Mapping flows for a model city

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Content

Summary

This report is an output of the FanpLESStic-Sea project. FanpLESStic-Sea ran during the period January 2019 to December 2021 and was financed by the Interreg Baltic Sea programme. FanpLESStic-Sea worked with preventing and mitigating the pollution of microplastics in water and the Baltic Sea and for increasing the understanding about sources and pathways of microplastics. This report specifically concerns one of the work packages (2.3) within the project. The aim of the work package was to understand, visualise, and communicate the sources, pathways, and recipients of microplastics in a flow model for a hypothetical model city.

The report is divided into two main parts. The first part gives insights into how different sources of microplastics in urban waters can be calculated and what contextual information is needed to be able to perform such assessments. The report is accompanied with an Excel-based tool to ease the process of estimating each source. The report further brings up other information related to each specific source, such as the polymers and the shapes one can expect. The second part of the report focuses on estimating flows of microplastics in a semi-hypothetical model city in the Baltic Sea area. The insights on source estimates from the first part of the report are combined with the measurements taken in the project to assess flows to urban waters in a city. In addition, a number of control measures are introduced to the initial flow assessment to evaluate the impact of different interventions.

The results show that the largest source to urban waters in terms of mass are tyre wear particles. For wastewater the highest load came from laundry. Tap water, dust, and roof runoff all made a small contribution to the overall load to urban waters. The emissions to receiving waters were approximately 7 kg/year for treated wastewater, 1kg/year for CSOs and 13 000-17 000 kg/year for stormwater in terms of microplastics and 2 100 tonnes/year from tyre wear particles. If only the sources that were sampled are included, the load to the receiving water from stormwater was 82 kg/year of microplastics. The measurement-based estimates are based on a few measurements that often showed large variations in concentrations. More samples over a longer time period can give a more comprehensive view of the flows of microplastics.

Several policy interventions have been proposed, mostly for wastewater sources. If a ban on all intentionally added microplastics was to be introduced, consumers stopped rinsing paint brushes in the sink, and filters in washing machines were mandatory, there is the potential to cut emissions to the wastewater treatment plant with 30-50%. A large-scale facility for aggregated stormwater was tested in the project and if that was implemented to the model city it would lower the emissions of microplastics with over 90% and tyre wear particles in half. More research is needed on the technologies that would be most efficient for treatment of microplastics and tyre wear particles in stormwater, and under what circumstances treatment at source and centralised treatment in large scale facilities is preferred. More research is also needed to increase the understanding of differences between source estimates and measured values, as well as concentrations in other receiving compartments than water, such as urban soils.

Sammanfattning

Denna rapport är ett resultat av projektet FanpLESStic-Sea som finansierades av Interreg Baltic Sea Region och pågick under perioden januari 2019 till december 2021. FanpLESStic-Sea syftade till att förebygga och minska föroreningen av mikroplaster i vatten generellt och till Östersjön specifikt, samt öka förståelsen för mikroplastens källor och spridningsvägar. Denna rapport berör ett av projektets arbetspaket (2.3). Syftet med arbetspaketet var att förstå och visualisera källor, spridningsvägar och recipienter av mikroplast genom en flödesmodell för en modellstad.

Rapporten är uppdelad i två övergripande delar. Den första delen handlar om hur olika källor till mikroplast i urbana vatten kan beräknas och vilken information som behövs för att kunna utföra sådana uppskattningar. Till denna del av rapporten finns ett Excel-baserat beräkningsverktyg som är tänkt att underlätta källuppskattningarna. I rapporten finns även annan information relaterat till varje specifik källa, såsom vilka polymerer och vilken morfologi på partiklarna som kan förväntas från källan. Den andra delen av rapporten fokuserar på att uppskatta flöden av mikroplast i en semi-hypotetisk modellstad i Östersjöområdet. Kunskapen om källuppskattningar från rapportens första del kombinerades med mätningar som gjorts i projektet med syfte att skapa en överblick över flöden av mikroplast i en stad. Till den initiala flödesmodellen introducerades även ett antal både förebyggande och reningstekniska åtgärder med syfte att utvärdera effekten av olika insatser.

Resultaten visar att den största källan (baserat på massa) till urbana vatten är utsläpp från däck. För avloppsvatten var den största källan utsläpp från klädtvätt. Mikroplast i kranvatten, damm och avrinning från tak hade alla en förhållandevis liten belastning. Utsläppen till recipienten var cirka 7 kg/år för renat avloppsvatten, 1 kg/år från bräddning och 13 000–17 000 kg/år från dagvatten för mikroplast och 2 100 ton/år från däckspartiklar. Om endast de källor som provtogs i projektet tas med i beräkningen för dagvatten var belastningen till recipient från dagvattnet 82 kg/år för mikroplast. Flödesuppskattningarna baseras på ett fåtal prover som ofta visade på stor variation. Fler prover under en längre tidsperiod kan ge en mer heltäckande bild av flödet av mikroplast.

Flera åtgärder har föreslagits för att minska mängden mikroplast, främst för källor till avloppsvatten. Om ett förbud mot all avsiktligt tillsatt mikroplast skulle införas, samt att konsumenterna slutade skölja färgpenslar i diskhon och filter i tvättmaskiner var obligatoriska, finns det potential att minska utsläppen till avloppsreningsverket med 30–50%. En storskalig anläggning för dagvattenrening testades i projektet och om en sådan implementerades i modellstaden skulle det minska utsläppen av mikroplast med över 90% och däckspartiklar till hälften. Mer forskning behövs dock kring de tekniker som skulle vara mest effektiva för att minska utsläppen av mikroplast och däckspartiklar i dagvatten, och under vilka omständigheter rening vid källan eller centraliserad rening i storskaliga anläggningar är att föredra. Mer forskning behövs också för att öka förståelsen kring skillnader mellan källuppskattningar och uppmätta värden, samt koncentrationer i andra recipienter än vatten, till exempel städers jordar.

1. Introduction

This report is an output of the FanpLESStic-Sea project. The project was supported by the Interreg Baltic Sea Region programme and ran from January 2019 to December 2021. FanpLESStic-Sea aimed at contributing to the prevention and mitigation of the pollution of microplastics in water and to the Baltic Sea, and to increase the understanding about sources and pathways of microplastics. This report specifically concerns one of the work packages (2.3) within the project.

The purpose of work package 2.3 was to estimate flows of microplastics in an urban area. This report focuses on sources and pathways found in the literature as well as the results of the sampling performed as part of the project. The estimates are applied to a semi-hypothetical model city. The characteristics of the model city are presented in chapter 3.1 of this report. There are also some control measures that are introduced to the model city, which are described in chapter 3.2. These control measures are based on policy suggestions, literature values on removal efficiencies of certain technologies, and a treatment technique that was tested within the project. Another purpose of work package 2.3 was to give insights into how sources of microplastics can be estimated and aid in the process of making such assessments. An Excel-based calculation tool has been developed to assist the calculation of loads of microplastics to stormwater and wastewater.

The report is structured as follows: First, previous studies on sources to microplastics in wastewater and stormwater, and a way to estimate each source is presented. Second, the semi-hypothetical model city is described, as well as a description of the introduction of the control measures. Last, the results in terms of flows in the semi-hypothetical model city and the potential impact of the control measures is presented.

2. Source estimates

This chapter presents an overview of how different sources of microplastics in wastewater and stormwater can be estimated. In some literature stormwater sources are estimated to end up in a wastewater treatment plant (WWTP) due to combined sewers. As there can be large differences regarding duplicate and combined systems between cities, this is not assessed in this report. Stormwater is assessed as the final compartment for stormwater sources. These general source estimates are connected to a calculation tool in Excel, which can be adapted to local conditions. There is a section below each source description named “information needed as input to the tool” which is the information that depend on contextual factors, and which needs to be put into the tool by the user. Information about the shape of the particles and the predominant polymer types, as well as the predominant pathways are also presented in connection to each source. Figure 1 gives an overview of the sources and pathways related to urban waters that have been identified in this study.

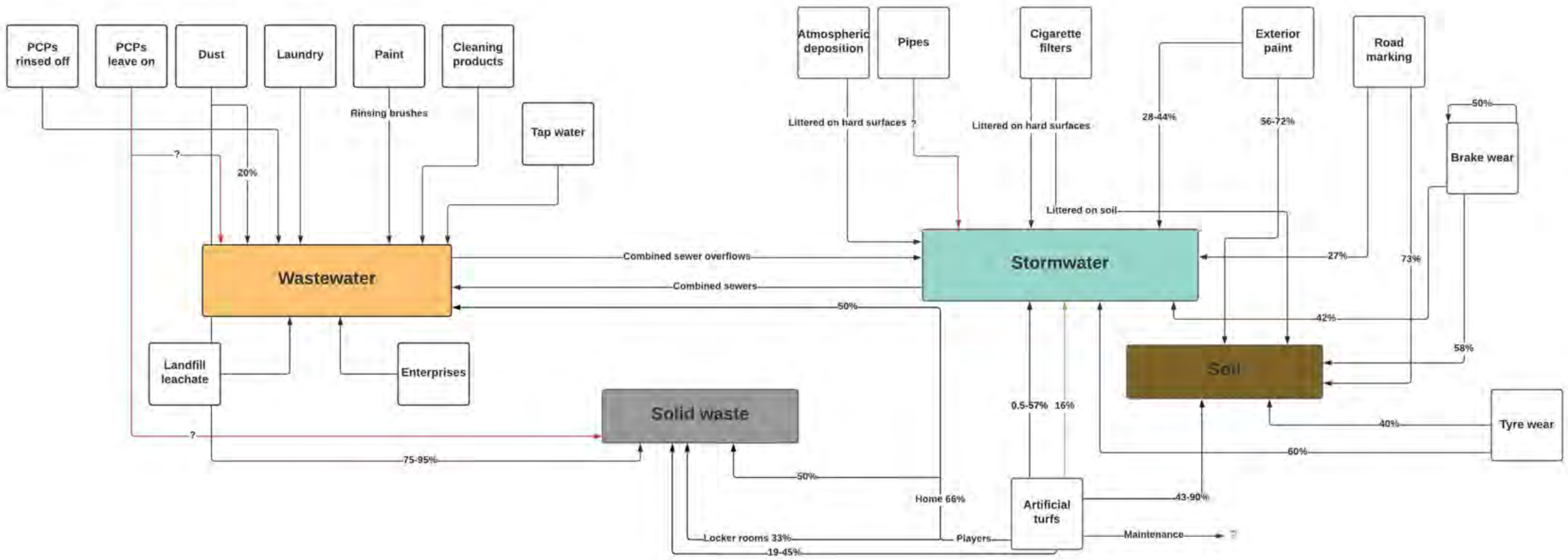

Figure 1: Overview of sources and pathways of microplastics in urban areas. PCPs stands for personal care products. For artificial turfs, the black arrows represent granulate and the green arrow represents pile.

2.1 Sources and pathways of microplastics to wastewater

The two main sources of microplastics in wastewater are households and enterprises. Microplastics may also occur in the tap water that is used by both households and enterprises.

2.1.1 Impact of tap water

Microplastics may occur in drinking water either from insufficient removal from raw water or by addition of microplastics from the treatment process. Microplastics can also be added to the water in the distribution system from storage tanks and pipes (Mintenig et al., 2019). However, this is uncertain, Shen et al. (2021) for example observed similar quantities of microplastics in treated water and tap water and Kirstein et al. (2021) found no clear pattern in microplastic abundance between water works, pumping stations and fire posts and did not find an increase in polyethylene (PE) after the water had passed 5 km of PE-pipes.

Source estimate

The measurements at the outlet of the drinking water plant taken in this project can be used as in-put to the wastewater source estimates. The water in the outlet of the drinking water plant contained an average of 12 µg microplastics per m^3.

Information needed for input to the tool

- Amount of wastewater generated by households that end up at the WWTP per year or the water consumption/person/year.

Pathways

A human is assumed to consume approximately 2-3 litres of water per day. It is the smallest particles that risk uptake by the human body. Based on their findings, Kirstein et al. (2021) conclude that the annual uptake of microplastics in human bodies would be less than 1 particle per year. The rest will pass with excretes. The majority of the microplastics in tap water will enter the sewer directly via the water used for personal hygiene, flushing the toilet, and washing dishes, as this is what most of the tap water is used for.

2.1.2 Households

The identified sources of microplastics from households were: laundry, dust, personal care products, cleaning products, and paint originating from the rinsing of paint brushes in water.

Laundry

Microplastics, in the form of small fibres, are released from synthetic textiles when they are washed. The reports on emissions of synthetic fibres during wash vary greatly between studies. Some studies also include fibres of natural origin (e.g., wool and cotton) or use fabric blends, which makes it challenging to derive the synthetic share. Most of the studies on emissions from laundry have studied polyester, but there are a few studies that have investigated other materials such as polypropylene (PP), acrylic and polyamide (PA).

Source estimate

The emissions from laundry ($E_{Laundry}$) can be calculated in the following way:

$$E_{laundry} = (T_{washed} \times S_{share}) \times EF \tag{1}$$

where T_{washed} is the textiles washed in kg/capita/year, S_{share} is the synthetic share of the textiles washed, and EF is the emission factor. The EF can be obtained from different scientific studies that quantify release of fibres from laundry. If priority is given to studies that simulate real washing conditions in terms of loads, temperature, use of detergent, and cycle duration, as well as studies that report the results in mass (either using gravimetric methods or mass calculations), an EF of 33–399 mg/kg is obtained (Dalla Fontana et al., 2020; De Falco et al., 2019; De Falco, Gentile, et al., 2018; De Falco, Gullo, et al., 2018; Hernandez et al., 2017; Kelly et al., 2019). This EF is for polyester and can be used as a proxy for laundering of other textiles as well. PP was shown to release less fibres than polyester in one study (De Falco, Gullo, et al., 2018), but within the interval described above. Acrylic has been shown to release both less (Belzagui et al., 2019) and more (Cesa et al., 2020; Napper & Thompson, 2016) fibres than polyester. A reason for this discrepancy can be the large differences between polyester fabrics. Carney Almroth et al. (2018) showed that a knitted polyester released less fibres than acrylic and PA, but a polyester fleece released more.

Information needed for input to the tool

- Amount of textiles washed per capita per year. In Europe one washing load is estimated to be 3–4 kg (Pakula & Stamminger, 2010) and the number of cycles washed per capita per week are between 1.2 and 1.5 dependent on country (Schmitz & Stamminger, 2014). See Table “input washing behaviour” in the Excel file for details.
- The share of household textiles that are of synthetic origin. Hann et al. (2018) report that 34% of the clothes sold in Europe are of synthetic material. This does not give the full picture as other textiles (such as towels and bedlinen) are also washed frequently but can provide a rough estimate.

Polymer types and shapes

The polymer composition is dependent on the types of fabrics washed. In general, polyester is the most common synthetic material used in clothes in Europe, followed by acrylic and PA (Hann et al., 2018). The microplastics released during laundry is in the form of fibres.

Pathways

The emissions from laundry during wash is expected to, in full, be released to the wastewater.

Dust

In general, there are few studies that have investigated microplastics in settled household dust. Based on the available studies from households and other indoor environments, there seem to be several contextual factors that influence the amount of microplastics found in indoor dust. For example, the deposition was higher during weekends than weekdays in a dormitory and 50% higher deposition occurred in an office during weekdays compared to weekends. This pattern indicate that the presence of people has a large impact (Q. Zhang et al., 2020). There were also differences between countries (J. Zhang et al., 2020). Fibres seem to dominate the microplastics in dust and account for up to 99% (Soltani et al., 2021). The polymer composition resembles the textiles found at the indoor environment (Q. Zhang et al., 2020). Despite the contextual factors, the synthetic share was reported to between 30 and 40% in several studies (Dris et al., 2017; Soltani et al., 2021; Q. Zhang et al., 2020).

Source estimate

$$MP_{dep.} = Dust_{dep.} \times S_{Share} \tag{2}$$

where $Dust_{dep}$ is the number of particles deposited per m^2/year and S_{Share} is the synthetic share of the dust. The study by Soltani et al. (2021) is used in the calculation tool as they provide the possibility for conversion to mass.

Information needed for input to the tool

- The total living area connected to the wastewater treatment plant.

Polymer types and shapes

The polymer composition is dependent on the textiles used at the site, which mirror use in society. The shape is predominantly in the form of fibres.

Pathways

All settled dust will not be added to the wastewater, it will also reach solid waste from vacuuming. According to Ewers et al. (1994), approximately 75-95% of dust is removed by vacuuming ($\bar{p}_{vacuum}$ in Equation 3). The final wet washing step will remove 20% of the remaining dust (w.w. in Equation 3) and this is the share that will end up in the wastewater. This share can be calculated with the following equation:

$$E_{wastewater} = (MP_{dep.} \times \bar{p}_{vacuum}) \times w.w. \tag{3}$$

Personal care products

Personal care products (PCPs) may contain microplastics. There are generally two types of PCPs that contain microplastics: rinse-off (e.g., facial scrubs, toothpaste, shower gel) and leave-on (e.g., lotions and make-up). Rinse-off products, containing microplastics, have already been prohibited in several countries (Kentin & Kaarto, 2018) and a restriction of all intentionally added microplastics (which will include both rinse-off and leave on PCPs) is currently under consideration in the European Union (European Chemicals Agency, 2020).

There are several empirical studies that have investigated different types of rinse-off PCPs, primarily facial scrubs. In summary, there are large differences both between different product categories, and between products within the same category. For example, facial cleaners were reported to have 2.95-124 particles/gram of product (Jemec Kokalj et al., 2018; Lei et al., 2017) and facial scrubs 5 219-50 391 particles/gram of product (Cheung & Fok, 2016:2017).

To the author's knowledge only one report (Amec Foster Wheeler, 2017) presents an estimation of microplastics in products that are left on the skin. By dividing the amount of microplastics added to leave-on products in Europe in 2015 with the number of inhabitants in the region the same year, the contribution was estimated to be 1.1-2.2 g/cap./year. The same was carried out for rinse-off products, which were estimated to 1.4-1.6 g/cap./year.

This is slightly lower than previous estimates for rinse-off products, which can be due to a voluntary substitution by the industry (Amec Foster Wheeler, 2017). Galafassi et al. (2019) summarise the findings from several source quantification studies and on a European level, estimates were 7.9-24 g/cap./year. However, Galafassi et al. (2019) stress that these data are from 2012 and need to be reviewed based on current events with increased restrictions.

Source estimate

$E_{wastewater} = \text{Consumption} \times \text{inhabitants}$

where *consumption* is the yearly emissions per person and *inhabitants* are the number of people connected to the WWTP.

Information needed for input to the tool

- Number of people connected to the wastewater treatment plant.

Polymer types and shapes

PE has been put forward as the most common polymer used in PCPs (Q. Sun et al., 2020) but reports from the industry suggest that polyurethane (PU) is the polymer that is most often added (60%) followed by PE (40%) (Amec Foster Wheeler, 2017). No information is available on the polymers added to leave-on products. Microplastics added to rinse-off PCPs are commonly spherical beads or irregular particles (Amec Foster Wheeler, 2017; Q. Sun et al., 2020).

Pathways

All microplastics in rinse-off products are expected to end up in wastewater. For leave-on products it is more uncertain as it can also enter solid waste via dry removal.

Cleaning products

Microplastics can be added to cleaning products used to, for example, polish surfaces. Three studies were found that estimate emissions from cleaning products (Amec Foster Wheeler, 2017; van Wezel et al., 2016; Verschoor et al., 2016). The estimates from these studies were divided into a per capita share to allow for comparison (Table 1). van Wezel et al. (2016) and Verschoor et al. (2016) only included abrasive agents, while as Amec Foster Wheeler (2017) included all cleaning, maintenance and detergents in both households and industry, where abrasive agents are the most common (Amec Foster Wheeler, 2017). All three estimates are provided in the calculation tool.

Table 1: Estimations of emissions from cleaning products.

Reference	Minimum (g/capita/year)	Maximum (g/capita/year)
Amec Foster Wheeler (2017)	0.40	0.40
Verschoor et al. (2016)	0.14	0.15
Van Wezel et al. (2016)	0.02	0.67

Source estimate

$$E_{wastewater} = \text{Consumption} \times \text{inhabitants} \tag{4}$$

where *consumption* is the yearly emissions per person and *inhabitants* is the number of people connected to the WWTP.

Information needed for input to the tool

- Number of people connected to the wastewater treatment plant.

Polymer types and shapes
According to Amec Foster Wheeler (2017), the most common polymer is PU in the form of a fine powder, with particles smaller than 600µm, but how much PE that is added to the products is not known.

Pathways
This source can be assumed to, in full, reach the wastewater.

Rinsing paint brushes
Paint contains polymer and the process of rinsing paint brushes after the use of water-based paints release residues to the wastewater. This has primarily been put forward as a household-related source, as the professional sector has other routines for handling equipment after use (Hann et al., 2018).

Source estimate

$$E_{Paint} = Paint_{purchased} \times Paint_{used} \times MP_{content} \times EF \tag{5}$$

where $Paint_{purchased}$ is sales data of water-based paint for private use and $Paint_{used}$ is the estimation of how much is used (75-90%). The $MP_{content}$ can be handled in two different ways. Some studies take the polymer content of paint (usually set to 5-30%) (Hann et al., 2018; van Wezel et al., 2016; Verschoor et al., 2016). However, it has been questioned if all polymer in paint can be considered microplastics (Swedish Chemicals Agency, 2018). According to industry estimations the microplastics content[1] in water-based paints are less than 1% (Amec Foster Wheeler, 2017). This difference leads to large variations in emission estimates on a European level: 220 tonnes/year (Amec Foster Wheeler, 2017) and 3 500 tonnes/year (Hann et al., 2018) from this source. EF is the emission factor from rinsing painting equipment (1.6%), which has been determined after a rinsing-test (Verschoor et al., 2016).

Information needed for input to the tool

- Amount of water-based paints purchased by the private sector in the country or region.

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[1] Defined as solid non-biodegradable polymeric particle with physical dimensions between 1µm-5 mm originating from anthropogenic sources.

Polymer types and shapes
The polymers that were reported by the industry were small (5-80µm) acrylic particles with a spherical shape and fibres of PA or polyacrylonitrile, with varying length (4-50mm) and 10µm in diameter (Amec Foster Wheeler, 2017).

Pathways
The emissions from paint by rinsing painting equipment can be assumed to, in full, reach the wastewater.

Other potential sources
There are a few potential sources where the information is too limited to make estimations at this point, but it might be revised in the future.

Microplastics may enter the wastewater from incorrect flushing. Wet wipes are one such personal care product that is wrongly flushed into toilets. Lee et al. (2021) found that wet wipes gave rise to 693-1066 particles/sheet when submerged into water. Another source that has been suggested is wrong disposal of contact lenses. Rolsky et al. (2020) found that approximately 21% of contact lens users flush their lenses and this equals a load of 44 000±1700 kg of dried contact lens material (silicon hydrogels) into US wastewater yearly. There is further one study that has investigated microplastics in human stool and found approximately 2 pieces (fragments and films in the size range 50-500 µm) per gram stool (Schwable et al., 2019). However, the results are still uncertain as the sample size was very small.

2.1.3 Enterprises and other non-household related sources
The contribution of microplastics from enterprises and industries can vary greatly dependent on the types of industries connected and their internal treatment facilities. In general terms, enterprises that use plastic in their production, workshops that use a large amount of paint, which may be washed into sewage via overspraying, workshops using hand soap with microbeads, pre-washings at textile industries, and plastic pre-production are industries that are of specific interest. Commercial laundries are also a source of interest. For laundries, Equation 1 can be used by adapting T_{washed} to the loads of laundry that is being washed in the connected laundries and the share of synthetics they wash.

Leachate from landfills may contribute to the load of microplastics to wastewater if there is leachate connected to the WWTP. A study of eleven landfills in three Nordic countries found 0.16-4.51 items/L and 0.08-2.36 µg/L (van Praagh et al., 2018). However, it should be noted that no microplastics were detected at two of the landfills. A Chinese study (J. Sun et al., 2021) report slightly higher values (235.4 ±17.1 items/L and 11.4±0.8 µg/L). This variation can be due to contextual differences, J. Sun et al. (2021) only sampled one landfill. He et al. (2019) found large differences between the six Chinese landfills studied (0.42-24.58 items/L). Different cut-off values may also impact the results. van Praagh et al. (2018) analysed down to 50µm and J. Sun et al. (2019) down to 10µm. In the calculation tool, the landfills from the Nordic countries were chosen as it was determined as most representative for the Baltic Sea region.

Source estimate

$$E_{wastewater} = Leachate_{connected} \times MP_{conc.}$$

where $Leachate_{connected}$ is the amount of leachate connected to the plant (L/year) and $MP_{conc.}$ the microplastic concentration in untreated leachate (µg/L).

(6)

Information needed as input to the tool

- Amount of leachate added to the WWTP per year.
- If the leachate is untreated or has undergone some pre-treatment and, in that case, what kind and its efficiency.

Pathways

If leachate is connected to the WWTP, it is assumed to, in full, be added to the wastewater.

2.1.4 Further pathways

Most of the wastewater in the countries in the Baltic Sea catchment area is collected and treated at a WWTP, often with secondary treatment (Baresel & Olshammar, 2019). The retention capacity can differ dependent on treatment. In general, the first part of the treatment process can retain more than half of the microplastics, and WWTPs with secondary or tertiary treatment report a retention of 96-99% and 90-99.9%, respectively (Habib et al., 2020). The particle shape can affect the retention. There are indications that fragments, granulate and spherical shapes are more easily retained than fibres. While PE and PP are common polymers in influent, they are more rarely detected in effluent (Ngo et al., 2019). Despite the high retention capacity of WWTPs, an increased concentration of microplastics has been measured outside WWTPs (Estahbanati & Fahrenfeld, 2016). There are some specific treatment technologies that have been investigated for microplastic removal, such as sand filter, disc filter, tertiary biofilter, membrane bioreactor (MBR) and dissolved air flotation (DAF). All these techniques increase the retention (see Table 1 in Vahvaselkä & Winquist, 2021 for summary).

Combined sewer overflows (CSOs), where untreated or partially treated wastewater is released to receiving waterways, can be a source of microplastics to the environment. Baresel and Olshammar (2019) distinguish between three types of CSOs: technical (caused by failures in the collection system), weather-related (occur when the hydraulic capacity of the sewer system is exceeded) and at the WWTP (limited hydraulic capacity at the WWTP). Weather-related CSOs had the largest contribution, and the contribution of microplastics to the environment was similar to that of WWTP effluent.

2.2 Sources and pathways of microplastics to stormwater

There are approximately the same number of sources to stormwater and wastewater, but the pathways are more complex in stormwater (Figure 1).

2.2.1 Atmospheric deposition

Airborne microplastics may reach the stormwater. There are a few studies that have investigated European cities, primarily large cities. The results of the studies are displayed in Table 2. One can see that the larger cities seem to have more microplastics. This is similar to the pattern found by Dris et al. (2016), who found more fibres at the urban site compared to the sub-urban. There are also several other factors that can influence the deposition. Precipitation is one factor, Szewc et al. (2021) observed twice as many particles on a wet day compared to a dry day. However, Klein and Fischer (2019) found no such difference. Higher wind speed may also lead to more deposition (Klein & Fischer, 2019; Szewc et al., 2021; Wright et al., 2020). Further, Szewc et al. (2021) found that wind direction had an impact, where winds from land led to more microplastics compared to marine winds. As seen in Table 2, the studies have different cut-off sizes, which can be the reason why deposition seems higher in London than in Paris and Hamburg.

Table 2: Summary of the studies that have investigated atmospheric deposition of microplastics in European cfities. *Results from the urban site. #Range is based on mean values from three urban sites.

Reference	**Location**	**Cut off size**	**Results** (mean no. particles/m^2/day)
Dris et al. (2016)	Paris	50µm	110±96* of which 29% contained petrochemical polymers.
Klein and Fischer (2019)	Hamburg	63µm	29.1±14.9-66±43.6#
Szewc et al. (2021)	Gdynia (coastal city in Poland with approximately 250 000 inhabitants)	Not stated, but report size from 5µm	10±8
Wright et al. (2020)	London	20µm	712±162 fibrous 59±32 nonfibrous Total average: 771±167

Source estimate

The contribution of microplastics from atmospheric deposition to stormwater can be estimated by assuming that all microplastics that are deposited on impervious surfaces in the city will reach the stormwater (Sörme & Lagerkvist, 2002). Hence, it can be calculated as follows:

$$E_{\text{urban water}} = MP_{\text{dep.}} \times \text{impervious surfaces} \tag{7}$$

where $MP_{\text{dep.}}$ is the deposition rate (see column 4 in Table 2). Note that to achieve a yearly estimate, the deposition rate should be multiplied with 365, which is based on the assumption that the results are representative for the whole year. This calculation will give a result in terms of particles/year. Dris et al. (2016) estimated the weight by multiplying fibre length with a basal area of 80µm and the densities of two polymers, one light (PA) and one heavy (PET). From these assumptions an estimate of 1.2-4 kg fibres per km^2 was derived.

Information needed for input to the tool

- Area that is covered by impervious surfaces in the city.

Polymer types and shapes

There were differences in the most common polymers and shapes between the studies. Szewc et al. (2021) and Wright et al. (2020) found mostly fibres (60% and 92%, respectively), of mainly polyester and polyacrylonitrile, respectively. Klein and Fischer (2019) instead report that 89-97% were fragments and these were most commonly PE. Of the non-fibrous category, Wright et al. (2020) identified eight polymers with similar shares. Like with indoor dust, the polymers in atmospheric deposition can be linked to local sources (Szewc et al., 2021).

Pathways
If deposited on impervious surfaces the deposition can be assumed to, in full, enter stormwater.

2.2.2 Artificial turfs

There are primarily two sources of microplastics associated with artificial turfs: the rubber infill and the synthetic pile.

Infill
Several quantification studies have estimated that the yearly refill is equal to yearly losses to the environment, but this has been criticized for not taking compaction into account (Verschoor et al., 2021). Two studies consider compaction explicitly, in two different ways. Løkkegaard et al. (2019) assume that all the flows that are not accounted for in the mass balance is a result of compaction and estimate it to be 67-86% based on a yearly refill of 2 200 kg/year. Verschoor et al. (2021) bases their estimates on a compaction of 2.7%, which accounts for 50-83% based on a yearly refill of 600-1200 kg/year. Another approach is to base the estimation on the amount of infill on the field and an emission factor for yearly loss. The amount of infill has been reported to be 10-16 kg/m^2 (Bø et al., 2020; Hann et al., 2018) and Hann et al. (2018) use an emission factor of 1-4% to account for both good and poor maintenance practices.

Source estimate
If the amount of yearly infill added to the field is known, the first approach can be used and the emission of infill from the field calculated as follows:

$$E_{infill} = refill \times (100\% - compaction) \tag{8a}$$

where *refill* is the amount of granulate added to the field yearly and *compaction* is what is left on the field due to compaction in percentage.

If the yearly refill is not known, but the size of the pitch is, the other approach can be used, and the emissions calculated as follows:

$$E_{infill} = pitch\ size \times infill \times EF_{infill} \tag{8b}$$

where *infill* is the amount of infill on the field (10-16 kg/m^2) and EF_{infill} is the emission factor (1% for good maintenance and 4% for poor).

Pathways

There can be large differences in dispersal of granulate. Different studies have also investigated different parts of this chain. Dispersal routes that have been brought up in the literature are summarised below.

1) Removal with players

Two studies (Forskningskampanjen, 2017; Regnell, 2019) have investigated granulate attachment to players during wet and dry conditions. Forskningskampanjen (2017) found that the average amount attached to a player was 1.4 ml in dry conditions and 3.7 ml if it was raining. The games were between 10-100 minutes. Regnell (2019) estimated 0.91 g per player for dry conditions and 2.7 g/player in rainy conditions for games that were between 60-120 minutes. This fraction can either be brought home or enter the locker rooms. The share that is brought home can either be swept up or added to the wastewater when the clothes are washed. The share that enters locker rooms can either be swept up and enter solid waste or go into the drains in locker rooms and be added to the wastewater. Wallberg et al. (2016) estimate that about two thirds follow the player home. In the homes, 50% enter wastewater and 50% solid waste. The remaining third is emitted in the locker rooms and was estimated to, in full, be swept up and enter solid waste. However, this is dependent on the maintenance practice of the arena.

2) Field maintenance

The maintenance practice can impact emissions to the environment. In regions with snowfall, granulate can be removed with the snow when ploughing the fields. According to Løkkegaard et al. (2019) the loss can be 0-11% dependent on the practice. Snow blowers led to the highest loss. Wallberg et al. (2016) also estimated a loss from snow clearance, but also that, if the snow is stored, 20-50% can be returned to the field. In their study, the rest would enter solid waste. In addition, leaf blowing can also remove granulate from the field (Verschoor et al., 2021). Maintenance vehicles can also capture granulate. Regnell (2019) found that under dry conditions cleaning by blowing gave rise to 15 g of granulate per machine, while under wet conditions it was 177 g/vehicle. Further, if the machine was also brushed, the emissions were 178 g in dry and 510 g in wet conditions per machine. Based on these results, Regnell (2019) suggests cleaning all maintenance vehicles before they leave the facility.

3) Surrounding environment and stormwater

The share that is estimated to end up in the water compared to soil largely differ between studies. Hann et al. (2018) and Sieber et al. (2020) both report that around half will enter soil (45% and 43%, respectively), but while Sieber et al. (2020) then estimate the remaining to enter water, Hann et al. (2018) only estimate that about 5% will enter water and the larger fraction to enter solid waste. Kjølholt (2018) instead estimate that the large majority will enter surrounding soils (85-90%) and the rest water. Løkkegaard et al. (2019) estimate that only 11% enter soil and paved areas and an even smaller share to water. The conclusion that can be drawn is that the pathways probably largely differ between fields and is dependent on, for example, maintenance routines and the location of stormwater wells. With the correct preventive measures Regnell (2019) found that emissions to water could be lowered to 0.1 kg/year.

Pile

The emissions of synthetic pile from the fields have not been studied to the same extent as the infill.

Source estimate

Based on Hann et al. (2018) emission from pile can be estimated in the following way:

$$E_{pile} = \text{Pitch size} \times \text{pile} \times EF_{pile} \tag{9}$$

where *pile* is the amount of pile on a field (0.6-1.6 kg/m^2) (Bø et al., 2020) and EF_{pile} is the emission factor, which was set to 0.5-0.8% (Hann et al., 2018).

Pathways

There is, to the author's knowledge, only one grey report (Olshammar et al., 2021) that have investigated infill-less pitches and focused on the dispersal of pile. Olshammar et al. (2021) estimate that on average 0.19 g/m^2/year[2] of pile disperse from fields and that 0.03 g/m^2/year[3] will end up in stormwater wells. This means that 16% of the pile that leaves the field would end up in stormwater.

Information needed for input to the tool

- The generation of artificial turf (1G-4G) and the type of infill.
- Number of pitches in city or region.
- Size of the pitch (es) or the yearly refill of granulate.
- How often the field is used.
- Number of players that utilise the field per unit of time.

Polymer types and shapes

The material used for the pile and infill is dependent on the age of the field. The first generation (1G) have PA fibres and second generation (2G) used PP fibres and sand as infill. The third generation (3G) uses PE fibres and most commonly styrene-butadiene rubber (SBR) granulate, but ethylene propylene diene monomer rubber (EPDM), thermo plastic elastomer (TPE) or organic materials such as cork can be used. The fourth generation (4G) have PE fibres and can use sand as infill or no infill at all (Bø et al., 2020). The pile has an elongated shape, and the infill consists of irregularly shaped particles.

2.2.3 Littering of cigarette filters

Cigarette butts are the most common litter in urban areas and the filters consists of cellulose acetate (Belzagui et al., 2021). A citizen science study conducted in seven nature types in Denmark in 2019 found that 30% of the plastic litter were cigarette butts (Syberg et al., 2020). In 2020, a national litter-quantification campaign was carried out in urban areas in Sweden for one week. Based on these results, 25 cigarette butts are littered per m^2 per year. However, there are some uncertainties related to how well the streets were cleaned before the start of the litter collection campaign, which means that some litter that had been littered before the sampling week started may have been included. This will impact the estimation of litter/year. Yet, if the results presented is in fact representing weekly littering in an urban area and this is assumed to be representative for the whole

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[2] 0.08-0.32 g/m^2/year
[3] 0.00078-0.07583 g/m^2/year

year, the cigarette filters will contribute with 4.3 g/m²/year, when using a filter weight of 0.17 g (Register, 2000).

A filter is in its original size is approximately 15mm long and 8.1mm in diameter, which means that there is not much fragmentation needed for the filter to be regarded as microplastics. There are some scientific studies that have investigated the fragmentation of cigarette filters. Gerritse et al. (2020) observed a 15% weight loss per year in a laboratory that was set up to simulate marine conditions and Belzagui et al. (2021) found that approximately 100 microfibres (below 2 mm) detached from a cigarette filter per day and that one cigarette filter contains over 15 000 such fibres. This means that a filter will be fragmented in less than a year, assuming that the detachment rate is constant. Bonanomi et al. (2020) found that there is a rapid fragmentation the first 30 days (15% mass loss), both in presence and absence of soil. After the first period, the fragmentation was slower, especially in the absence of soil.

There are several uncertainties related to the fragmentation in a real urban setting compared to laboratories, such as UV-exposure and mechanical abrasion. Further, previous research has not identified cellulose acetate in any large quantities in stormwater (Liu, Olesen, et al., 2019) or surface water (Kanhai et al., 2017; Vianello et al., 2018). Cellulose acetate has a higher density than water and can be expected to sink, but it does not seem prevalent in stormwater sediments (Liu, Vianello, et al., 2019; Olesen et al., 2019) or in marine (Cheang et al., 2018) or freshwater (Klein et al., 2015) sediments.

Source estimate
Assuming that the cigarette filters present in an urban area will become microplastics in a year and that all cigarette filters that are littered on impervious surfaces will enter the stormwater, the contribution can be calculated as follows:

$$E_{\text{urban water}} = CF_{\text{littered}} \times \text{impervious surfaces} \tag{10}$$

where $CF_{littered}$ is the amount of cigarette filters littered per m² in the urban area per year (after street sweeping, if applicable) and *impervious surfaces* is the area of impervious surfaces in the city.

Information needed for input to the tool
- Area that is covered by impervious surfaces in the city.
- If there is street sweeping and its efficiency.

Pathways
It can be assumed that all items that are littered on impervious surfaces and that are not swept up during street sweepings will enter the stormwater.

2.2.4 Exterior paint

Painted surfaces may release microplastics from both removal of old paint layers and wear, but despite that it is found in various environments, it has not been subject to much research (Gaylarde et al., 2021).

Source estimate

To estimate the emissions from exterior paints, like rinsing of paint brushes, sales data can be used:

$$E_{Paint} = Paint_{purchased} \times Paint_{used} \times Paint_{exterior} \times P_{degradation} \times EF_{wear/removal} \tag{11}$$

As seen in Equation 11, there are a number of aspects to take into account. First, the paint sales data ($Paint_{purchased}$) need to be divided between professional and private as the latter leave more unused, 15-25% compared to 1.5-3%, according to Hann et al. (2018). Second, the sales data need to be divided between exterior and interior application. This can differ between European countries. In some countries it is estimated that 25% is used for exterior applications and in others up to 63%, which has been suggested to be because of climate-related differences, where the Nordic countries have a harsher climate that requires more painting (Hann et al., 2018). Third, the polymers in the paint might be degraded when they have been applied ($P_{degradation}$). The polymer degradation was set to 67% by Hann et al. (2018). Fourth, two emission factors are needed, one for removal and one for wear of the surface. Hann et al., (2018) and Verschoor et al. (2016) report several wear losses dependent on type of paint, but most often it is between 2.5 and 3% and removal losses 1-4% (6.4% for some paint used by private consumers).

Information needed for input to the tool

- Paint purchased for professional and private use, respectively.
- The share of the sales that is used for exterior purposes.

Polymer types and shapes

Even if the whole paint particle does not consist of polymer (between 20 and 95% dependent on the type of paint (Hann et al., 2018)), the whole paint particle is commonly assessed as microplastics (Hann et al., 2018; Verschoor et al., 2016). Paint particles are usually sharp-angled fragments and the polymer used are PU, polyester, polyacrylates, polystyrene (PS), alkyls and epoxies (Gaylarde et al., 2021).

Pathways

The paint flakes that are emitted from the surfaces are often estimated to primarily enter soil (Gaylarde et al., 2021). Verschoor et al. (2016) estimated that 56-58% will enter soils. If assuming that the amount removed by road cleaning would also enter soil in the absence of road cleaning, 72% will end up in soil (Hann et al., 2018). The rest was estimated to reach the water.

2.2.5 Road related emissions

There are three main sources of microplastics related to road traffic addressed in this study: tyre wear, brake wear, and road marking.

Tyre wear

Two different approaches are typically used to estimate emissions from tyre wear. One approach uses emission factors per vehicle-km multiplied by the total mileage, and the other uses the number of tyres multiplied by the weight loss of these tyres during use (Kole et al., 2017). As wear is expected to be particularly high in urban areas due to more braking and accelerating (Knight et al., 2020), the emission-based method with emission factors for urban driving was determined as most appropriate for estimating city emissions.

Source estimate

Emissions from tyre wear can be calculated as follows:

$$E_{tyres} = (T.A._{vechicle\ type} \times \text{urban share}) \times EF_{urban} \quad (12)$$

where $T.A._{vechicle\ type}$ is the traffic activity for different types of vehicles and *urban share* is the share of that vehicle type that is driven in urban areas. This information can be found in national or European databases. The European average of shares of different vehicle types driven in urban areas is presented in the Excel file. EF_{urban} is the wear rate on urban roads and is set to 0.06-0.85 dependent on vehicle type (details in the Excel file) as reported by Hann et al. (2018). Further, sometimes the whole particle can be considered as microplastics and sometimes only the rubber content is considered. The rubber content in tyres is between 40 and 60% (Wagner et al., 2018).

Information needed as input to the tool

- The traffic activity for different types of vehicles in the city.

If this is not available, Sieber et al. (2020) made an approximate calculation of 1.29±0.45 kg/cap./year for Switzerland. It should be noted that this value is only for the rubber part of the particle and not the whole tyre particle.

Polymer types and shapes

A tyre can contain both synthetic and natural rubber. Heavy vehicles, such as buses and trucks usually have tyres with more natural rubber, while passenger cars have more synthetic rubber (Goßmann et al., 2021; Wagner et al., 2018). SBR is a common synthetic rubber. In terms of shapes, tyre wear particles commonly show a slightly elongated shape (Knight et al., 2020).

Pathways

The pathways of tyre wear particles can be context dependent and therefore it can be challenging to draw general conclusions. For example, in the Netherlands porous asphalt is very common, which led Kole et al. (2017) to estimate that over 50% is trapped in asphalt, but porous asphalt is not common in the rest of Europe (Hann et al., 2018). In general, the majority of the particles released will be deposited close to the road. For the studies that take the pathway to air into account, the share has been estimated to be 1-7% (Kole et al., 2017; Sieber et al., 2020; Verschoor et al., 2016). However, Sieber et al. (2020) point to that these particles will probably enter soil or water in a short period of time and hence did not see this as the final compartment. For urban areas the share to stormwater has been estimated to 60% and soil 40% (Verschoor et al., 2016).

Brake wear

Brake wear originates from friction during the braking process in the vehicles. Emissions from this source can be estimated in a similar manner as emissions from tyres.

Source estimate

$$E_{brakes} = (T.A._{vechicle\ type} \times \text{urban share}) \times EF_{vehicle\ type} \times \text{coarse fraction} \quad (13)$$

where T.A.$_{vehicle\ type}$ is the traffic activity for different vehicles (passenger cars, goods vehicles and lorries) and EF$_{vehicle\ type}$ is the emission factor for the different vehicle types. Specific emission factors for urban areas were not obtained. Some studies report urban emission factors, but only for PM_{10}, which includes particles up 10µm and the detection limit for the measurements in this project is 10µm. Based on a review by Ntziachristos & Boulter (2016) the coarse fraction of brake wear (10µm-5mm) has been estimated to between 2 and 38% (Hann et al., 2018).

Polymer types and shapes

The polymer share of a brake wear particle has been estimated to 20-40% (Grigoratos & Martini, 2015; Sommer et al., 2018), but commonly the whole particle is assessed as microplastics (Hann et al., 2018; Verschoor et al., 2016).

Pathways

The predominantly small size of brake wear particles means that they can pose a risk to humans because of inhalation, and that they are primarily airborne. Of the non-airborne fraction, approximately 50% has been estimated to be trapped in the vehicle (Ntziachristor & Boulter, 2016). Hann et al. (2018) estimate that most of the coarse fraction will enter soils (56%) followed by stormwater (26%). However, this estimate can be difficult to generalise as it includes all types of roads, not only urban. The pathway "directly to surface waters" has been argued to not be a pathway in urban areas (Verschoor et al., 2016). If this share and the share that is estimated to end up at the WWTP via combined sewers are added to stormwater and the amount captured by road cleaning is assumed to enter soils in the absence of road cleaning, this will result in 42% to stormwater and 58% to soils.

Road marking

Road marking is worn off when vehicles drive on the roads. This source is often recognised in relation to road-related microplastic emissions, but there are not so many studies that have aimed to quantify this source (Galafassi et al., 2019).

Source estimate

Hann et al. (2018) estimate the emissions from road markings in a similar manner as for other paint emissions. Sales data for road markings were used and the amount that remains unused (3%) and the amount that is used on new roads (20%) were subtracted. Further, it is recommended that the marking is repainted when 30% has been removed on urban roads. Similar to other exterior paint there is a polymer degradation (50-64%) and it was estimated that 19% of the roads are urban. Based on the findings from Hann et al. (2018) a share of emissions per kilometer road can be obtained. This was estimated to 19 kg/km/year.

Information needed as input to the tool

- Kilometer road in the city.

Polymer types and shapes

Several polymers can be used in road marking, such as PE, PU and PA (Gaylarde et al., 2021). There are different kinds of road markings, and these can contain different polymers (Andersson Sköld et al., 2020) (Table 3). The shape is in the form of irregular particles. These can be differentiated from other similar particles as they commonly contain glass beads (Horton et al., 2017).

Table 3: Summary of polymers used in different types of road marking paint (from Andersson Sköld et al. (2020), p.54).

Type of road marking product	Composition
Thermoplastic systems	Binding agents: e.g., pentaerythritol rosin ester, C5-hydrocarbon resin, or a mix of these. Some markets also use a proportion of EVA (ethylene vinyl acetate copolymer), or styrene block copolymers
Water-based paints	Binding agent: thermoplastic acrylic resins
Solvent-based paints	Binding agent: mainly thermoplastic acrylic reins, but styrene acrylic mixes are also used. Solvents: e.g., esters or ketones, aromatic solvents are still permitted in some countries.
2-component systems	*Acrylic systems:* Binding agent: thermosetting acrylic resins (e.g., methyl methacrylate, MMA) Curing agents: e.g., dibenzoyl peroxide (BPO) Solvents *Epoxy systems:* Binding agent: Epoxy resins (a reaction product of bisphenol A and epichlorohydrin is the most commonly used resin) Curing agents: e.g., amines Solvents
Road marking tape	Binding agents: polyurethane and flexible polymer Adhesive To add reflective properties, either glass beads or micro-crystalline ceramic beads are used

Pathways

Similar to other road related emissions and building paint, road markings are estimated to primarily enter soil. Hann et al. (2018) have estimated pathways for road markings. This estimate can be difficult to generalise as it includes all types of roads, not only urban. However, if the category “directly to surface water” is added to stormwater and the amount captured by road cleaning is assumed to enter soils in the absence of road cleaning, this will result in 27% to stormwater and 73% to soils.

2.2.6 Other sources to stormwater

There are a few potential sources that are considered small or where the information is too limited to make estimations at this point, but it might be revised in the future as the knowledge basis grows.

Littering of plastic

Littering of larger plastic items can become microplastics when the litter fragments into smaller pieces. The challenges for quantification are twofold, both related to estimating the amount of litter and related to the fragmentation into microplastics. The fragmentation process is dependent on a number of contextual factors such as UV-exposure, mechanical abrasion (e.g., by sand or people), and biofilm formation on the littered material (Song et al., 2020).

In order to estimate flows of macroplastics associated with littering, Kawecki and Nowack (2019) first estimated the products that are consumed on-the-go, and then divided these between three different environments (residential, natural areas, car). The results showed that most littering took place in the residential areas and that the majority still ended up in solid waste. The second largest receiving compartment was soil, closely followed by stormwater.

The results of the national litter-quantification campaign in Sweden found approximately 0.07 plastic items per m^2, which means that 3.64 plastic items are expected to be littered per m^2/year with a corresponding weight of 9 g/m^2/year.

For the littered plastic to become microplastics it needs to be fragmented into pieces below 5 mm. Galafassi et al. (2019) calculated formation of microplastics in marine environments by using a degradation rate of 1-5%, although they also mention that a degradation rate of 0.5% has been estimated for the North Sea. However, fragmentation can differ in aquatic environments compared to urban areas and there can also be differences between plastic types. For example, Svedin (2020) found that PP fragmented faster than PE and that fragmentation was slower in water than air. Song et al. (2017) found that after one year of UV exposure and abrasion with sand, there was nothing left of the original particle of expanded polystyrene (EPS), while over 90% was left of the original PE particle. Further, Song et al. (2020) estimated that 5% of an EPS box would be fragmented per month. However, it should be noted that this UV exposure does not correspond to a "real" year as the plastic was exposed 24h a day. There are also other aspects in natural environments that can lower the UV exposure. The plastic item can for example be buried under some other material in the urban area, and in water, biofilms can form on the material, or the plastic can have some additives that halt UV effects (Song et al., 2017). Further, the fragmentation process might not be constant, and, for example, a tear, can speed up the fragmentation.

Stormwater pipes

Stormwater pipes may contribute to microplastics. Sang et al. (2021) investigated microplastics in rainwater pipes, both the rainwater and pipe sediment. They did find polyvinyl chloride (PVC) (4.2% in water and 12.5% in sediment) and the majority of the pipes were of PVC. However, it was not dominating, and it is possible that the PVC can originate from other sources.

Other sources

Automotive paint was considered a source by Hann et al. (2018) and was estimated to be below 100 tonnes for Europe and was therefore assumed negligible on a city level. The use of plastic abrasive media (e.g., for cleaning or removing graffiti) has also been suggested as a source, but it is uncertain both to the extent this is used and how much spill there might be (Magnusson et al., 2016).

2.2.7 Further pathways

Some cities have combined sewer systems and the microplastics in the aggregated stormwater from the area with combined sewers will be a source to the WWTP. Cities can also have a separate sewer system. The stormwater can enter receiving waters untreated or first be treated in various stormwater treatment facilities. Most of the stormwater is today not treated prior to release to receiving waters (Vahvaselkä & Winquist, 2021).

3. The semi-hypothetical model city

To allow for the use of several measurements that have been taken in different countries in this project, the flows of microplastics are estimated for a semi-hypothetical model city. The characteristics are primarily set to the real cities in Norway, Poland, and Sweden where the samples were taken. When literature values were used, a qualitative estimate was made by the author, either to choose data from studies that were deemed most similar to already set characteristics of the model city or to base on a European average. An overview of the characteristics of the model city are presented in 3.1 and the estimates and assumptions for each source estimate are described in chapter 3.3.

3.1 Characteristics of the model city

The model city has 110 844 inhabitants and is located in the Baltic Sea region in northern Europe. The yearly average temperature is 9 °C. Most of the precipitation falls as rain (90%), while 10% falls as snow. The city centre has an area of 26 km^2. This area consists of 44% impervious surfaces and the remaining are green areas, agricultural areas, or surface water. Out of the impervious surfaces, most are buildings (37%), followed by roads (26%), and parking lots (11%). The rest consists of miscellaneous impervious surfaces. The distribution among vehicle types in the city are summarised in Table 4.

Table 4: The share of different vehicle types in the city.

Passenger cars	84%
Goods vehicles/vans (>3.5 tonnes)	10%
Mopeds/motorcycles	5%
Lorries/trucks	1%
Buses	0.2%

There are 12 artificial turfs in the area that all use SBR granulate infill and PE pile. There is no snow ploughing of the fields in the winter. Half of the fields are used 30h/week and the other half are used 25h/week. The fields are used 40 weeks per year and there are, on average, 16 players per game.

The city receives drinking water from a large drinking water plant that also supplies other cities in the region. All inhabitants in the city are connected to a WWTP. The WWTP has mechanical treatment, an activated sludge process, and post-precipitation with ferric chloride. The WWTP treats about 11 million m^3 per year. The receiving water for the WWTP is a river. The combined sewers cover 9% of the city area and 20% of the wastewater at the inlet of the WWTP is inflow and infiltration. Approximately 3000 m^3/year is discharged to the receiving water without treatment due to combined sewer overflows (CSOs). This water consists of 91% stormwater, 7% grey water and 2% black water. There are some industries connected to the WWTP, but none that release water that can be expected to contribute with microplastics.

3.2 Description of control measures

To gain insights on ways to prevent and mitigate the pollution from microplastics, control measures were introduced to some of the flows in the model city after the initial flow assessment had been performed. Two types of measures were introduced in the city: preventive and treatment. In this study, preventive measures are those that avoid introduction to the system (e.g., bans) and those that avoid introduction to the urban waters (e.g., more efficient solid waste management). Treatment can either be decentralised and target a specific source or pathway or centralised such as at a large WWTP. Table 5 gives an overview of the types of control measures that was introduced to the model city.

Table 5: Summary of the control measures that are introduced to the model city.

Category	Measure	Source	Builds upon
Preventive	Ban on microbeads	Rinse-off personal care products	Swedish legislation
Preventive	Ban on all intentionally added microplastics	Personal care products (rinse-off and leave-on), cleaning products, behaviour change for paints	EU restriction proposal
Preventive/decentralised	Limit dispersal to 7g/m^2/year	Artificial turfs	EU restriction proposal
Decentralised	Filter in laundry machine	Washing of synthetic textiles	Literature
Centralised	Large stormwater treatment facility	Stormwater	Summary of emissions + measurement in the project
Centralised	Disc filter at WWTP	Wastewater	Literature + measurement in the project.
Centralised	Tertiary biofilter at WWTP	Wastewater	Literature + measurement in the project

3.3 Method for estimating flows in the semi-hypothetical model city

The estimations for flows in the model city was primarily based on the source estimates described in chapter 2 of this report. Some assumptions were made when estimating each source, which is summarised in Table 6. There is no standardized method for sampling and analysis and no established definition of microplastics. Therefore, when using different studies for a source estimate, aspects such as cut-off sizes, analytical methods, and what is considered microplastics for each study needs to be taken into account.

Table 6: A display of background information of the estimates of flows in the semi-hypothetical model city.

Source	Cut-off size	Year	Method	Reference	Assumptions and other notes related to calculating flows in the model city
Tap water	10 µm	2019	µFTIR whole sample (10-500 µm) ATR-FTIR (500-5000 µm)	Measurement in this project	• Transport from the outlet of the drinking water plant to the tap was assumed to not affect microplastics concentrations. • Assumes that all microplastics in the tap water enters the wastewater, i.e., the uptake in humans and the amount that might be used for purposes where it does not become wastewater (e.g., watering plants) was assumed negligible.
Laundry	40 µm	N/A	Light microscope, scanning electron microscope, ATR-FTIR Gravimetric mass estimate	(Dalla Fontana et al., 2020)	• Number of washing cycles and weight of a load was based on the country where the sampled WWTP was located. • European average of synthetic textiles in clothes were used.
	20 µm pore size	N/A	Gravimetric mass estimate	(Kelly et al., 2019)	
	5 µm pore size	N/A	Scanning electron microscope and image software Calculated mass estimate	(De Falco, Gullo, et al., 2018)	
	20 µm pore size	N/A	Gravimetric mass estimate	(De Falco et al., 2019)	
	20 µm pore size	N/A	Gravimetric mass estimate	(De Falco, Gentile, et al., 2018)	
	0.45 µm pore size	N/A	Microscope and image software Calculated mass estimate	(Hernandez et al., 2017)	
Dust	50µm	2019	Stereomicroscope, fluorescent microscope, FTIR	(Soltani et al., 2021)	• Average living area was derived from the city where the sampled WWTP was located.
PCPs	1 µm	2015	Information from industry	(Amec Foster Wheeler, 2017)	• Amec Foster Wheeler (2017) provide the most recent data, include all products put on the market, and provide estimations on a European level, why this data was used for the source estimates.

					• For leave-on products, it was assumed that 50% enters wastewater and 50% enters solid waste. The amount that will wear off when the product is used was assumed to be negligible.
Cleaning products	1 µm	2015	Information from industry	(Amec Foster Wheeler, 2017)	• Amec Foster Wheeler (2017) was chosen as they include the most products.
Rinsing paint brushes	1 µm	2015 Sales data: 2019	Information from industry	(Amec Foster Wheeler, 2017) (Sveff, 2021)	• Sales data for private paint consumption was taken from the country where the sampled WWTP was located and for the same year as when the WWTP was sampled.
Artificial turfs	N/A	2018	Material flow analysis µFTIR whole sample (10-500 µm) ATR-FTIR (500-5000 µm) Py-GCMS	(Hann et al., 2018) Measurements in the project	• The annual refill was not known for the fields sampled within the project. Therefore, the method based on pitch size was used. • Size was derived from the sampled fields.
Cigarette filters	N/A	2020	Litter quantification campaign	(Swedish Environmental Protection Agency, 2020).	• The sampled week was assumed representative for the whole year.
Exterior paint	N/A	Sales data: 2020	Material flow analysis	(Hann et al., 2018; Verschoor et al., 2016). (Sveff, 2021)	• Sales data were taken from the country where the stormwater samples were taken and for the year when the stormwater samples were taken.
Road related emissions	10 µm	2020	µFTIR whole sample (10-500 µm) ATR-FTIR (500-5000 µm) Py-GCMS	Measurement in this project	• Average yearly precipitation (732 mm) for the sampled city was multiplied with the area sampled and the runoff coefficient 0.8 (Swedish Water and Wastewater Association, 2016)
Parking lots	10 µm	2020	µFTIR whole sample (10-500 µm) ATR-FTIR (500-5000 µm) Py-GCMS	Measurement in this project	• Average yearly precipitation (732 mm) for the sampled city was multiplied with the area sampled and the runoff coefficient 0.8 (Swedish Water and Wastewater Association, 2016) • One of three gully pots was sampled, which was assumed to receive one third of the water.

Roof runoff	10 µm	2020	µFTIR whole sample (10-500 µm) ATR-FTIR (500-5000 µm)	Measurement in this project	• Average yearly precipitation (732 mm) for the sampled city was multiplied with the area sampled and the runoff coefficient 0.9 (Swedish Water and Wastewater Association, 2016)

Microplastics are commonly defined as particles smaller than 5 mm, but the lower limit can differ between studies (Lee & Chae, 2021), which can be a challenge for comparison of results from different studies. In Table 6 the different cut-off sizes of the studies used for the source estimates for the model city are summarised. The measured values in this project include particles down to 10µm, but this is not the case for all literature values. However, we hypothesize that such an uncertainty is a larger issue for estimations of particle numbers, as these often increase with decreasing size (Lee & Chae, 2021) but make a smaller difference for mass. As mass is a more stable measurement than particle number and because Pyrolysis-Gas Chromatography-Mass Spectrometry (Py-GCMS) only gives results in mass, mass was used for the flow assessment. In some studies, as well as the µFTIR analysis performed in this project, provides an estimate of the mass rather than an absolute value, which increases the uncertainty.

In the initial stages of microplastics research, particles were typically counted in a microscope (Hidalgo-Ruz et al., 2012), but concern have been raised about the uncertainties with this method as it can be difficult to distinguish the plastic from other small particles in the sample (Li et al., 2018). For the literature values used in the source estimates, the minimum requirement was that at least a subsample was verified with another method (such as Raman or FTIR) to the extent possible. The analytical method was determined as more important for environmental samples than for source samples, as it can be more challenging to determine plastic materials when there are interferences from many other materials in the sample. Analytical methods are also summarised in Table 6.

The definition of microplastics can differ between studies (Hartmann et al., 2019). For example, some studies on textile emissions include rayon as a synthetic fibre, while other do not. Another example is that sometimes the whole emitted particle from a tyre is considered microplastics and sometimes only the rubber part of the tyre. The same has been noted for paint particles. In addition, it can be difficult to determine how much of the polymer added to a product that can be defined as microplastics. The definitions used from the measured data is applied when calculating the source estimates. In practice, this means that the whole particle is considered as microplastics for tyre- and paint particles and that the materials that are detected with the method used are included. Table 7 lists the materials included in the FTIR analysis of the samples taken in this project. This analysis was complemented with Py-GCMS to detect tyre wear particles.

Table 7: Materials included in the automatic FTIR analysis

Material group
Acrylic
Acrylic paints
Acrylonitrile butadiene styrene (ABS)

Alkyd
Aramid
Cellulose acetate
Diene elastomer
Epoxy
Ethylene propylene diene monomer rubber (EPDM)
Ethylene-vinyl acetate (EVA)
PEBAX
Phenoxy resin
Polyamide (PA)
Polycarbonate (PC)
Polyester
Polyethylene (PE)
Polyethylene glycol (PEG)
Polyimide
Polylactic acid (PLA)
Polyoxymethylene (POM)
Polypropylene (PP)
Polystyrene (PS)
Polytetrafluoroethylene (PTFE)
Polyurethane (PU)
Polyurethane paints
Polyvinyl Alcohol/ (PVA)
Polyvinyl Acetate (PVAC)
Polyvinylchloride (PVC)
Polyvinylidene Chloride (PVDC)
Styrene Acrylonitrile (SAN)
Styrene Butadiene Rubber (SBR)
Vinyl copolymer

4. Microplastics in the semi-hypothetical model city

The results from the flow estimates in the model city are shown in Figure 2. The focus is primarily on urban waters, but solid waste and soil were also included as receiving compartments. There are some interactions between wastewater and stormwater as the model city has some combined sewer system, which leads to some stormwater being added to the wastewater and some microplastics being discharged with CSOs. A number of control measures were introduced to the flows presented in Figure 2 and the effect of these are presented in chapter 4.2.

4.1 Flows of microplastics

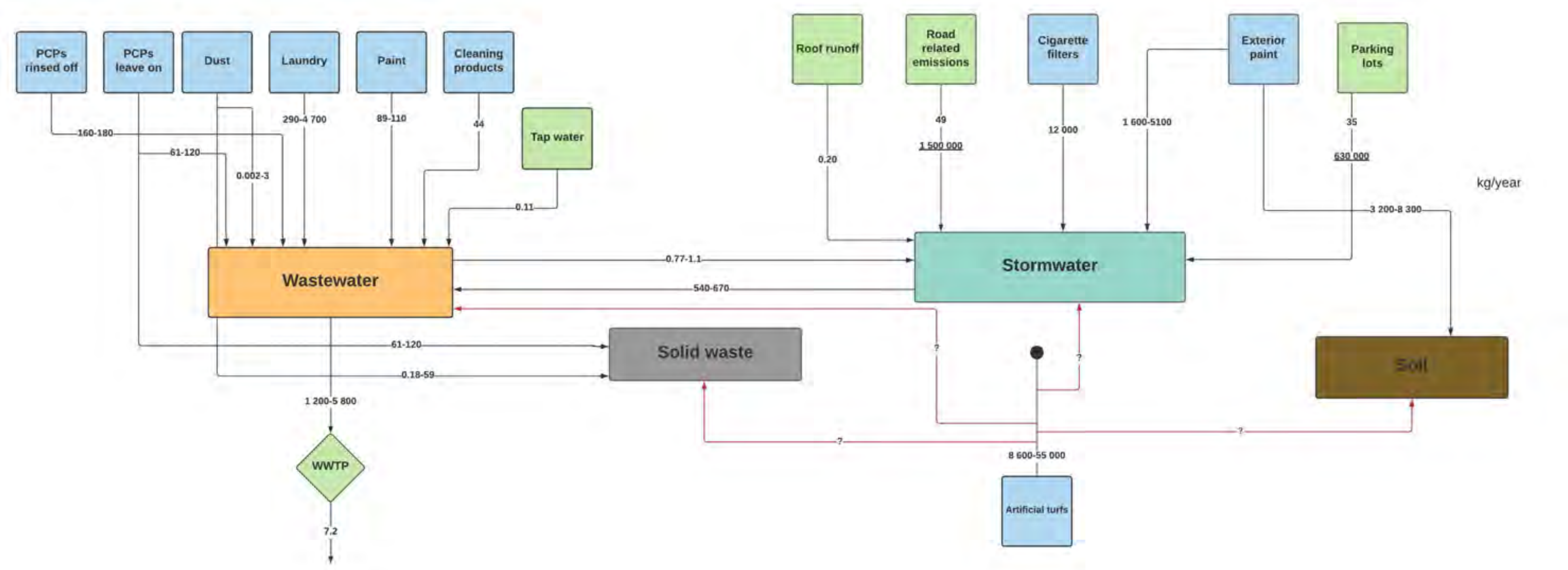

Figure 2: Flows of microplastics in the semi-hypothetical model city in kg/year. The underlined values are tyre wear particles in kg/year. The blue boxes are source estimates, and the green boxes are extrapolated measured values. PCPs stands for personal care products and WWTP stands for wastewater treatment plant.

For raw wastewater, laundry had the highest contribution of microplastics, and tap water and dust had the lowest contribution. For dust, the majority was instead estimated to end up in solid waste. The total load to the WWTP was estimated to 1 200-5 800 kg/year of which the stormwater from the combined sewers contributed with 540-670 kg/year (excluding car tyre particles). The load to receiving waters from CSOs was estimated to approximately 1 kg/year.

The effluent water from a WWTP was sampled within the project and an extrapolation of this measurement gives a yearly load of 7.2 kg microplastics. However, this estimate is uncertain as it is based on only one sample. Other studies that used the same analytical method and the same analysis lab report effluent values from 2.6-73 kg/year (Ljung et al., 2018; Tumlin & Bertholds, 2020). These WWTPs are larger than the one in the model city. However, there is no consistent pattern between size and load to receiving waters. For example, a plant with over 500 000 personal equivalents had lower yearly emissions than the plant in the model city, although this plant had less than 100 000 personal equivalents. The characteristics of the water and the treatment technology likely also impact the effluent load.

The influent at the WWTP in the model city was estimated using source estimates, while microplastics in the effluent were based on measurements. If the influent estimates and effluent value are compared, it suggests a retention of over 99%. There is a risk that some of the sources for the source estimates are overestimated. Fältström et al. (2021) showed that the measured values at the inlet of a WWTP were at the lower end of intervals for the source estimates. Nonetheless, a retention capacity of over 99% is not uncommon (Habib et al., 2020).

Tyre wear particles were estimated to contribute with approximately 1 500 tonnes/year from roads in the city and about 600 tonnes/year from parking lots. However, it should be noted that these values are based on three samples and the concentration varied largely between samples. The roads in the city cover a larger area than parking lots, but if the emissions are compared per square metre, the average emissions from parking lots and roads around 480 g/m^2/year (478 and 481 g/m^2/year, respectively). The similar emissions of tyre wear particles between the roads and the parking lot are surprising. It can be expected that wear on roads is higher due to higher speed and more vehicles in motion. For other microplastics, the parking lots had a higher contribution per square metre than roads (0.026 compared to 0.016 g/m^2/year). One reason for the discrepancy can be that some sources have not been quantified (for example littering), which might be higher on a parking lot than on a road. These results indicate that parking lots need to be further considered as a pollution source of microplastics both in future research and for pollution mitigation measures. The results also show that tyre wear particles had a much higher contribution than other microplastics, which corroborates the findings in road dust by Goßmann et al. (2021).

The theoretical contribution of microplastics and tyre particles from the sampled road was compared to the measured values in the road runoff. The theoretical contribution was made up of tyre wear, road marking, and brake wear. For microplastics, the theoretical contribution was higher than what was found in the samples (400-1 300 g/year compared to 7 g/year) while it was opposite for tyre particles. For tyre wear particles, the theoretical contribution was estimated to 32 kg/year and the measurement-based estimate was 208 kg/year.

Artificial turfs were estimated to have high emissions at source (8400-53 000 kg/year from infill and 250-1000 kg/year from pile) but the potential emissions to stormwater and wastewater was determined as too uncertain to assess. Within this project, samples were taken in a river upstream and downstream (60 and 100 metres, respectively) the location of two artificial turfs. A comparison of the upstream and downstream samples showed no increased concentration of neither PE nor granulate, which suggest that the particles does not travel far.

For the sources where a load to stormwater was estimated, cigarette filters had the highest load, followed by exterior paint, if tyre wear particles were excluded. However, when investigating the polymers in the stormwater sampled in this project, neither cellulose acetate nor polymers commonly associated with paint were prevalent. In an aggregated stormwater from an urban area sampled in this project, two samples were dominated by PP (93-97%) and the other two by PE (60-91%), of which PP had the rest of the share. PP also dominated the road runoff. PP and PE, together with polyester were also prevalent in roof runoff and parking lot runoff. Neither brake wear nor road marking are expected to contain PP (see section 2.2.5 in this report), but some automotive parts can be made up of PP (FanpLESStic-sea, 2019) and paint can contain polyester. The paint on the sampled roof and gutter was tested, but it did not contain PP nor polyester. The roof paint was primarily made up of polyvinylidene fluoride (PVDF) and the gutter paint of acrylate. Neither of these were detected in the roof runoff samples. There is athe possibility that some sources have been missed. PP and PE are the two most common synthetic polymers and are used for a variety of purposes, such as food packaging, pipes, and shopping bags (FanpLESStic-sea, 2019). Plastic litter that fragment into microplastic-sizes has been highlighted as a potentially large source (Galafassi et al., 2019), but the quantification of this source can be challenging (see section 2.2.6 in this report) and therefore it has not been estimated for the model city.

In total, the emissions to the receiving water were approximately 7 kg/year from treated wastewater and 13 000-17 000 kg/year from stormwater (excluding tyre wear particles). If only the sampled pathways for stormwater were considered, the emissions to the receiving water were 82 kg/year for microplastics and 2 100 tonnes/year for tyre wear particles. Scherenewski et al. (2021) also found stormwater runoff (together with CSOs, which were not separated in their study) was the largest urban pathway to the Baltic Sea.

4.2 Effect of control measures on flows of microplastics in the semi-hypothetical model city

Preventive and treatment measures addressing both stormwater and wastewater (see chapter 3.2) were introduced to the flow assessment presented in Figure 2.

4.2.1 Wastewater

PCPs that are rinsed off have been prohibited in several countries. If this measure was implemented in the model city, it would decrease the microplastics load from households with 3-22% and the total load at the inlet of the WWTP with 3-12%. There is also a proposal to limit all intentionally added microplastics, which would impact PCPs that are left on the skin and cleaning products, in addition to PCPs that are rinsed-off. Microplastics in paints would not be prohibited under this restriction. Instead, consumers should be informed on how to clean equipment without rinsing. If all inhabitants in the city would stop rinsing the equipment in water, and the other uses would be prohibited, this would decrease the total load from households with 9-49% and the total load at the inlet of the WWTP would be reduced with 8-28%.

The largest wastewater-related source in the model city was synthetic fibres from laundry (Figure 2). The emissions from laundry were estimated to stand for 40-90% of the total load from households and 23-80% of the total load in the influent water to the WWTP. There are several technologies that have been developed to reduce the emissions from laundry. Napper et al. (2020) tested different devices, both filters and devices that are put into the washing machine. The variety in performance was large, between 21% (a washing bag) and 78% (a filter) retention. Browne et al. (2020) report a similar retention for a filter (74%). If one of these filters would be installed in all washing machines, the laundry emissions would decrease from 290-4 700 kg/year to 64-1 200 kg/year.

To sum up, if all the above-described measures were implemented and the inhabitants continued to vacuum before they wet wash their floors, the total load to the WWTP would decrease from 1 200-5 800 kg/year to 670-2 000 kg/year. As conventional treatment at WWTPs are efficient and the load in effluent often low even without any interventions, this change might not make a big difference for the load to receiving waters. However, the benefit of preventive measures compared to treatment options are that the introduction to the system is avoided and, hence, the pollution is not shifted from one compartment to another. In other words, even if these microplastics would have been captured in the treatment processes at the WWTP, the introduction to other compartments, mainly the sewage sludge, are avoided.

There is the possibility of additional treatment at the WWTP. There are two treatment options that have been investigated with the same analytical method as used in this project (and performed by the same laboratory): a biofilter for tertiary treatment (Liu, Nord et al., 2020) and a disc filter (Simon et al., 2019). Both treatment technologies showed an increased retention, 76% for disc filter and 89% for the biofilter, based on mass. For the total load to the receiving water, the yearly emissions from the WWTP would decrease from 7.2 to 0.8 kg/year if installing the biofilter and to 1.8 kg/year if installing a disc filter.

4.2.2 Stormwater

Artificial turfs are a large source, but how much will enter the stormwater is uncertain. If the use of rubber-based granulate was prohibited, it would remove all the emissions from the infill, which stand for the large majority of the emissions (97-98%). The 250-1000 kg/year that was estimated to be emitted from the pile will remain. There has also been a proposal to not impose a ban, but to limit the dispersal of microplastics to a maximum of 7g/m^2 (European Chemicals Agency, 2020), which would mean that the total emissions from all fields in the model city would be 550 kg/year, a reduction of 93-99%. Regnell (2019) showed that a combination of traps and change in practice could almost entirely avoid the introduction of granulate to the environment.

A nature-based filtration treatment technology for stormwater was tested within the FanpLESStic-Sea project[4]. It showed a high retention of microplastics (93%) and a moderate retention for tyre wear particles (47%) when one test was performed, but more tests are needed to confirm the retention. The pilot technique has the possibility to treat all stormwater in the model city, which means that there is the potential to lower the microplastics emissions to the receiving river from 13 000-17 000 to 940-1 200 kg/year for microplastics and from 2100 to 1100 tonnes/year for tyre wear particles. However, even if the technique theoretically can handle the water flow, urban stormwater generally has a large number of discharge points and installing it will require large reconstruction of the stormwater collection system. A system for handling the sediment would also be required.

5. Conclusions

One of the goals of the FanpLESStic-Sea project was to increase the knowledge on where microplastics come from and their transport pathways. By combining strategic measurements with source estimates, the flows were visualised for a model city in the Baltic Sea region. The, by far, largest source to urban waters were tyre wear particles. For wastewater the highest load came from laundry. Tap water, dust, and roof runoff all made a small contribution to the overall load to urban waters. The emissions to the receiving water were higher from stormwater than wastewater, even if only pathways where measurements were taken were considered.

Several policy interventions have been proposed, mostly for wastewater sources. If all control measures for wastewater were to be implemented there is the potential to cut emissions to the WWTP with 30-50%. For stormwater, more research is needed on the techniques that would be most efficient for treatment of microplastics and tyre wear particles, and under what circumstances treatment at source and centralised treatment in large scale facilities is preferred.

There are still several uncertainties related to source estimates and the agreement between expected polymers and the polymers found in the samples were sometimes not consistent. This raises the question if some sources are missed, while other might be overestimated. Microplastics in water has been studied more than microplastics in other environments, such as urban soils. Therefore, it can be difficult to assess if the emissions at source are overestimated or if it is the split between compartments that is not accurate.

[4] More information on the treatment technology can be found here: Copy of Template FanpLESStic-sea fact sheet (swedenwaterresearch.se)

In the future, microplastics in other compartments, such as urban soils, should receive increased attention to receive a more comprehensive view of the flows. Further, the flows based on extrapolated measured values were based on a few measurements at each sampling point and these showed large variations in concentration. Taking more samples over a longer time period that capture for example seasonal variations will also advance the understanding of the flows.

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