (2021), which depicts the flows of PolyFR in EPS and XPS where the red flows indicate the unknown quantities of PolyFR and, therefore, EPS and XPS particles throughout their lifecycle.

### 2.4.7 Tire Wear Particles (TWP)

Tires are a complex elastic rubber product and are the only component of a vehicle that comes into contact with the ground (Sommer et al., 2018). The structure of a tire includes tread, tire shoulder, tire side, belt ply, cord ply, inner lining, and steel chafer (Grigoratos & Martini, 2014; Baensch-Baltruschat et al., 2021). Tire tread wear occurs due to the contact between tires and road surface during acceleration and braking when driving (Wagner et al., 2018; Sieber et al., 2020; Baensch-Baltruschat et al., 2020; Österlund et al., 2023). Heat and friction alter the original chemical composition of the wear particles, and material from the road surface is incorporated into the tire wear particles along with other particulate traffic-related emissions (Sommer et al., 2018; Panko et al., 2013). These aggregates are defined as tire and road wear particles. The generation of TWRP can potentially have stronger toxicity than the initial tire material (Camatini et al., 2001; Tian et al., 2021). However, the additional mass collected from roads is difficult to quantify as it varies depending on each particle generated. Therefore, this current study only calculates the mass of TWPs generated from abrasion of tire tread material and roads. The composition of TWPs is mainly synthetic and natural rubber, filler, process oils, and additives (Grigoratos & Martini, 2014; Wagner et al., 2018; Dillon Consulting Limited and Oakdene Hollins, 2021). TWP emissions come as both airborne and non-airborne emissions, where the majority of TWPs are non-airborne and are deposited on the road or roadside (Panko et al., 2013; Unice et al., 2018; Wagner et al., 2018; Mian et al., 2022). While most traffic-related pollution focuses on particulate and gaseous pollutants emitted from the exhaust, particulars from non-exhaust emissions remain to be understudied; this includes research regarding TWP emissions (Wik & Dave, 2009; Kocher et al., 2010; Kole et al., 2017; Sommer et al., 2018; Wagner et al., 2018; Fussell et al., 2022).

The main flows of TWPs to the environment include stormwater, effluent from WWTPs and water runoff (Wagner et al., 2018; Sieber et al., 2020; Osterlund et al., 2023). While research regarding the ecological impacts of TWPs has been ongoing for 30 years, there remain few studies investigating their mass flows or total emissions into the environment (Mennekes & Nowack, 2022). Thus far, two studies have conducted MFAs of TWPs in Switzerland and Austria. Sieber et al. (2020) provide more granularity in their study by identifying the specific receiving environmental compartments of TWPs (i.e., surface water, soils, or roadsides), whereas Prenner et al. (2021) simply encapsulate the loss of TWP to the environment. In addition, Sieber et al. (2020) considered the portion of TWPs that ends up in different sinks depending on the type of road they are deposited on, i.e., highway, urban or rural road. Urban roads tend to have high traffic, and more efforts for road cleaning to reduce air pollution; therefore, a fraction is removed from the roads and sent to landfill or incineration (Boller et al., 2006; Vogelsang et al., 2019; City of Toronto, 2015). In comparison, Prenner et al. (2021) just applied a general estimate of TWP deposition to surface waters, stormwater drains or roadside across all road types. Most TWPs are deposited in surface waters directly, sent to stormwater drains or accumulated on the roadside (Peano et al.., 2020; Unice et al., 2018; Wagner et al., 2018). Depending on the sewage system that is connected to the stormwater drain, TWPs can either end up directly in surface waters, sent to a stormwater management facility or sent to WWTP facilities which are further described below (Statistics Canada, 2016). In contrast, Baensch-Baltruschat et al. (2021) calculated TWP emissions for Germany utilizing another method utilizing emission factors of the mass of generated tire wear per vehicle kilometre, vehicle type and total annual mileage. Like Sieber et al. (2020), they mapped out the flows of TWPs across different road types, sewage systems and environmental compartments and estimated a total of 98,400 tonnes/year of TWPs being generated (Baensch-Baltruschat et al., 2021).

## 2.5 Wastewater Management Systems in Canada

As described above, a significant pathway for microplastics to end up in environmental compartments such as surface water, agricultural soil or other final compartments like landfill or incineration are stormwater drains or wastewater (Kawecki & Nowack, 2018; Hale et al., 2020; Hoseini & Bond, 2022). The type of wastewater management plays an important role in determining what fraction of microplastics end up in which environmental compartments compared to which are disposed of in landfill or incinerators (CCME, 2012; Mohajerani & Karabatak, 2020).

In Canada, three types of WWTPs exist, primary, secondary, and tertiary (ECCC, 2017). Primary WWTPs remove some contents, such as rags, sticks, floatable, grit and grease, which may inhibit any downstream processes (Iyare et al., 2020). Next, fine screening, grit and grease removal may occur along with skimming, primary settlement of the wastewater, and removal of the organic matter or suspended solids through chemical processes (Statistics Canada, 2020; Iyare et al., 2020). Secondary WWTPs further treat wastewater post the primary treatment through biological processes, including activated sludge processes, biofiltration, trickling filters, and solid contact tanks (Iyare et al., 2020). These processes further reduce the residual suspended solids and dissolved solids as they can be entrapped by solid flocs, sedimentation in secondary clarifiers or ingestion by microorganisms (Iyare et al., 2020; Statistics Canada, 2020). Tertiary WWTPs remove specific inorganic or organic substances such as solids, nutrients or contaminants after the secondary treatment (Government of Canada, 2020; Iyare et al., 2020). Various technologies, such as depth, surface and membrane filtration or dissolved air floatation, are used to achieve this. As the treatment level of the WWTP increases, fewer suspended solids and pollutants are released in the effluent; however, more is accumulated in the sewage sludge/ biosolid (Iyare et al., 2020; Parashar & Hait, 2023). It is estimated 86% of the Canadian population is served by WWTP systems, where primary WWTPs comprise 26.4%, secondary WWTPs comprise 48.7%, and tertiary WWTPs comprise 24.8% (Government of Canada, 2020). Not all the wastewater collected in the municipal systems is sent to a WWTP. A survey conducted by ECCC found 4.4% is lost directly to surface waters through combined sewage overflows (CSO) or to the environment without treatment purposefully (Government of Canada, 2020).

Wastewater systems in Canada are made up of separate and combined sewage systems (ECCC, 2017). Combined sewage systems combine sanitary sewage and stormwater collection such as rainfall, snowmelt, and wastewater from residential and businesses. These mainly exist in older parts of Canada as these were the initial systems put in place, some were built nearly a century ago (City of Toronto, 2023). However, they can overflow directly into creeks, rivers and other surface waters during heavy rainfall or increased input into the combined sewage systems. This prevents overflows into properties, public spaces, or sewage treatment plants (City of Toronto, 2023). In separate sewage systems, sanitary sewage from businesses and residences is separate, where sanitary sewage is sent directly to a WWTP, and stormwater drains are connected to storm sewers. Some storm sewers are connected to stormwater management facilities where pollutants are filtered out through different mechanisms in place, such as bioretention, filtration, retention ponds, and wetlands (Stang et al., 2022).

Numerous studies have investigated the presence of microplastics in the effluent and sludge of WWTPs, their different efficacies of removal and how they act as both a source and pathway for microplastics (Murphy et al., 2016; Mintenig et al., 2017; Iyare et al., 2020; Gao et al., 2023). However, their methods of testing the occurrence, characteristics, and retention of microplastics are varied, and the general theme of studying microplastics has no standardized method for testing these objectives. Therefore, averages for the efficacies of removals have been used based on what is accepted in the literature (Iyare et al., 2020). The occurrence and removal of microplastics from the stormwater drain and stormwater management facilities are much less investigated and often neglected in comparison to WWTPs (Symth et al., 2021). As per a literature review conducted by Stang et al. (2022), 16 studies have investigated various removal methods by stormwater management facilities and their efficacies of microplastic removal, one of which was conducted in Vaughan, Ontario, Canada (Smyth et al., 2021). In Canada, stormwater management facilities consist of ponds and wetlands as per the inventory of publicly owned stormwater assets (Infrastructure Canada, 2016). They were used to predict microplastics' efficacy and retention rate in the Canadian context (Infrastructure Canada, 2016).

## 2.6 Current Canadian Plastics Policy and Integration of Plastics Circular Economy

Studies have shown that countries in the Global North produce the most plastic waste per capita (Law et al., 2020). To address the plastic waste generated in Canada, the government has launched several initiatives to tackle the plastics and microplastic pollution issue.

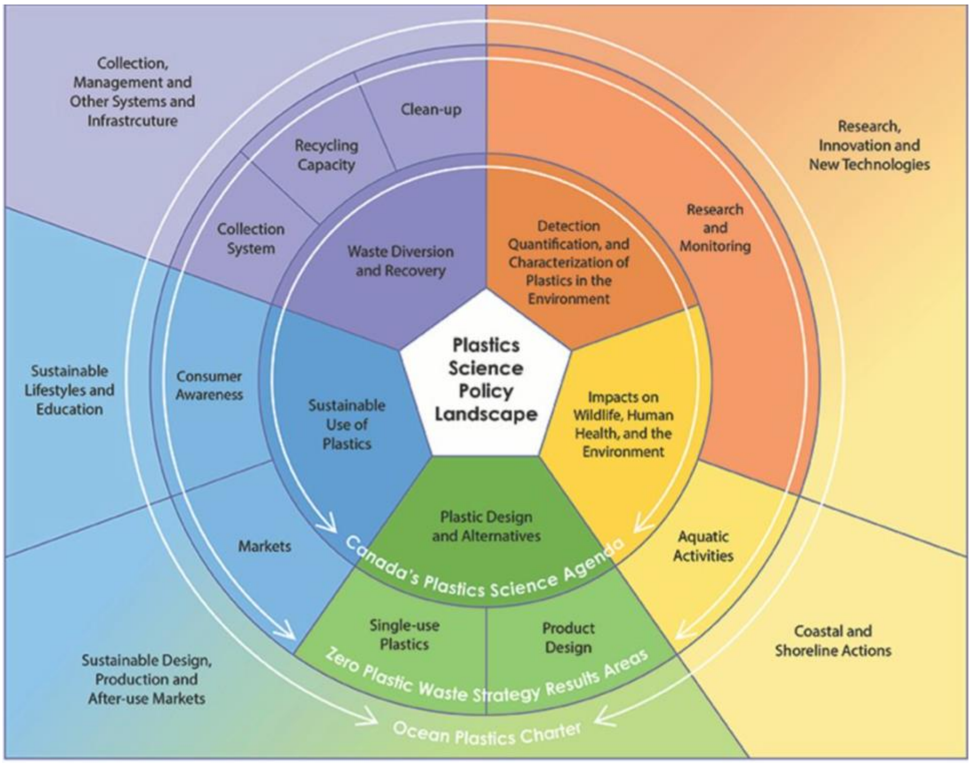


Figure 5: Canada’s plastic policy landscape (ECCC, 2021)

Some of these initiatives include regulatory policies to improve waste management, promote recycling, and reduce or prohibit the use of certain plastic products (Figure 2) (Government of Canada, 2018). The first initiative was the Microbeads in Toiletries Regulation which was announced in 2018. The regulation stated a ban on the manufacturing, importing, and selling of toiletries used to exfoliate or cleanse that contain plastic microbeads (Government of Canada, 2017a). Following this, Canada, as part of the G7 presidency, became a part of the Ocean Plastics Charter in 2018, which commits to targets toward preventing plastic waste and its flows to the environment (Figure 7) (Government of Canada, 2021). It aims to unite governments, businesses, and civil society organizations to support its objectives and commit to a more sustainable approach to managing plastic and plastic waste through many grants, funds, and signing international legally binding agreements (Government of Canada, 2021). Building off the Charter, the Canadian Council of Ministers of the Environment (CCME) developed a national approach to tackling plastic waste in Canada called the *Canada-wide Strategy on Zero Plastic Waste* in 2018 (Government of Canada, 2021). The strategy was developed in two phases to set out tangible actions and clear timelines to better prevent, reduce, reuse, recover, capture, and clean up plastic waste and pollution in Canada (Government of Canada, 2021). The strategy takes a circular economic and lifecycle approach to plastics, providing a framework for action (Government of Canada, 2021). However, given that plastics research is fragmented and inconsistent, Canada launched the *Canada Plastics Science Agenda* (CaPSA) to address these challenges by identifying current and future research across various disciplines (Government of Canada, 2021). A major target of (CaPSA) is to align the priorities of plastics research in Canada. For microplastics research, CaPSA outlines the need for established uniform procedures for collecting, measuring, and assessing the impact of microplastics (Government of Canada, 2019). The goal is to gain insight into their environmental toxicity, potential harm to humans, and effects on human health. This includes regular monitoring efforts being extended to encompass previously overlooked areas such as soil (Government of Canada, 2019).

The results of these initiatives include adding plastic-manufactured items to the Canadian Environmental Protection Act Toxic Substances List (Schedule 1) to prevent and control plastic waste throughout its life cycle (Government of Canada, 2018). Once added to Schedule 1, Canada announced a ban on six single-use plastic products deemed harmful: plastic bags, cutlery, food service ware, ring carriers, stir sticks and straws; it has been in effect since 2022 (Government of Canada, 2017b). The Plastics Innovation Challenge was another output offering grant opportunities for small to medium-sized businesses to fund cost-effective technologies to tackle tire wear particles, which has just entered Phase 2 (Innovative, Science and Economic Development Canada, 2022).

In tandem with the action plan presented by the Government of Canada, a separate initiative operated by a national charity, The Natural Step Canada, launched the Canada Plastics Pact (CPP). CPP brings together businesses, NGOs, and the government to rethink how to design, use, and reuse plastic packaging across Canada (Canada Plastics Pact, 2023). Stakeholders who have joined this initiative agreed to achieve four targets regarding reducing plastic packaging and waste by 2025, which aligns with the national effort to reduce plastic waste to zero by 2030 (Canada Plastics Pact, 2023).

While these targets will provide valuable information for product development and improve material recovery, it is important to identify where the leakages are occurring, as well as the quantity and rate of release. Canada has already completed an assessment on macroplastics, as described in section 3.2.3; however, in this assessment, microplastics were only mentioned once when discussing the estimated leakage of macroplastic to the environment (Deloitte & ChemInfo Services, 2019; Government of Canada, 2019). To gain a complete picture of the plastics problem, an assessment of the flows and stocks of microplastics is also necessary.

## 2.7 MFA and Plastics

### 2.7.1 Application of MFA to Macro and Microplastics

There is emerging literature on the flows and stocks of macroplastics, which have been conducted at national and global levels (Kawecki et al., 2018; Mutha et al., 2006; Sieber et al., 2020; Wang et al., 2021; Prenner et al., 2021; Liu & Nowack, 2022; Hoseini & Bond, 2022). Wang et al. (2021) state that 28 peer-review studies have been published on this topic covering plastic emissions in several countries. The studies often only focused on specific plastics (i.e., one or two polymer types) or only a few plastic products (i.e., consumer packaging, automotive, construction). For example, Heller et al. (2020) conducted an MFA for plastic products across the United States, only identifying seven major plastics products, while ten products were identified as 'unknown.' Some significant markets excluded are agriculture, textiles and medicine, significant sources of single-use plastic emissions (Bergmann et al., 2015).

On the other hand, Luan et al. (2021) conducted an MFA of plastics in China. They included these missing plastic products but failed to provide the wide range of popular polymers that make up plastic waste, such as low-density polyethylene, linear low-density polyethylene, and high-density polyethylene flowing their system, providing a more holistic and accurate representation of the major plastic products being used across their system and their compositions in the end-of-life treatment. Across these 28 studies on the MFA of plastics, when addressing the environmental leakage of plastics, they fail to include all environmental compartments in which plastics may leak, which include aquatic, terrestrial, biota and air and instead clump them into one category called "environment."

Given that microplastic particles are in much higher number and presence than macroplastics within the environment due to their ubiquity, it is vital to understand their various source points, intermediary stocks, different processes and inevitable sinks across their life cycle. This ensures the prevention of 1) loss of primary microplastics and 2) risk of degradation of macroplastic products into microplastics across the plastics value chain (Lebreton et al., 2018). A better understanding of microplastic flows can help to identify areas of inefficiency and potential leakage to natural systems, thereby informing effective solutions across the plastics value chain to reduce or even eliminate the potential for microplastic pollution (Wang et al., 2021).

### 2.7.2 MFA and Microplastics

Few studies have conducted an MFA on primary microplastics with varying coverage and resolution of spatial and temporal scales, polymers, microplastic applications, and life cycle stages (Cholewinski et al., 2022). Boucher & Friot (2017) were the first to quantify primary microplastic leakage into oceans and showcase that primary microplastic constitutes a significant source of global plastic pollution and provide optimistic, central, and pessimistic estimations of microplastic emissions from seven potential sources based on their applications. They identified the significant sources of primary microplastics as tire dust, synthetic textiles, pre-production pellets, city dust (synthetic soles of footwear, synthetic cooking utensils, household dust, artificial turfs, harbours and marina, building coatings, and detergents), road markings, personal care products and road markings (Boucher & Friot, 2017). Their analysis found that 15-31% of all plastics (by mass) in the oceans may originate from primary microplastics. This assessment only utilized publicly available data to produce their estimates and only considered flows of microplastics to oceans. Ryberg et al. (2019) also estimated global microplastic emissions but widened their geographic scope to include more regions and included microplastic emissions into the terrestrial environment, but overall were faced with the same limitations as Boucher & Friot (2017). Both studies stated the lack of data regarding the loss of primary microplastics during production, processing and transport and were limited to using three sources to predict these emissions. Boucher & Friot (2017) state that their models could have been more accurate by using fee-based proprietary data on regional plastic quantities and improving underlying regional assumptions on behaviours. These studies lacked detail on specific release processes as the data is scarce and subject to high variability.

In contrast, Kawecki and Nowack (2018) conducted an MFA of macro- and microplastics in Switzerland and were the first to publish one of the most comprehensive and complete assessments thus far. This study includes most plastic products and terrestrial, aquatic, and air environmental compartments in their models. In addition, it provides a complete map of microplastic flows and stocks for each common polymer used in Switzerland and their flows to the aquatic and terrestrial compartments. This largely had to do with data availability, as the authors state there was high data availability in the region (Kawecki & Nowack, 2018).

Building from the dataset and method proposed by Kawecki & Nowack (2018), another global assessment of microplastic distribution was conducted; and is the most comprehensive assessment thus far (Hoseini & Bond). Hosieni & Bond (2022) take the initial environmental sinks from Kawecki & Nowack (2018) and expand them to include subcompartments for the freshwater, ocean, and terrestrial environments such as shoreline, surface water, water column, the floor of aquatic beds, industrial soil agricultural soil and residential soil (Hoseini & Bond, 2022). This is the most comprehensive study thus far, including a detailed breakdown of the potential final fates of microplastics. In addition, dynamic MFAs of specific polymers and their associated microplastics have appeared using this same methodology for China and Switzerland however do not provide the same level of granularity and final fates as Hoseini and Bond, (2022) (Liu & Nowack, 2022; Chen et al., 2022; Chen et al., 2023). Otherwise, remaining studies which investigate the flows and stocks of microplastic emissions do not provide a systemic model of the types of microplastic polymers and their associated plastic markets, which would greatly benefit researchers and relevant stakeholders as different polymers will have different toxicities and additives within their life cycle may have different exposures enhancing their toxicity levels, and ability to break down into smaller pieces (Ryberg et al., 2019; Prenner et al., 2021). A bigger picture of the different plastic sources to the environment, identification of specific polymers and clarity of where action must occur is needed.

Thus far, each study has also neglected including secondary microplastics in their models. It is crucial to target macroplastics and life cycle stages most susceptible to degradation and to be a source of these microplastic emissions to the environment (Boucher & Friot, 2017; Waldman & Rillig, 2020). However, the lack of studies regarding the degradation rate of different plastic products in the environment prevents studies from including it in such assessments (Boucher & Friot, 2017; Horton, 2022). Therefore, it is not included in the scope of this investigation (Waldman & Rillig, 2020).

### 2.7.3 Plastics MFA in Canada

In 2019, a report on Canada's plastics industry was published and included a simplified MFA indicating the mass of plastics throughout the plastics lifecycle (Deloitte & ChemInfo Services, 2019). Deloitte & ChemInfo Services (2019) only focused on macroplastics but provided a comprehensive overview of the polymer types and plastic products which dominate the Canadian plastics value chain. In addition, they provide insight into how Canada can divert 90% of plastic waste, thereby avoiding leakage into the environment, through different end-of-pipe solutions such as chemical recycling, mechanical recycling, incineration with energy recovery etc. Nevertheless, this report fails to provide an analysis of microplastics, which for the Global North is the larger concern as primary microplastics represent 79% of mismanaged waste emitted into aquatic bodies compared to 21% of macroplastics (Boucher & Friot, 2017). Therefore, for Canada to have an impact on its plastics waste issue and turn away from the problematic linear plastics problem, Canada must widen their attention to include the leakage of microplastics and begin to think of technological and legislative solutions. An MFA can be conducted to help inform these solutions, where the quantification of microplastic emissions based on an analysis of the plastics' life cycle should be completed.

This literature review investigated the research conducted over the past decade on microplastics and indicates the lack of attention toward quantifying its flows and stocks. Researchers have primarily been focused on researching the ecological effects of plastics, their presence in different regions and, more recently, their transport. Here this literature review argues that given the plethora of research on the quantities of microplastics across different sectors and environments, connecting the flows and providing a holistic picture of the plastics cycle through an MFA is necessary. Until now, research has focused on measuring the output of macroplastics during various stages of their life cycle. Although some studies have been done to comprehend the flows of microplastics, they have yet to disclose the precise mechanisms/points of discharge or the specific environmental compartments that microplastics can infiltrate. Only macroplastic flows and stocks have been quantified in Canada, while microplastics remain uninvestigated. An MFA of microplastics in Canada can provide a comprehensive road map and identify major points of intentional and unintentional releases of microplastics across the Canadian economy and environment. Plastic stakeholders in Canada can then have a clear outline of the seven microplastic types and their estimated outputs into the environment and guidance on where policy or technological solutions need to be put in place to prevent further environmental harm.

# Chapter 3: Methods

## 3.1 Methodological Background of MFA

MFA is one of the central methodologies in industrial ecology. It is an integral tool that enables researchers to map the mass movement of a material within a defined system and time period (Brunner & Rechberger, 2009; Greadel et al., 2019). It operates on the principle that "materials cannot be lost," drawn from the first law of thermodynamics entailing the conservation of matter and energy (Makarichi et al., 2018). In resource, waste, and environmental management, MFA is used as a decision-support tool and arguably has been utilized for the past 200 years (Brunner & Rechberger, 2016). As a simplified model of real-world systems, it acts as a tool to analyze the transformation of materials, their transportation and storage within a system, thereby acting as a valuable method for researchers and stakeholders interested in material efficiency (Allesch & Brunner, 2015). As MFA uses the law of conservation of matter, its results can also be controlled by simple material balance comparing all inputs, stock, and outputs of a defined process (Islam & Huda, 2019).

Historically, MFAs have been used for many goods and substances, such as municipal solid waste, cement, nutrients, and heavy metals (Greadel et al., 2019). In addition, MFAs have been applied for regional and industry analysis, aiding in optimizing material flows and waste streams in production processes and, thereby, cost savings (Binder et al., 2007). At a regional scale, the results from an MFA can aid in the optimization of resource exploitation, consumption, and environmental protection within the constraints of a region or company (Fet & Deshpande, 2023). As of late, its application in measuring electrical and electronic equipment has been instrumental in uncovering the flows of the substances associated with e-waste and their products, providing economic assessments of material recovery, and identifying the roles of responsible authorities (Islam & Huda, 2019).

When conducting an MFA, a few factors need to be established before conducting the analysis:

1. The definition of a problem must be defined along with the study's goals.
2. Relevant goods, substances, processes, and system boundaries must be established to ensure consistency and scope for the study and sufficient data collection.
3. The substance flows and stocks must be calculated and balanced so that the inputs of a system match its outputs.

Equation 1 is MFA's general mass balance equation (Brunner & Rechberger, 2016). If the inputs and outputs do not balance, one or several flows are missing or miscalculated (Brunner & Rechberger, 2016).

$$Sum\ of\ input = Sum\ of\ outputs + Change\ in\ stock \qquad (1)$$

The implications of an MFA are vast and allow researchers to conduct performance evaluation, system analysis, and waste management system comparisons, allow for early recognition of harmful substances or goods in a system or conduct scenario analyses to optimize the systems better (Allesch & Brunner, 2015). MFA is an effective tool for addressing sustainability issues with materials as they provide a systematic account of a defined system to support relevant decision makers – thereby allowing researchers to best inform the federal government on potential strategies.

### 3.2 Mass Flow Analysis Model

A static MFA was employed to evaluate the mass of microplastics generated and released into the environment. The geographic boundary is Canada, while 2016 was the temporal boundary chosen as that is the corresponding year that Deloitte and Cheminfo Services conducted their static MFA of macroplastics across Canada in 2016. An MFA can be conducted at two levels: goods and substances. Analyzing the flow of goods is crucial for understanding the overall system functioning while assessing substances helps gauge the quality of material flows, including resource flows and environmental emissions. Both goods and substances were considered in this study, as the primary microplastics examined were derived from macroplastics, which are classified as goods. This approach assesses the transformation, transport, and storage of substances, identifying resource potentials and risks to health and the environment (Allesch & Brunner, 2015).

As the flows produced for each microplastic type can be expansive and hard to follow, simplified versions of each microplastic type and their flows were created using the software tool e!Sankey Pro 4.5.3 can be found in the results section. The Sankey diagrams created in STAN can be found in Appendix C.

### 3.3 Quantifying the Flows of Microplastics

Secondary and tertiary data were collected through peer-reviewed literature, market reports, national statistical databases, and industry reports to provide values for model inputs and transfer coefficients between compartments. Input data for pre-production pellets, building paint, and agricultural plastic film were obtained from Deloitte and Cheminfo Services (2019). Input data for microfibres and construction foam were obtained through reports by Cheminfo Services, microbead input data was obtained from an analysis by the Government of Canada, and lastly, road marking paint input data was derived from a technical document from the Government of Quebec (Government of Canada, 2015; Ministry of Transportation Quebec, 2019; ChemInfo Services Inc, 2022). To determine the transfer coefficients for the flows for each type of microplastic, peer-reviewed literature was used from previous studies done across different regions, as there have been little to no studies conducted regarding the release of microplastics in the Canadian context.

Below, the methods to calculate microplastic generation and subsequent release into the environment are described for pre-production pellets, microbeads, microfibres, paint from road markings and buildings, construction foam, agricultural plastic film, tire wear particles, and other categories.

### 3.3.1 Pre-production Pellets

Pellet loss/spills have been most studied in Europe (Sundt et al., 2014; Lassen et al., 2015; Sherrington et al., 2016; Hann et al., 2018). A study in the UK and Norway conducted a comprehensive study of pellet loss across UK and Norwegian industries and their loss rates during various pellet processes (Sundt et al., 2014; Sherrington et al., 2016). They achieved this by analyzing previous reports that recorded pellet loss within the industries and consulting with industry representatives and stakeholders (Sundt et al., 2014). The results from these studies and what has been universally used as the pellet loss rate across multiple studies is between 0.001-0.01% with an average of 0.01% for production, transport, processing and waste management (Sundt et al., 2014; Lassen et al., 2015; Sherrington et al., 2016; Hann et al., 2018). Therefore, this study assumes that the pellet loss rate would be similar to that of Europe. However, it is noted that there are researchers conducting research specific to pellet industries in Canada to gain a better picture of the North American scene (U of T Trash Team, 2023). Based on the calculation of the pellets lost, based on the literature, it is assumed that 25% is released to stormwater drains, 73% is released to soils, and 2% is lost to air (Hann et al., 2018; Kawecki & Nowack, 2018; Peano et al.., 2020). From here, the fate of pellets in the different environmental compartments depends on the sewage system systems, which are described further below. It is important to note that these pellets lost are assumed to be never recovered.

### 3.3.2 Microbeads

As described in Chapter 2, microbeads can be found in industry abrasives and personal care cosmetic products. Microbeads found in PCCP have been widely studied, and average loss rates have been established based on available studies. Similar to pellets, these studies have mainly been conducted in Europe and the US, which provides a more accurate representation of what is happening in Canada as it is assumed consumer behaviour is more similar to the US than that of Europe (Napper et al., 2015; Carr et al., 2016; Kalcikova et al., 2017). Based on a survey conducted by the Canadian Environmental Protection Act (CEPA) of importers and manufacturers of toiletries that contain microbeads in 2012, they estimate that 100 tonnes of microbeads were imported while 10 tonnes were domestically produced (Government of Canada, 2015). It was also assumed that the total quantity of microbeads produced was used entirely when input into the system, and this input would be the same for 2016. Kawecki and Nowack (2018) state that 5% of microbeads remain in the container when thrown away/finished by the consumer – therefore, this was also applied to the current study. The remaining fraction of microbeads was assumed to be washed down the drain directly to the WWTPs, ending up in each of the six final compartments. This process is described below in section 3.4. The polymer composition of the microbeads was not reported, and as this information is often proprietary, it is assumed that 50% was comprised of polyethene and the other 50% PMMA (Beat the Microbead, 2018). Although microbeads in cosmetic products have been banned in Canada as of 2018 under the Canadian Environmental Protection Act of 1999, depicting the mass of microbeads flowing through Canada's economy and environment showcases why this ban was a positive action and provides insight into the minimum quantity prevented from entering the environment (Environment Canada Climate Change, 2023).

### 3.3.3 Microfibres

Microfibre shedding from clothing is a widely researched source within the microplastics field (Athey & Erdle, 2022). Therefore, the shedding rate of microfibres has been heavily studied for multiple different types of clothing pieces and different washing machine types (Rathinamoorthy & Raja Balasaraswathi, 2020). To calculate the microfibre generation for Canada, first, the mass of clothing being washed per year was calculated. This was done by extracting data from a report by Cheminfo Services

(2022), which investigated clothing disposal across Canada. The quantity of clothing in use (stored in wardrobes and actively worn) for 2016 was calculated assuming an average lifespan of three years. Therefore, it was also assumed clothing from 2014 and 2015 had the same mass input as 2016 (Deloitte & ChemInfo Services, 2019). Therefore, the total input of clothing is 2,089,382 tonnes. Next, to calculate the microfibre shedding rate, a total of 14 peer-reviewed articles and reports were analyzed to obtain an average shedding rate of 96 g/tonnes for synthetic microfibres (Belzagui et al., 2020; Almroth et al., 2018; Dalla Fontana et al., 2021; De Falco et al., 2018, 2019; Hartline et al., 2016; Hernandez et al., 2017; Kelly et al., 2019; Lant et al., 2020; Napper & Thompson, 2016; Pirc et al., 2016; Sillanpää & Sainio, 2017; Vassilenko et al., 2021; Zambrano et al., 2019). To calculate the tonnes of microfibres released through washing machines in Canada, the equation developed by Geyer et al. (2022) was used:

$$MF(t) = AS(t) * 0.75 * f * 96\ \frac{g}{tonnes}$$

Where:

- $AS(t)$ is Canada's apparel stock in the year 2016
- $f$ is the washing frequency
- 0.75 is the fraction of clothing that is purchased and sits in the wardrobes of wardorbes that is worn and washed (Laitala et al., 2018; Geyer et al., 2022)
- 0.25 remain in wardrobes unworn (Laitala et al., 2018; Geyer et al., 2022)

Geyer et al. (2022) conducted an MFA on microfibres in California, USA and calculated the washing frequency to be eight times per piece of clothing per year. Given similar behaviour between the US and Canada, it was assumed Canadians have the same washing frequency.

The microfibres released from the dryer were also quantified. The following equation was used to calculate the total mass released:

$$MF(t) = \big((AS(t) * 0.75 * f) * 0.39\big) * 0.216\,\frac{g}{tonnes} \qquad (3)$$

The same parameters are used as described above for washing machine microfibre release. However, the percentage of Canadians who dry their clothing using the dryer is 36%, according to a Statistics Canada survey (Statistics Canada, 2009). Few studies focus on the shedding of microfibers from dryers. Moreover, the existing studies, which have different methodologies, report the shedding rate using different units similar to the loss rate from washing. Therefore, 0.216g/tonnes was used as the only study to record the shed rate in units of weight (Yousef et al., 2021).

Similar to microbeads, the fate of microfibres from washing machines is directly to WWTPs, where their fate to the environment is heavily dependent on the types of WWTPs available in the region (further described in section 3.4). As for microfibres from dryer vents, it is assumed that microfibres generated in the dryer are thrown away and, therefore, end up in the landfill.

### 3.3.4 Road Marking Paint

As the frequency of paint applied to roads varies between road type and by municipality, there was no national data on this, so instead, the guidelines provided by the Government of Quebec were used to estimate the mass of paint that enters the system boundary for the year 2016. This technical report also listed the types of paint used on roadways, which affect the wear and tear rate of the paint and, thereby, the microplastics generated (Hann et al., 2018). In the report, it is stated that depending on the type of paint, between 35-66 L/km is applied to roads. Given the harsh climate in Canada, roads are repainted yearly, while others are on a cyclical basis based on the priority determined by the municipality. However, for this study, it is assumed that all roads are repainted, as there is no other data available. Therefore, to calculate the mass of paint applied to all roads in Canada, the following equation was used:

$$Road\ length\ (km) * \frac{tonnes\ of\ paint}{km} * wear\ tear\ rate * polymer\ content \qquad (4)$$

The total road lengths across Canada were obtained from Statistics Canada and were broken down by highways, rural highways, urban roads, and rural roads, which is helpful to help predict where paint fragments from road markings end up once in the environment (Infrastructure Canada, 2016). Statistics Canada also reported that 40% of rural roads are paved, and therefore, 60% are unpaved, which is then assumed to have no paint applied (Statistics Canada, 2018).

### 3.3.5 Building Paint

To calculate microplastics that are created through the paint applied to residential buildings across Canada, the calculation method from Verschoor et al. (2016) was applied (Figure 7).

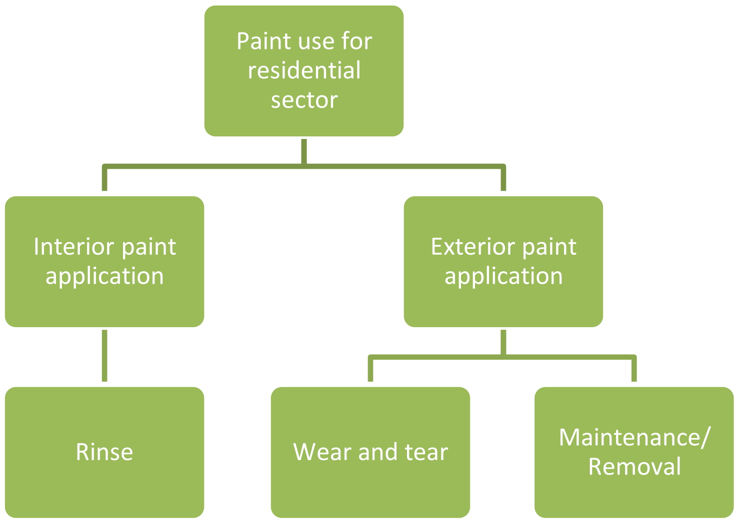


Figure 6: Process and fate of paint emissions from building application.

Two applications are considered: paint from interior and exterior applications. This is important as the majority of the paint market is driven by interior paints and their application. Due to national regulations on proper disposal of hazardous waste, it is assumed professional painters dispose of paint properly, whereas individuals who decide to paint themselves (do-it-yourself) would rinse and wash their brushes in the sink to remove the paint. As for exterior paints, emissions from paint come from the wear and tear of exposure to the environment and from maintenance or removal of paints on exterior surfaces.

Below are the following equations to calculate paint emissions from interior paints and exterior paints:

$$Paint\ lost\ due\ to\ interior\ application\ through\ DIY = Paint\ in\ use * f_{sector} * f_{used} * f_{interior} * EF_{rinising} \qquad (5)$$

$$Paints\ lost\ during\ Maintenance\ and\ removal: \qquad (6)$$

$$= Paint\ in\ use * f_{sector} * f_{used} * f_{exterior} * EF_{removal}$$

$$Paint\ lost\ due\ to\ wear\ and\ tear:$$

$$= Paint\ in\ use * f_{sector} * f_{used} * f_{exterior} * EF_{wear\ and\ tear} \qquad (7)$$

Where:

- $f_{sector}$ is the fraction of paints sold to the professional or DIY sector
- $f_{used}$ is the fraction of paints that is actually used (some is assumed to be left in the can or not entirely used for a job)
- $f_{exterior/interior}$ is the fraction of paints that enter the system that are made for interior or exterior use specifically
- $EF_{removal}$ $EF_{wear}$ $EF_{rinising}$ are emissions factors from Verschoor et al., (2016) of paint for these activities

To understand the fate of paint particles once in the environment, assumptions regarding what fraction of paint applied is for rural vs. urban areas. It is assumed that 57% of paint is applied to urban areas and 42% is applied to rural areas utilizing the calculation method from Verschoor et al. (2016), which calculates it based on the sides of homes that need painting, the length of house sides, the average number of floors, the height of one floor, and outside surface area that needs painting. The assumption for the fraction of paint used for DIY purposes vs. professional was based on a survey conducted by the Royal Bank of Canada in 2006, which reported that 52% of Canadians use professional contractors while 48% do it themselves (Royal Bank of Canada, 2006). The calculations and assumptions can be found in Appendix A.

### 3.3.6 Construction Foam

Here, the types of construction foam chosen are XPS and EPS foam, as they are increasingly identified in aquatic environments (Turner, 2021; Gao et al., 2023). The production of XPS and EPS was calculated based on the mass of hexabromocyclodecane (HCBD) manufactured in Canada in 2012 (ChemInfo Services Inc, 2012). HBCD is primarily found in XPS and EPS foam. Studies have stated that 90-96% of HBCD in EPS and XPS foam is for the building and construction sector (Environmental Protection Agency, 2014; Stubbings & Harrad, 2019). Therefore, an average of 93% was used for this study. An expert from ChemInfo Services consultancy group has stated that not all XPS foam has HBCD present, i.e., in packaging for consumer goods. However, it is assumed that the fraction of XPS used in building and construction does. The range of HCBD, which is incorporated into EPS and XPS, is between 0.5 and 3% (ChemInfo Services, 2012; Environmental Protection Agency, 2014). Based on the literature available, an average composition of 1.75% is assigned to HCBD to account for the total weight of EPS and XPS. Therefore, the total mass of XPS and EPS foam that is produced for Canada is calculated with the equation below:

$$\frac{HCBD\ mass * HCBD\ used\ in\ construction\ foam}{Composition\ of\ HCBD\ used\ in\ XPS\ and\ EPS\ foam}$$

Next, to estimate the mass of EPS and XPS lost during construction processes, estimates from Minet et al. (2021) were used and can be found in Appendix A. It is assumed all foam released directly enters surface waters or soils or is captured and sent to a landfill and does not enter stormwater drains or WWTPs (Minet al., 2021).

### 3.3.7 Agricultural plastic film

In Canada, it is estimated that 45,900 tonnes of plastic mulch were used in 2016. To estimate the mass of microplastics generated from applying plastic film to agricultural soils, it assumed 1.1% of microplastics generated from the application, wear and tear and removal of the mulch to the agricultural soils (Albertsson et al., 1987; Kawecki & Nowack, 2019).

### 3.3.8 Tire Wear Particles (TWPs)

Two approaches are described in the literature to calculate TWP emissions. The first is by taking into account total consumption of tires and their weight loss due to abrasion per year. (Mennekes & Nowack, 2022). The second method which is most used in the literature is based on emission factors of different vehicle types and the total annual vehicle km (Baensch-Baltruschat et al., 2020). For this current study the second method was applied using the following equation (Peano et al., 2020):



(9)

$$D_{Car\_vhc}[vhc * km] * Loss_{car\_tires} \frac{kg\ tread}{vhc * km}$$

The way TWPs are generated reflect a variety of factors: tire characteristics (type, age, composition, size, chemical composition) properties of the road surface, characteristics of the vehicle, and vehicle operations (Wagner et al., 2018). The emission factors from passenger cars and trucks were obtained from Deltares and TNO, an independent knowledge institute from The Netherlands (Deltares and TNO, 2016; Table 3). Statistics Canada provide data on annual mileage of vehicles registered in Canada and has them differentiated based on municipality and road type (Statistics Canada, 2019; Table 4). This allows for the distinction of TWP emissions to different regions which then allows their different fates to be determined (i.e., emission to roadside vs stormwater drains).

Table 3: Emission factors for vehicle types and road types to calculate TWP generation (Deltares and TNO, 2016; Baensch-Baltruschat et al., 2021).

| Vehicle type | Road type (mg/km) | | |
|---|---|---|---|
| | Rural | Urban | Highway |
| Passenger Car | 85 | 132 | 104 |
| Truck | 423 | 658 | 517 |
| Lorry | 546 | 850 | 668 |

Table 4: Distance travelled by passenger cars, trucks and lorries on main road types in Canada.

| | Road type and vehicle-kilometres (million km) |
|---|---|

| Vehicle Type | Highway | Rural highways | Urban roads | Rural roads |
| --- | --- | --- | --- | --- |
| Passenger cars | 55,000 | 102,000 | 33,000 | 113,000 |
| Trucks | 1,500 | 2,770 | 893 | 3,070 |
| Lorry | 5,400 | 9,980 | 1,320 | 4,540 |

Once the mass of the TWPs was calculated, the fate of TWPs to the environment was determined. The different trajectories are based on the road type the TWPs are generated on i.e., flows to roadside, stormwater drains, WWTPs, or stormwater management facilities (Baensch-Baltruschat et al., 2020; Sieber et al., 2020). Below, Table 5 depicts the parameters and assumptions applied to calculate TWP emission onto roads and its distribution to the environment.

Table 5: Parameters and assumptions applied to calculate tire wear emissions on roads in Canada and its distribution in the environment.

| Parameters | Data Applied | Unit | Reference |
| --- | --- | --- | --- |
| Annual mileage of vehicles registered in Canada | Table 4 | km | Statistics Canada, 2009 |
| Emission factor | Table 3 | mg/vehicle km | Deltares and TNO, 2016 |
| Atmospheric emission | 2.2% | Mass % | Charbouillot et al., 2023 |

| | | | | |
|---|---|---|---|---|
| Fraction of TWP on roads to roadside | Highway | 90% | Mass % | Kaufmann et al., 2007<br>Ten Broeke et al., 2008 |
| | Urban | 46.7% | | |
| | Rural | 85% | | |
| Fraction of TWP removed by municipal street sweepers | Urban | 36% | Mass % | City of Toronto, 2015 |
| Fraction of TWP on roads to water runoff | Highway | 10% | Mass % | Kaufmann et al., 2007<br>Ten Broeke et al., 2008 |
| | Urban | 53.3% | | |
| | Rural | 15% | | |
| Fraction of TWP from water runoff to surface water | Highway | 50% | Mass % | Unice et al., 2019 |
| | Urban | 37.5% | | |
| | Rural | 70% | | |
| Fraction of WTP from water runoff to | Highway | 50% | Mass % | Government of Canada, 2013; Unice et al., 2019 |
| | Urban | 62.5% | | |

| | | | | |
|---|---|---|---|---|
| stormwater drains | Rural | 30% | | |
| Percentage of stormwater collected in Combined wastewater systems | | 7% | Mass % | ECCC, 2017 |
| Percentage of stormwater collected in Separate wastewater systems | | 93% | Mass % | ECCC, 2017 |
| Combined system overflow (CSO) in separate wastewater systems | | 4.4% | Mass % | ECCC, 2020 |

Modelling the emissions of TWPs to different road types is crucial to depict best which environmental compartment they accumulate in. For TWPs that end up on highways it is assumed only 10% are washed off by water runoff while the remaining 90% end up on roadside to accumulate (Table 5). These factors are applied both to rural and urban highways. For urban roads it is assumed 36% of TWPs are removed by street sweepers, and from this remaining fraction 53.3% are taken by water runoff to surface waters or stormwater drains, while 46.7% end up on roadsides (Table 5). The fraction that ends up in stormwater drains either are received by separate wastewater systems or combined wastewater systems which is further explained below.

## 3.4 Stormwater Runoff and Wastewater Treatment

Canada has two types of sewage systems: combined and separate as described in Chapter 2. To calculate the mass of microplastics that end up in separate sewage systems, data was collected from a survey by Statistics Canada and a report by ECCC (Infrastructure Canada, 2016; ECCC, 2017). The survey identifies the types of stormwaters drains and systems for publicly owned assets, including open culverts, stormwater pipes, open ditches, and stormwater management facilities such as wetlands and ponds (Infrastructure Canada, 2016). This assessment excluded any combined sewage systems; therefore, only includes wastewater outfalls from separate sewage systems (Infrastructure Canada, 2016; ECCC, 2017). Based on the survey it was calculated that 25% of stormwater ends up in a stormwater management facility while 75% ends up in surface water (Infrastructure Canada, 2016). In addition, to determine the fraction that is retained in a stormwater management facility and released back to surface waters a study by Stang et al. (2022) found 74% is retained due to bioretention and filtration mechanisms while and 26% of stormwater is filtered and sent back to surface waters. The 74% retained is effectively a stock of microplastics in the system. The survey by ECCC found that 93% of sewage systems are separate sewage systems, while 7% are combined sewage systems (ECCC 2021). To calculate the fraction of wastewater that overflows from combined sewage systems data was collected by a municipal wastewater treatment assessment from 2013-2017 and found to be 4.4% (Statistics Canada, 2019).

The next step is to calculate the fraction of wastewater ends up in primary, secondary or tertiary WWTPs. This is crucial as they each vary in their ability to remove microplastics. The percentage of primary secondary and tertiary WWTPs in Canada are 26.4%, 48.7% and 24.8% respectively (Statistics Canada, 2019). The removal rates were extracted from a review conducted by Iyare et al. (2020), as only one study in Canada has been conducted looking at a secondary WWTP of microplastics in Vancouver (Gies et al., 2018). The removal rates of microplastics for primary, secondary, and tertiary WWTPs can be found in Table 6. The two by-products from WWTPs include effluent and sludge, where effluent is sent to surface waters, and sludge can be applied to agricultural soils, sent to landfill, or sent for incineration. These estimations were based on a study conducted by the Government of British Columbia, which included the following provinces: Ontario, Manitoba, Newfoundland and Labrador, Quebec, Alberta, and British Columbia. The sewage sludge application rates were averaged from these provinces and applied across Canada (Government of British Columbia, 2016).

Table 6: Fraction of microplastics found in effluent and sludge of each WWTP type.

| Type of WWTP | Fraction of microplastics in effluent | Fraction of microplastics in sludge |
|---|---|---|
| Primary | 28% | 72% |

| | | |
|---|---|---|
| Secondary | 12% | 88% |
| Tertiary | 6% | 94% |

## 3.5 Stocks

The stocks of microplastics are calculated in the software tool STAN, and for this current study only exist in stormwater management facilities as described in section 3.4. However, stocks of macroplastics exist for each plastic product, as it is from these macroplastic products that the microplastics are generated from. However, the flows of the microplastics are the focus of this assessment, as the stocks of microplastics in the environment would be a separate study, through in-field monitoring data obtained through sampling various environments or by conducting an environmental transport fate, or accumulation and dispersion model (Chen et al., 2022; Kedzierski et al., 2023; Schwarz et al., 2023)

# Chapter 4: Results

## 4.1 The Flows of each Microplastic

Given the complexity of polymer and product types that can generate microplastics, each microplastic modelled through an MFA has significantly different flows based on their production, generation, and location (urban vs. rural). However, once microplastics have been generated through intentional production or use of the initial macroplastic product; they tend to end up in similar trajectories, released to sewage systems and wastewater treatment plants and/or to surface waters, air, soils, roadsides or agricultural soils.

There are two sets of material flow results illustrated. The first is the simplified flows presented below, created in e!Sankey software with the intention of easy communication and depiction for relevant stakeholders. The second set is a detailed version created in STAN, which includes every flow involved for microplastics to be released to their final compartments.

The mass of microplastics released to Canadian aquatic and terrestrial systems in 2016 was 19,500 tonnes (Figure 7). However, the total mass of microplastics released into both Canada's bio and technosphere (i.e., landfill, incineration, and roadsides) in 2016 was 60,000 tonnes. Based on the seven microplastic types modelled, TWPs were generated the most, releasing 50,300 tonnes in 2016 onto highways, urban roads, and rural roads (Figure 14). Here, it was found that 39,700 tonnes of TWPs were deposited onto roadsides. Another significant flow for TWPs is water runoff, which acts as a mechanism of transportation, carrying 4,840 tonnes to creeks, rivers, lakes, or oceans. Lastly, stormwater drains were significant flows, receiving 3,140 tonnes to separate sewage systems or combined sewage systems, which directly leads to a WWTP or stormwater management facility and eventually to surface waters.

In contrast, out of the seven microplastics studied, construction foam released the lowest quantity of microplastics at 24.6 tonnes per year (Figure 7, 13). The final compartments that receive the most foam particles are surface water, at 12.6 tonnes and soils, which receive 7.7 tonnes (Figure 14).

The results and leakages to different flows below are described in order of the literature review and methods to maintain consistency.

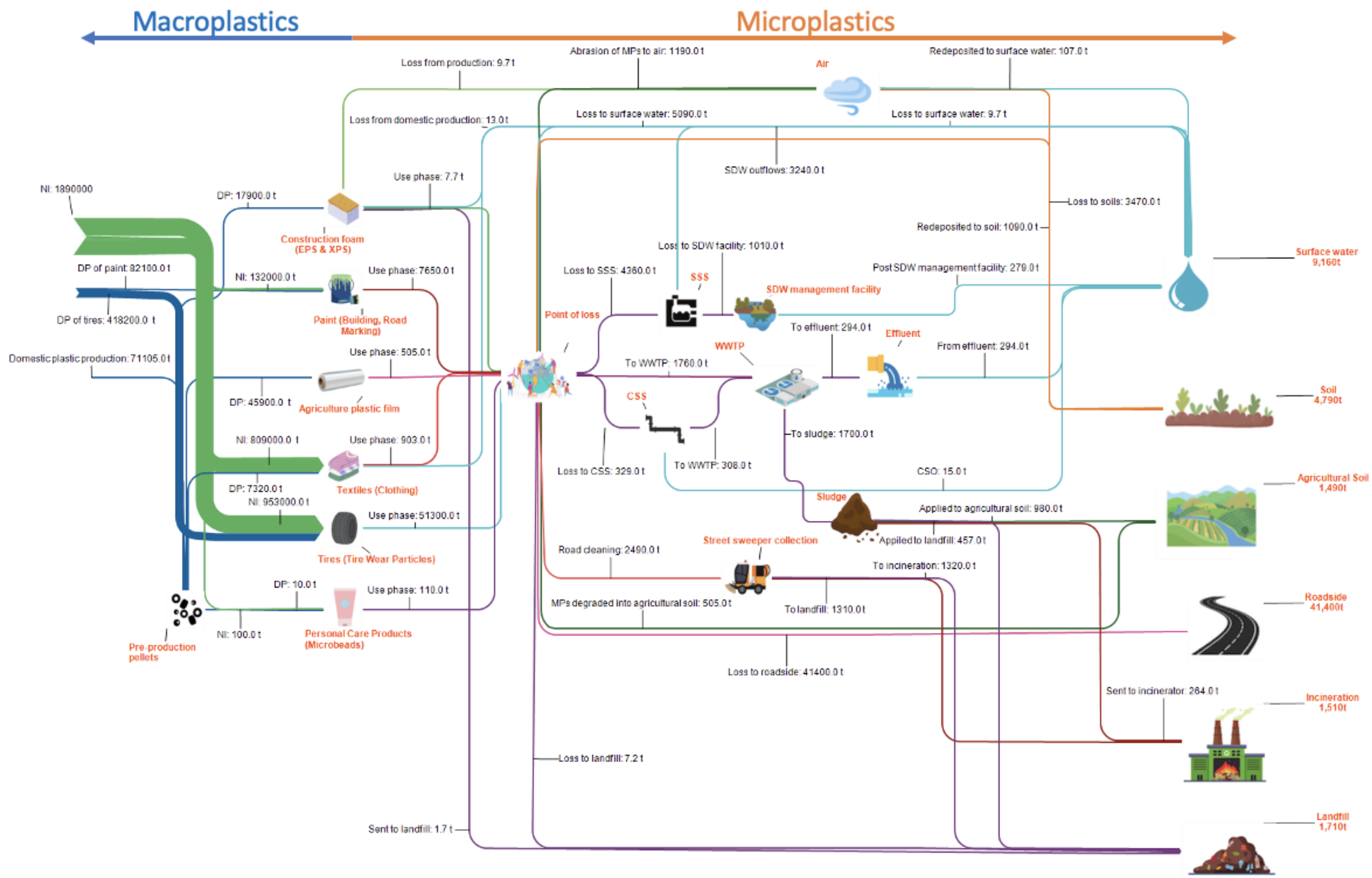


Figure 7a: Aggregated emissions flows for seven types of microplastics in tonnes per year in Canada in 2016. Each flow begins as macroplastics and is then transformed into a microplastic during its usage form (except for paint and microbeads). The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flow qualitatively represents its magnitude.

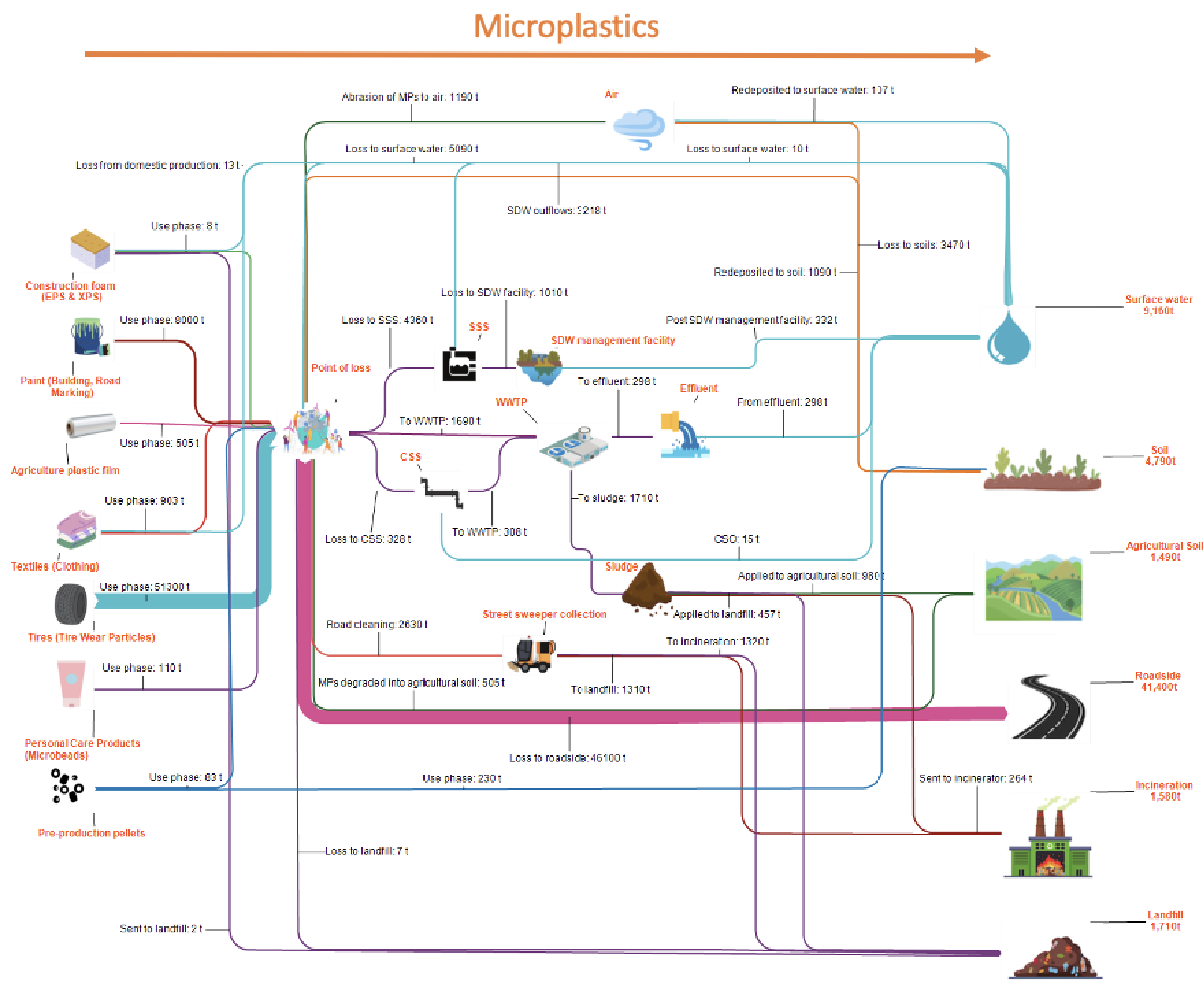


Figure 7b: An extension of Figure 7a to showcase the aggregated microplastics emissions flows for each of the seven types in tonnes per year in Canada in 2016.

### 4.1.1 Pre-production Pellets

In 2016, 308 tonnes of pellets were estimated to be lost in Canada during production, manufacturing, and transport (Figure 8). Based on the MFA results, most pellets were lost directly to soils at 224 tonnes during redistribution, packing, transport, or disposal, as described in Chapter 2. Surface waters received 60 tonnes, with the majority released from stormwater outfalls through the separate sewage systems. Only 5 tonnes of pellets were expected to have entered the WWTP, which is then distributed to effluent (0.7 tonnes) or released to sludge, which is then applied to either agricultural soils, sent to landfill or for incineration. In the case of pellets, emissions of pellets to agricultural soil were higher than that of surface water from WWTPs (2.2 vs. 0.7 tonnes) (Figure 8). In addition to WWTPs, it was found that stormwater drains and air are indirect flows for microplastics. 13 tonnes were found to be stored in stormwater management facilities, and 6 tonnes were redeposited from air to soils.

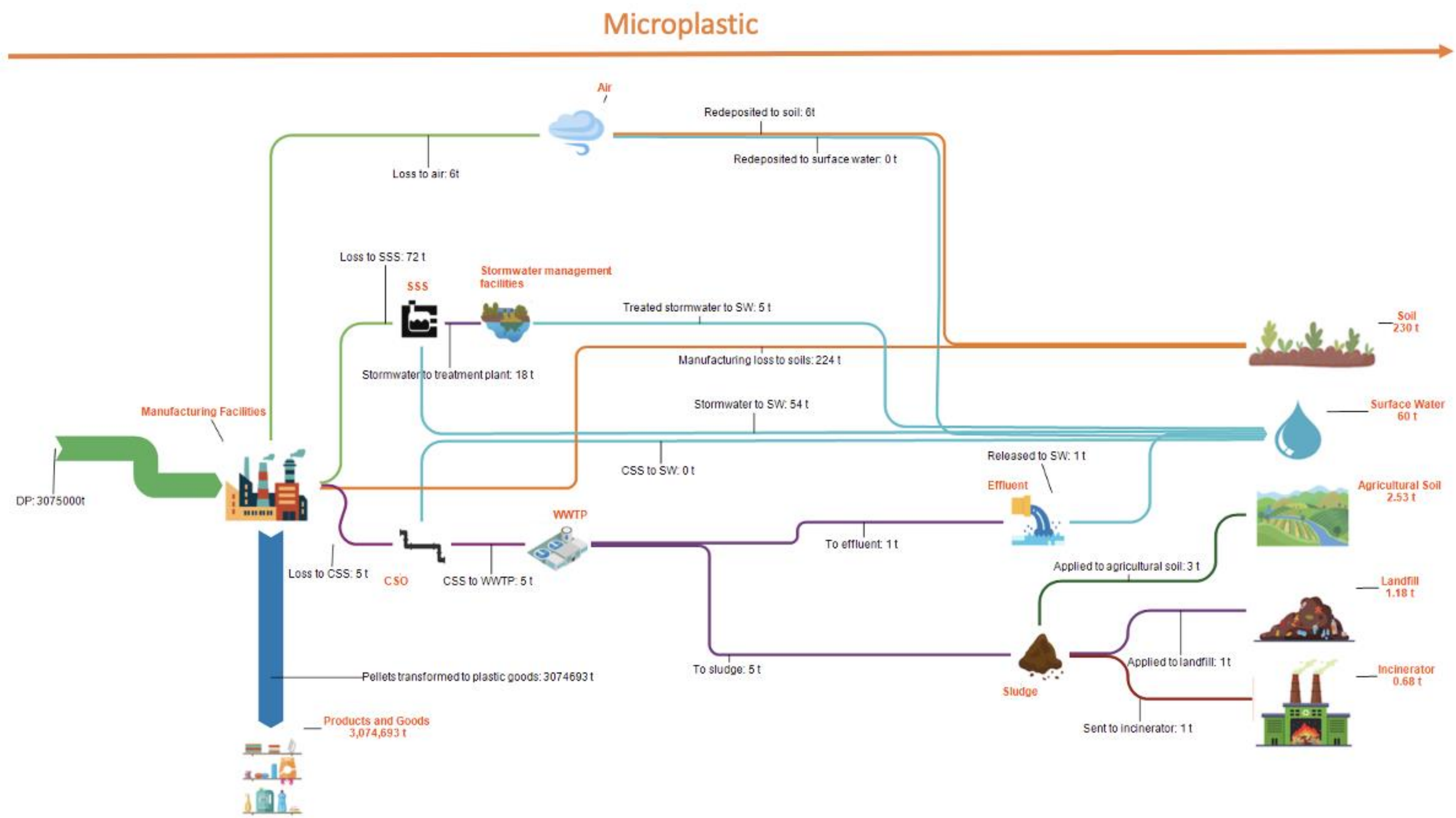


Figure 8: Emissions flows for pre-production pellets from domestically produced thermoplastic products in Canada in 2016. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The colour of the flows is representative of the receiving compartment, and the width of the flows represents its magnitude.

### 4.1.2 Microbeads

Similar to pre-production pellets, microbeads are intentionally produced at a size of <5mm. However, microbeads are intended to be washed down drains as personal cosmetic care products and should be rinsed off after use. The flows of microbeads are straightforward compared to that of TWP or paints as they are released to WWTPs, which thereby act as the only source of microbeads from personal care products to the environment. Based on the findings of the 110 tonnes of microbeads that entered the Canadian system, 6 tonnes were sent to landfill as left over in the bottle of the personal care product (Figure 7; Figure 9). Meanwhile, 104 tonnes were sent directly to WWTPs, where 14 tonnes were lost after effluent treatment by primary, secondary or tertiary WWTPs to surface waters. 49 tonnes were applied to agricultural soils through sludge generation, while 23 tonnes were incinerated, and 19 tonnes were sent to landfills (Figure 9).

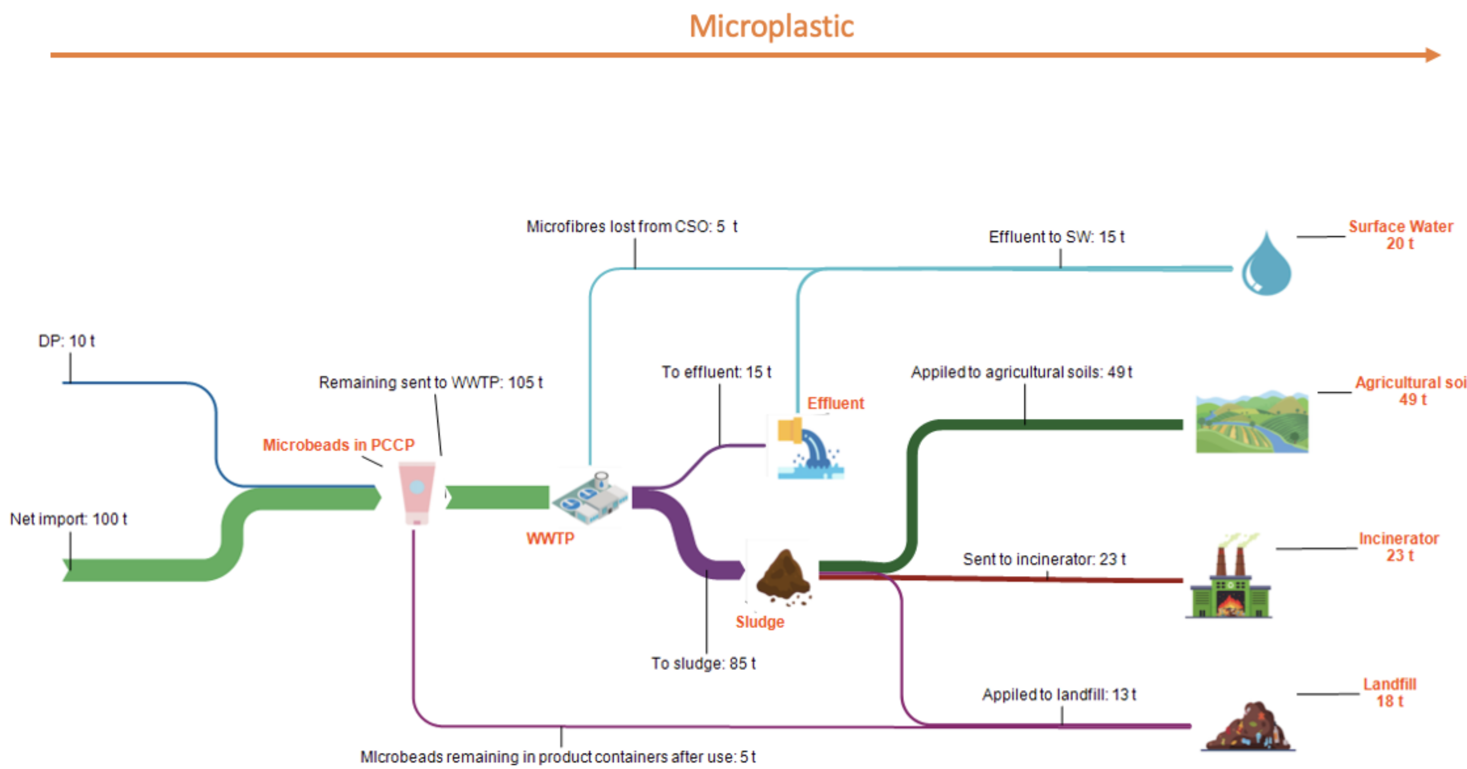


Figure 9: Aggregated emissions flows for microbeads domestically produced and imported in Canada in 2016. Microbeads are similar to pre-production pellets, are manufactured as microplastics already and, throughout the life cycle, are only in microplastic form. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

### 4.1.3 Microfibres

Microfibers are considered an unintentional loss of microplastics as they are created through the wear and tear of clothing through use, washing and drying. As Canada mainly imports its clothing, loss of microfibres is minimal from domestic production of clothing, and results show that 14 tonnes were lost directly to surface waters. In Canada, the majority of microfibers were lost through the washing process in the washing machines; approximately 903 tonnes of microfibers were released directly to WWTPs, where 127 tonnes were released to surface waters by treated effluent. The remaining mass is concentrated in sludge which is then distributed into agricultural soils, landfilled or incinerated (Figure 7; 10). Similar to microbeads, as their only outlet to the environment is through WWTPs, they act as the largest source of microfibre emissions. Agricultural soils receive 424 tonnes of microfibres, while landfill receives 197 tonnes, and 115 tonnes are incinerated. Microfibres are also released from dryers, which the average person who owns a dryer has actively touched and thrown away as it collects as lint in lint traps (Figure 10). The mass was estimated to be minimal compared to microfibres released from the washing machine with an estimated 0.13 tonnes, which is assumed to be directly thrown away and sent to the landfill. Microfibres are largely in stock as intact clothing, either stored in wardrobes or not worn, although purchased (522,000 tonnes). The largest stock of clothing is the portion that was actively worn and washed by the average Canadian in 2016, which is 955,00 tonnes, while the remaining mass is the portion that has been washed and dried in the washing machines and dryers, which is 611,000 tonnes. Lastly, the model depicts 113,000 tonnes of clothing sent to landfills, while 4,180 tonnes are chemically recycled or incinerated (Figure 10).

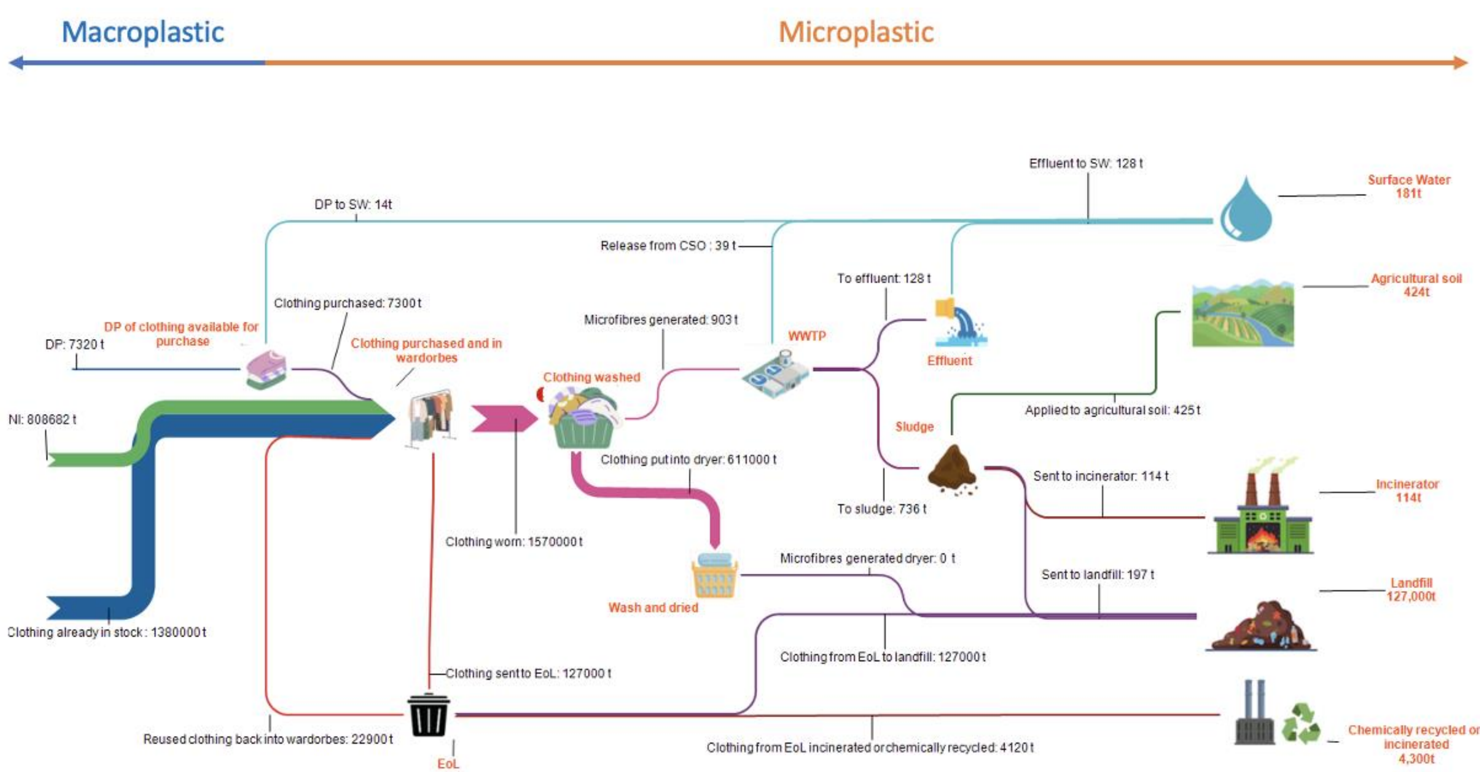


Figure 10: Aggregated emissions from microfibres from apparel textiles in Canada in 2016. Each flow begins as macroplastics and is then transformed into a microplastic during its usage stage at the washed and drying stage. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

### 4.1.4 Road Marking Paint Fragments

Unlike pre-production, microbeads and microfibres, paint particles from road marking have more processes and variables when released to the environmental compartments. Paint emissions were divided up by road type: highways, urban and rural roads. Rural roads (which includes rural highways) receive the highest paint emissions at 1,520 tonnes as they are used the most by passenger and truck vehicles. Urban roads receive 570 tonnes of paint emissions per year, while urban highways received the least mass of paint emissions at 294 tonnes (Figure 11). Once paint fragments are released to roads, they are either collected and transported by water runoff, street sweepers or collected on the roadside, which is defined as soils or land present at the edge of roads (Figure 11). Results show 72% of paint fragments end up on roadsides of highways, urban or rural roads, receiving a total of 1,700 tonnes. Water runoff is the alternative to roadside collection, where most paint fragments transported by water runoff end up directly into water bodies, totalling 402 tonnes. The other significant flow that leads to surface water is paint fragments to stormwater outfalls and stormwater management facilities. These outfalls are shown to release 140 tonnes directly to surface waters. The flows that carry the least mass of paint fragments to surface water are intuitively effluent from stormwater management facilities and WWTPs (Figure 11). Within this flow, a new process is introduced, which is the street sweepers. It is estimated that only urban roads have street sweepers to remove urban dust from roads, where 101 tonnes are collected and sent to landfills or incinerated. Lastly, in this flow, due to the street sweepers as well as most paint fragments ending up in stormwater sewage, the mass of microplastics that are applied to agricultural soils through sludge disposal is less than that of other microplastics like microfibres or microbeads totalling to 6 tonnes (Figure 11).

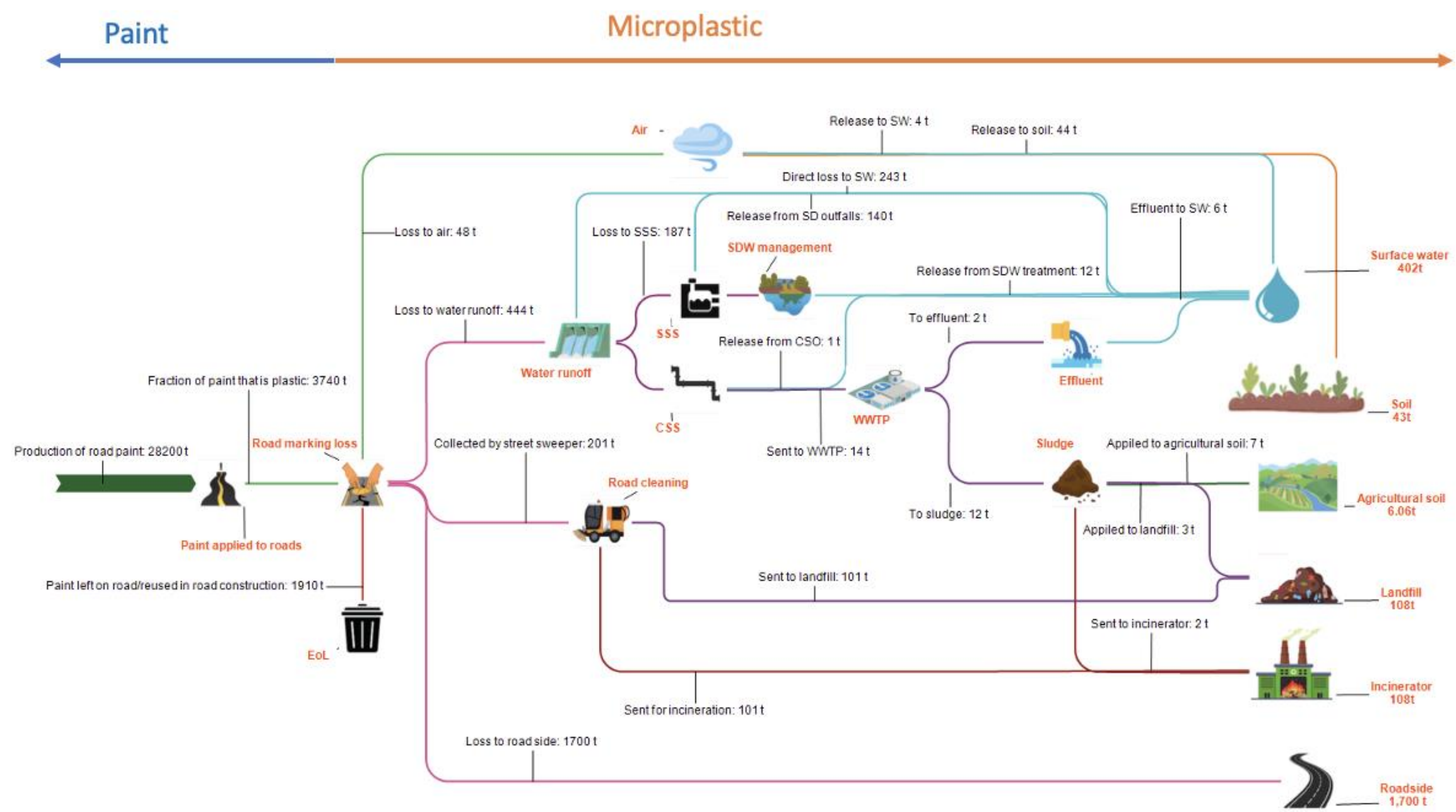


Figure 11: Aggregated emissions of paint fragments from road marking in Canada in 2016. Paint isn't a macroplastic and therefore is just considered as paint and then microplastic once fragmented from the wear and tear/usage. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

### 4.1.5 Building Paint Fragments

Like road marking paints and TWP, microplastics from this source are directly released to the environment due to wear and tear or removal of paint, which are deposited onto soils or are released to stormwater systems. Most paints produced are for interior use (116,000 tonnes); however, due to weathering conditions, wear and tear, and how paints are removed on the exterior, they generate the most microplastics released into the environment. Figure 13 depicts the highest emissions due to the wear and tear of exterior paints at 3,430 tonnes, where 1,960 tonnes are released from urban households and 1,470 tonnes are released from rural households (Appendices A and B). Meanwhile, emissions from the removal of building paint are estimated to be 613 tonnes from urban households and 458 tonnes. Microplastics that originate from the wear and tear and paint removal end up directly onto soils near the households or in nearby stormwater drains, which, similar to road marking paints, end up in separate sewage systems or combined sewage systems. It was then found that 2,660 tonnes were released from the wear and tear of exterior paints, and 826 tonnes were released from the removal of paints to soils. The remaining portion is sent to stormwater drains where separate sewage systems are more common and have been established across Canada; the majority of the microplastics end up in these pipes where it is found that 718 tonnes are released directly into bodies of water from stormwater outfalls (Figure 13). However, the portion of stormwater that ends up in stormwater management facilities, the treated water, only contains 62 tonnes.

In contrast, many microplastics are released directly to WWTPs as those who paint the interior of their homes DIY style are assumed to wash their brushes/rollers down the drain. This is estimated to be 433 tonnes from urban households and 324 tonnes from rural households. Combined with microplastics that stem from combined sewage systems and interior paint loss, microplastic content in the effluent of WWTPs is estimated to be 117 tonnes. Due to the large portion of microplastics released to WWTPs, the discharge of microplastics to sludge is significant, where it is shown 390 tonnes are applied to agriculture, 181 tonnes are applied to landfill, and 105 tonnes are sent for incineration (Figure 13). Here, it was found that building paint was the largest source of microplastic to soils, contributing 3,470 tonnes, and in comparison, it only contributed 934 tonnes to surface waters (Figure 13).

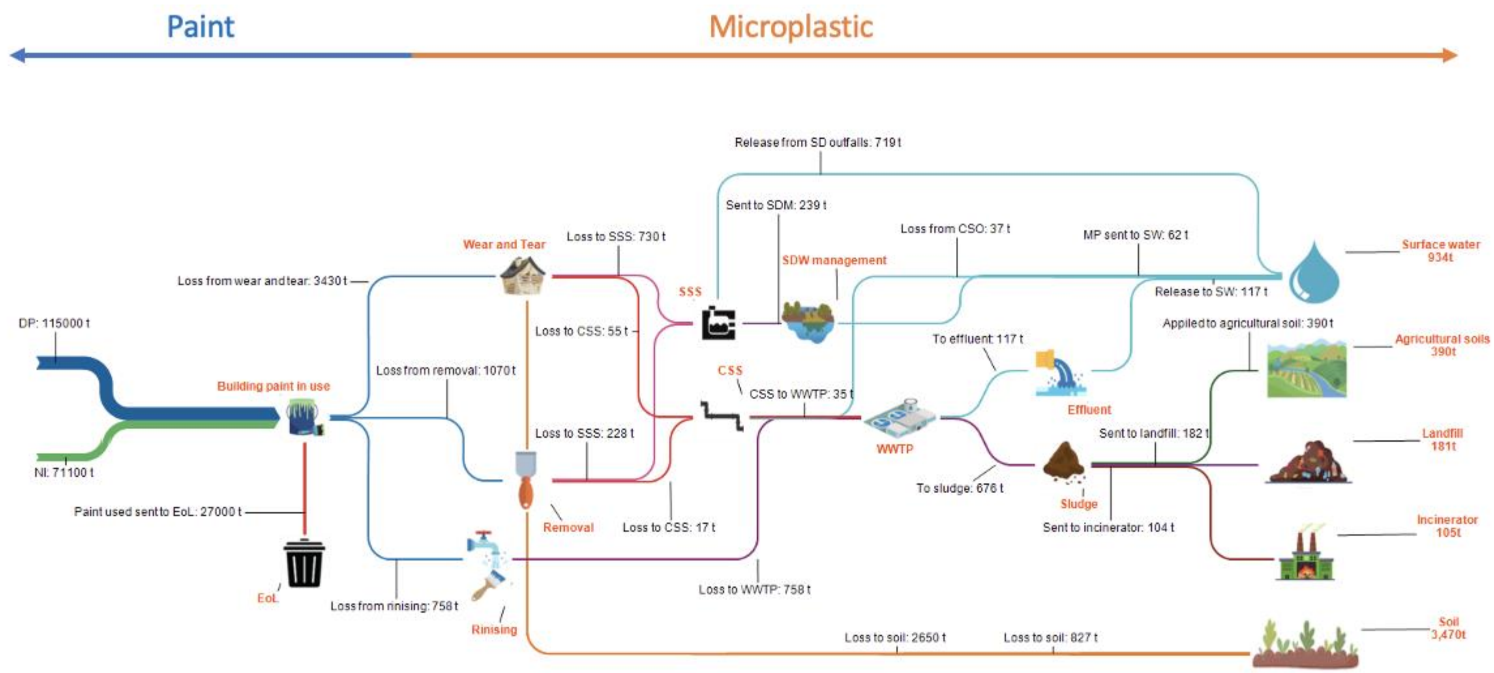


Figure 12: Aggregated emissions of paint fragments from buildings in Canada in 2016. Paint isn't a macroplastic and therefore is just considered as paint and then microplastic once fragmented from the wear and tear/usage. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

### 4.1.6 Construction Foam

The release of construction foam is considered unintentional as it's a by-product of construction through shaving and cutting foam boards. Given the lack of information on the pathways and flows of construction foam, it was assumed any microplastic emissions from EPS and XPS foam were released directly into the environment, as shown in Figure 14. Based on the available data, this model showed microplastics were equally released to surface waters and soils, where surface waters received 13.2 tonnes through direct release or redistribution from atmospheric transport. Soils received 13.8 tonnes through the release of foam during construction activities and being deposited on soils nearby. As for technological compartments, it was found that the landfills received 2,590 tonnes of foam through the demolition process.

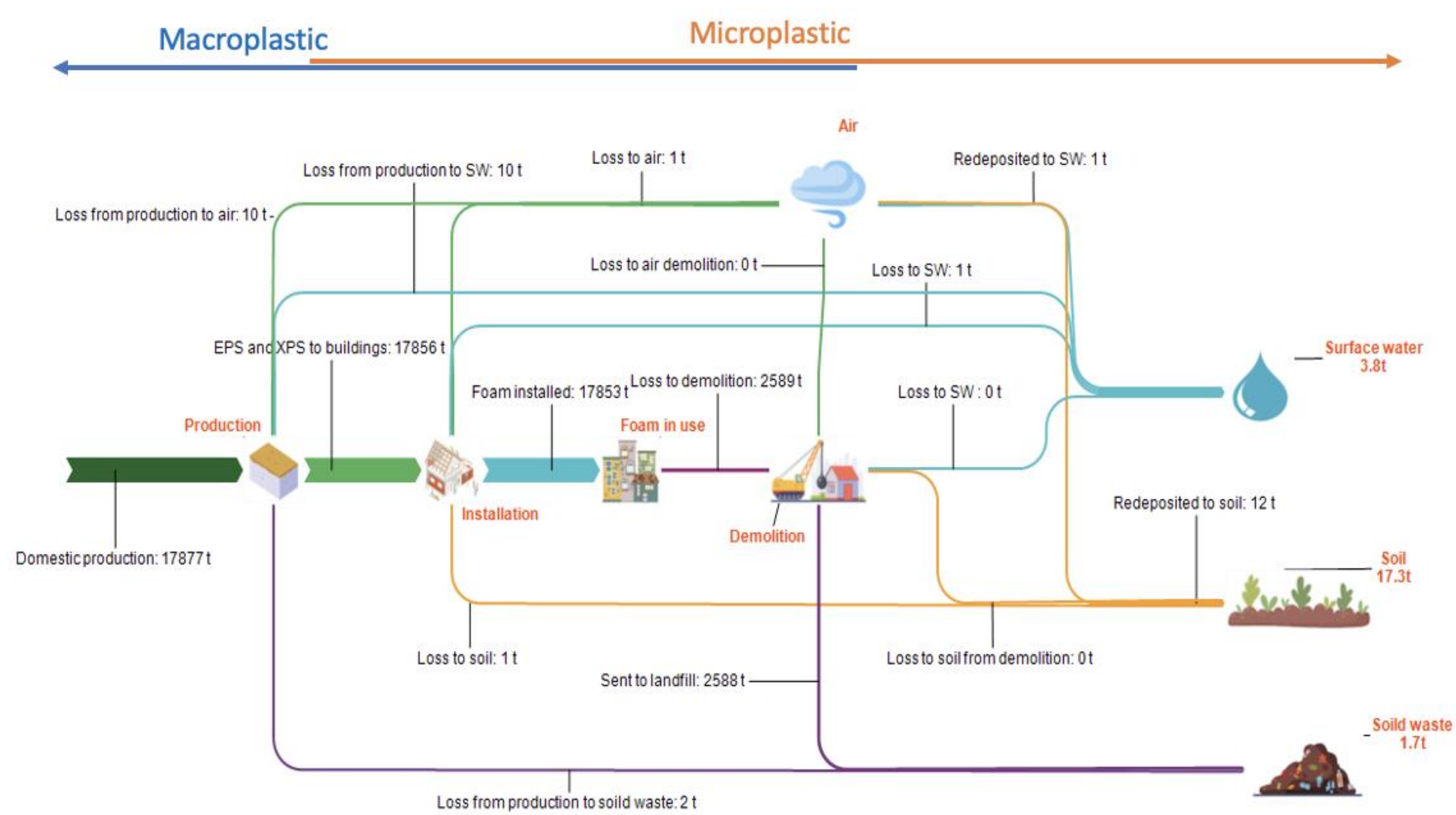


Figure 13: Emissions from EPS and XPS in construction foam in Canada in 2016. The green flow begins as macroplastics and is then transformed into microplastics during different processes. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

### 4.1.7 Agricultural Plastic Film

Like construction foam, agricultural plastic film was released directly into the environment. However, it is initially applied as a macroplastic and, therefore, is intentionally placed onto soils. Due to the lack of diversity in flows, this microplastic source doesn't have its own Sankey diagram but is depicted in Figure 8. In 2016, 505 tonnes were released directly into agricultural soils.

### 4.1.8 Tire Wear Particles

Vehicle tires generated the most microplastics, releasing 51,300 tonnes to highway, urban and rural roads. The flows of TWPs follow the same trajectory of road marking paint fragments and are dependent on the frequency of the types of roads used. Highways, specifically rural highways, are the most frequently used type of road in Canada and, therefore, received the most TWPs at 33,400 tonnes. Meanwhile, highway and urban roads received 10,400 and 6,360 tonnes, respectively. Once TWPs are released to roads, their flows to the environment are based on their behaviour and densities, which may vary depending on what the TWPs are composed of. Here, results show that 78% end up along roadsides (soil/land near the edge of roads) (Figure 14). The remaining fractions were transported by water runoff. A total of 4,820 tonnes were released directly to bodies of water through water runoff of the roads, while 3,380 tonnes were sent to stormwater drains (Figure 15). For TWPs that were generated on urban roads, 36% were removed from roads by street sweepers and collected a total of 2,290 tonnes. Once collected by the street sweepers, TWPs are assumed to be disposed of in landfills or incinerated. A small portion of TWPs is sent to WWTPs through combined sewage systems. A total of 226 tonnes were released to WWTPs, where 33 tonnes were released back to surface waters through the effluent, and 193 tonnes were sent to sludge. Through sludge management, 58% of sludge was applied to agricultural soils, 16% was incinerated, and 26% was sent to landfill. Lastly, vehicles were the largest contributor to microplastic emissions to air, where it was found that 102 tonnes were redeposited to surface waters while 1,030 tonnes were redeposited to soils.

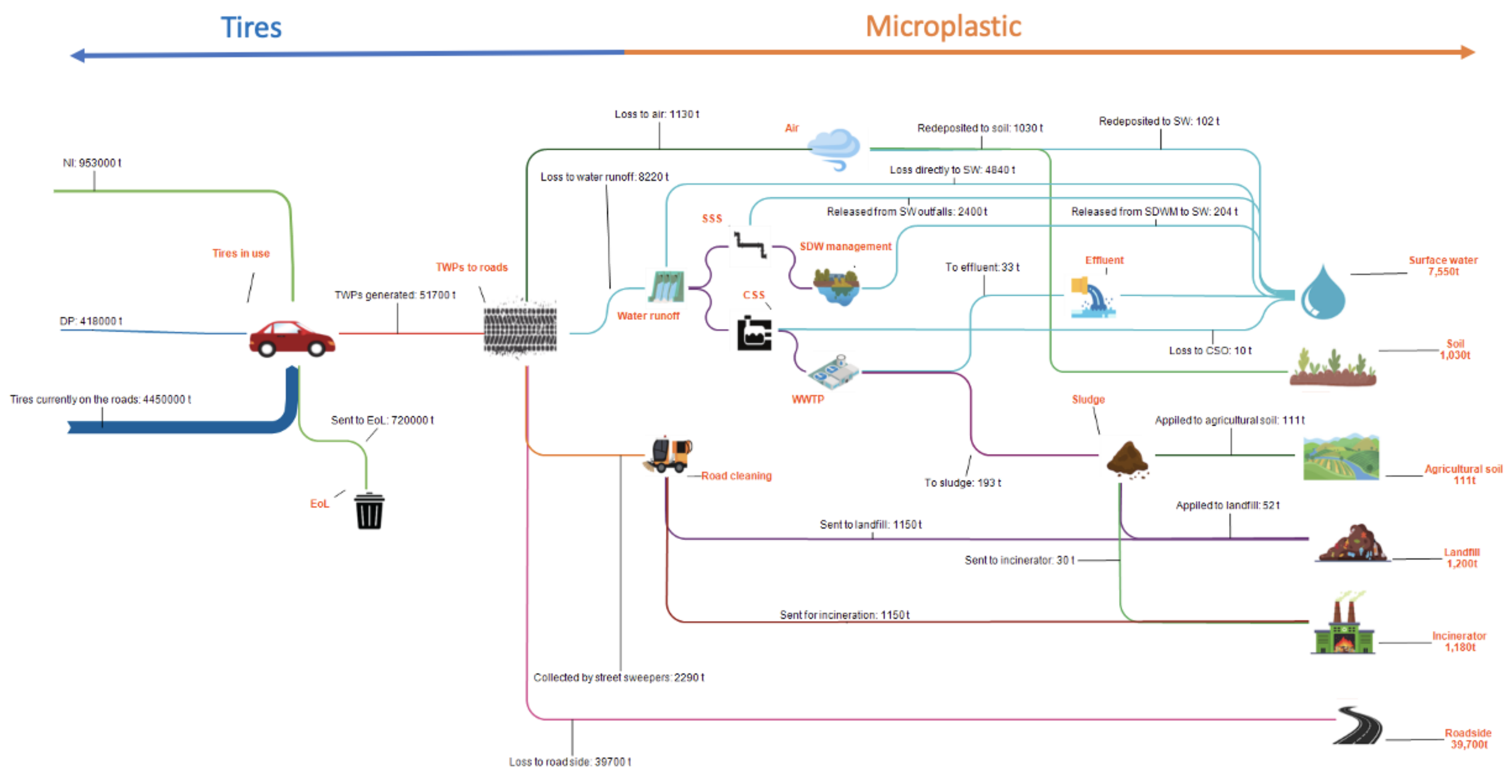


Figure 14: Aggregated emissions flows for tire wear particles from domestically produced tires in Canada in 2016. Each flow begins as macroplastics and is then transformed into a microplastic during its usage stage (except for domestically produced pellets which enter the system as pellets, whereas imported plastic is already in the form of a plastic product. The processes and flows are aggregated for visualization purposes to depict the general flows of each type. The flow's colour represents the receiving compartment, and the width of the flows qualitatively represents its magnitude.

## 4.2 Polymer Composition of Each Microplastic Flow

The polymer composition of microplastics can be found in Table 7. Vehicle tires were the highest emitters of microplastics; therefore, synthetic and natural rubber were the most common polymer types found in the environment, representing 85% of polymers released in 2016. The polymer type that was the second highest emitter was PMMA. This polymer type was comprised of microbeads, microfibers, road marking and building paint. PMMA represented 5% of polymers generated and released into the environment. The third and fourth highest emitters were epoxy and polyester, which comprised 3.6% and 2.6% of the total polymers, respectively that end up in the environment. Epoxy is found in paint on roads and buildings, while polyester is found in microfibers and road marking paint. The remaining polymers comprised less than 1% of microplastics found in the environment, including microplastic types such as paint application, microfibers, agricultural plastic mulch, microbeads, and pre-production pellets.

Table 7: Breakdown of each polymer type and its associated microplastic and the quantity emitted as a microplastic during production or use phases.

| Polymer Type | Type | Microplastic Type | Quantity (tonnes) |
|---|---|---|---|
| Synthetic rubber<br>Natural rubber | Elastomer | TWP | 51,300 |
| Poly methyl methacrylate (PMMA, Acrylic) | Thermoplastic | Building paint fragments<br>Road marking paint fragments<br>Microbeads<br>Microfibres | 3229 |
| Epoxy | Thermoset | Building paint fragments<br>Road marking fragments | 2155 |
| Polyester | Thermoplastic | Microfibres<br>Road marking | 1559 |
| Polyurethane (PUR | Thermoplastic | Road marking paint fragments | 1015 |
| Unsaturated Polyethylene | Thermoplastic | Building paint fragments | 798 |
| Ethylene-vinyl acetate | Thermoplastic | Pre-production pellets<br>Building paint | 158 |
| Polyethylene | Thermoplastic | Pre-production pellets | 148 |

| | | | |
|---|---|---|---|
| | | Agricultural plastic mulch | |
| | | Microbeads | |
| | | Building paint | |
| Other polymers | | Pre-production pellets | 114 |
| Polypropylene | Thermoplastic | Pre-production pellets | 82 |
| | | Microfibres | |
| Polyamide | Thermoplastic | Microfibres | 81 |
| | | Pre-production pellets | |
| Polystyrene | Thermoplastic | Pre-production pellets | 40 |
| | | Foam | |
| Elastane | Elastomer | Microfibres | 15 |

## 4.3 Analysis of the Final Compartments

This study considered six final compartments:

1. Surface water which includes freshwater and oceans
2. Soil which encompasses residential soil or nearby land excluding agriculture
3. Agricultural soil
4. Roadside, which indicates explicitly land next to roads.
5. Landfill
6. Incineration

According to the findings, roadsides received 68% of microplastics produced and generated, totalling 41,400 tonnes (Figure 7). The microplastics released to roadsides were TWPs and road marking paint. Surface waters were the second largest receiving compartment, and received a total of 9,160 tonnes, representing 15% of all microplastics produced and generated (Figure 7). Here, TWPs, building paint and road marking fragments were responsible for the majority of microplastic emissions (Figure 7, 15). The microplastics that contributed the lowest quantity to the final compartments included construction foam and microbeads (Figure 15).

Agricultural soil received the least quantity of microplastics compared to all the possible environmental compartments of microplastics and received 5% of microplastics. This was mainly dominated by the application of sludge to agricultural fields and the direct release of microplastics from agricultural plastic films (Figure 7, 15). Soil received 20% of microplastics, which was mainly due to the loss of building paint fragments from wear and tear and removal processes to soil and the redistribution of microplastics that were released to the atmosphere from TWPs, pre-production pellets and road marking paint fragments. Landfills and incinerators accounted for only 6% of the total microplastics exiting the system, and therefore, microplastics that were properly managed (Figure 16). Sources of microplastics to landfills or incinerators were from the disposal of sludge from WWTPs, collection and disposal from street sweepers, or direct entry to landfills through either the municipal waste system or industrial waste disposal.

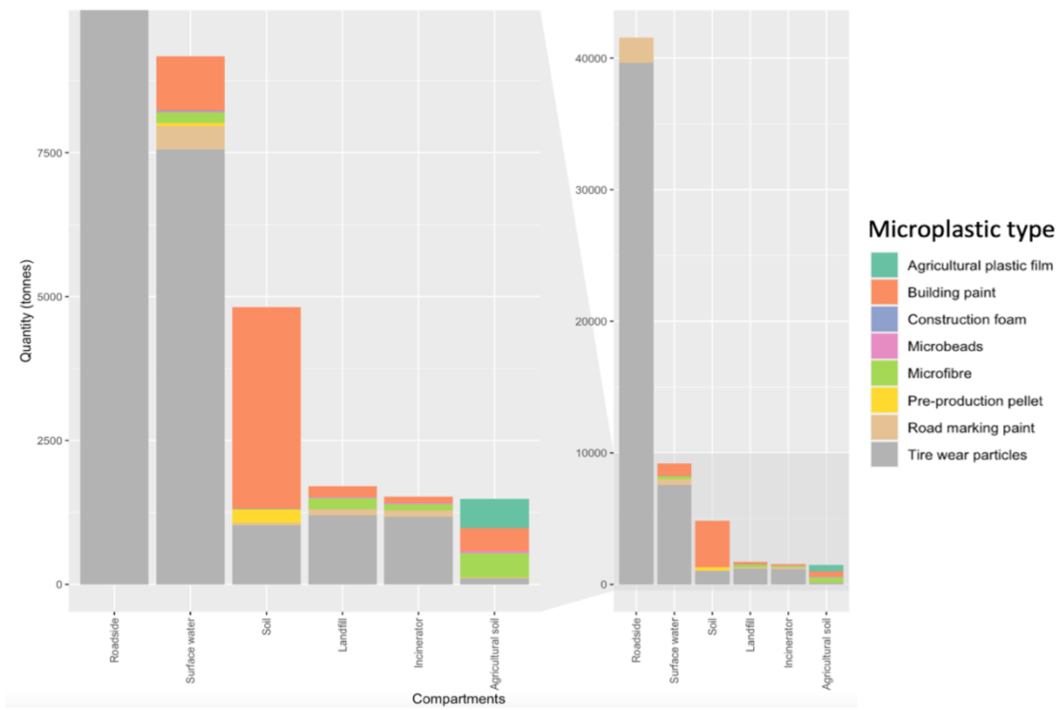


Figure 15: Comparison of each final compartment and within each bar depicts the microplastic type. Zoomed-in bar chart of Figure 15 focusing on pre-production pellets, microfibres, construction foam, agricultural plastic film, building paint, microbeads and road marking paint, reducing the scale to better visualize the quantities leaked/assigned to the environmental or technological compartments.

# Chapter 5: Discussion

The objectives of this study were to:

1. Quantify the flows of primary microplastics in Canada.
2. Identify the hotspot sources, flows, and sinks of microplastics in Canada.
3. Provide recommendations for microplastic mitigation strategies and how they can be managed in a circular economy.

This study provides the first assessment of primary microplastic flows in Canada, providing detailed Sankey diagrams of seven types of microplastics for the year 2016. The generation and release of these microplastics were then modelled across several processes and showcased various flows to the environment and technological compartments. To provide more granularity, the environment was divided into three compartments: surface water, soils, and agricultural soil. Lastly, based on existing literature, various mitigation strategies for managing microplastics are discussed, along with ways to incorporate them into the circular economy of plastics.

## 5.1 Flows of Microplastics in Canada and Applicability of the Model Outcomes

In 2016, 60,100 tonnes of microplastics were generated in Canada and released to six final compartments. The output to roadsides was 41,400 tonnes, surface water received 9,120 tonnes, soils received 4,790 tonnes, agricultural soils received 1,490 tonnes, landfill received 1,750 tonnes, and lastly, 1,540 tonnes was incinerated. A total of 16,400 tonnes of microplastics were estimated to be released to the environmental compartments. Building off Deloitte & ChemInfo Services (2019), who conducted an MFA of macroplastics across Canada for 2016, emissions of plastic waste may be greater than predicted as the original estimate of plastic waste to the environment was stated as 29,000 tonnes. However, combining the results of these studies, an estimated 45,000 tonnes of plastic waste was released into the environment in 2016. Interestingly, as per the findings of Boucher & Friot (2017), North America has a higher loss of microplastics than macroplastics, which is not what this study has found, however not all the sources addressed in Boucher & Friot (2017) were modelled for this current study, such as city dust (discussed in section 5.4), which could lead to an increase in the mass of micro vs macroplastics. While the model isn't an exact representation of the flows of microplastics in 2023, it provides a general overview of what is most likely entering terrestrial and aquatic bodies currently. Since 2016, the Canada's Zero Plastic Waste Agenda has been implemented along with two bans on plastic products, the first on the production and imports of microbeads in personal care products, and the ban on six single-use plastics (Government of Canada, 2017;

Government of Canada, 2021). In addition, general awareness of the plastics and microplastics problem has grown, and studies have shown that the public is concerned about their negative impacts on environmental and human health (Deng et al., 2020; BFR-Verbraucher, 2020; Catarino et al., 2021). However, other than the microbead ban, these developments would not impact the production or generation of primary microplastics. While general awareness of the issue may influence the public to change their behaviours, the awareness in Canada has been mainly focused on macroplastics and reducing plastic packaging items (Government of Canada, 2021; Walker et al., 2021). The results of this study are relevant to the current day, as the production of the macroplastics which these primary microplastics stem from has only increased in production as demand continues to increase. The flows and processes of the sankey diagrams remain the same. The only variable that would change is the width of the flows as the input of plastics increases.

## 5.2 Comparative Analysis of the Literature

This study found that, in Canada, TWPs were the most common type of microplastics released to the environment, contributing 8,690 tonnes of microplastics to surface waters, soils and agricultural soils. Boucher & Friot (2017) published one of the first reports on the global emissions of microplastics to the environment. Here they calculated that microfibres were the highest concentration of microplastics released to the environment, comprising around 35%. In comparison, TWPs comprised 28% of all microplastics released (Boucher & Friot, 2017). However, a more recent global estimation by Ryberg et al. (2019) found that TWPs were the largest contributors, comprising 47% of all microplastic emissions to the environment. This claim has also been made by other studies investigating microplastic emissions to the environment (Sundt et al., 2014; Magnusson et al., 2016; Bertling et al., 2018; Sieber et al., 2020). This current study found that TWPs comprised 56% of microplastics released into the environment. Mian et al. (2022) investigated the release of TWPs to surface waters using a freshwater transport model for the Okanagan Valley, British Columbia, Canada region. Here, Mian et al. (2022) found TWP emissions were between 4-23 tonnes per year for a 20km stretch of highway in the region. If this emission rate was extrapolated to the total length of highways found in Canada, TWP emissions to surface waters alone could be between 452,540 – 2,602,105 tonnes per year. In comparison, Dillon Consulting and Oakdene Hollins Limited (2021) were commissioned by ECCC to explore the circular economy for rubber and calculate TWP emissions for Canada using the emissions rate per capita method. They estimated that 35,900 tonnes of TWPs were released for the year 2020.

Recent studies have suggested that TWPs are the largest emitters of terrestrial soils rather than aquatic systems (Wagner et al., 2018; Hann et al., 2018; Knight et al., 2020; Baensch-Baltruschat et al., 2020; Sieber et al., 2020; Ding et al., 2022). The results produced from this current study do not agree with these findings, as surface waters received 7,550 tonnes while soils and agricultural soils received 1,140 tonnes. However, other studies consider roadsides to be included in the terrestrial soils category, whereas this study does not and categorizes roadsides as a technological compartment and therefore finds that roadsides receive the most TWPs (Wagner et al., 2018; Sieber et al., 2020; Prenner et al., 2021). Wagner et al. (2018) calculated TWP emissions for the USA using per capita emission factors. They found that the USA generates 1,120,000 – 1,800,000 tonnes of TWPs per year, as transportation largely depends on passenger vehicles rather than public transport (Malik, 2022). Although Canada is also car-dependent, the TWPs generated in the US are nearly 20 times that of Canada. Sieber et al. (2020) conducted a dynamic MFA from 1988-2018 and found that 41,500 tonnes of TWPs accumulated in surface waters, 9,600 tonnes accumulated in soils and 167,000 tonnes accumulated along roadsides. In comparison, Canada's TWP emissions could surpass the accumulation of TWPs in Switzerland over the past 20 years in only five years. Lastly, Prenner et al. (2021) conducted a static MFA of TWPs in Austria in 2018 and found that 4,835 tonnes entered the environment. Here, Prenner et al. (2021) did not differentiate between soils along roadsides or terrestrial soils. However, these comparisons highlight the difference in TWP emissions in North America compared to European countries, given the difference in population size, transportation infrastructure and driving habits. These differences in results also emphasize the need for localized mitigation strategies, as the release of microplastics can vary depending on the region.

A comparison of TWP emissions per kg/per capita/year can be found in Table 8. This study of Canada's TWP emissions is lower than that of Switzerland, Germany and the US. However, in comparison to other studies done for TWPs emissions in Canada, it is slightly higher than the estimation by Dillon Consulting and Oakdene Hollins Limited (2021), who calculated their TWP emissions using an average emission factor from 13 countries calculated by Kole et al. (2017). In comparison, Mian et al. (2022) calculated TWP emission using the vehicle emissions model, similar to this current study. The emissions calculated from this current study sit in the range of TWP emissions that Mian et al. (2022) have estimated, however on the lower end as they have predicted emissions to be as high as 4.2kg/per capita/year depending on road type, speed, and weather conditions. Therefore, given the methodologies and assumptions, the calculations from this current study and those from Mian et al. (2022) may be more reliable.

However, the lack of investigation of TWP generation and release from vehicles to roadsides, surface waters and soils requires further investigation. Most studies to date are using models to predict TWP generation and release rather than collecting empirical data (Mennekes & Nowack, 2022). The wide range of emissions reported emphasizes the need for standardization in how TWPs are measured, along with the need for research on the ecological effects of TWPs on terrestrial soils and roadsides to be investigated.

Table 8: Comparison of the quantity of TWPs from vehicle tires

| Geographic scope | TWP emissions (kg/per capita/year) | Source |
|---|---|---|
| Canada | 1.42 | This study |
| Canada | 0.95 | Dillon Consulting and Oakdene Hollins Limited (2021) |
| Canada | 0.8 - 4.2 | Mian et al. (2022) |
| Switzerland | 0.8 - 1.7 | Sieber et al. (2020) |
| Austria | 2.4 | Prenner et al. (2021) |
| USA | 5.5 | Kole et al. (2017) |

Pre-production pellets have received increasing media attention as they are tied directly to plastics production and manufacturing facilities (Little, 2019; McVeigh, 2021; Fawcett-Atkinson, 2021). Although these are considered unintentional losses, they are still lost with often little effort for recovery (Corcoran et al., 2020). The results of this current study found that a total of 308 tonnes of pre-production pellets are lost from the production, manufacturing, and transportation stage. In comparison, Boucher & Friot (2017) estimates a global release of 960,000 tonnes, while Kawecki &

Nowack (2018) found their pre-production pellet emissions to be approximately 110 tonnes for 2014 in Switzerland. Hann et al. (2018) estimated the loss of pellets as 16,888-167,431 tonnes of pellets per year in the EU. A comparison of pellet loss to a more similar socioeconomic region, such as the US, has yet to be conducted. The model produced in this current study suggests the largest source of pellet emissions to the environment is soils nearby the production site or transportation route, as this is where spills most commonly occur (OSPAR, 2018; Policy Manager from Chemistry Industry Association Canada, personal communication August 9th, 2022). This is likely to be the case for other regions with pellet spills.

Researchers investigating the pollution of pre-production pellets in Canada have focused their attention on the redeposition of pellets to beaches and coasts of the Great Lakes and nearby tributaries (Ballent et al., 2016; Corcoran et al., 2020; Earn et al., 2021; Arturo & Corcoran 2022). Corcoran et al. (2020) conducted a study across three Great Lakes. They found that across the 66 beaches along Lake Huron, Superior, and Ontario, 42 contained pollution from plastic pellets; on average, 19 pellets/m2 were present. Based on the results of this current model, spills to surface water only represent 21% of the loss to the environment and further emphasizes how terrestrial environments remain to be under-investigated, especially for the Great Lakes region (de Souza Machado et al., 2017; Dissanayake et al., 2022).

Surprisingly, microfibre emissions across Canada did not fall under the top three microplastics released to the environment, contradicting findings across other MFA studies (Boucher & Friot, 2017; Ryberg et al., 2019; Xu et al., 2020). Based on the findings of this current study, microfibres comprised 1% of the total microplastics generated and released. In contrast, Gavigan et al. (2020) conducted a global dynamic MFA of microfibres from clothing and for North America in 2016 estimated 49,000 tonnes of microfibres were released. They found that most microfibres ended up in terrestrial environments at 22,000 tonnes, while surface waters received the second highest quantity at 11,000 tonnes, 10,000 tonnes were sent to landfill, and 6,000 tonnes were incinerated. Gavigan et al. (2020) found North America had the highest in-use stock per capita of synthetic fibre apparel, at 0.062 tonnes/per capita. This is mostly likely due to consumers located in the USA, which is corroborated by the findings of a study conducted in California (Geyer et al., 2022). Geyer et al. (2022) determined that the in-use stock of microfibres in clothing was 2,600,000 tonnes in 2019. In contrast, the model in this current study estimated 609,000 tonnes in 2016. This comparison is important as socioeconomic factors between the US and Canada are relatively similar, and the population of California and Canada has always been similar, thereby accentuating the difference in clothing consumption and, therefore, microfibre release.

Geyer et al. (2022) estimated that the state of California in 2019, had a total of 2,220,000 tonnes of microfibres shed into the environment, where 1,650,000 tonnes were applied to terrestrial soils, 751,000 tonnes were incinerated, 385,000 tonnes were sent to landfill while 110,000 tonnes were released in effluent and then to surface waters. Although this current study was conducted for the year 2016, the microfibre emissions were significantly lower, generating a total of 903 tonnes of shedding from washing machines and even including emissions from dryers which was not included in Geyer et al. (2022) or Gavigan et al., (2020). This suggests, as expected, the USA is a much larger contributor to the North American emissions of microfibres. However, the release of microfibres to different environmental compartments was similar to that of Geyer et al. (2022), where most microfibres were applied to land and released in effluent to surface waters. However, in comparison to other studies such as Kawecki and Nowack (2018) reported in 2014, Switzerland had 33 tonnes of microfibres emitted to their environment, while Wang et al. (2019) estimated 2,120 million tonnes of synthetic fibres to be emitted into the environment in 2015 in China, which relative to each country's population seems probable. While microfibre emissions based on this current study are not one of the top microplastics released to the environment, the literature provides insight into a source of emissions which can be better controlled through technological solutions and prevented relatively easily (Henry et al., 2019; McIlwraith et al., 2019; Gaylarde et al., 2021).

Building and road marking paint rank as Canada's second and third largest source of microplastics. Research into paint fragments is limited and often overlooked where in the literature; the justification for this is due to the limited polymer content present in paints (Horton et al., 2017; Turner, 2021). In 2022, Environmental Action released a worldwide report stating that six sectors producing paint emissions are the largest emitters, potentially releasing 1,857,000 tonnes of microplastics annually (Paruta et al., 2022).

The results of this current study suggest that building paint is responsible for the majority of microplastic loss to soils; as shown in Figure 12, most paint fragments remain on soil nearby instead of being transported in runoff and represent the major source of microplastic emissions to soil. Compared to TWPs, pre-production pellets, and microfibres, there is a lack of research regarding the emissions from paint fragments to the environment (Turner, 2021). Environmental Action released the first global report on paint emissions from seven sectors (Paruta et al., 2022). From the architectural and road marking sector, Environmental Action estimated globally that 3,748,000 tonnes of paint fragments are released into the environment annually (Paruta et al., 2022; Appendix B). In their study, seven paint sectors were investigated, and for the North American region, they found that between 56,000 and 280,000 tonnes of microplastic paint fragments were released (Paruta et al., 2022). The findings from Paurta et al. (2022) are similar to what was found in this current study, as they determined that 74% of paint fragments from the architectural sector end up on land, while this study found that 68% of paint fragments end up on land.

As for road marking paint, Paruta et al. (2022) found that globally, 172,000 tonnes of road marking paint were used, with 91,000 tonnes lost to oceans and 81,000 tonnes to land. Within North America, Paruta et al. (2022) estimated that 28,000 tonnes are released into the environment. In comparison, Boucher & Friot (2017) measured global emissions for road marking fragments and found that 15,400 tonnes are released yearly in North America. Magnusson et al. (2016) estimated that 504 tonnes of microplastics per year are generated and potentially lost to the environment in Switzerland. This current study found that 2,340 tonnes were released to the final compartments; however, 541 tonnes were released to the environment. Canada is a car-reliant country, and according to a survey conducted by an insurance company, it ranks 4th in the world, indicating why the generation of road marking fragments is significantly higher compared to Switzerland; however, the estimated quantity released to the environment is quite similar (Malik, 2022). Although the mass generated is not comparable to that of the findings of Boucher & Friot (2017) and Paruta et al. (2022), this may be due to most of the generation and release of road marking paint fragments being dominated by the production from the US.

Lastly, agricultural plastic film and construction foam have only been investigated by Kawecki and Nowack (2018). However, the mass released from agricultural plastic film and construction foam is unclear as they also investigated microplastics from other construction and agricultural activities. However, Gao et al. (2023) investigated the presence of foam across multiple beaches in Toronto, Canada and found that 58% of samples were suspected to be construction foam. As for emissions from agricultural mulch, further studies are needed to provide more granular flows and pathways for these sources (Kawecki & Nowack, 2018).

## 5.3 Uncertainties and Limitations of MFA Model Outcomes

MFAs require a wide range of data and, in turn, involve an assortment of assumptions. Researchers agree that MFAs are insightful in principle, but their reliability has been questioned due to data limitations and inherent uncertainty in the analysis (Laner et al., 2014). Many of the limitations in this study have already been echoed in the literature, especially by Wang et al. (2021) and Mennekes and Nowack (2022), who stated that missing data, conflicting data, and reuse of existing studies to determine microplastic emissions instead of producing new shed/production rates of different microplastic types to gain a larger set of data and increase its reliability. It is well noted that the methods to gather comprehensive data on microplastic sources, their pathways, flows, and quantities can be challenging given their diversity of materials (Cesa et al., 2017; Stock et al., 2020; Shruti et al., 2021). This inevitability is a common limitation of microplastic research.

This current study is the first attempt to quantify primary microplastic emissions within Canada, which can be further improved with more accurate and Canadianized data. However, calculating the mass of microplastics with the MFA approach was based on data and assumptions with various geographic and temporal differences, resulting in large spreads and high uncertainties. The results of this sort are challenging to analyze for uncertainty due to a lack of a robust methodology to quantify the uncertainties associated with each flow and microplastic type. Therefore, the values should not be treated as approximations to show comparisons of microplastic types and their relative contribution to environmental emissions. For more robust results, further research on microplastic emissions, transport and degradation within the Canadian context and globally is required, which would strengthen model estimates and provide validation.

The main drivers of uncertainty for this investigation regard the reliability of the data, geographic and temporal relevance, and microplastic type. The current model relies heavily on reports and literature which determined the loss and generation rates of microplastics on expert estimates and little on empirical data (Hann et al., 2018; Mennekes & Nowack, 2022; Horton, 2022). For example, building paint emissions were based on expert assumptions pulled from an OECD report and based on expert assumptions from the European Council of the Paint, Printing Ink, and Artist's Colours manufacturers (Hann et al., 2018). Loss of pre-production pellets from manufacturing facilities is based primarily on one study conducted on a Norwegian processing facility, and the remaining estimates are based on expert estimates from Operation Clean Sweep or Plastics Europe (Hann et al., 2018). In comparison, microfibre shedding rate estimation from clothing is based on over ten studies which collected primary data (Peano et al., 2020; Geyer et al., 2022). In addition, most of the literature used is from Europe and Asia. There needs to be more research regarding microplastic generation in Canada (Ryberg et al., 2019; Hoseini & Bond, 2022; Paruta et al., 2022). As for the temporal correlation, much of the input data collected was within a 10-year range of 2016, where some data was from 2022 or as far back as 2009 (Cheminfo, 2022; Statistics Canada, 2009). However, most of the data collected was between 2010-2016, consistent with the increased research conducted on microplastics.

Through this investigation, agricultural plastic film, TWPs and paint from road marking are subject to greatest uncertainty due to a lack of data regarding their fate and transport throughout the environment from source and non-source points; in addition, educated guesses were made without verifiable assumptions. This was done specifically for the redeposition of TWPs and road marking fragments from air to surface waters and soil, and their disposal from street sweepers to landfills or incinerators. In comparison, results on microfibre emissions are more robust, as most assumptions are based on temporally and geographically relevant sources, except for assumptions regarding the current stock of clothing in Canadian wardrobes and the use frequency of dryers.

The uncertainty associated with TWPs has been discussed more, and researchers are concerned that current studies continue to utilize expert assumptions with little empirical data to support them (Mennekes & Nowack, 2022). Knight et al. (2020) claim a considerable lack of empirical data regarding TWP abundance and distribution across roads, drains and the environment. In addition, Mennekes and Nowack (2022) highlight the lack of TWP emission measurements across reports, and that country-based emission calculations are based 12% of the time on experimental data, while 63% of the time, they are based on reviews. In addition, over half of these studies are over 50 years old, severely affecting the integrity of estimates produced by researchers (Mennekes & Nowack, 2022). This current study's input data for TWPs was based on vehicle kilometre data from 2009, and emission factors were based on one of the few empirical studies of TWPs conducted in the Netherlands (Statistics Canada, 2009; Baensch-Baltruschat et al., 2020). Improving the quality and quantity of TWP emissions for Canadian vehicles will enhance the validity of the data, therefore, lower the uncertainty. Measurement studies of TWPs are necessary for Canada to understand the various characteristics and parameters that impact the emissions rate onto different road types and seasons. A field study conducted in the Greater Toronto Area in Canada took samples of WWTP and stormwater effluent and quantified and characterized the microplastics present (Grbic et al., 2020). Based on their findings, WWTPs effluents contained less than 1% of TWPs, and of this, 90% were microfibres; whereas, stormwater samples contained around 22% TWPs, and 41% microfibres. The results from this current study do not reflect the findings from Grbic et al. (2020) and suggest that this study underestimated the microfibre generation and release as well as overestimates the release of TWPs to surface waters. The results of this study provide a practical, informative starting point toward identifying the major leakage points of TWP to the environment; however, given the data available and the uncertainty regarding the data, they echo the same sentiments as Knight et al. (2020) and Mennekes and Nowack, (2022).

Another significant source of uncertainty was assumptions of the behaviour of each type of microplastic. For example, the removal efficiencies of different WWTPs have been investigated; however, the majority of the effluent samples quantified and characterized have found microfibres, fragments and microbeads rather than TWPs. However, given the lack of data regarding the fate of TWPs in WWTPs, it is assumed they will behave similarly to that of other microplastics regardless of the difference in polymer type and density.

The need for measurement data regarding TWPs and paint emissions from road markings and building paints is echoed as researchers begin to state that paint may be the greatest source of microplastic emissions to the environment (Paruta et al., 2022). A deeper mechanistic understanding supported by direct experimental measurements of environmental processes including degradation, fragmentation, biofouling, resurfacing, and sedimentation, is needed. This is especially important as currently researchers assume all microplastics behave the same way, regardless of varying polymers and, therefore, densities.

In addition, the confidence in the results of this study would be greatly enhanced if more primary data collection was conducted. Canada lacks a detailed account of the mass of plastics throughout their lifecycle, in addition to the plastics stocks for each category. The sources and accumulations of microplastics vary depending on their microhabitats, and although global and national assessments are useful in pinpointing where the general areas of work lie, assessments on a provincial or municipal level may be more helpful in addressing specific areas to manage and implement solutions.

Based on the data collected and the uncertainty associated with each flow, the study is confident in the results produced for the release of pre-production pellets, microfibres, microbeads, and construction foam. Subsequently, for pre-production pellets and microfibres, technological solutions have been tested and shown to reduce microplastic emissions to waterways. For example, for pre-production pellets, a trash capture device called LittaTrap was installed in four stormwater drains at an operating facility, and in 289 days, they successfully captured and removed 34,766 pellets from entering stormwater drains (Tiddy et al., 2021). As for microfibres, two devices have been tested which are installed or placed into a washing machine and have removed from 41-87% of microfibres from effluent drains (McIlwraith et al., 2019; Erdle et al., 2021).

## 5.4 Other Sources of Microplastics

The eight sources discussed in this current study are the most investigated and best understood in the literature and, therefore, are most commonly included in other microplastic MFA transport models studies (Boucher & Friot, 2017; Hann et al., 2018; Kawecki & Nowack, 2019; Schwarz et al., 2023). However, there are several other sources mentioned in the literature that lack comprehensive investigation and have yet to be quantified (Magnusson et al., 2016; GESAMP, 2019). In Table 9, a summary of the sources investigated in this study is listed, along with potential sources of microplastics which have input data but lack data or literature regarding their generation rate. For example, there is data regarding the mass of plastic packaging produced in Canada and released to the environment; however, the rate that leaks into each specific environmental compartment and the degradation rate once in the terrestrial soils or surface waters are unavailable. This is due to the various conditions that influence plastic degradation in the environment, i.e., sunlight, moisture, pH, temperature, oxygen, and biota present (Kale et al., 2015; Bacha et al., 2023).

Table 9: Microplastics investigated in this study compared to sources with no input data or data regarding generation rates during use or loss.

| **Microplastics investigated in this study** | **Known sources of microplastics with input data but no generation rate** | **Known sources of microplastics no input data and no generation rate** |
|---|---|---|
| Pre-production pellets | Marine paints (commercial and recreation boats) | Abrasive blasting with microbeads |
| Microfibres from clothing in washing machines and dryers | Household textiles | Street sweepers from road cleaning |
| Construction foam | Agricultural plastics (pots, pipes, bales, nutrient pellets) | Artificial turf |
| Agricultural plastic mulch | Construction dust (pipes, window frames, floors etc.) | Pharmaceutical products |
| Tire Wear Particles | Automobile plastic parts | Compost |

| | | |
|---|---|---|
| Paint from buildings | Footwear | Fishing gear |
| Paint from roads | Plastic packaging | Cigarettes |
| Microbeads from personal care products | Landfills | Litter |
| | Microplastics from recycling facilities | Household dust (toys, foam rubber, kitchen ware, electric wiring, electronics, cleaning agents) |

Similarly, the shedding of microplastic fibres from household textiles has only recently gained recognition as a significant source, with insufficient evidence so far to adequately quantify fibre release rates and dispersion patterns. To quantify this source would require surveys regarding the behaviours of Canadians on how often they wash household textiles, the shedding rates of the various types of furniture over time and how much is vacuumed compared to mopped, as this would determine their flows to the environment (Magnusson et al., 2016).

The agricultural sector's role in microplastic generation remains relatively unexplored, with a lack of comprehensive data on the degradation rates of different plastic items and their subsequent contribution to soil and water contamination while in use. Environment Canada commissioned a non-profit environmental stewardship program, Clean Farms, to investigate the plastic generated from Canadian farming (Clean Farms, 2021). However, due to the lack of literature regarding the degradation rate or generation rate of microplastics from agricultural plastics in use, microplastics could not be quantified.

The construction industry's impact on microplastic pollution has often been overlooked, even though it is responsible for 14% of yearly plastics output (Schwarz et al., 2019). A US Environmental Protection Agency study found that the production process for microplastics in the construction sector might be one of the most hazardous sources overall (Mahon et al., 2014). It is clear this source may be associated with high risk; however, again, there is a lack of empirical studies investigating their generation rate and potential flows to the environment. Kawecki and Nowack (2018) attempt to quantify microplastics from construction pipes; however, it is based on expert assumptions and their own educated guesses. Furthermore, the emission of microplastics from rubber footwear and plastics from automobiles has not been thoroughly studied. Lassen et al. (2015) assumed that 10% of footwear soles are worn off over their lifetime, while the literature only discusses macroplastics from automobiles.

Automotive parts are suspected to generate microplastics (Lassen et al., 2015; Kawecki & Nowack, 2018; Forster et al., 2020). Microplastics generated from recycling facilities have gained some attention since 2022, and a recent publication by Stapleton et al. (2023) provided one of the first-generation rates of microplastics for various polymers in a facility in Australia. Recycling facilities were not included in this study as it was published weeks prior to the completion of this study; however, it can now be estimated in future research.

In the last column, there is a list of potential sources which lack input data and generation rates for Canada. The release of microbeads from abrasive blasting with microbeads has been investigated in Europe; however, they are based on assumptions from interviews with industry stakeholders (Verschoor et al., 2016). Other unsuspecting sources include street sweepers, artificial turf, pharmaceutical products, and compost. Each has not been extensively investigated and was only mentioned as a potential source of microplastic generation was calculated based on unverifiable assumptions (Kawecki & Nowack, 2018). Research for these sources has not been conducted in Canada. The gaps in knowledge extend to overlooked areas like discarded fishing gear and litter, which may act as significant sources of secondary microplastics to surface waters (Boucher & Friot, 2017). The lack of literature and investigation across these sources emphasizes the pressing need for expanded research efforts and a holistic approach to comprehensively understanding and addressing microplastic pollution.

## 5.5 Implications for Canada's Zero Plastic Waste Agenda and Circular Economy Initiative

To support the efforts of the strategy and agenda, the results of the study suggest Canada focus on the following microplastics in the ranking below based on the quantity entering the environment:

1. TWPs
2. Paint
3. Microfibres
4. Agricultural plastic film
5. Pre-production pellets
6. Construction foam

Microbeads have been removed from the assessment since the ban was enacted in 2017. The results from the investigation fill a large gap missed in the initial release of the Canada-wide Strategy Zero Plastic Waste Action Plan, one which can help better focus researchers on which ecotoxicological effects may be relevant for nearby surface waters, roadsides, soils and agricultural soils. It can also help inform branching initiatives such as the Canadian Plastics Innovation Challenge, which are seeking sectors that generate the most plastic waste/pollution to provide solutions to reducing plastic waste while creating economic growth (ECCC, 2021a). Here the results from this study show that the investment for mitigating the release of microplastics from TWPs is a priority. As a next step, an investigation should be directed toward paints, from road painting and buildings and mitigating their release as it is second on the priority.

The investigation into microbeads may seem repetitive, given the ban put in place in 2017 (Government of Canada, 2017). However, the results of this study showcase the success of this ban in that it mitigates approximately 69 tonnes of microplastics to the environment (Figure 9). While bans such as these cannot be put in place for other primary microplastics investigated in this study due to their method of generation, the MFA showcases areas of leakage where policy or technology may have a greater impact now. For example, microbeads in abrasive materials such as plastic blasting and automotive moulding were disregarded. While these potential sources were not covered in this analysis due to the sheer lack of data, the presented flows from other microplastics showcase their ability to surpass WWTPs and quantities that escape through stormwater systems, which would be the more likely route of microbeads from these sources.

Canada's strategy towards a zero plastic waste economy does not address how microplastics can be prevented, collected, or recovered and, therefore, what their role is in a circular economy (CCME, 2018). Syberg et al. (2022) propose two concepts for implementing a circular economy to include microplastics. First, future development in product design and the use of new or existing polymers should focus on being more durable with less toxicity than those currently existing for plastic products. In addition, they should be designed with the reuse and recycling potential in mind. Furthermore, Syberg et al. (2022) specifically state that the potential loss of microplastics to the environment must be accounted for throughout the life cycle of a given product, and the next generation of polymers should have much shorter environment turnover times compared to existing polymers (Syberg et al., 2022). Futhermore, they highlight that when new materials and products are developed, an assessment of their environmental fate should be done; this current study does report the types of microplastics released into each environmental compartment, the scope was to assess what may occur within the year 2016 and it is clear over time the accumulation in microplastics in different compartments such as agricultural soil or roadside may not remain entirely, and may be transported to surface waters or nearby land through different means such as groundwater or water runoff (Gnecco et al., 2005; Huang et al., 2023; Luo et al., 2021). Moreover, when microplastics reach surface waters, they can be remobilized onto land (Earn et al., 2020; Anik et al., 2021; Kallenbach et al., 2022). This study can act as a starting point for further environmental fate models to be conducted to understand the final fate of microplastics throughout different ecological cycles (Schwarz et al., 2023).

Lastly, the results of this work can aid in pinpointing where technologies and remediation solutions can be implemented along the life cycle of different types of plastic products. Although the methodologies and technologies are still underdeveloped, methods such as absorption, ultrafiltration, membrane technology, and degrading/mineralizing microplastics into harmless compounds such as CO2 or H2O through photocatalysis, advanced oxidization process or microbial degradation offer the opportunity to utilize microplastics as a feedstock in creating new bioplastics or biodegradable polymers (Cholewinski et al., 2022; Chen et al., 2022). In tandem with these remediation solutions, plastic clean-up technologies should be deployed, which are gaining popularity among community groups and municipalities (Schmaltz et al., 2020; Sherlock et al., 2023; Brouwer et al., 2023). Using plastic clean-up technologies as a method of accumulation can provide another avenue and potential to provide a substantial feedstock for the remediation solutions. It is important to explore these potential solutions to reverse the current value of microplastics. As these solutions are further explored, the processes explored in this study should be used to map where remediations and technologies should be implemented.

## 5.6 Future Directions for MFA Work on Microplastics

Microplastic MFAs have only been conducted on a national or global level. The results from these studies provide an estimate of where leakage points are occurring throughout the plastics life cycle; however, more granular flows should be conducted to provide better-localized recommendations for regions or municipalities. Before this is possible, it is important for primary data collection regarding the estimates of microplastic releases or losses across the various compartments depicted in the Sankey diagrams above. For example, the estimate that 0.01% of pre-production pellets are lost to the environment is based on a study conducted in Norway (Hann et al., 2018). However, it is used to estimate pellet loss globally and in various microplastic emission or

MFA studies (Sherrington et al., 2016; Kawecki & Nowack, 2018; Peano et al.., 2020; Liu & Nowack, 2022; Hoseini & Bond, 2022).

Localized studies for TWP, construction foam, paint and agricultural plastic film are highly recommended to gain a better insight into microplastic generation and release to the environment, as Canadian behaviours are different from that of Europeans in terms of car dependence, rate of construction and agricultural production. In addition, a better understanding of microplastic accumulation and release from sewage systems, combined and separate, is necessary. The results of this current study suggest that stormwater outfalls are a significant pathway for microplastic leakage to surface waters, which is consistent with the literature (Shruti et al., 2021). There is only one Canadian study regarding the number of microplastics in biosolids and how much is being applied yearly to agricultural soils (Gies et al., 2018). However, these studies, similar to the majority of the microplastic studies, do not record the number of microplastics found in samples by mass, rather by count due to difficulty in obtaining an accurate weight, but there have been calls to standardize reporting for different microplastic contaminations in different environments (Mai et al., 2018; Dioses-Salinas et al., 2020). A solution can be to report whatever measurement is possible based on the samples collected from the study, which allows the results to be utilized for different investigations and comparisons.

Flows that should be further investigated for Canada include the mass and number of microplastics removed from municipal street sweepers and whether their removal of microplastics on roads is discounted by their generation of microplastics while in use. Here, an estimate from the City of Toronto street sweepers was used to determine how much is removed from urban roads (City of Toronto, 2015). However, this estimate was from a report conducted in 2014 and may vary across the nation depending on the municipality budget and the quality of street sweepers. This has not been investigated as a microplastic removal technology. In addition, given the climate in Canada, snow removal is a common activity in both urban and rural areas, more so urban. It is assumed to play a large part in how TWP and road marking fragments end up in each environmental compartment (Vijayan et al., 2022). Based on conversations with snow removal companies and municipalities, when snow is removed from roadways, it is either piled to the side of the road onto land nearby or, if it is too much, it will be relocated to a designated snow dump site. Suppose municipalities need to dump snow at a designated site. In that case, they depend on the soil to reduce pollution from heavy metals and phosphorus and a likely sink and accumulation zone for microplastics over time. (Ministry of the Environment, Conservation and Parks, 2021). Lastly, a source of microplastics which may act as a substantial source is the marine coating from boats and ships (Paruta et al., 2022; Turner, 2021).

Canada has a large fishing industry representing 6 billion dollars, the second largest food export in 2015, and a recreational boating industry representing a 5-billion-dollar industry and an uninvestigated direct source of microplastics to marine and freshwater systems. (Fisheries and Oceans Canada, 2020).

Potential sources such as recycling, littering, other agricultural plastic waste and mismanaged waste were not included in this assessment; however, they have been covered in previous studies (Nizzetto et al., 2016; de Souza Machado et al., 2017; Kawecki & Nowack, 2018; Hoseini & Bond, 2022; Liu & Nowack, 2022). In a future study, this could be assessed once input data for these sources has been collected.

The results of this study help assess the ecological hazards of microplastics in Canada. The MFA has provided insight as to which microplastic types are being released into the environment the most. For example, based on this study's results, further investigation regarding the toxicity of TWP leachate is recommended, as 39,000 tonnes is estimated to end up along roadsides. Thus far, research has focused on the toxicity of TWPs through leachate to aquatic environments, while the effects of TWP emissions on terrestrial or roadsides have been neglected (Ding et al., 2022). Studies that have investigated TWP effects on aquatic systems have only been short-term studies and have investigated the ingestion and retention of TWPs, the effects of leachates on organisms, and reproduction and growth (Halle et al., 2020). However, these studies mostly used unrealistically high concentrations and leachates of laboratory-produced tire particles rather than real TWP, given the difficulty of obtaining it (Wagner et al., 2018). Therefore, direct effects of TWP have rarely been studied (Wagner et al., 2018). However, this leaves an important question: What is the implication of 39,000 tonnes of TWPs to roadsides on nearby soils and surface waters, and what is their fate over time? It is recommended that similar assessments be carried out for paint fragments, as, to the best of my ability, no studies investigating their ecological effects on biota.

Lastly, this study can be viewed as a foundation for more detailed MFAs for provinces or municipalities and environmental fate models, as microplastics are affected by runoff, sedimentation, fluvial transport, fragmentation, and degradation. For provinces or municipalities, conducting such an MFA would be extremely beneficial as each has its regulations regarding the disposal of biosolids, application to agricultural soils and emissions through stormwater drains.

# Chapter 6: Conclusion

This research quantified stocks and flows of various microplastics across Canada in 2016 using an MFA. The results of this study indicate that Canada's roadsides receive the most microplastic pollution, followed by surface waters, soils, incinerator, landfill, and agricultural soils. Regarding environmental compartments, surface waters received the most at 9,160 tonnes. This study found TWPs are responsible for most microplastic releases in Canada, a total of 50,700 tonnes, followed by building paint fragments at 5,070 tonnes, road marking paint fragments at 2,650 tonnes, microfibres at 913 tonnes, agricultural plastic film at 505 tonnes, pre-production pellets at 294 tonnes, microbeads at 110 tonnes and construction foam at 22.8 tonnes.

Primary microplastics are released during the use phase of a macroplastic product and are mostly generated unintentionally due to the wear and tear of different plastic products. Significant flows that determine microplastics' fate to the environment are largely determined by water runoff from roads to roadsides and stormwater systems. The findings of this study do not confer with the findings from previous literature, as researchers have suggested microplastic accumulation occurs more in terrestrial environments than in surface waters. However, the lack of granular data for microplastic flows in Canada has been a large limitation of this study. Further investigation regarding the generation and flows of TWPs and paints in Canada is necessary to provide sound data regarding their risk and prevalence in Canada.

The discussion of how microplastics should be integrated into a circular economy is still in its infancy (Sadeghi et al., 2021; Syberg et al., 2022; Sarkar et al., 2022; Fuschi et al., 2022). Most studies state microplastics can be mitigated with better product design and utilizing better quality polymers to reduce the wear and tear of plastic products currently used. In addition, how legislation such as the microbead ban has directly prevented the intentional release of microplastics to surface waters in Canada. However, similar bans cannot be put in place for the majority of microplastics as their release is unintentional. Cholewinski et al. (2022) are one of the few studies that consider how microplastics currently polluting terrestrial and aquatic environments can be dealt with in a circular economy and utilized as a feedstock in creating new plastic products through microbial degradation. However, the technology to do so remains in the preliminary stages in theme with microplastic research in other disciplines, lacking standardization and universally accepted methodological protocols.

Horton (2022), in her recent perspective article regarding plastic pollution, has posed the question, "When do we know enough to act?". Canada has made large strides in addressing the plastics and microplastics problem. However, significant knowledge gaps and uncertainties remain regarding the production and generation of microplastics and their various flows to each environmental compartment. Researchers continue to argue that despite these uncertainties, ample literature showcases the threats plastics can pose in various conditions, which justifies action to be taken immediately (Besseling et al., 2019; Horton, 2022). This study can help inform which microplastic types are a priority to better understand their potential. Here, TWPs are released significantly compared to the remaining; therefore, research regarding realistic particle types, concentrations and chronic exposures should be addressed in terrestrial and aquatic environments. More importantly, the results of this work provide a starting point for more detailed investigations of primary microplastics in Canada, where uncertainties in the model can be addressed to provide a more accurate representation of the leakage occurring in Canada. As the Government of Canada continues to roll out its action plan, the model above can assist in understanding the dominant sources of primary microplastics and echoes the need for cross-disciplinary collaboration between biologists, chemists, material scientists, economists, industry, and community members to consider localized strategies to mitigate microplastic generation (Wang et al., 2021; Horton, 2022).

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# Appendices

## Appendix A: Building Paint Calculations

Table 1: Parameters adapted from Verschoor et al., 2016 to calculate the paint on residential buildings across Canada in 2016

| | Urban | Rural |
|---|---|---|
| Number of houses in Canada | 9,835,655 | 42,364,425 |
| Sides of the house that need painting | 3 | 4 |
| Length of house sides (m) | 8.2 | 8 |
| Average number of floors | 1.5 | 2 |
| Height of one floor (m) | 3 | 3 |
| Outside surface area that needs painting (m2) | 1,088,807,009 | 813,393,600 |
| Percentage of paint used | 57% | 43% |

Table 2: Parameters and total quantities in kilotonnes of paint lost from wear and tear and removal of paints from residential buildings in Canada adapted from Verschoor et al., (2016). To deduce the quantity of paint that goes to exterior vs interior parts of the building we assumed 73% as per stated by Verschoor et al., 2016, Hann et al., 2018, Paruta et al. (2022) of paints are for interior use while 27% is for exterior.

| Product group | *fused* | *fexterior* | $EF_{removal}$ | $EF_{wear}$ | $E_{removal}$ | $E_{wear}$ | $E_{removal}$ total (kt) | $E_{wear\ tear}$ total (kt) |
|---|---|---|---|---|---|---|---|---|
| Professional use in the building and construction sector (exterior) | 52% of paints are used by professionals (Royal Bank of Canada, 2006) | | | | | | | |
| Concrete paints | | 0.5 | 0 | 0.03 | 0 | 0.32 | | |
| Lacquer, varnish, undercoats and primer | | 0.4 | 0.03 | 0.03 | 0.28 | 0.26 | | |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Wood stains | 0.97 | 0.25 | 0.03 | 0.03 | 0.17 | 0.16 | 1.07 | 3.43 |
| Wall paints | | 0.07 | 0 | 0.03 | 0 | 0.65 | | |
| Plastered | | 0.03 | 0 | 0.03 | 0 | 0.02 | | |
| Other paints | | 0.1 | 0 | 0.03 | 0 | 0.07 | | |
| Paint used in pre-made wooden products | | 0.25 | 0.03 | 0.03 | 0.17 | 0.16 | | |
| Do-it-yourself (exterior) | 48% of paints are used by professionals (Royal Bank of Canada, 2006) | | | | | | | |
| Lacquer, varnish, undercoats and primer | 0.85 | 0.4 | 0.064 | 0.03 | 0.4483953 | 0.2101853 | “ | “ |
| Wood stains | | 0.6 | 0 | 0.15 | 0 | 1.58 | | |
| Wall paints | | 0 | 0 | 0.03 | 0 | 0 | | |
| Plastered | | 0 | 0 | 0.03 | 0 | 0 | | |
| Other paints | | 0 | 0.07 | 0.03 | 0 | 0 | | |
| Do-it-yourself (interior) | 48% of paints are used by professionals (RBC survey, 2006) | | | | | | | |
| | fused | | | finterior | EFrinsing | Erinsing | | |
| Wall paint | 0.85 | | | 1 | 0.02 | 0.76 | | |

Below are the parameters and assumptions used to determine the rates of loss for the MFA.

Table 3: Parameters for the calculations of microplastics emissions to different processes in the MFA.

| Category of loss | Urban (kt) | Rural (kt) |
|---|---|---|
| Wear and Tear | 1.96 | 1.47 |
| Removal | 0.61 | 0.46 |
| Rinsing | 0.43 | 0.32 |

## Appendices B: Comparison of microplastic flows and sinks across different environmental compartments from other global or regional MFAs.

| Study | Type of MFA | Geographic | Temporal | Type of MP | SW | Soil | Agricultural soil | Terrestrial soil | Landfill | Incineration | Roadside | ENV |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Current study (t) | Static | Canada | 2016 | Pre-production pellet | 60 | 230 | 2.53 | | 1.18 | 0.68 | | |
| | | | | Microfibres | 181 | | 424 | | 197 | 115 | | |
| | | | | Microbead | 20 | | 49 | | 19 | 23 | | |
| | | | | Building paint | 934 | 3,470 | 390 | | 181 | 105 | | |
| | | | | Road marking paint | 401 | 43 | 6.38 | | 103 | 102 | 1,700 | |
| | | | | Agricultural plastic mulch | | | 505 | | | | | |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | EPS and XPS foam | 13.2 | 13.8 | | | 2,590 | | | |
| | | | | TWPs | 7,550 | 1,030 | 111 | | 1,200 | 1,180 | 39,700 | |
| Boucher & Friot 2017 (t) | Static | Global | 2017 | Pre-production pellet | 4500 | | | | | | | |
| | | | | Microfibres | 5250000 | | | | | | | |
| | | | | Microbeads | 30,000 | | | | | | | |
| | | | | Marine coating | 55,500 | | | | | | | |
| | | | | Road marking | 105000 | | | | | | | |
| | | | | City dust | 360000 | | | | | | | |
| | | | | TWP | 420000 | | | | | | | |
| | | | | Pre-production | | | | | | | | 20,000 |
| | | | | Microfibres | | | | | | | | 200,000 |
| | | | | Microbeads | | | | | | | | 10,000 |
| | | | | Marine coating | | | | | | | | 50,000 |
| | | | | Road marking | | | | | | | | 600000 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ryber et al., 2019 (t) | Static | Global | 2015 | City dust | | | | | | | | 500000 |
| | | | | TWP | | | | | | | | 1,400,000 |
| Kawecki & Nowack, 2019 (tonnes) | Static | Switzerland | 2014 | All microplastics* | 14.9 ± 8.8 | 600 ± 110 | | | | | | |
| Sieber et al., 2020 (tonnes) | Dynamic | Switzerland | 1990-2018 | TWP from rubber tires | 41,500 ± 5700 | 9,600 ±1,500 | | | | 511,000 ± 75,400 | | |
| Prenner et al., 2021 (tonnes) | Static | Austria | 2018 | TWP from rubber tires | 1,365 ± 14% | 2,600 ± 14% | 250 ± 14% | | | | | |
| Baensch-Baltruschet al., 2021 (tonnes) | Not MFA<br><br>Emissions calculation | Germany | N/A | TWP from rubber tires | 15,510-19,770 | 68,190 | | | | | | |
| Geyer et al., 2022 (tonnes) | Dynamic | California, USA | 2008-2019 | Microfibres | 1100 | NA | 14,500 | NA | 5,600 | 800 | | |
| Gavigan et al., 2020 | Dynamic | North America | 1950-2016 | Microfibres | 22,000 | | | 11,000 | 10,000 | 6,000 | | |
| Paruta et al., 2022 (tonnes) | Static | Global | 2019 | Paint | 1,900,000 | 2,800,000 | | | | | | |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Hoseini & Bond, 2022 (tonnes) | Static | Global | 2015 | All microplastics* | 1,900,000 | 1,400,000 | | | | | | |

## Appendix C: Supplemental Figures

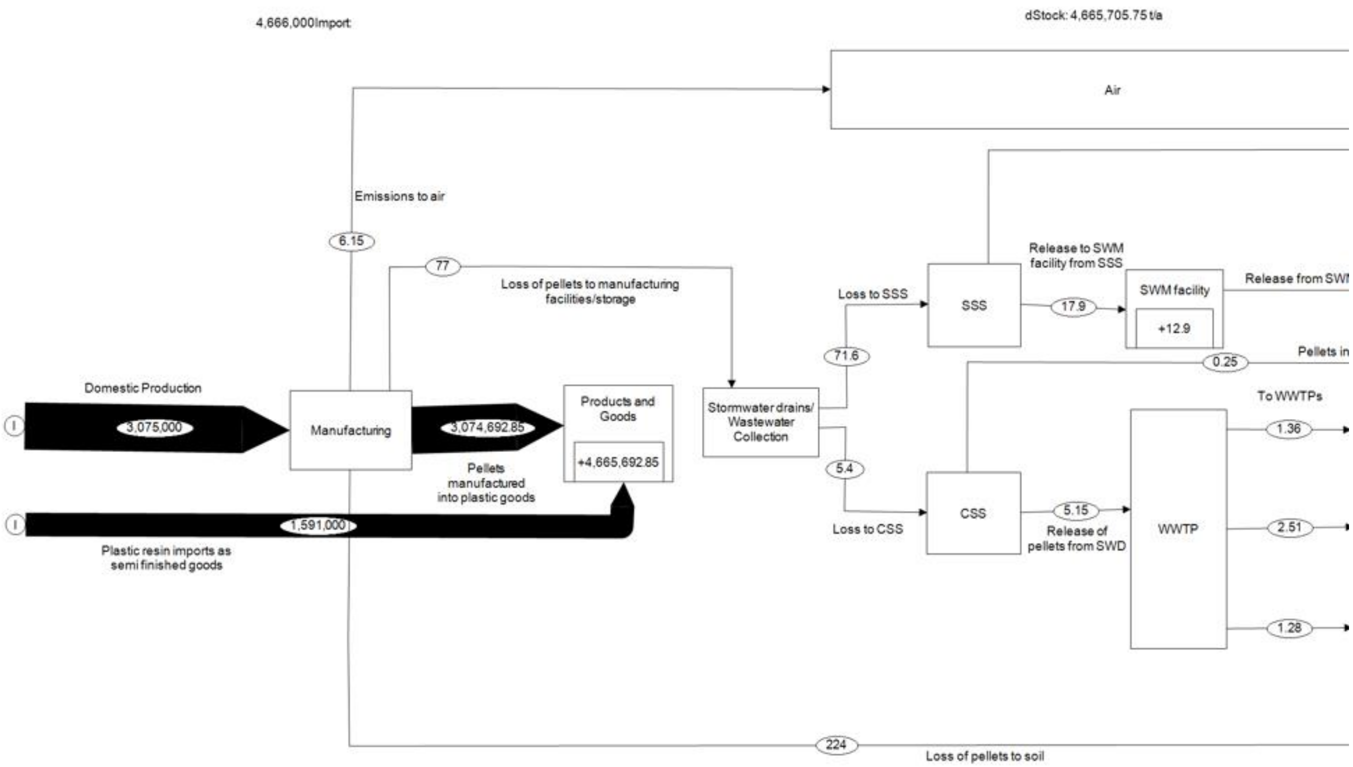
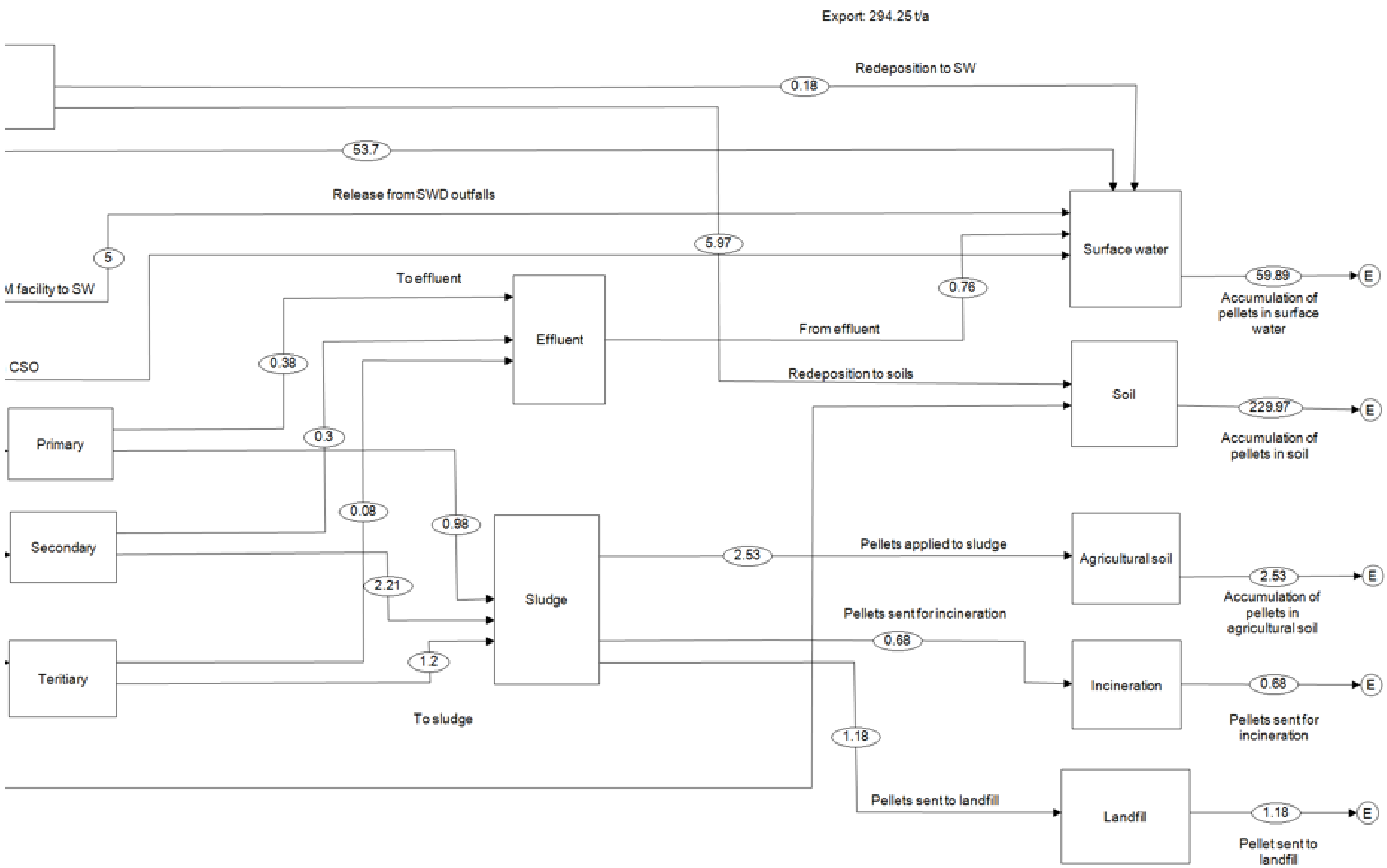



Figure S1: Expanded sankey diagram depicting the complete flows pre-production pellets and potential loss and release points in 2016 in tonnes.



Export: 294.25 t/a
Redeposition to SW
0.18
53.7
Release from SWD outfalls
5
M facility to SW
5.97
CSO
To effluent
0.38
0.3
0.76
From effluent
Redeposition to soils
Effluent
Surface water
59.89
E
Accumulation of pellets in surface water
Soil
229.97
Accumulation of pellets in soil
Primary
Secondary
Teritiary
0.08
0.98
2.21
1.2
To sludge
Sludge
2.53
Pellets applied to sludge
Agricultural soil
2.53
Accumulation of pellets in agricultural soil
Pellets sent for incineration
0.68
Incineration
0.68
Pellets sent for incineration
1.18
Pellets sent to landfill
Landfill
1.18
Pellet sent to landfill

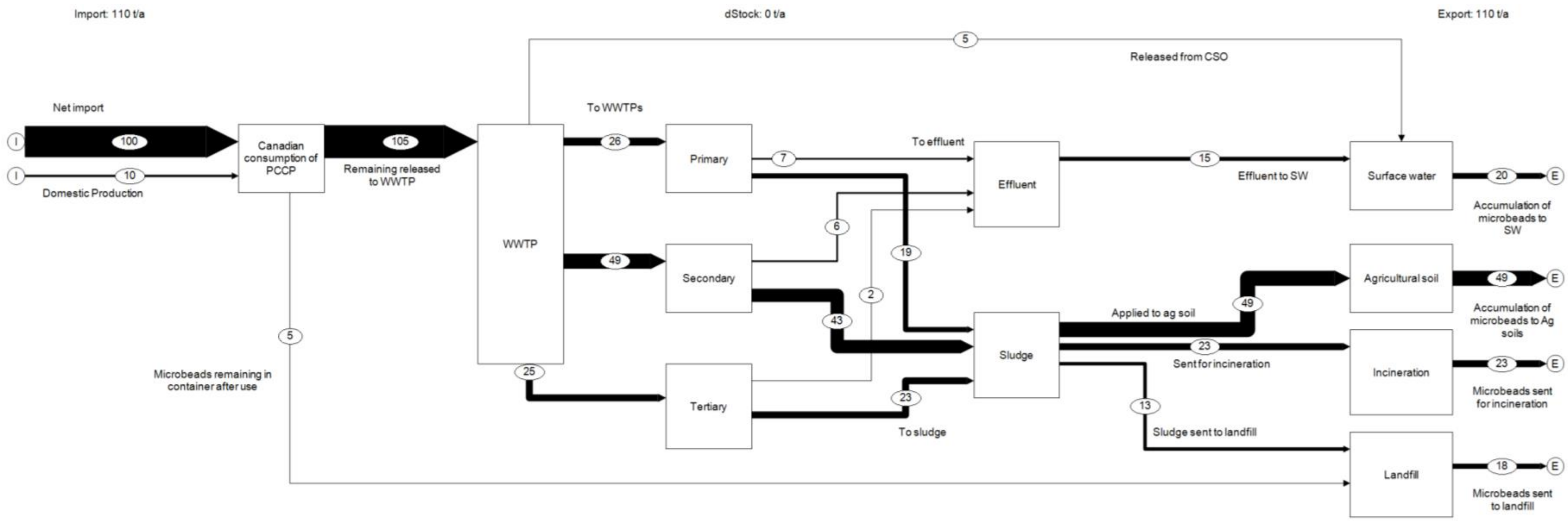



Figure S2: Expanded sankey diagram depicting the complete flows of microbeads and potential loss and release points in 2016 in tonnes.

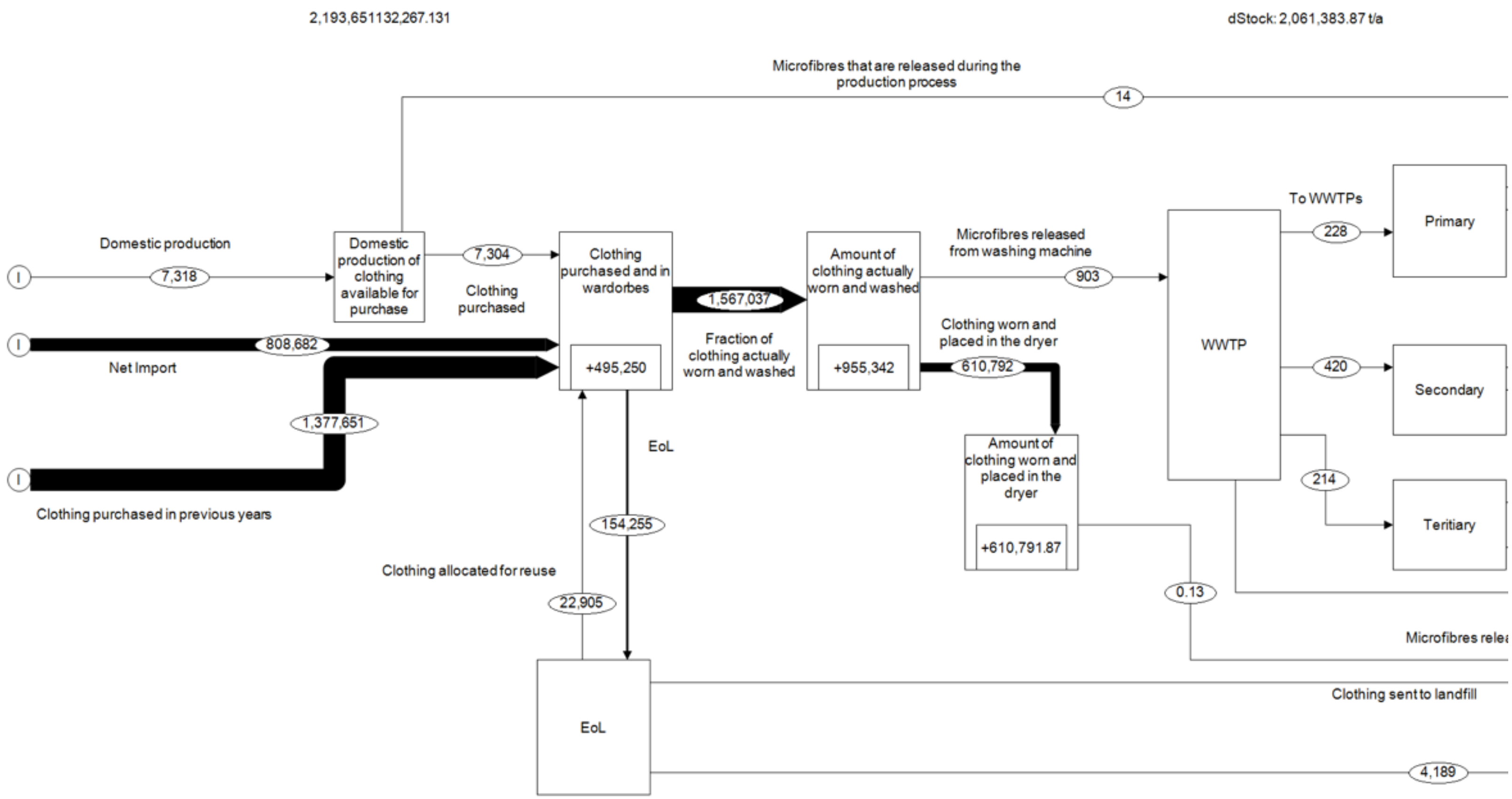
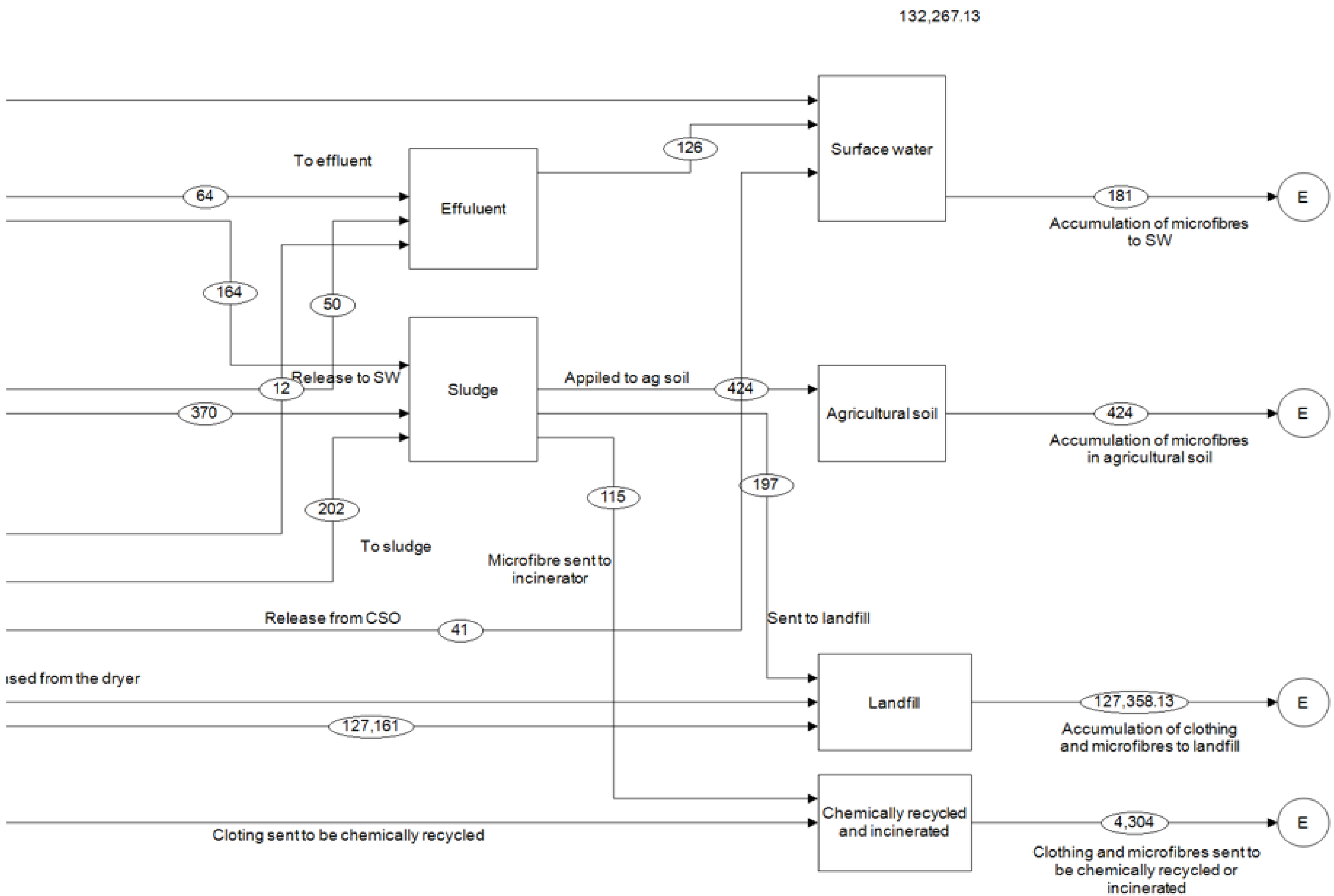



Figure S3: Expanded sankey diagram depicting the complete flows of microfibres and potential loss and release points in 2016 in tonnes.



132,267.13
To effluent
64
Effuluent
126
Surface water
181
E
Accumulation of microfibres to SW
164
50
Release to SW
12
370
Sludge
Appiled to ag soil
424
Agricultural soil
424
E
Accumulation of microfibres in agricultural soil
202
115
197
To sludge
Microfibre sent to incinerator
Release from CSO
41
Sent to landfill
ısed from the dryer
127,161
Landfill
127,358.13
E
Accumulation of clothing and microfibres to landfill
Cloting sent to be chemically recycled
Chemically recycled and incinerated
4,304
E
Clothing and microfibres sent to be chemically recycled or incinerated

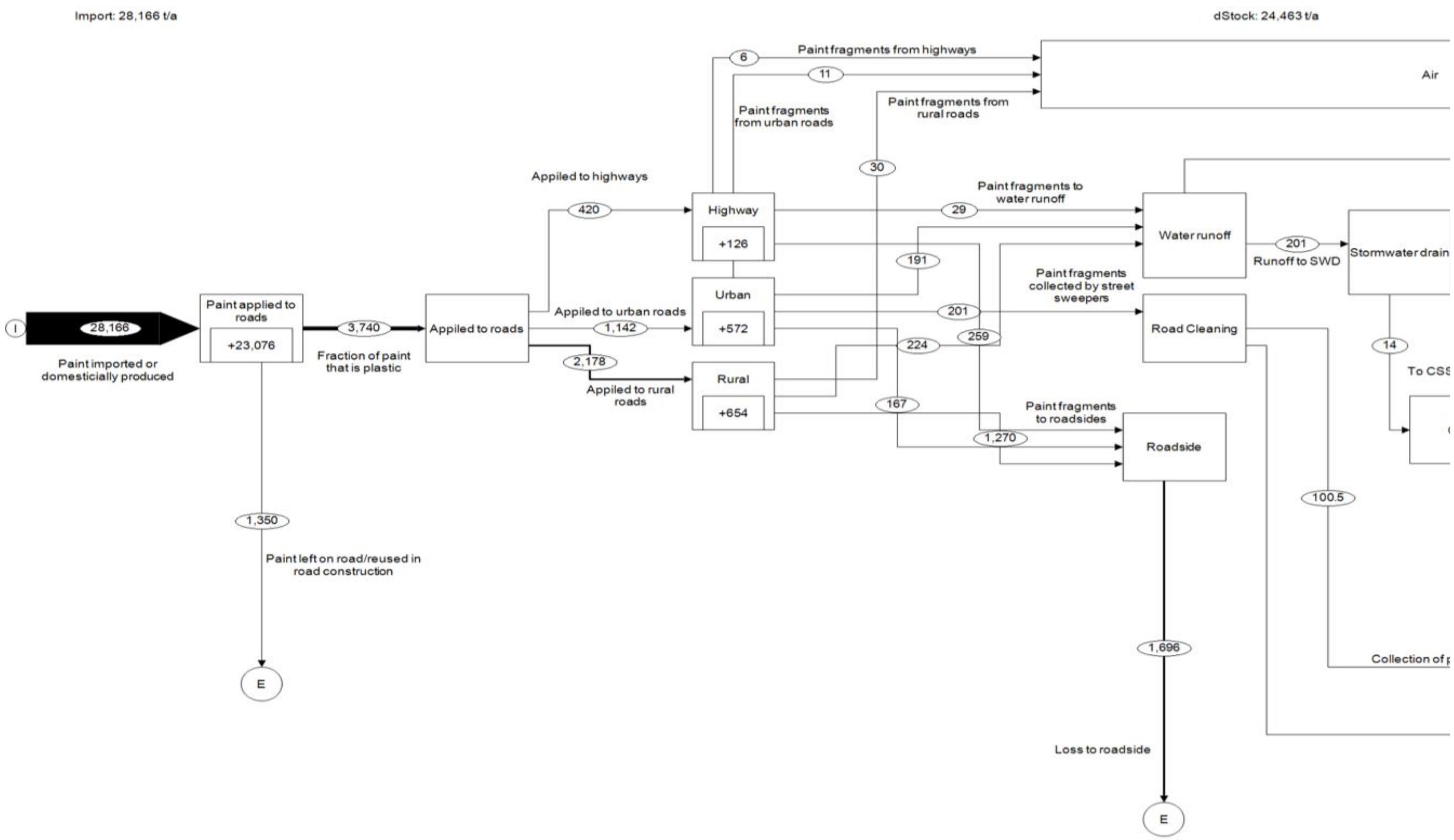
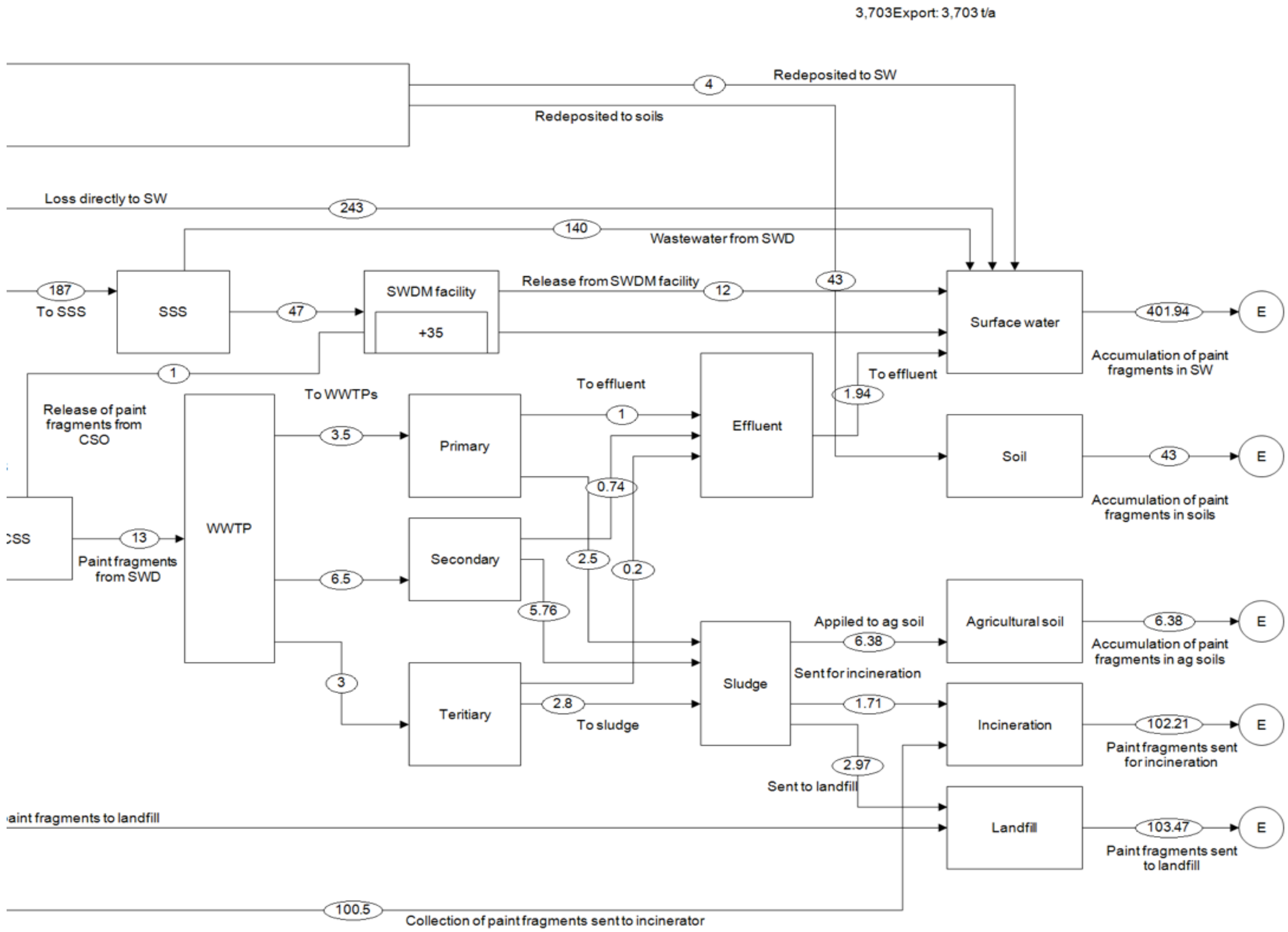



Figure S4: Expanded sankey diagram depicting the complete flows of road marking paint fragments and potential loss and release points in 2016 in tonnes.


3,703Export: 3,703 t/a
4
Redeposited to SW
Redeposited to soils
Loss directly to SW
243
140
Wastewater from SWD
187
To SSS
SSS
47
SWDM facility
+35
Release from SWDM facility
12
43
Surface water
401.94
E
Accumulation of paint fragments in SW
1
To WWTPs
To effluent
To effluent
1.94
Effluent
Release of paint fragments from CSO
WWTP
3.5
Primary
1
Soil
43
Accumulation of paint fragments in soils
0.74
CSS
13
Paint fragments from SWD
Secondary
2.5
0.2
6.5
5.76
Applied to ag soil
6.38
Agricultural soil
6.38
Accumulation of paint fragments in ag soils
Sludge
Sent for incineration
3
Teritiary
2.8
To sludge
1.71
Incineration
102.21
Paint fragments sent for incineration
2.97
Sent to landfill
aint fragments to landfill
Landfill
103.47
Paint fragments sent to landfill
100.5
Collection of paint fragments sent to incinerator

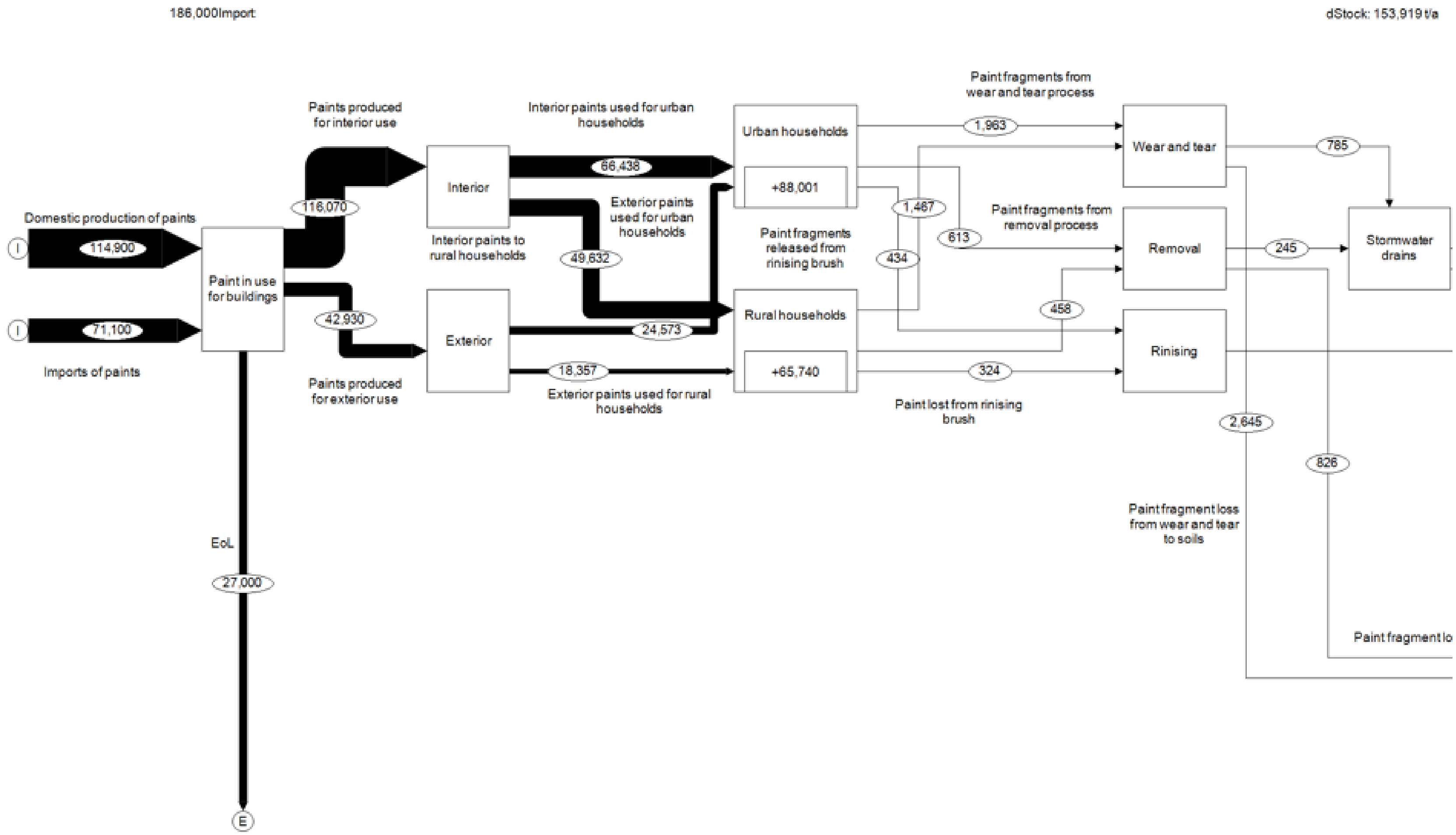
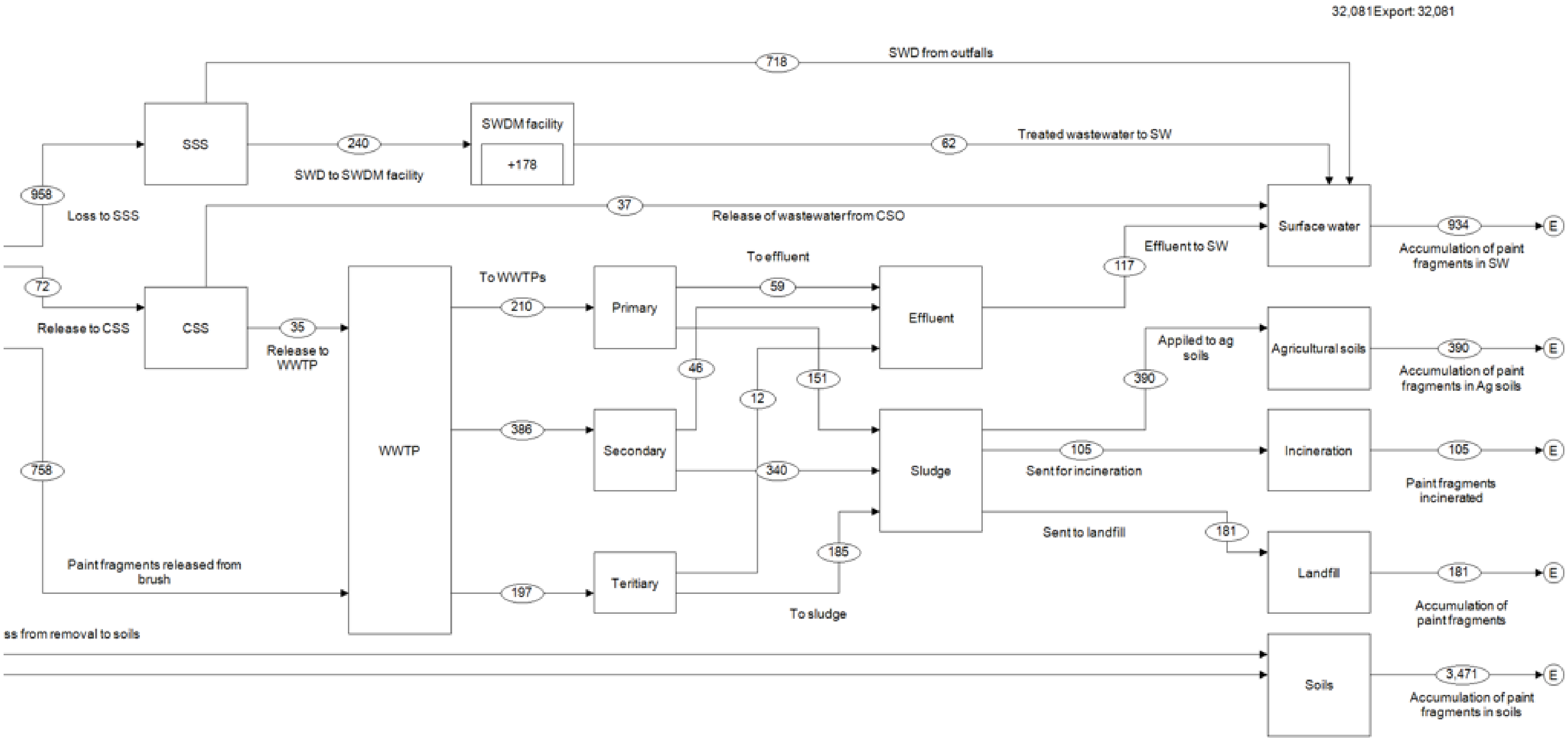



Figure S5: Expanded sankey diagram depicting the complete flows of building paint fragments and potential loss and release points in 2016 in tonnes.


32,081 Export: 32,081
SWD from outfalls
718
SSS
240
SWD to SWDM facility
SWDM facility
+178
62
Treated wastewater to SW
958
Loss to SSS
37
Release of wastewater from CSO
Surface water
934
Accumulation of paint fragments in SW
E
117
Effluent to SW
72
Release to CSS
CSS
35
Release to WWTP
WWTP
To WWTPs
210
Primary
To effluent
59
Effluent
46
12
151
Applied to ag soils
390
Agricultural soils
390
Accumulation of paint fragments in Ag soils
E
386
Secondary
340
Sludge
105
Sent for incineration
Incineration
105
Paint fragments incinerated
E
758
Paint fragments released from brush
197
Teritiary
185
To sludge
Sent to landfill
181
Landfill
181
Accumulation of paint fragments
E
ss from removal to soils
Soils
3,471
Accumulation of paint fragments in soils
E

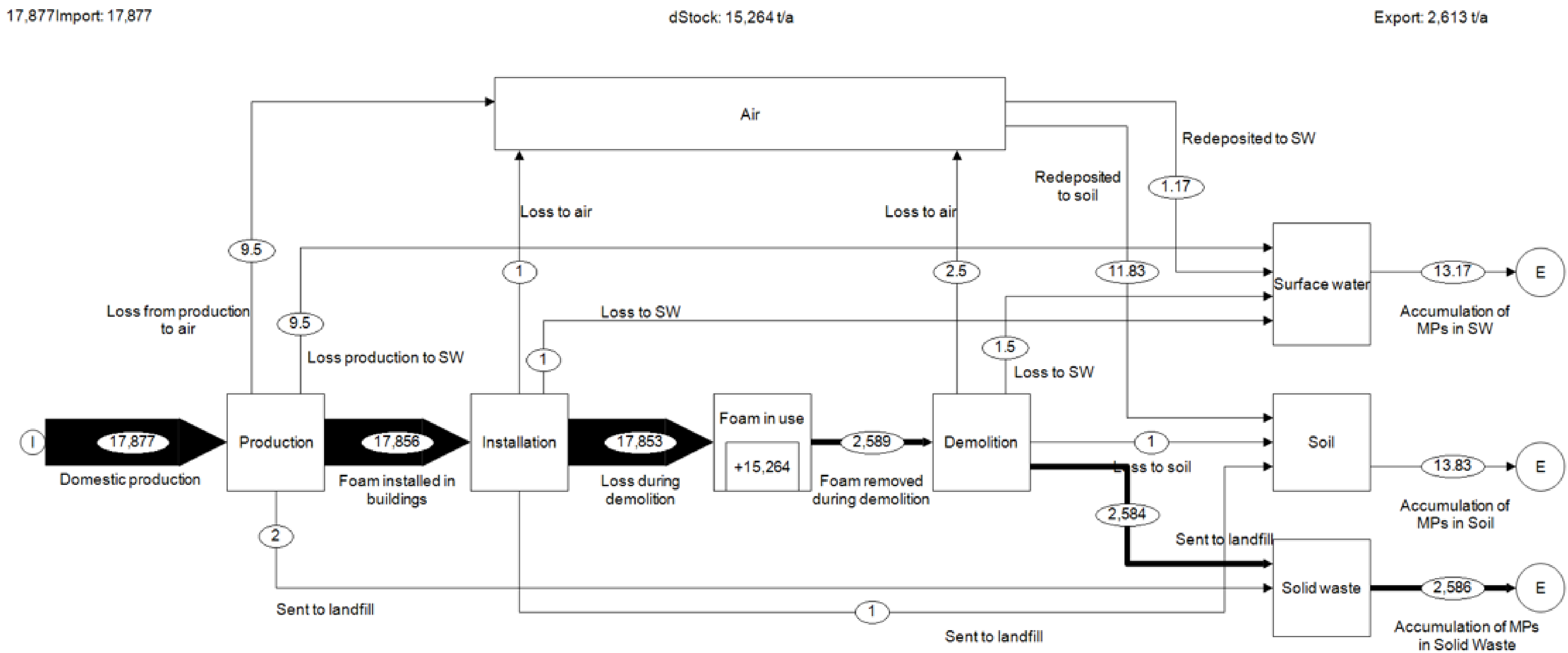



Figure S6: Expanded sankey diagram depicting the complete flows of EPS and XPS construction foam pieces and potential loss and release points in 2016 in tonnes.

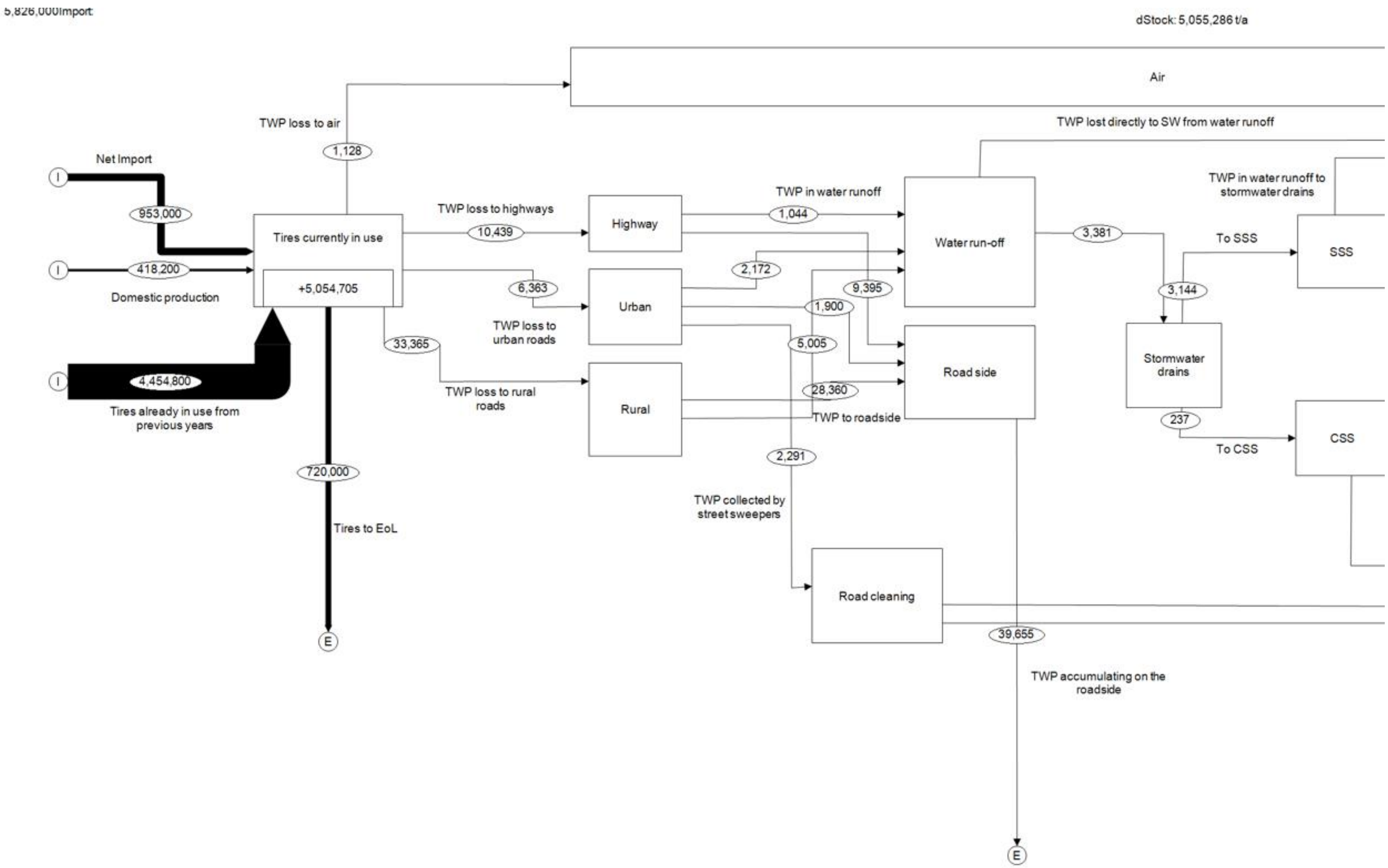
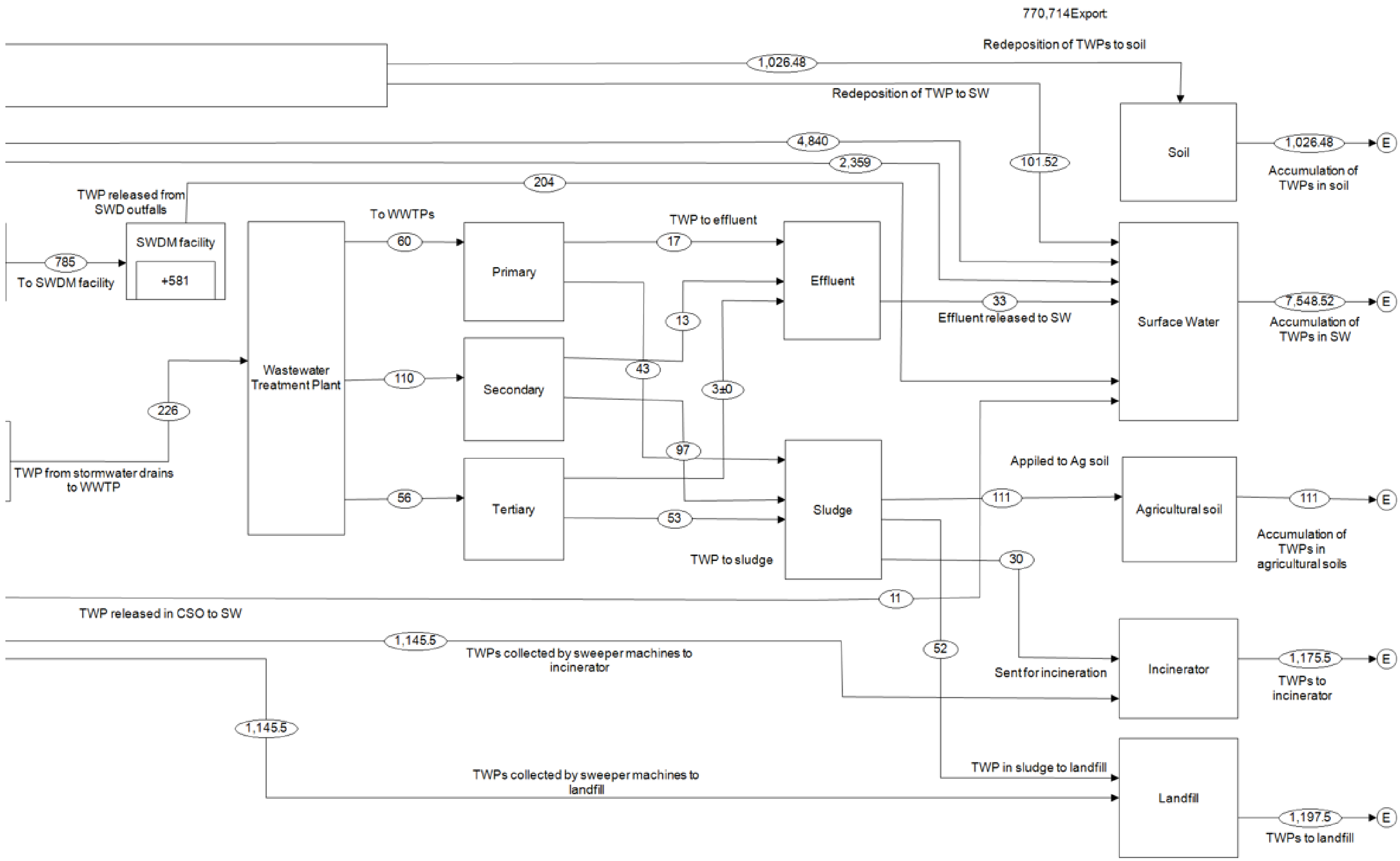



Figure S7: Expanded sankey diagram depicting the complete flows of TWPs and potential loss and release points in 2016 in tonnes.


770,714Export
Redeposition of TWPs to soil
1,026.48
Redeposition of TWP to SW
Soil
1,026.48
Accumulation of TWPs in soil
4,840
2,359
101.52
204
TWP released from SWD outfalls
To WWTPs
TWP to effluent
SWDM facility
+581
785
To SWDM facility
60
17
Primary
Effluent
13
33
Effluent released to SW
Surface Water
7,548.52
Accumulation of TWPs in SW
Wastewater Treatment Plant
110
Secondary
43
3±0
226
97
TWP from stormwater drains to WWTP
Applied to Ag soil
56
Tertiary
111
Agricultural soil
111
Sludge
53
Accumulation of TWPs in agricultural soils
TWP to sludge
30
11
TWP released in CSO to SW
1,145.5
TWPs collected by sweeper machines to incinerator
52
Sent for incineration
Incinerator
1,175.5
TWPs to incinerator
1,145.5
TWP in sludge to landfill
TWPs collected by sweeper machines to landfill
Landfill
1,197.5
TWPs to landfill
E