THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

Tyre wear particles in the near-road environment of a Swedish highway

Characterisation of concentrations and transport dynamics across snow, soil and through a stormwater system

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Water, Environment and Technology

CHALMERS UNIVERSITY OF TECHNOLOGY

Gothenburg, Sweden 2025

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ISBN 978-91-8103-257-4.

Doktorsavhandlingar vid Chalmers tekniska högskola
Ny serie nr 5715
ISSN 0346-718X

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Cover: Photo of highway E18 by Testsite E18 (Sweden) taken on 2022-12-16 (Elly Lucia Gaggini, 2022)

Printed by:

Chalmers Reproservice

Gothenburg, Sweden 2025.

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ABSTRACT

Tyre wear particles (TWP) are increasingly recognised as a major source of microplastic pollution in urban environments, particularly in areas with high traffic intensity. Several laboratory studies suggest toxic effects from TWP and chemical compounds found in tyres, that can leach into aquatic environments. However, the occurrence of TWP in various environmental compartments remains poorly quantified, especially in the finer size fraction <20 µm.

This thesis investigates the environmental concentrations, transport pathways, and morpho-chemical characteristics of TWP in the near-road environment of a Swedish highway. TWP were quantified across multiple environmental compartments—snow, soil, stormwater, sediments, and recipient waters—with special attention to the fine fraction (<20 µm). The methodology employed for the analysis of the samples was a quantification of styrene-butadiene rubbers with pyrolysis-gas chromatography-mass spectrometry using a combination of marker compounds (benzene, α-methylstyrene, ethylstyrene, and butadiene trimer) followed by Monte Carlo simulations for calculation of TWP concentrations. The work in this thesis also includes quantification of other microplastic polymers in stormwater and detailed morphological and chemical characterisation of TWP in snow.

TWP were consistently detected in all sampled compartments (snow, soil, stormwater, recipient waters and sediments) confirming their widespread distribution. Elevated concentrations were found in roadside snow and in stormwater during rain, with limited retention observed in the stormwater system. Fine particles (1.6–20 µm) represented a substantial proportion of total TWP, particularly in runoff, highlighting their relevance for monitoring and treatment.

Microplastic analyses revealed polyethylene, polypropylene and rubber polymers as the most common polymers in runoff from highway stormwater systems. Correlation analyses showed strong associations between TWP and suspended solids, organic content, and metals in waterborne sample matrices, while more complex behaviours were observed in soil. These findings provide new insights into TWP dynamics and support more effective risk assessment and mitigation strategies.

Keywords: Automatic sequential sampling, Gully pot, Microplastics, Pyrolysis-gc/ms, Road runoff, SEM-EDX, Snow, Soil, Stormwater pollutants, Tire and road wear

SAMMANFATTNING

Däckslitagepartiklar (Tyre Wear Particles – TWP) betraktas i allt större utsträckning som en betydande källa till mikroplaster i urbana miljöer, särskilt i områden med hög trafikintensitet. Flera studier har visat att TWP och de kemikalier som finns i däck kan ha toxiska effekter och lakas ut till vattenmiljöer. Trots detta är förekomsten av TWP i olika miljöer fortfarande otillräckligt kartlagd, särskilt för TWP med partikelstorlek <20 µm.

I denna avhandling undersöktes miljökoncentrationer, transportvägar samt morfologiska och kemiska egenskaper hos TWP i närmiljön av en motorväg i Sverige. TWP kvantifierades i flera typer av miljömatriser – snö, jord, dagvatten, sediment och recipientvatten – med särskilt fokus på partiklar av mindre storlek (<20 µm). Proverna analyserades genom kvantifiering av styrengummi med pyrolys-gaskromatografi-masspektrometri (Pyr-GC/MS), där markörämnen som bensen, α-metylstyren, etylstyren och butadientrimer användes. Monte Carlo-simuleringar tillämpades därefter för att beräkna TWP-koncentrationer. Arbetet omfattar även kvantifiering av andra mikroplastpolymerer i dagvatten samt en detaljerad karaktärisering av TWP i snö, med avseende på storlek, form och kemiskt innehåll av partiklarna.

TWP återfanns konsekvent i samtliga provtagna miljömatriser, vilket bekräftar deras utbredda förekomst. Särskilt höga halter uppmättes i snö från vägkanten och i dagvatten under regn, vilket tyder på att TWP endast fångades upp i begränsad utsträckning i dagvattensystemets rännstensbrunnar. Mindre partiklar (1,6–20 µm) utgjorde en betydande andel av den totala mängden TWP, särskilt i dagvatten, vilket understryker behovet av fortsatt miljöövervakning och reningsåtgärder.

Mikroplastanalyserna visade att polyeten, polypropen och olika gummipolymerer var de vanligaste polymererna i dagvattnet från motorvägen. Korrelationsanalyser påvisade starka samband mellan TWP och suspenderade partiklar, organiskt material samt metaller i miljöprover i vattenfas, medan mer komplexa mönster observerades i jord. Resultaten bidrar med ny kunskap om TWP:s förekomst och transport och kan stödja mer effektiva riskbedömningar och åtgärdsstrategier.

LIST OF PUBLICATIONS

This thesis is based on the work from the following papers:

I. Gaggini, E.L., Sokolova, E., Rødland, E.S., Strömvall, A.-M., Andersson-Sköld, Y., Bondelind, M., 2025. Characterisation and spatial distribution of tyre wear particles in Swedish highway snow: Loads into roadside ditches and risk of emissions with snowmelt. *Environmental Challenges,* 20, 101228. https://doi.org/10.1016/j.envc.2025.101228

II. Polukarova, M., Gaggini, E.L., Rødland, E., Sokolova, E., Bondelind, M., Gustafsson, M., Strömvall, A.-M., Andersson-Sköld, Y., 2025. Tyre wear particles and metals in highway roadside ditches: Occurrence and potential transport pathways. *Environmental Pollution,* 125971. https://doi.org/10.1016/J.ENVPOL.2025.125971

III. Gaggini, E.L., Polukarova, M., Bondelind, M., Strömvall, A.-M., Andersson-Sköld, Y., Sokolova, E., 2024. Assessment of fine and coarse tyre wear particles along a highway stormwater system and in receiving waters: Occurrence and transport. *Journal of Environmental Management,* 367, 121989. https://doi.org/10.1016/J.JENVMAN.2024.121989

IV. Gaggini, E.L., Sokolova, E., Rødland, E.S., Strömvall, A.-M., Andersson-Sköld, Y., Bondelind, M., 2025. Tyre wear particles in a highway stormwater system during rain: quantification by automatic sampling and Pyrolysis-GC/MS and correlations with metals, solids and turbidity. *Manuscript submitted.*

V. Gaggini, E.L., Strömvall, A.-M., Rødland, E., Sokolova, E.S., Andersson-Sköld, Y., Bondelind, M., 2025. Microplastics in a highway stormwater system: concentrations and variability during rainfall. *Manuscript submitted.*

The work in this thesis was supported by the FORMAS project (*Mikroplast i vägdagvatten: förekomst, egenskaper och transportmodellering* 2019-00284) and by the Norwegian Public Road Administration (Statens Vegvesen, B11191 Ferjefri E39). The sampling campaigns and analyses carried out were further supported by J. Gust. Richert foundation (2022-00807), Sveriges Ingenjörer Environmental fund (2022), Adlerbertska foundation (2022), and Åke and Greta Lissheds foundation (2022-00188).

The author has contributed in the following ways to the journal articles included in the thesis:

Paper I: Planned and performed the study. Procured field tools and conducted all field measurements, sample collection, and sample pre-treatment. Performed laboratory analyses (solids concentrations, nanoparticles, conductivity, and pH). Carried out data analysis and statistical analyses. Prepared data visualisations for the manuscript. Wrote the original draft, edited and reviewed the text in collaboration with co-authors and following external review.

Paper II: Assisted and participated in planning of field measurements, sampling, procurement of tools for field and laboratory use. Contributed to developing methodologies for sampling, laboratory work and data processing and analysis. Prepared data visualisations for the manuscript. Edited and reviewed the manuscript draft in collaboration with co-authors and following external review.

Paper III: Planned and performed the study. Procured field tools and conducted all field measurements, sample collection, and sample pre-treatment. Performed laboratory analyses (DS, LOI, TSS, and VSS). Carried out data analysis and data visualisation for the manuscript. Wrote the original draft, edited and reviewed the text in collaboration with co-authors and following external review.

Paper IV: Planned and performed the study. Procured field tools and conducted all field measurements, sample collection, and sample pre-treatment. Performed laboratory analyses (TSS, VSS, and turbidity). Carried out data analysis and statistical analyses. Prepared data visualisations for the manuscript. Wrote the original draft, edited and reviewed the text in collaboration with co-authors and following external review.

Paper V: Planned and performed the study. Procured field tools and conducted all field measurements, sample collection, and sample pre-treatment. Carried out data analysis and data visualisation for the manuscript. Wrote the original draft, edited and reviewed the text in collaboration with co-authors.

The author has contributed to the following work and publications, which are not appended to the thesis:

Gaggini, E., Bondelind, M., Andersson-Sköld, Y., Sokolova, E. (2022). *Numerical modelling of road related microplastics in urban gully pot.* Conference: SETAC Europe 32nd Annual Meeting (Poster).

Gaggini, E., Sokolova, E., Polukarova, M., Rødland, E.S., Strömvall, A-M., Andersson-Sköld, Y., Bondelind, M. (2024). *Tyre Wear Particles at a Swedish Highway– Occurrence in Stormwater, Sediments and Snow*. Conference: SETAC Europe 34th Annual Meeting (Poster).

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ACKNOWLEDGEMENTS

The long journey that this PhD project has been comes with its fair share of acknowledgements and gratitude for the support received along the way.

First and foremost, I would like to express my sincerest thanks to my main supervisors Associate Professor Mia Bondelind and Associate Professor Ekaterina Sokolova (Uppsala University), for taking on this journey with me and for their dedication and guidance. I also thank my co-supervisor Prof. Yvonne Andersson-Sköld for her support and insights, and my collaborators Prof. Ann-Margret Strömvall and Dr Elisabeth Rødland (NIVA) for their expertise and generous availability throughout the work.

The work carried out in this thesis would not have been possible without the assistance of Tobias Ulegård (Swedish Transport Administration), Andreas Asp (VAP). Mats Riehm (Swedish Transport Administration), and Björn Sundqvist (Swedish Transport Administration). I would also like to thank Dr Niels Markwat for assistance with the statistical analysis; Dr Albert Celma (SLU) and Aniruddha Kshirsagar for assistance in the field at Testsite E18; Dr Ida Järlskog (VTI) for equipment loan and valuable advice; Lene Sørlie Heier (Statens Vegvesen) for providing valuable guidance; Dr. David Jaramillo and Dr. Juanita Rausch (Particle Vision) for a good collaboration.

Special thanks go to Dr Amir Saeid Mohammad for the endless support, patience and encouragement in the lab. Many thanks also to my comrades in the PhD trenches: Maria Polukarova for the many insightful discussions and mutual support; Glenn Johansson for the many loans of appliances and fieldwork help; and Dr Gabriella Rullander (Uppsala University) for organising our journal club and the great discussions we shared there.

I would also like to thank my examiner Prof. Britt-Maire Wilén, and my colleagues at the division of *Water Environment Technology* both past and present, as well as those at the divisions of *Geology and Geotechnics* and *Building Technology*. Special appreciation goes to the colleagues I've worked with in teaching the civil engineering students. Thanks also to Anna, Cecilia, Divia, Jennifer, Marie, and Victor for generously sharing their wisdom and helpful advice.

I would like to recognize the organisers and teachers at the Water Research School for their excellent courses, my mentor Marinette Hagman (DHI), the Young Water Professional Sweden network for their interesting seminars, and Kretslopp och Vatten (Gothenburg municipality) for their assistance on numerous occasions.

Finally, I would like to send my appreciation to my family, sambos and friends for their support and to Per for pushing me through the final stretch of this journey.

Göteborg, July 2025

Elly Lucia Gaggini

ABBREVIATIONS

The following definitions and abbreviations are used in the thesis:

AADT	Annual average daily traffic
ABS	Acrylonitrile butadiene styrene
BR	Butadiene rubber
DS	Dry substance
ECD	Equivalent circular diameter
EMC	Event mean concentration
FTIR	Fourier-Transform Infrared Spectroscopy.
GC/MS	Gas Chromatography-Mass Spectrometry
ICP-MS	Inductively coupled plasma mass spectrometry
LOI	Loss on Ignition
MP	Microplastics
PA6	Polyamide 6
PA66	Polyamide 66
PB	Polybutadiene
PC	Polycarbonate
PE	Polyethylene
PET	Polyethylene terephthalate
PI	Polyisoprene
PMMA	Poly(methyl methacrylate)
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
Pyr-GC/MS	Pyrolysis-Gas Chromatography-Mass Spectrometry.
SBB	Styrene butadiene hybrid trimer
SBR	Styrene-butadiene-rubber
SBS	Styrene-butadiene-styrene
SEM-EDX	Scanning Electron Microscope with Energy-Dispersive X-ray spectroscopy.
TBWP	Tyre and bitumen wear particles
TWP	Tyre Wear Particles / Tire Wear Particles
4-PCH	4-phenylcyclohexene
4-VCH	4-vinylcyclohexene

1 INTRODUCTION

Microplastics (MP) have emerged as a growing environmental concern due to their widespread occurrence, persistence, and ecological and health impacts (Thompson et al., 2024). One of the most significant and diffuse sources of MP is tyre wear particles (TWP), which are generated on roads through friction between tyres and road surfaces. TWP emissions are estimated at around 6 million tonnes per year globally (Kole et al., 2017) with TWP emissions from Swedish road amounting to 11 000–12 000 tonnes/year (Polukarova et al., 2024), and they are believed to represent a dominant source of MP in urban water systems, particularly via stormwater runoff (Fältström, 2020). Despite their importance, considerable uncertainty remains regarding the emission rates and environmental fate, and concerns have been raised about the toxic effects of TWP to aquatic biota, primarily due to leaching of chemical additives and metals, which are key components in tyre formulations (Liu et al., 2022; Tian et al., 2021a; Wik and Dave, 2009, 2006a).

Due to several analytical challenges in detecting and quantifying TWP in environmental sample matrices (Mattonai et al., 2022; Rødland et al., 2023a; Wagner et al., 2018), their occurrence and spread in the near-road environment have not been extensively assessed. Reliable data on environmental TWP concentrations are needed to evaluate ecological risks under relevant exposure conditions and to better understand the transport patterns of TWP. Information about TWP in the finer size fraction <20 µm is lacking due to analytical challenges. Knowledge about the particle sizes and morphology is important for the environmental fate of TWP, as TWP <30 µm are not expected to settle in stormwater systems to a large extent (Vogelsang et al., 2020) and particle shape can affect settling behaviour (Egarr et al., 2004). Additionally, TWP can contain varying amounts of mineral encrustations (Sommer et al., 2018), that in turn can yield different particle densities (Wilkinson et al., 2023), which is another key factor affecting sedimentation, thus complicating predictions of their behaviour.

Understanding the distribution of TWP in compartments such as snow, water, sediments, and soils is important for evaluating their fate and potential environmental impact. The occurrence and mobility of TWP are influenced not only by particle properties (e.g. size, shape, density), but also by environmental factors (e.g. rainfall intensity, snowmelt, wind), and the surrounding infrastructure. The finer airborne fraction (PM_{10}), estimated to be about 10% of TWP can be transported for long distances (Evangeliou et al., 2020). TWP have been found to deposit due to wet and dry deposition with decreasing concentration per increasing distance from the road, but only few studies have assessed this (Cadle and Williams, 1978; Järlskog et al., 2022a). Snow has been proposed as a passive sampler for traffic-related pollutants (Alexandra Müller et al., 2022), which could help to better understand TWP deposition patterns. Roadside snowbanks have also been found to contain very high TWP concentrations, pointing out snowmelt as a risk for acute emissions (Rødland et al., 2022b). From the road surfaces, TWP can be mobilised with runoff and spread further. On most rural roads and highways, runoff is usually drained and infiltrated into roadside soils that also receive TWP depositing from air, and the TWP are suggested to be retained in the soils (Baensch-Baltruschat et al., 2021), but the knowledge on TWP occurrence in roadside soil is still lacking (Axel Müller et al., 2022). If a stormwater system is present, runoff can spread TWP through the system to receiving waters (Barber et al., 2024; Rauert et al., 2022). These dynamics highlight the importance of studying TWP across multiple environmental compartments to better understand their spread, accumulation, and potential risks, as well as to support the development of effective mitigation strategies.

Information of TWP occurrence, characteristics and transport patterns is needed in order to evaluate the removal efficiencies of stormwater treatment technologies, to help in the establishment of guideline values for environmental monitoring, and to prioritise where stormwater treatment solutions should be implemented to protect sensitive receiving waters. Due to the complexity and cost of TWP analyses, the possibility of utilising other well-established water quality parameters, such as total suspended solids, as indicator parameters has been proposed (Rødland et al., 2022a), to enable broader monitoring of TWP, though research on such parameters has been limited so far.

1.1 Research aim and objectives

The aim of this thesis was to assess the occurrence, characteristics and transport dynamics of TWP in near-road environments and stormwater systems. By conducting field sampling across several environmental compartments (snow, soil, runoff, sediments and natural recipients) along a Swedish highway, the work has improved the understanding of TWP distribution and transport processes both near the road and within a stormwater system during rainfall, while also providing insights into the abundance of other microplastic polymers in highway stormwater.

The research questions of the thesis were:

1) What are the concentrations of TWP across various environmental compartments of the near-road environment?
2) What are the characteristics of TWP in terms of sizes and morpho-chemical properties?
3) What are the transport patterns of TWP and MP in the near-road environment and in stormwater systems?

The research objectives were to:

- Assess the TWP concentrations in roadside snow and soils as a measure of TWP loads into highway roadside ditches and assess the morpho-chemical characteristics of TWP in roadside snow (Paper I and II)
- Investigate the concentrations of TWP in stormwater and sediments of a stormwater system to assess the transport of TWP, including presence in receiving waters (Paper III)
- Examine the occurrence of TWP in the fine particle size fraction 1.6–20 µm across multiple environmental compartments (Paper I, III and IV)
- Quantify the concentrations of TWP and other MP in the gully pots and wells of a highway stormwater system during rain events (Paper IV and V)
- Investigate the correlations between TWP and other water quality parameters and pollutants— such as solids concentration, LOI, TSS/VSS, metals, and ions—to better understand the transport behaviour of TWP (Paper I, II, III and IV).

1.2 Summary of papers

The thesis consists of five papers numbered from I to V of which an overview is given in Figure 1. A summary of each paper is given below.

Paper I: This study explores the occurrence, characteristics, and spatial distribution of TWP in snow in the near-road environment of Swedish highway E18 by a road research facility. Snow from the road, from the ploughed packed snow close to the road and from the roadside ditches was sampled 0–5 m from the road. Sample pre-treatment and sample analysis using Pyrolysis–Gas Chromatography/Mass Spectrometry (Pyr-GC/MS), targeted and quantified TWP in two size fractions: <500 µm and 1.6–20 µm. Morpho-chemical characterization of TWP and solids in the snow was performed through Scanning Electron Microscopy coupled with Energy Dispersive X-ray (SEM-EDX) for morphological and elemental analysis. Additionally, the study examines TWP transport behaviour by comparing it with selected stormwater-related parameters, including conductivity, nanosized particle concentration, total and dissolved carbon and organic carbon, ions, and total and dissolved metals.

Paper II: In this study, the distribution and transport of TWP and metals in roadside soils by the highway at Testsite E18 were examined. Soil samples were collected at varying depths and distances to assess TWP concentrations using Pyr-GC/MS and their correlations with metals. Additional analyses of soil size fractions through sieving and LOI were carried out to characterize the soil samples.

Paper III: In this study, the occurrence and transport of TWP through a highway stormwater system by Testsite E18 was investigated. Water and sediment samples, collected in the wells and ditches of the stormwater system, and in nearby streams, together with a sample of direct runoff from the road, were analysed through Pyr-GC/MS. By fractionating samples into fine 1.6–20 µm and coarse <500 µm size ranges, the study aimed to enhance the understanding of TWP pathways and their potential to spread through stormwater networks into receiving environments.

Paper IV: This study explored the transport dynamics of TWP in the highway stormwater system by Testsite E18 during rain events using automatic sequential sampling. Runoff samples were collected from gully pots and stormwater wells and analysed for TWP, metals, total suspended solids (TSS), volatile suspended solids (VSS), and turbidity. The analyses aimed to assess the temporal variability of TWP concentrations during rain, together with the outflowing event mean concentration (EMC) from the stormwater system, thus providing insights into TWP transport pathways, similarities to other pollutant behaviours and implications for stormwater treatment design.

Paper V: This study investigated the occurrence and distribution of microplastics (MP) in highway runoff during rain using automatic sequential sampling. Thirteen plastic polymers were analysed through Pyr-GC/MS. The samples analysed consisted of aggregated samples of stormwater collected in Paper IV, with the aim of quantifying a broader range of MP polymer types, and to assess the relative abundance of rubber-derived polymers. This was done using Pyr-GC/MS analyses by commercial laboratory on particles in size range 10–1000 µm.

Research questions:

- What are the concentrations of TWP in different parts of the near-road environment?
- What are the characteristics of TWP in terms of sizes, and morpho-chemical properties?
- What are the transport patterns of TWP and MP in the near-road environment and in stormwater systems?

	Paper I	Paper II	Paper III	Paper IV	Paper V
Aim	Investigate the characteristics, abundance and spatial distribution of TWP in the near-road environment of a highway.	Characterise occurrence and distribution of TWP and metals in roadside soils and to investigate the correlations between the contaminants	Assess the transport behaviour of TWP in a stormwater system during rain events and compare it to behaviour to other stormwater pollutants	Assess the transport behaviour of TWP in a stormwater system during rain events and compare it to behaviour to other stormwater pollutants	Assess the transport behaviour of MP in a stormwater system during rain
Sampling matrix	Roadside snow	Roadside soil	Standing water from stormwater system and sediments River water and sediments	Stormwater in stormwater system during rain	Stormwater in stormwater system during rain
Methods	TWP quantification (<500 µm) Quantification TWP fraction (1.6–20 µm) TWP morpho-chemical characterisation TWP correlation to water quality parameters	TWP quantification (<500 µm) TWP correlation to metals	TWP quantification (<500 µm) Quantification TWP fraction (1.6–20 µm) TWP correlation solids and organic matter	TWP quantification (<500 µm) Quantification TWP fraction (1.6–20 µm) Sequential sampling during rain TWP correlation to solids, organic matter and metals	MP polymers quantification (10–1000 µm) Sequential sampling during rain
Key findings	Strong decrease of TWP within 5 m from road. Similar TWP characteristics to TWP in road dust. High but varying proportion of TWP in fraction <20 µm High concentrations in snowmelt are cause of concern for acute emissions Strong correlation with particle-related pollutants indicates similar transport	TWP found at all soil depths TWP concentrations decreased with distance from road Traffic-related metals were the most abundant Varying correlation between metals of different types and TWP	TWP concentrations similar to other studies, also detected in natural waters. High amounts of TWP in fraction <20 µm. TWP concentration in sediments to do not decrease across the stormwater system, indicating spread	Large variation in concentrations in the gully pots during rain event Not first-flush according to definition. Strong correlation with particulate matter and metals. No difference in fraction <20 µm during rain	No pattern in terms of polymer release during rain PP and PE most common polymers Rubber polymers make up approx. one third of MP detected – important TWP contribution

Figure 1: Overview of the research question of the thesis and the research papers included in the work. The coloured boxes indicate which research questions are being addressed by the methodologies applied in each paper.

2 THEORETICAL BACKGROUND

2.1 Stormwater as a vector of traffic-related pollutants

Stormwater or runoff is water from precipitation or snowmelt that flows on paved or unpaved surfaces such as roads, parking lots, roofs, lawns and gravel (City of Gothenburg, 2025; United States Environmental Protection Agency, 2024). Paved impervious surfaces reduce the infiltration, interception and evapotranspiration of runoff, causing larger runoff volumes and greater risk of flooding, especially for high intensity rainfalls (Erickson et al., 2013), and extreme precipitations have been observed to increase globally (Westra et al., 2013). Stormwater can also be a vector for a large range of different types of pollutants, including sediments and particulate material, nutrients, metals and organic pollutants (Björklund et al., 2011; Davis et al., 2001; Legret and Pagotto, 1999; Markiewicz et al., 2017; Rossi et al., 2006; Vijayan et al., 2024a; Westerlund et al., 2003; Yang and Lusk, 2018). This is especially the case in the built environment, where diffuse pollutant sources such as traffic, construction and industrial sites, in combination with large amounts of impervious surfaces, can generate large amounts of contaminated runoff (Erickson et al., 2013). Runoff discharges can negatively affect the quality of receiving waters (Crabill et al., 1999; Mallin et al., 2009; Marsalek, 2002) and is one of the leading causes of waterbody impairment in the United States (United States Environmental Protection Agency, 2004). In many OECD countries, efforts targeting point sources of pollution, such as wastewater treatment, have improved the situation, but challenges persist due to diffuse sources, such as runoff, which have been more difficult to target due to their diversity and spatial and temporal variability (OECD, 2017).

According to a survey answered by Swedish municipalities, only 8% of stormwater from Swedish urban areas receives some form of treatment. As a result, large volumes of runoff reach recipient waters untreated with potentially high pollutant loads (Swedish Environmental Protection Agency, 2017). Historically, conventional stormwater management has been mainly focused on storing or conveying runoff away from developed areas to prevent flooding (Arya et a., 2023), with limited attention to water quality. Since around 2012, research has increasingly focused on more integrated stormwater management approaches, such as green infrastructure, low impact development, nature-based solutions, and sustainable urban drainage systems (Ferrans et al., 2022). These approaches can address both quantity and quality aspects of stormwater by providing runoff retardation, improvement of stormwater and groundwater quality, urban climate regulation, water cycle restoration, and recharge of aquifers (Arya et al., 2023; Gogate et al., 2017; Maglia et al., 2025). Within sustainable urban drainage frameworks, remediation of stormwater contaminants is addressed close to the pollutant source (Tuomela et al., 2019), and solutions such as reduction of traffic intensity and street sweeping can be effective for reducing pollution from traffic (Polukarova et al., 2020; Wang et al., 2023). Other remediation strategies can target the pollutants further downstream from the sources through end-of-pipe solutions (Vezzaro et al., 208), like stormwater ponds, wetlands, swales, biofilters and infiltration systems (Blecken, G.T.,2016; Erickson et al., 2013).

2.2 Microplastics and tyre wear particles as emerging contaminants

In recent years, concerns have been raised about the presence of microplastics (MP) in the environment. The global plastic production increased dramatically from the 1950s to amounts over 350 million tonnes in 2015 (Geyer et al., 2017; PlasticsEurope, 2016, 2006). Estimates on the fate of the plastic waste suggest that 79% ends up in landfills or in the environment, whereas only 9% is recycled and 12% incinerated (Geyer et al., 2017). Research literature dealing with MP has increased exponentially (Zhou et al., 2021) and microplastics have been reported in a multitude of environmental compartments, such as oceans, freshwater, stormwater, sediments, soils, and air (Bäuerlein et al., 2022; Cho et al., 2023; Goßmann et al., 2023b, 2023a, 2022; Heinze et al., 2024; Koelmans et al., 2019; Laju et al., 2022; Lange et al., 2022; Liu et al., 2019; Öborn et al., 2024b), and also in habitats and animal species (GESAMP, 2016). Knowledge of the effect of MP on ecosystems and on human health remains limited, and ecological risks may occur locally, but can escalate if microplastic emissions are constant or increasing (GESAMP, 2016; SAPEA, 2019). Stormwater has been identified as a major vector of MP into the marine environment (Boucher and Friot, 2017), and has been estimated to be the major MP source in urban water systems, outweighing wastewater by 6–13 times (Fältström, 2020). Many types of plastic polymers have been identified in stormwater, which originate from diffuse sources such as air-deposition, traffic, littering and construction and building materials (Österlund et al., 2023).

Plastic debris is categorised based on the size of the material, with MP usually defined as particles having sizes between 5 mm and 1 µm (Frias and Nash, 2019), but the size range definitions are not standardised (Hartmann et al., 2019). MP can be divided into primary or secondary microplastics depending on the state in which they entered the environment. Primary MP denotes the debris that was manufactured to be in that size prior to release (e.g. pellets used in the industry and content in cosmetical and pharmaceutical products), while secondary MP indicates it originated from fragmentation and deterioration of larger debris (GESAMP, 2016), e.g. wear of plastic products, tyres, road paint, fishing gears, fibres from laundry (Magnusson et al., 2016). There are also different interpretations of what materials should be considered as MP. Synthetic petroleum-based polymers with thermo- plastic or thermo-set properties are generally included in the definition, but Hartmann et al. (2019) suggest a broader definition also encapsulating elastomers (e.g. synthetic rubber), heavily-modified natural polymers (e.g. vulcanised natural rubber) and inorganic/hybrid polymers (e.g. silicone).

Tyre wear particles (TWP), i.e. particles generated from the wear of tyres during use, are also considered as MP when utilising the broader definition. Recent studies have suggested tyre wear as an important emission source of MP. Flow analysis of MP in a model city found that tyre wear particles, together with cigarette butts, were the largest sources of MP in urban water systems (Fältström, 2020). Emissions of TWP are estimated to approximately 6 000 000 tonnes/year globally, with global average per capita emissions of 0.81 kg/year (Kole et al., 2017). In Norway the annual estimation of emissions is 8 000 tonnes of microplastics (1.6 kg per capita), of which tyre and road marking wear accounts for 5 000 tonnes (Sundt et al., 2014). In Sweden, TWP emissions from roads are estimated to 11 000–12 000 tonnes/year (Polukarova et al., 2024), and road and tyre wear is believed to be the largest source of MP (Magnusson et al., 2016). However, all these figures remain uncertain, as the scientific literature is lacking measured emissions factors of these particles from tyres (Mennekes and Nowack, 2022). In sampling studies on stormwater, the most often detected MP is TWP and road wear particles, followed by textile fibres, films, fragments, and paint particles (Österlund et al., 2023).

2.3 Properties of tyre wear particles

Tyre wear particles are generated when the tyre tread, the outer part of the tyre, interacts with the road surface due to friction during driving. The generation of TWP is affected by many factors, such as road surface characteristics, driving conditions, tyre pressure, vehicle load, etc (Alexandrova et al., 2007). Tyres are a complex product, both from a structural and chemical perspective (Mattonai et al., 2022). The materials composing tyres vary not only among different types of tyres (winter tyres, all-season tyres, studded tyres and tyres for heavy vehicles), but also among different brands (Mattonai et al., 2022; Mattsson et al., 2023). Sommer et al. (2018) report that natural and synthetic rubber are the main compounds making up ca 50% of the tyre mass, followed by filler materials (e.g. black carbon, silica and chalk; ca 30% of the mass), oils and resins used as softeners (ca 15% of the mass) and sulphur and zinc oxide as vulcanizing agents (ca 5% of the mass). The chemical complexity of tyres was also stressed by K. Müller et al. (2022), that found more than 200 organic samples in tyres and tyre, of which 145 were deemed to be leachable.

A lot of the concerns regarding the environmental hazards of TWP, stem from the additives in the tyres, which are added to enhance the properties of the material (Rauert et al., 2022). Tyres can contain aromatic oils and tyres were suggested as a source of PAHs, but their use has been banned in the European Union since 2005 (European Commission, 2005). Toxic effects of leachates from TWP on several aquatic species have been reported (Liu et al., 2022; Tian et al., 2021a; Wik and Dave, 2009, 2006a), and tyre additive chemicals have been detected in surface water from urban sites in Australia, including hexa(methoxymethyl) melamine (HMMM; a tyre bonding and cross-linking agent), vulcanisation accelerators 1,3-diphenylguanidine (DPG) and benzothiazoles/benzotraizoles, and 6PPD-quinone (transformation product of tyre rubber antioxidant N-(1,3-Dimethylbutyl)-N′-phenyl-p-phenylenediamine) (Rauert et al., 2022). In addition to numerous organic compounds, TWP has also been suggested as a source of Zn, Cu and Pb, and also of other metals (Zn > Cu > Pb > Sn > Sb > Ni > Cr > As > Cd, in decreasing concentration order; Jeong, 2022). Hence, TWP is a source of both heavy metal pollution and of leachable organic compounds which can be mobile in aqueous environments, which underlines the gravity of tyre wear as a stormwater pollutant and as a risk for aquatic organisms.

2.3.1 Size ranges and density of TWP

The particle sizes of TWP have been reported to range from a few nanometres to several hundred micrometres, but size distributions vary considerably between studies (Wagner et al., 2018). The variability is likely due to differences in formation conditions (e.g. driving behaviour), as well as variations in sampling and analytical methods, and environmental processes that can further affect the particle characteristics (Kole et al., 2017; Wagner et al., 2018). Several studies suggest that TWP are predominantly present in size fractions below approximately 200 µm (Klöckner et al., 2021a; Kreider et al., 2010). Studies that have generated and measured TWP in laboratory settings have found the majority of TWP to be in very fine airborne size fractions: 60–100 nm (Park et al., 2017; Zhong et al., 2024). Road simulator studies employing scanning electron microscopy (SEM) observed larger size fractions, with the size distributions of TWP showing modes at 30–80 µm (Kovochich et al., 2021; Kreider et al., 2010). The type of tyre also appears to influence the size distribution of TWP. Wilkinson et al. (2023) analysed TWP generated in a road simulator using light microscopy (>30 µm) and found that studded winter tyres predominantly produced finer particles (30–80 µm), while most summer tyres generated larger particles (150–2000 µm), with the exception of one summer tyre that also produced finer TWP in the 30–80 µm range.

Environmental sampling studies also highlight the prevalence of finer TWP. In samples of stormwater, sweepsand and road washwater analysed using stereo-microscopy, TWP were most frequently found in the size fraction 20–100 µm rather than ≥100 µm (Järlskog et al., 2021, 2020). Rausch et al. (2022) analysed TWP (1–80 µm) in air samples and found most TWP to be >10 µm, but with a considerable percentage in size fraction <10 µm, especially close to roads with heavy traffic. Similarly, Järlskog et al. (2022a) showed that samples of deposition particles, road dust, and water and sediments from a runoff system, had higher concentrations of tyre and bitumen particles in the size fraction 2–20 µm, compared to TWP in the size range 20–125 µm. This suggests that a large fraction of TWP might be finer than <20 µm, and studies employing analytical techniques or sample pre-treatments not allowing quantification in the finest size fractions may underestimate total TWP concentrations. The size fractions in which environmental TWP can be found are important also for the assessment and design of suitable remediation techniques, as for example stormwater treatment processes relying on sedimentation are expected to be less efficient for TWP <30 µm (Vogelsang et al., 2019).

Different densities have been reported in the literature for TWP, which is likely dependent on the constituents of the particles, such as the mineral encrustations, as well as the formation and wear processes that the particles are subjected to. The rubber polymers present in tyres (polyisoprene, butadiene rubber and styrene butadiene rubber) are reported have densities of 0.9–0.97 g/cm^3 (Andersson-Sköld et al., 2020), whereas clean rubber crumbs from tyres are reported to have densities of 1.05–1.2 g/cm^3 (Degaffe and Turner, 2011; Sofi, 2018; Tang et al., 2006), which is lower than most sediments indicating they would have lower settling velocities (Degaffe and Turner, 2011). Environmental TWP have been found to have densities closer to 1.8–1.9 g/cm^3 (Järlskog et al., 2020; Klöckner et al., 2019), which is due to the encapsuled minerals from the road material. Wilkinson et al. (2023) made a physicochemical distinction of TWP generated in a road simulator into two classes: sub-elastic and firm-elastic. The sub-elastic type, which was the most abundant, showcased more amounts of embedded mineral particles, yielding higher densities than previously reported, between 2.0–2.2 g/cm^3, indicating solutions denser than 2 g/cm^3 should be used when density separation is part of the sample pre-treatment process (Kovochich et al., 2021; Vogelsang et al., 2020; Wilkinson et al., 2023). The firm-elastic type had lower densities, more similar to pristine tyre crumb, likely due to the mineral encrustations only being superficial (1.2 g/cm^3; Wilkinson et al., 2023).

2.3.2 Morphology of TWP

The shape of a particle can significantly influence its settling behaviour in water, as particles with shape factors that deviate from one (i.e. non-spherical particles) experience greater drag and, consequently, lower settling velocities (Egarr et al., 2004). Several studies have tried to assess the morphology and appearance of TWP in different media, using single particle analysis methods such as Scanning Electron Microscopy (SEM). Comparisons of pristine tyre tread particles generated artificially by cryogenic milling, and TWP generated from road simulators and collected from cars on driving circuits, which are more representative of environmentally found TWP, showed that these differ in characteristics. Road simulators generated TWP that were more elongated in shape and contained mineral encrustations (Knight et al., 2020; Kovochich et al., 2021; Kreider et al., 2010). The mineral encrustations arise from brake and road wear, together with other material in the road dust, and the amount of mineral encrustations can vary greatly, ranging from about 10% of the particle volume to over 50% and is likely affected by traffic characteristics such as traffic volume and speed (Sommer et al., 2018).

Kreider et al. (2010) described the TWP generated from road simulators and collected from cars on driving circuits to be very similar in shape and morphology, with aspect ratio showing a mode at 1.56-1.58. Similarly, TWP generated in a road simulator, with addition of chalk and stone dust as third bodies, had 49% of particles with an aspect ratio between 1.5 and 2.5, but also contained particles of more spherical shape, with 34% having an aspect ratio <1.5 (Kovochich et al., 2021). A mix of elongated and spherical particles was also described in airborne dust: 79% <2, 47% <1.5 and 14% <1.2 (Rausch et al., 2022). Morphological differences were also found between the TWP classes sub-elastic and firm-elastic defined by Wilkinson et al. (2023). The sub-elastic type was elongated in shape and brittle with a mean aspect ratio of 2.48. It also showcased more amounts of embedded mineral particles, yielding higher density as mentioned before. The firm-elastic type was firm, more elastic, with superficial mineral encrustations, and the shape was more rounded with a mean aspect ratio of 1.68 (Wilkinson et al., 2023).

2.3.3 Elemental composition of TWP

Pristine tyre rubber mostly presents few elements: C from the rubber polymers and carbon black, SiO_2 from filler materials, S and ZnO from vulcanisation and catalyst additives (Rausch et al., 2022), but these compounds are not always included in all tyre formulations (Sommer et al., 2018). Additionally, the elemental composition of TWP varies greatly depending on which road they were generated on (Sommer et al., 2018), meaning that the elemental fingerprint of TWP is mostly affected by the encrustation from the road (Kreider et al., 2010; Rausch et al., 2022; Sommer et al., 2018; Wilkinson et al., 2023).

The most abundant elements in TWP have been found to be O, C and Si (Järlskog et al., 2022b; Rausch et al., 2022), making up 30, 42 and 12% of the estimate particle weight, respectively (Järlskog et al., 2022b). Silicon can originate from the filler material in the rubber, but is also a common element and can originate from road material encrustations (Kreider et al., 2010). Elements Fe, Cu, Zn and Al, likely originating from brake wear, were also found on TWP (Sommer et al., 2018). Additionally Järlskog et al. (2022b) found that the other most abundant elements (Fe, Al, K, Ca, Na, Mg, S and Ti) were the same as found in mineral particles in the road dust.

Kreider et al. (2010) found Zn and S in TWP from road simulator and in TWP collected from tyres in driving circuits, but the concentrations were lower than in tyre tread, indicating dilution from encrustations. Also, in a field study by Järlskog et al. (2022b), Zn was almost completely absent in TWP from highway road dust. Those findings are in contrast to what has been reported in tyre and road wear particles from a road simulator laboratory (Kovochich et al., 2021) and in relation to what is reported in tyre tread, i.e. 0.78–1.2% of tyre tread weight consisting of Zn (Wagner et al., 2022). SEM-EDX results on TWP in air did also not confirm clear and abundant presence of Zn in TWP (Sommer et al., 2018).

2.4 Analytical methods and their challenges for TWP analysis

Analysis and quantification of TWP is challenging due to the heterogenous and fine particle sizes, the density and the chemical composition (Mattonai et al., 2022), and is further hindered by absence of robust standardized techniques and global data on tyre compositions (Ma et al., 2025). Methods developed for analysis of clean tyres are overall not suitable as environmental TWP concentrations are more diluted and sample matrices are more complex (Wagner et al., 2018). Suitable marker compounds for the quantification of TWP in environmental samples need to meet certain criteria including similar abundance of the markers across different types of tyres, and that the markers should be stable, non-leachable and specific for tyre material (Wagner et al., 2018).

Analytical methods for TWP can be categorised in bulk-based methods or single particle methods (Rødland et al., 2023a). Bulk-based methods provide mass quantification of the TWP content, and can be based on marker quantification using element composition or organic composition, such as Zn, organic additives and decomposition products such as benzothiazoles, DPG, oleamide, and transformation products of the antioxidant 6-PPD (Chae et al., 2021; Klöckner et al., 2021b; Parker-Jurd et al., 2021; Tumwet et al., 2025; Wik et al., 2008), with the latter compounds being more promising as markers, since they are stable, specific for tyres and are found in concentrations above environmental background concentrations (Rødland et al., 2023a).

Thermal decomposition methods for TWP quantification are the most common bulk-based method in the literature for analysis of environmental samples (Rødland et al., 2023a), with several studies employing thermal extraction desorption (TED-GC/MS) or pyrolysis (Pyr-GC/MS) with gas chromatography mass spectrometry (Barber et al., 2025, 2024; Eisentraut et al., 2018; Klöckner et al., 2020; Axel Müller et al., 2022; Rødland et al., 2023b, 2022a, 2022b, 2022c; Rosso et al., 2023; Unice et al., 2012, 2013). These techniques measure the thermal products of rubber polymers in the tyre tread as markers, and various markers are proposed in the literature (Rødland et al., 2023a). The rubber types found in tyre are natural rubber (isoprene) for which pyrolysis markers dipentene or isoprene have been used (Unice et al., 2012), and synthetic rubber (styrene butadiene rubber SBR and butadiene rubber BR) for which markers 4-vinylcyclohexene (4-VCH), 4-phenylcyclohexene (4-PCH), or styrene butadiene hybrid trimer (SBB) have been employed (Goßmann et al., 2021; Ma et al., 2025; Miller et al., 2022; Rødland et al., 2023a; Unice et al., 2012). The ISO technical specifications ISO/TS 21396:2017 and ISO/TS 20593:2017(E) suggest Pyrolysis GC/MS with 4-VCH and dipentene markers for quantification of TWP based on Unice et al. (2013, 2012), however they rely on assumed proportions of total rubber content of 50% in tyres, with 44% SBR and BR in tyres for personal vehicles, and 45% natural rubber in tyres for heavy-duty vehicles (Rauert et al., 2021). Rauert et al. (2021) found large variability in the synthetic and natural rubber content of different tyres treads quantified using 4-VCH, 4-PCH and SBB. Therefore a combinatorial marker approach of four markers has been suggested by Rødland et al. (2022c), which has shown less variation in tyres than 4-VCH. Other studies have employed single marker 4-PCH successfully (Barber et al., 2025; Goßmann et al., 2021; Miller et al., 2022), and further research is needed to assess the suitability of various chemical markers for complex sample matrices, and to achieve more comprehensive data on the chemical composition of tyres (Ma et al., 2025). Also, thorough sample pretreatment may be required for certain environmental sample matrices in order to dry and concentrate the samples before Pyr-GC/MS analysis, and to reduce matrix interference (Eisentraut et al., 2018).

Single particle methods, such as light microscopy, Fourier-transform infrared spectroscopy-based methods (FTIR), Raman spectroscopy, and scanning electron microscopy (SEM), yield information about size and morphology of individual TWP, from which mass estimates can be made (Rødland et al., 2023a). Light microscopy, sometimes combined with tactile assessment, has been previously employed for the analysis of TWP in coarser size fractions (Goehler et al., 2022; Järlskog et al., 2021, 2020; Lange et al., 2021; Vijayan et al., 2022), but is no longer recommended for assessment of microplastics in scientific publications (Science of the Total Environment, 2024). Some studies have employed FTIR-based methods for analysis of TWP and rubber polymers (Lange et al., 2022; Mengistu et al., 2019, 2021; Rosso et al., 2023; Yano et al., 2021; Ziajahromi et al., 2020). Overall, the extensive sample preparation needed, and the presence of mineral encrustations and of carbon black in tyres limit the applicability of Raman spectroscopy and FTIR methods for TWP quantification (Eisentraut et al., 2018; Mattonai et al., 2022; Wagner et al., 2018), and FTIR methodologies also have limitations in analysing finer particles <10 µm (Rødland et al., 2023a). Several studies have employed SEM coupled with Energy Dispersive X-ray (SEM-EDX) to analyse TWP based on size and chemical and morphological features, sometimes together with estimation of the relative mass of TWP in the sample (Gao et al., 2022; Järlskog et al., 2022a, 2022b; Kovochich et al., 2021; Kreider et al., 2010; Rausch et al., 2022; Rosso et al., 2023; Sommer et al., 2018). As bulk-analysis methods do not provide information about TWP sizes and morphology, there is still a need for both bulk-analysis and single particle methods as complementary analyses (Rødland et al., 2023a) for assessment of TWP quantities and characteristics in different environmental compartments.

2.5 Measured TWP in environmental compartments

An overview of sampling studies quantifying TWP mass concentrations in different types of environmental compartments can be seen in Table 1. Studies employing light microscopy or other stereoscopic techniques, such as FTIR, without presenting a mass-estimate are not included, as the TWP amounts presented as number of particles concentrations is not easily comparable to mass concentrations, but these studies can still provide relevant insights into occurrence, spread, particle behaviour and especially particle characteristics (Lange et al., 2022, 2021; Yano et al., 2021; Ziajahromi et al., 2023, 2022; Vijayan et al., 2022).

Table 1: Summary of studies addressing TWP mass- quantification in different types of environmental compartments.

Study	Sample matrix	Size fraction	Analysis method	TWP concentration
Road dust				
(Goßmann et al., 2021)	Road dust over drain covers	<1 mm	Pyr-GC/MS	n.d.–11.3 mg/g
(Järlskog et al., 2022a)	Highway road dust (Testsite E18)	20–125 µm	SEM-EDX	10–43 000 mg/m^2
(Rødland et al., 2022a)	Road dust from tunnel	<1.6 µm	Pyr-GC/MS	25.3–4820 mg/m^2
(Wang et al., 2025)	Urban and highway road dust	-	Pyr-GC/MS	0.25–25 mg/g
Stormwater samples				
(Dröge and Tromp, 2019)	Runoff from highway roadside	No reported size range	TED-GC/MS	51–59 mg/L
(Dröge and Tromp, 2019)	Standing runoff in well (Testsite E18)	No reported size range	TED-GC/MS	1.0 mg/L
(Järlskog et al., 2022a)	Standing runoff in wells (Testsite E18)	20–125 µm	SEM-EDX	10–130 mg/L
(Järlskog et al., 2022a)	Standing runoff in wells (Testsite E18)	2–20 µm	SEM-EDX	10–150 mg/L (TBWP[a])
(Lindfors et al., 2025)	Runoff from road and parking lot	10–500 µm	Pyr-GC/MS	0.07–2.26 mg/L
(Lindfors et al., 2025)	Runoff from roof	10–500 µm	Pyr-GC/MS	0.03–2.26 mg/m^3
(Rauert et al., 2022)	Stormwater drain water	>1 µm	Pyr-GC/MS	0.23–0.67 mg/L[b]
(Rauert et al., 2022)	Retention pond water	>1 µm	Pyr-GC/MS	0.072–0.24 mg/L[b]
(Rasmussen et al., 2024)	Stormwater ponds	>10 µm	Pyr-GC/MS	n.d. –0022 mg/L
(Rødland et al., 2022a)	Untreated tunnel wash water	>1.6 µm	Pyr-GC/MS	14.5–47.8 mg/L
(Parker-Jurd et al., 2021)	Effluent stormwater drain		Pyr-GC/MS	~0.4–8.2 mg/L
Sediment samples				
(Dröge and Tromp, 2019)	Stormwater wells (Testsite E18)	No reported size range	TED-GC/MS	13 mg/g
(Järlskog et al., 2022a)	Stormwater wells (Testsite E18)	20–125 µm	SEM-EDX	3.5–14.5 mg/g
(Järlskog et al., 2022a)	Stormwater wells (Testsite E18)	2–20 µm	SEM-EDX (TBWP[a])	21.4–71 mg/g
(Klöckner et al., 2020)	Sedimentation ponds	<500 µm	TED-GC/MS	0.95–11 mg/g[b]
(Mengistu et al., 2021)	Urban gully pot	<5 mm	STA-FTIR + PARAFAC	0.8–150 mg/g sediment d.w.
(Rasmussen et al., 2024)	Stormwater ponds	10–450 µm	Pyr-GC/MS	0.018–0.860 mg/g
(Rødland et al., 2022a)	Gully pots in road tunnel	<1 mm	Pyr-GC/MS	4.8–53 mg/g
(Öborn et al., 2024a)	Stormwater ponds	20–450 µm	Pyr-GC/MS	n.d. –0.0693 mg/g

Table continues on the next page

Soil samples				
(Cadle and Williams, 1978)	Freeway roadside soil (0-30 m), 1 cm top layer only	<0.59mm	Extraction-IR technique	0–4.5 mg/g
(Axel Müller et al., 2022)	German highway roadside soil 0.3-5m, depth 0-20 cm	<50 µm, 50–100 µm, 100–500 µm, 500–1000 µm	TED-GC/MS	0.16–16 mg/g (SBR based) 0.41– 45 mg/g (zinc based)
(Rødland et al., 2023b)	Roadside soil from rural roads, Norway	<500 µm	Pyr-GC/MS	2.0–26 mg/g
(Beaurepaire et al., 2025)	Soil from biofiltration swale by highway	<500 µm	Pyr-GC/MS	0–15 mg/g TWP[c]
Snow samples				
(Rødland et al., 2022b)	Roadside snow	<1 mm	Pyr-GC/MS	76–14500 mg/L
(Chand et al., 2024)	Urban snow dumping sites	>10 µm	Pyr-GC/MS	0.044–3.0 mg/L
Surface waters				
(Dröge and Tromp, 2019)	Surface waters	No reported size range	TED-GC/MS	0.001–0.011 mg/L
(Rauert et al., 2022)	Water from creeks and rivers in urban centres	>1 µm	Pyr-GC/MS	n.d.–0.48 mg/L[b]
(Barber et al., 2024)	River sediments	<5 mm[b]	Pyr-GC/MS	0.045–1.2 mg/g[b]
(Klöckner et al., 2020)	Lake sediments	<500 µm	TED-GC/MS	0.085 mg/g[b]
(Unice et al., 2013)	River sediments	<1 mm	Pyr-GC/MS	0.013–3.7 mg/g d.w.[b]

[a] TBWP indicates tyre and bitumen wear particles, as SEM-EDX technique employed does not distinguish between TWP and bitumen in very fine size fractions (see chapter 3.3.2)
[b] The presented reported values from the studies have been adjusted by dividing by two to convert from tyre and road wear particle concentration to tyre tread concentration.
[c] SBR+BR concentration converted to tyre tread based on conversion factor of 3.87 that they propose

The overview of sampling studies found in the scientific literature (Table 1) shows that reported concentrations vary greatly, even between studies assessing similar sample matrices. This could be due to differences in the local conditions, as e.g. direct runoff collected from a road surface (Dröge and Tromp, 2019) would be expected to show greater TWP concentration than stormwater downstream of a more heterogeneous catchment (Rauert et al., 2022). The wide range of different particle sizes analysed, along with differences in pre-treatment methods such as digestion and ultrasonication, are also likely to affect the outcome of TWP quantification. Overall, the literature is still lacking extensive information on environmental concentrations in several environmental compartments, with snow and soil concentrations in particular assessed by only a few studies.

2.6 Transport processes of TWP

There is very little knowledge of how much TWP is transported to different environmental compartments (Andersson-Sköld et al., 2020). Even if more studies have attempted to quantify TWP in environmental studies in the last years (Table 1), the knowledge gap is still large, starting from the actual emissions of TWP on roads (Mennekes and Nowack, 2022). The spread and occurrence of TWP in the environment are dependent on many factors, e.g. particle size, density and shape, weather conditions, such as rain events and intensity, snow melt and pathways into the environment (Andersson-Sköld et al., 2020). Based on a review by Andersson-Sköld et al. (2020) of different pathways that TWP can take, relevant paths and processes that apply to TWP on Testsite E18 are outlined in this chapter.

Tyre wear is generated on the road when driving, and the particles can be emitted directly to air or attach to the road and vehicles. Air emission estimates vary between 0.1% and 10%, and up to 30% in some studies (Kole et al., 2017). Particles smaller than PM_{10} are estimated to be about 10% of emitted microplastics from cars (Boulter et al., 2006) and this is the size range usually considered as airborne (Evangeliou et al., 2020). Suspended TWP in air can deposit due to wet and dry deposition and have been found to do so with decreasing concentration per increasing distance from the road between 3.1

and 27.1 m (Järlskog et al., 2022a). A large proportion of the emitted TWP have been found to be larger than 20 µm, and are expected to settle on/close to the road (Andersson-Sköld et al., 2020; Grigoratos and Martini, 2014). Tyre wear particles accumulate on the road surface, where they can be subjected to a number of deteriorating processes due to traffic and radiation. As was found by Järlskog et al. (2022a), the particles tend to accumulate close to the kerb, rather than in between wheel tracks.

Rain events can mobilise TWP from the road if the rain intensity is sufficiently high. If the infrastructure is minimal, the MP and TWP in road runoff is transferred directly to adjacent soil or to a close waterway (Wagner et al., 2018). The presence of many impervious surfaces, such as in urban stormwater systems, would require less rainwater for achieving major transport with surface runoff, whereas vegetation, gravel and soil in rural areas can be expected to retain large amounts of particles (Vogelsang et al., 2019).

If more complex infrastructure is present, the contaminated stormwater can be released through a drainage system to water recipients (Wagner et al., 2018). Processes that would affect transport of tyre wear in the stormwater system include sedimentation (Vogelsang et al., 2019) and resuspension, which both are heavily dependent on particle characteristics, aside from rain load and flow properties. Additionally, aggregation with other particles (Wijesiri et al., 2016), fragmentation due to abrasion, UV rays, oxidation and biofouling can cause changes in buoyancy (SAPEA, 2019), which would modify the settling conditions. This indicates a need for detailed modelling of particle interaction with fluids and other particles in order to predict transport, spread and occurrence.

3 METHODOLOGY

In this chapter the overall methodology applied during the work of the thesis is presented and summarised. Details about the sample preparation and analytical methods can be found in the research papers and in their respective supplementary information (SI).

3.1 Study area: test site by highway E18

Testsite E18 (59°38'01.9"N 16°51'18.7"E) is a road research facility owned by the Swedish Road Administration located by highway E18, between the cities of Västerås and Enköping, Figure 2. The land area surrounding the location is open and flat, mostly exploited for agricultural purposes. The highway was built in 2010, and has two lanes in each direction with a speed limit of 120 km/h. This part of highway E18 has a traffic load of about 10 000 AADT (average annual daily traffic) per highway direction, of which 1 400 are heavy vehicles (Swedish Transport Administration, 2024). The current pavement type is stone mastic asphalt (SMA 16, B 70/100), which was the same type used in the original cover. The current pavement also includes traces of polymer-modified bitumen (PMB 45/80-55), used for filling the right lane wheel tracks in 2016, that was recycled into the current cover using warm-remixing in 2021. It is estimated that the current pavement contains a maximum of 0.7% of PMB after the warm-remixing, compared to the previous 6.6% (Polukarova et al., 2025).

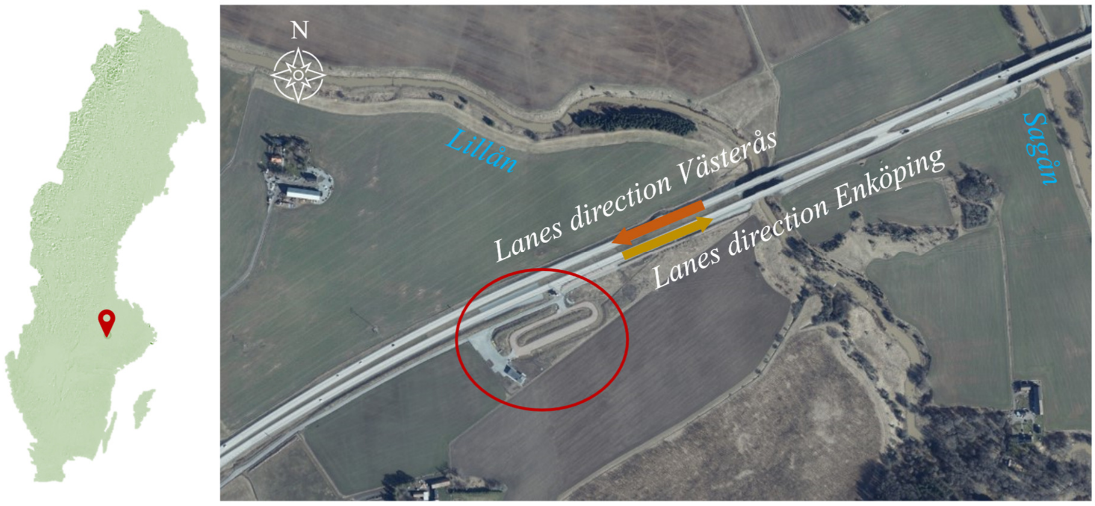

Figure 2:Left: Map of Sweden with the location of Testsite E18. Right: Map showing Testsite E18 marked with a red circle, and directions of the highway towards Västerås and Enköping indicated by arrows. The nearby streams of Sagån and Lillån are marked out. Lillån is a natural stream that debouches into the larger stream Sagån, which in turn flows into lake Mälaren. Maps modified from Swedish Land Survey (2024).

The climate in the area (Västerås weather station) is humid continental climate Dfb (Beck et al., 2023), with average yearly precipitation 606 mm during the normal period, with 154 days with precipitation ≥0.1 mm (SMHI, 2024a). Monthly weather details can be seen in Figure 3. In winter, the region is on average covered by snow for 75 days/year (SMHI, 2024b), with average largest yearly snow depth of 30 cm in normal period 1961–1990 (SMHI, 2024c).

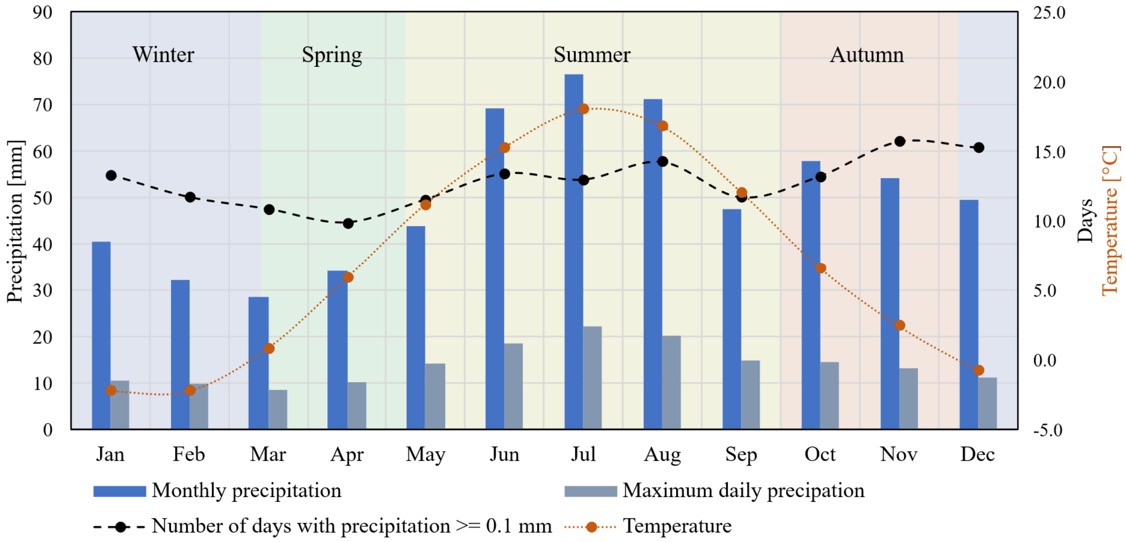

Figure 3: Weather data for normal year period 1991–2020 for the weather station in Västerås (SMHI, 2024a). The bar charts show average monthly precipitation and maximum daily precipitation is shown over the year (left y-axis). The line plots show average number of days with minimum precipitation 0.1 mm and average temperature over the year (right y-axis). The coloured fields in the background show average start and duration of the meteorological seasons in the normal period according to (SMHI, 2024a).

3.1.1 The stormwater system at Testsite E18

At Testsite E18, in the direction towards Enköping, the road surface is enclosed by a 100 m long kerbstone, and runoff is dewatered by two gully pots, locations GP1 and GP2 in Figure 4. The gully pots are serially connected, with gully pot (GP1) draining into gully pot (GP2). Gully pot GP2 is in turn connected to a well (location WA) equipped with a Thomson weir. The well WA drains into a final collecting well WF. The collecting well WF also receives stormwater from two additional wells WB and WC, which receive water infiltrated from the geo-textile lined ditches at the sides of the highway lanes in the direction of Enköping. The collecting well WF finally discharges runoff through a pipe (location D) into a ditch (Ditch 2), which runs along the highway. The ditch runs along the highway up until the bridge over the stream Lillån (location L, Figure 5). The bridge is dewatered by drains that discharge over the banks of the stream (location R, Figure 5). Lillån is a natural stream that debouches into the larger stream Sagån, which in turn flows into lake Mälaren. Apart from Testsite E18, along most of highway E18, runoff from the road is generally drained freely into the ditches at the side of the road, which is also the case at location Ditch 1 (Figure 5).

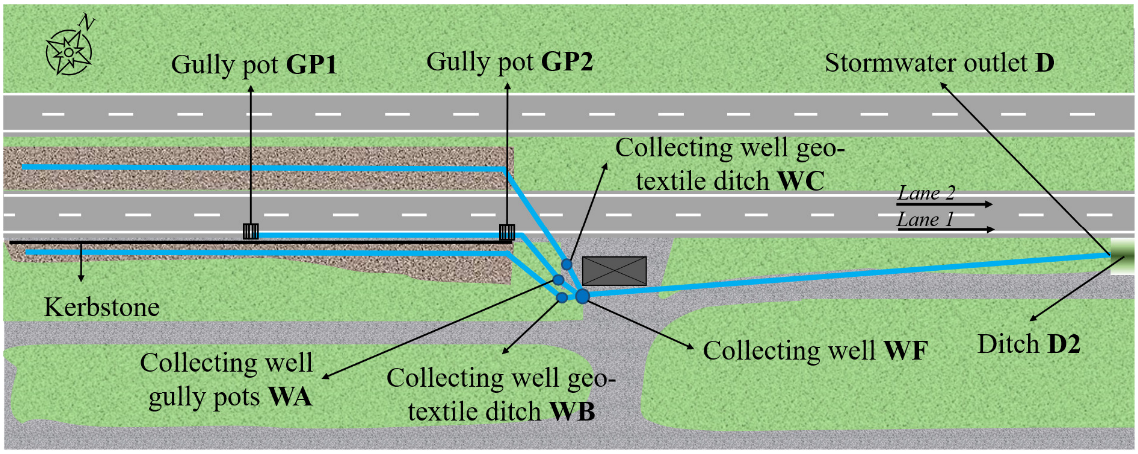

Figure 4: Illustration of the stormwater system by Testsite E18.

3.2 Sampling campaigns

In total five sampling campaigns were conducted during the thesis, targeting different types of environmental compartments. On three campaigns, each carried out over one day, snow (Paper I), soil (Paper II) and grab samples of water and sediments from the stormwater system (Paper III) were collected. The locations sampled during the sampling campaigns are illustrated in Figure 5 and the time periods in Figure 6. Two prolonged sampling campaigns of automatic sampling of runoff during rain from the gully pots and wells of the stormwater system (Paper IV and V) were carried out in autumn/winter 2022 and in spring 2023, Figure 6. Only the spring sampling campaign was successful in sampling runoff during rain due to dry weather, technical issues with the equipment and eventually cold weather with snow during the autumn/winter campaign. The first rains of the spring campaign had therefore to be used for calibrating the trigger settings for the rest of spring campaign, instead of doing this in the autumn campaign.

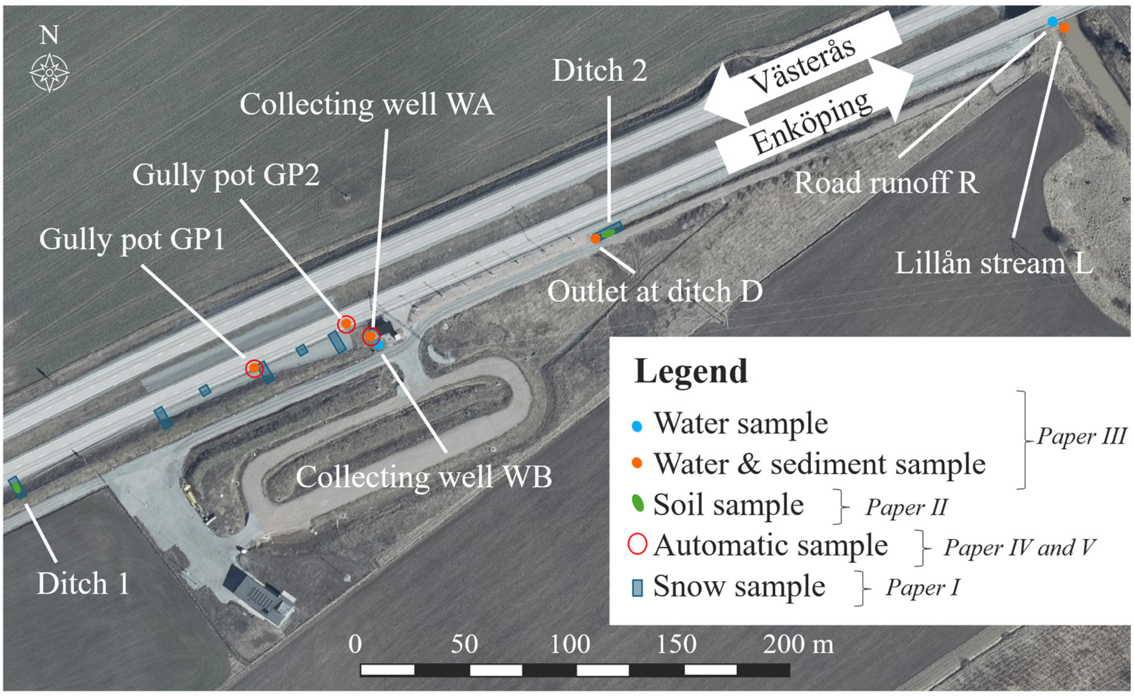

Figure 5: Samples on Testsite E18 collected between November 2022 and June 2023. Map adapted from the Swedish Land Survey (2024).

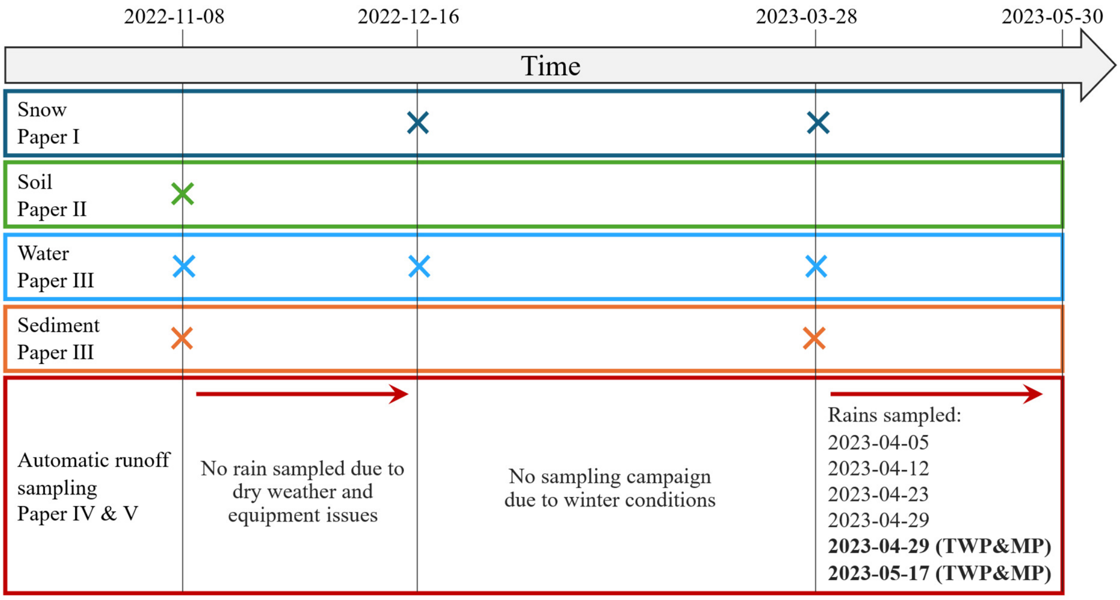

Figure 6: Time periods of sampling campaigns carried out on Testsite E18. For automatic runoff samples, dates when rains were collected are listed, with (TWP&MP) indicating if the samples collected were analysed for TWP and MP.

A general overview of the procedures followed while collecting samples from each sample matrix assessed in the theses is given below. More details of sampling can be found in the respective paper.

Snow sampling: Sampling of snow sampling was carried out by Ditch 1 and at several locations by the kerbstone at different distances from the road, and by Ditch 2 at different distances from the stormwater outlet. The snow was sampled in measured-out sections at defined distances from the road. The sampled areas were adjusted to obtain desired snow volumes from the different sections. Sampling was carried out by removing as much snow as possible from the measured-out area using a metal trowel. The snow was collected in LDPE freezing bags.

Soil sampling: The samples of soil from Ditch 1 and Ditch 2 were taken at different distances from the road and at different depths, using a powered auger drill, with removal of the top 10 cm with grass and roots.

Water samples: Grab water samples were collected using glass bottles submerged into the standing water volumes of the wells, ditch and stream (GP1, GP2, WA, WB, D, and L, Figure 5). The direct runoff sample from the road collected when a light rain started (R) was collected in a bucket and the outflowing water from wells WA and WB with glass bottles (OA and OB).

Sediment samples: The sediments from the gully pots and well A were collected using an Ekman sampler. Sediments from the stormwater outlet by Ditch 2 (D) were instead collected with a metal trowel, and from Lillån (L) with a core sampler.

Automatic stormwater sampling: The automatic samplers during rain were collected using ISCO automatic samplers (model 6712 and 6712c), with inlet hoses installed into the standing water volumes of the gully pots (GP1, GP2 and WA). The samplers were triggered using a level meter in the gully pots, due to their narrow shape, and using a flow meter installed in the outlet pipe in WA. The ISCO samplers were equipped with 24 bottles and collected 1-liter per trigger signal in 1-liter bottles in GP2, and 1-liter per trigger signal in 0.5-liter bottles in GP1 and WA.

3.3 Analytical methods

The samples collected were subjected to pre-treatment in the WET-laboratory at Chalmers. The pre-treatment steps varied depending on the type of sample matrix and the analysis type to be performed. An overview of the pre-treatment steps and analysis methodologies applied is given in this chapter, and details can be found in the respective papers and SI. An overview of the analytical approaches applied in the different papers is summarised in Table 2.

Table 2: Overview of analysis procedures employed on the environmental samples during the thesis.

Analysis	Matrix	Paper	Size fraction	Method
TWP quantification	Water, Sediments, Snow, Soil	I, II, III and IV	<500 µm, 1.6–500 µm, 1.6–20 µm	Pyr-GC/MS for SBR+BR quantification + Monte Carlo simulation
Chemical and morpho-textural particle characterisation	Snow	I	2–20 µm, 20–125 µm	SEM-EDX coupled to machine-learning algorithm by commercial laboratory
MP quantification (PE, PP, PS, ABS, PMMA, PC, PVC, PET, PA6, PA66, PB, PI and SBR)	Water	V	10–1000 µm	Pyr-GC/MS by commercial laboratory
TSS	Water	III, IV	<1.6 µm	SS-EN 872:2005
VSS	Water	III, IV	<1.6 µm	SS 02 81 13
Solids concentrations	Snow	I	-	Drying of measured volume of melted snow in a jar and weighting of dry material
DS	Sediments	III	-	SS 02 81 13
LOI	Sediments, Soil	II, III	-	SS 02 81 13
Total metals	Water, Snow, Soil	I, II, IV	-	ICP-MS, SS-EN ISO 15587-2:2002 and SS28311:2017mod/SS-EN and ISO 11885:2009 by commercial laboratory
Dissolved metals	Snow	I	<0.45 µm	ICP-MS, SS-EN ISO 17294-2:2016 by commercial laboratory
TOC/DOC and TC/DC	Snow	I	None/ <0.45 µm	Shimadzu TOC-V CPH total organic carbon analyzer
Ions	Snow	I	-	Ion chromatography Thermo-Fisher Dionex ICS-900
Nano-particles quantification	Snow	I	<0.45 µm	Malvern NanoSight NS300
Grain size distribution	Soil	II	>0.63 µm (sieving) and 0.002–0.063 µm (sedimentation analysis)	SS027123 mod and SS027124 by commercial laboratory
pH	Snow	I	-	pH-meter VWR pH110
Conductivity	Snow	I	-	Conductivity meter VWR HCO 304

3.3.1 TWP quantification: Pyr-GC/MS

The mass quantification of TWP in the different environmental samples was the main purpose of Paper I, II, III and IV. Pre-treatment steps were necessary to prepare the samples for Pyr-GC/MS analysis in order to obtain dry samples in volumes suitable for the pyrolysis equipment. The different quality of the sampled matrices, heterogeneity of the samples and expected levels of TWP in the samples, all posed different challenges, and the pre-treatment steps had to be adapted accordingly.

3.3.1.1 Pre-treatment approaches for the different sample matrices analysed

The overall sample pre-treatment is illustrated in Figure 7, and details of the pre-treatment steps can be found in the respective paper. Water samples, in all size fractions, as well as snow and sediments in size fraction 1.6–20 µm, had the shortest pre-treatment procedure, consisting of retrieval of a measured volume of sample, wet sieving for removal of coarse material and vacuum filtration. Depending on the quality of the samples, two adjustments in the pre-treatment steps were made for selected samples. Total Suspended Solids (TSS) analyses were performed before the TWP-preparation to have an estimate of the particle content in each sample. For samples with particle content above 70 mg/L, a centrifugation step in Falcon tubes for easier filtration, with the supernatant filtered first and the sedimented solid material resuspended on the filter, was added as proposed by Rødland et al. (2022a). The second adjustment was that for samples with high SBR+BR concentration, the volume of filtered samples had to be adjusted to low volumes to avoid going over the detection limit during pyrolysis, and in order to obtain more exact and homogenised subsamples, collection was performed by pipetting from a shaken volume. This precaution was applied to samples with an expected high TWP content: snow and runoff samples collected during rain.

For the snow and sediments in size fraction <500 µm, the very high amount of particulate material in the samples did not allow easy filtration, so the samples were wet sieved directly into glass jars and dried in oven at 90°C. For the sediment samples in both size fractions, dry substance (DS) analysis was performed in order to be able to re-calculate the SBR+BR concentrations over dry weight (dw) <500 µm and dw 1.6–20 µm, as SBR+BR concentrations over total dry weight of the unsieved samples (referred to as SBR+BR [mg/ g dw] henceforth). This allows for a better comparison of TWP content between sediments e.g. from the gully pots, that contained large amounts of coarser material >500 µm, and finer sediments from well WA, stormwater outlet pipe D and Lillån L. This step however ads more uncertainty due to the variation of DS between replicates (see Paper III SI). Another possibility was to first measure DS on the collected subsample for Pyr-GC/MS analysis, but this was not preferred in order to avoid aggregation of material during drying before the wet-sieving which could have affected the size fractioning.

The soil samples were the most challenging in terms of sample pre-treatment due to heterogeneity of the material, the necessity to use large volumes of distilled water for removal and washing of the coarse material, and the hardening of the material after drying due to the presence of clay. Additionally, the soil samples showed high concentrations of benzene from competing sources upon Pyr-GC/MS to a larger extent than other matrices. For samples with high benzene levels, the SBR+BR quantification was performed excluding the marker benzene (marker combination *b*, see chapter 3.3.1.2). Ulterior digestion pre-treatment has been recommended for complex samples with lower TWP presence (Rødland et al., 2023b).

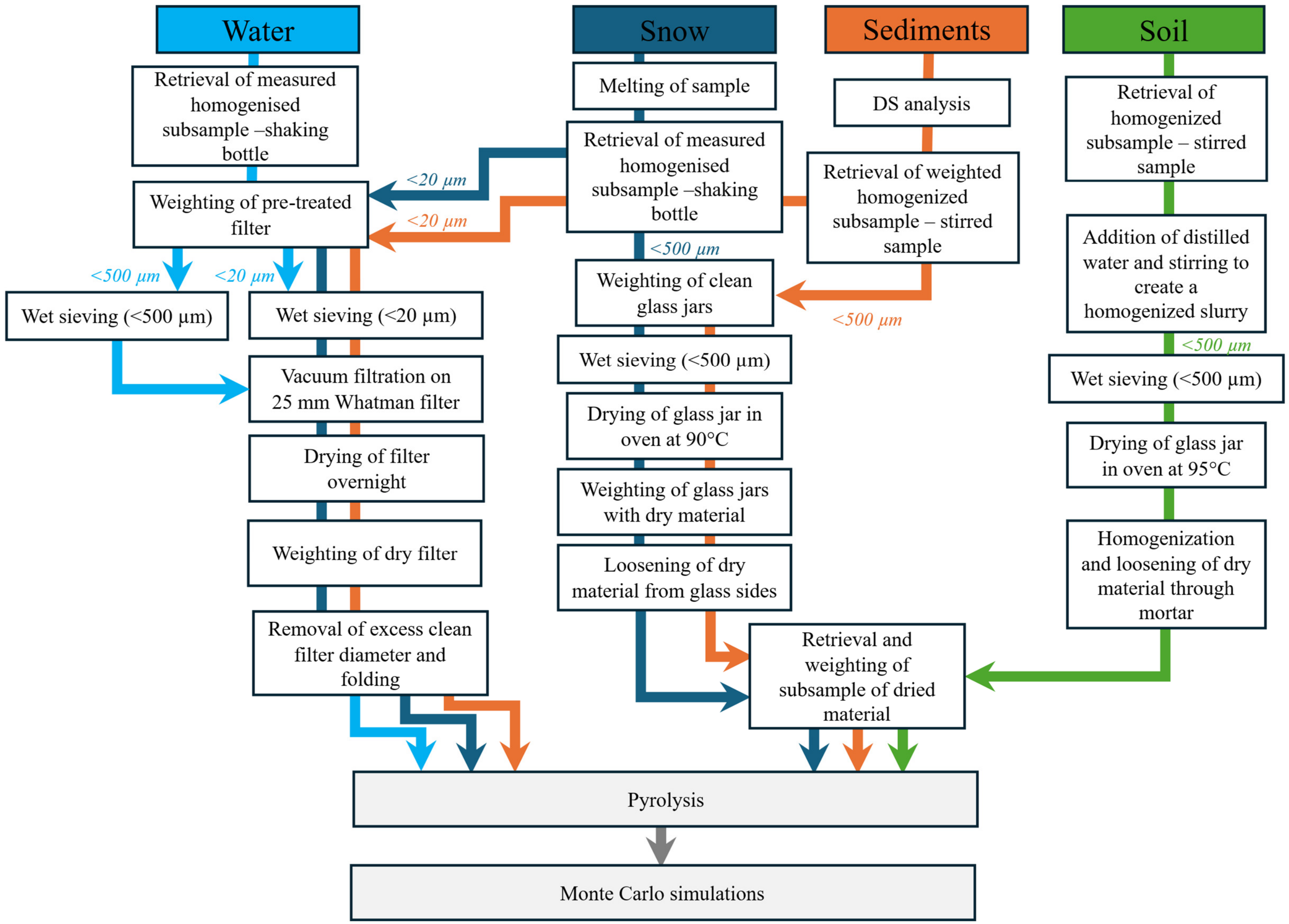

Figure 7: Flowchart of the pre-treatment and analysis steps for the Pyr-GC/MS analysis of TWP in the methodology employed in Paper I, II, III and IV. Steps coloured in grey were not performed by the author of the thesis.

3.3.1.2 Pyr-GC/MS analysis and TWP quantification

The SBR+BR analysis using Pyr-GC/MS was performed by Elisabeth Rødland following the procedure outlined by Rødland et al. (2022c), using marker combinations of m/z 78 Da for benzene, m/z 118 Da for α-methylstyrene, m/z 117 Da for ethylstyrene, and m/z 91 Da for butadiene trimer (marker combination *a*). The Pyr-GC/MS settings were the same for all studies and can be found in the respective paper. Calibration was performed using SBR1500 (Polymer Source Inc., Canada) in solution (chloroform, Sigma Aldrich) with calibration points 1, 5, 20, 60, 120 and 140 µg/cup, as well as internal standard (deuterated polybutadiene, d6- PB, in chloroform solution). The ratio between the four markers was monitored in each sample and compared to the ratios in the SBR1500 calibration samples and to ratios found in reference tyres (Rødland et al., 2022c). In the procedure outlined by Rødland et al. (2022c), the calibration and TWP calculation procedures can be adjusted to also account for the styrene-butadiene styrene (SBS) content of PMB asphalt, which would be identified as SBR+BR in the Pyr-GC/MS analysis. However, due to the low estimated content of PMB asphalt in the pavement of Testsite E18 (chapter 3.1), quantification was performed without adjusting the methodology for SBS content.

The combination of four markers (marker combination *a*) for SBR+BR quantification was preferred as it has been shown to give lower variation in SBR + BR concentrations compared to other markers (4-VCH, 4-PCH, and SBB trimer; Rødland et al., 2022b). 4-VCH was not used for quantification due to higher variability in commercial tyres (Rauert et al., 2021), and natural rubber markers were not used due to the interference of marker compounds from eventual plant material in the sample (Eisentraut et al., 2018). Additionally, findings by Rauert et al. (2021) show that heavy-duty vehicle tyres, also contain synthetic rubber SBR+BR; and not only natural rubbers. In samples with high organic matter content and complex matrix, there is a possibility of marker interference from additional sources of benzene (Rødland et al., 2023b). By using the expected ratios as a control, in samples where the benzene contribution is higher than expected, benzene is removed as a marker for that specific sample and quantification is made using marker combination *b*. The 4-VCH marker for butadiene rubber (SBR + BR) was monitored as a control.

The variability of SBR+BR content in different tyres is large (<0.05%- 28%), and for quantification of the TWP in the sample, relying on standard ratios as in the ISO specifications (ISO/TS 21396:2017(E) and ISO/TS 20593:2017(E)), can underestimate the TWP concentration (Rauert et al., 2021). Extensive datasets of tyre contents and formulations are needed for more accurate TWP quantification (Rauert et al., 2021), and they also need to be adapted to local conditions, to reflect the tyre types in use in specific geographical locations. Therefore, a database of SBR+BR content in Norwegian summer and winter tyres (studded and non-studded) of which 18 tyres from personal vehicles (PV) and 13 tyres from heavy-duty vehicles (HV), developed by (Rødland et al., 2022c), was utilized for TWP quantification, Figure 8. It can be seen that SBR+BR content varies between different types of tyres, with especially HV tyres showing large variation. The TWP quantification methodology employed based on the tyre database consisted of calculating the mean TWP concentration based on Equation 6 (Rødland et al., 2022c), combined with Monte Carlo simulations (100 000 simulations) of the distributions of SBR+BR content in PV and HV-tyres.

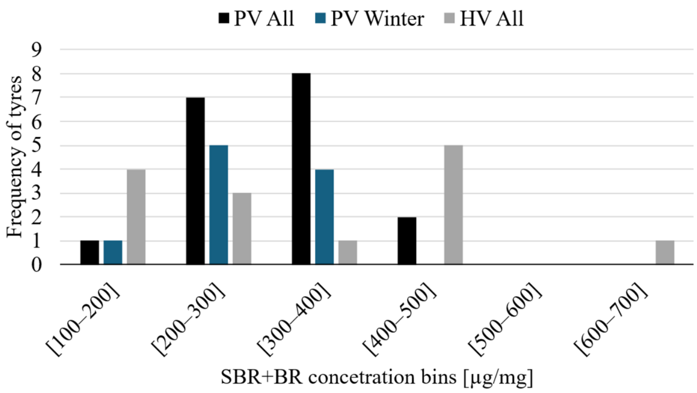

Figure 8: Histogram of measured SBR+BR concentrations in different types of tyres in use in Norway (Rødland et al., 2022c).

3.3.2 TWP characterisation: SEM-EDX

In Paper I, the particulate material, including TWP, were also characterised through Scanning Electron Microscopy coupled with Energy Dispersive X-ray coupled to a machine-learning algorithm (SEM-EDX+ML), based on the methodology by Järlskog et al. (2022b) and Rausch et al. (2022). The analysis was applied to four melted snow samples. Based on the elemental compositions and the morphology and texture, the method classifies the particulate material into the categories minerals, TWP, bitumen, organic particles, metallic particles, and road marking paint (Ti-rich). In the analysis workflow, Figure 9, the samples are sieved into two size fractions (20–125 µm and 2–20 µm) and filtered onto membrane filters. The material from the filters is dispersed on boron substrates and analysed. The size fraction >125 µm was excluded due to issues with obtaining good results with SEM-magnification for the low number of particles in that size range (Järlskog et al., 2022b). Due to the similarities between elemental composition and characteristics of tyre and bitumen wear particles, these two categories are not easily distinguishable for sizes <20 µm and are therefore classified into a common category (TBWP). Based on densities for the material categories and particle sizes, the relative mass of each classified material can be estimated. As the estimated mass of TWP from SEM-EDX+ML is based on the appearance of the particle, it includes both the mass of tyre material and mineral encrustations, whereas the Pyr-GC/MS results in chapter 3.3.1, report the mass of tyre material only.

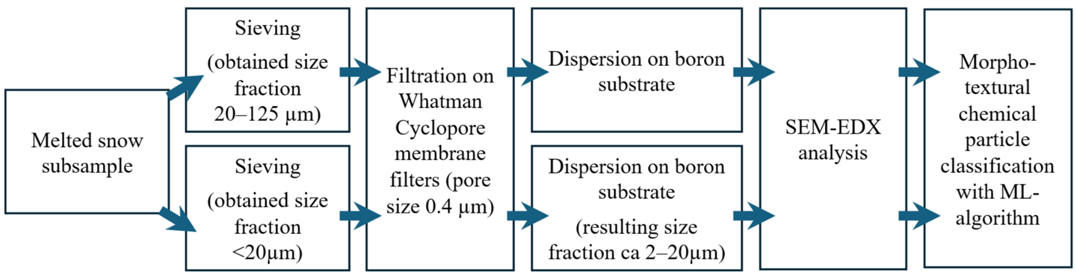

Figure 9: Flowchart of pre-treatment and analysis workflow for SEM-EDX+ML.

3.3.3 Quantification of microplastic polymers in stormwater samples

In Paper V, analyses of thirteen microplastic polymers in stormwater samples collected during the rains of 2023-04-29 and 2023-05-17 were performed using a Pyrolysis-GC/MS methodology developed by a commercial lab. Analysis of microplastics was performed on 19 selected aggregated samples of ca 1 L. The aggregated samples were made by combining equal homogenised volumes from different automatically collected bottles during rain. The plastic polymers analysed were: acrylonitrile butadiene styrene (ABS), polyamide 6 (PA6) and polyamide 6-6 (PA66), polycarbonate (PC), polyethylene (PE), polyethylene terephthalate (PET), polymethylmethacrylate (PMMA), polypropylene (PP), polystyrene (PS), polyvinylchloride (PVC), together with rubber polymers: polybutadiene (PB), polyisoprene (PI) and SBR.

The extensive pre-treatment of the samples involved filtration through stainless-steel filter (10 µm), ultrasonication, oxidation with 10% potassium hydroxide (KOH), decantation, and filtration on 1.6 µm GF/A filters. Samples with large quantities of particles were subjected to additional filtration with stainless-steel filter (10 µm), ultrasonication, and density separation ($CaCl_2$; density 1.4 g/mL). Information of which samples these additional procedures were applied was not available from the commercial lab, but it is likely that they were not applied to the samples from Paper V, due to the relatively low particle content. More information about the procedures can be found in Paper V SI, including marker compounds for the polymer analysed, and in Johansson et al. (2024). Due to the sieving processes and removal of particles >1000 µm, the analysis results should represent microplastics in size fraction 10–1000 µm, but it is likely that microplastic particles larger than 10 µm are lost as well due to the ultrasonication and oxidation steps possibly breaking-up microplastics into finer particles, that are lost in subsequent decantation and filtration steps. This was seen during the laboratory work for Paper III, during which the effect of ultrasonication pre-treatment step applied to the water samples was further investigated. The ultrasonication was applied to homogenise the samples and it was found that it increased the TWP content in the size fraction 1.6–20µm relative to sample that were only shaken, likely due to break-up of TWP particles and their mineral encrustations.

4 RESULTS AND DISCUSSION

In this chapter, the main results of the research papers included in the thesis are presented and discussed. First, the quantified TWP concentrations are examined, both across the various environmental compartments analysed and in relation to findings from other studies. This is followed by the characterisation of TWP in terms of proportion in size fraction 1.6–20 µm and SEM-EDX+ML results. Subsequently, the TWP transport behaviour in a stormwater system during rain is assessed, along with correlations between TWP and other measured parameters, and the quantified MP polymers in the stormwater system during rain. Finally, the broader implications of the findings are discussed, along with potential mitigation strategies.

4.1 TWP concentrations across sampled environmental compartments

The quantified concentrations of TWP in the various environmental compartments analysed in Paper I, II, III and IV can be seen in Figure 10.

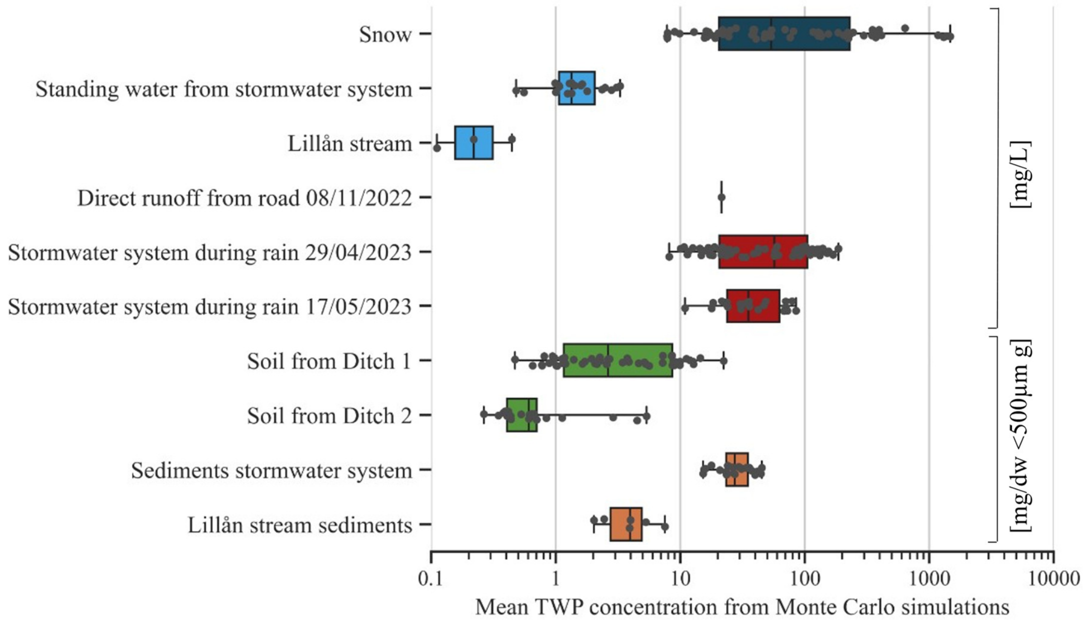

Figure 10: Boxplots of TWP concentrations (<500 µm) showing median, 25th and 75th quartiles and whiskers indicating minimum and maximum values, along with scatter plots of individual samples. The TWP values presented are grouped by sample type and coloured by study: Paper I (dark blue boxplots), Paper II (green boxplots), Paper III (light blue and orange boxplots) and Paper IV (red boxplots). Units are indicated on the right-hand side.

For the environmental compartments belonging to the water-based samples (marked with unit mg/L in Figure 10) the highest median TWP concentration was observed in stormwater samples collected during the rain event on 2023-04-29 (median ± standard deviation = 57±51 mg/L, Paper IV), followed closely by snow samples (54±340 mg/L, Paper I). The stormwater samples collected during the rain on 2023-05-17 also showed high median TWP concentration (35±22 mg/L, Paper IV). The samples from the rain events on 2023-04-29 and 2023-05-17, and especially the snow samples, presented high variability in TWP concentrations. This is due to the variation of TWP amounts in snow depending on road distance (chapter 4.1.1) and to the dynamic behaviour of TWP in the gully pots and wells during rain events (chapter 4.3). The high TWP concentrations in snow and stormwater during rain were in the same magnitude as runoff from a highway and untreated tunnel wash water (Table 1). This underscores that snow and runoff can be potentially large sources of TWP into the environment, as this section of highway E18 is of relatively moderate traffic loads. Reported TWP concentrations in direct runoff from a road and a parking lot (Table 1) were lower than in snow and runoff from Paper I and IV, likely due to lower traffic intensity at those sites. The standing water throughout the stormwater system—collected as grab samples in Paper III—had lower TWP concentration (1.3±0.81 mg/L, Paper III), with median values 16–42 times lower than those observed during rain events. Concentrations in the standing water were similar to previously reported values from stormwater drains and stormwater ponds (Table 1). The lowest concentrations of TWP were found in grab samples from the Lillån stream (0.11–0.44 mg/L, Paper III). Notably, these concentrations are higher than reported values in other surface waters, Table 1. The SBR+BR concentrations found using the 4-VCH marker also confirm the presence of TWP in the Lillån samples, and the measured concentrations are likely not due to cross-contamination between samples as the levels were higher than in the blank samples.

Among the solids matrices, the highest TWP concentrations were found in the sediments from the stormwater system (Paper III), with a median TWP concentration of 27±9.1 mg/g dw <500 µm (see chapter 3.3.1.1 for units), Figure 10. These concentrations are comparable to reported values from sediments of other gully pots and road dust, but higher than what has been reported for stormwater ponds (Table 1). The soil samples from Ditch 1 (the ditch dewatering the road) and Ditch 2 (the ditch also receiving the outflow from the stormwater system) presented lower median concentrations but more variability: 2.6±4.7 and 0.60±1.4 mg/g dw <500 µm, respectively. These values are overall similar to reported concentrations in roadside soil (Table 1). Finally, the sediments from Lillån stream contained somewhat higher median TWP concentration (3.9±2.0 mg/g dw <500 µm) than the soil samples. The values are in the same range as other reported TWP concentrations from river sediments (Barber et al., 2024; Unice et al., 2013). The main reason for the Lillån samples presenting higher TWP concentrations than the soil samples, which are located closer to the road and would be expected to receive higher TWP loads, is believed to be that the top 10 cm of soil, including roots and grass, was not analysed and this part likely contained the majority of TWP. Also, the sediments from Lillån were collected right below the highway bridge from which runoff is drained close to the riverbank, meaning that the sampled location is likely a TWP hotspot in the stream. The stream is also classified as being affected by transport and infrastructure (Water Information System Sweden, 2024), indicating that traffic-related pollutants could impact its water quality.

4.1.1 TWP in snow: concentrations and spatial patterns

In Paper I, the TWP concentration in the snow from the various collection locations (Ditch 1, Ditch 2 and the roadside ditch in the section with the kerbstone Kerb 1–5) are described more in detail. The highest concentrations were found on the road surface (1300 mg/L) by Ditch 1 and Kerb 2 in March 2023. The lowest TWP concentrations were instead found in December at Kerb 1 and 3 at some distance from the road (4.4 and 3.4 m, respectively), see Paper IV Figure 2. The measured TWP concentrations in snow were substantially higher than those reported in urban dumping sites, but lower than what was found in highway-adjacent Norwegian snowbanks, Table 1. These discrepancies can be attributed to different traffic loads, road types and meteorological conditions, and one likely reason for the higher concentrations in the Norwegian study is that the snowbanks had undergone several melting and freezing episodes.

The snow from Testsite E18 sampled in March 2023, presented significantly higher concentrations than in December 2022, despite the December snow cover being older, thus likely accumulating more pollutants. A possible reason for this is the partial thawing weather on March 2023, concentrating the TWP in the snow. These findings highlight the complexity of factors influencing TWP concentrations in snow beyond snow age and traffic load, and are consistent with observations by Vijayan et al. (2022), who also found no relationship between traffic load and number of tyre and road wear particles detected in roadside snowbanks.

The concentration of TWP in the snow samples clearly decreased with road distance, following an inverse power relationship (Paper I). Decreasing concentrations of TWP and tyre and road wear particles have previously been described between snowbanks at 3 m from the road and 0 and 1 m, and between 0.5 m and 2.5 m, but not between 1 and 0 m from the road (Rødland et al., 2022b; Vijayan et al., 2022). The results confirm that TWP behaves similarly to other traffic-related pollutants found in snowbanks, with decreasing concentration within 5 m from the road (Gjessing et al., 1984; Kuoppamäki et al., 2014; Reinosdotter et al., 2006).

To improve comparability with other studies, TWP deposition was also expressed as mass load per square metre of surface area (mg/m²), following the approach of Vijayan et al. (2024b). This removes the influence of snow depth on the TWP concentration and offers a more consistent basis for cross-study comparison. However, the snow collection methodology applied in this study generated additional uncertainty in mass load estimates, due to incomplete retrieval of all snow from the sampled surface, and due to uncertainties in the area measurements. For future studies involving the sampling of loose snow, the installation of containers of known surface area prior to snowfall is recommended to ensure complete collection of the snow.

4.1.2 TWP in soil and sediments: spatial variation

Spatial variations of TWP in roadside ditches were investigated in Paper II. The analysis of soil samples from Ditch 1 revealed that TWP concentrations were significantly higher closer to the road (1.7 m) compared to further away at 3.1 m. This suggest a decreasing trend of TWP concentration with road distance, similar to what was seen for the snow (Paper I), however this trend was not consistent as soil sampled at 2.5 m from the road had higher concentrations than at 2.1 m. The presence of TWP even at the furthest distance (3.1 m) indicates some degree of horizontal mobility, either through air deposition or through soil, as soil closer to the road consisted of more permeable soil types with gravel and dry-crust clay. Decreasing concentration of TWP in soil with road distance has been reported for German highways, but was not found in Norwegian roadside soils (Table 1). Ditch 2 showed no significant decrease of TWP concentration with distance from the stormwater outlet. This could indicate higher horizontal mobility due to transport with free-running water on the ditch surface, even if the soil in Ditch 2 consisted of more clay compared to Ditch 1 and is likely less permeable.

Neither Ditch 1 nor Ditch 2 showed significant difference in TWP concentrations with sampling soil depth. The apparent vertical mobility of TWP can have several explanations based on the soil type. In Ditch 1 closer to the road, where the soil is more permeable, the infiltration of TWP with runoff is more likely, but also cross-contamination due to sampling methodology could have occurred from upper-to-lower layers. Further from the road in Ditch 1 and in Ditch 2, where the soil consisted more of clay, which is a low-permeability soil, the more likely transport mechanism is preferential flow of infiltrated runoff through roots or cracks formed during dry weather. Similarly to Paper II, no significant difference in TWP concentration between 0–0.1m and 0.1–0.2 m depths was observed in Norwegian roadside ditches (Rødland et al., 2023b), whereas a decrease was reported in a roadside biofiltration swale, where TWP concentrations were lower between 0.05–0.35 m depth compared to the topsoil layer at 0–0.05 m (Beaurepaire et al., 2025).

Regarding TWP in sediments (Paper III), based on the concentrations over the total dry weight (mg/g dw), higher TWP concentrations were found in the sediments of the stormwater system collected in March 2023 compared to November 2022. Furthermore, TWP concentrations were found to increase along the stormwater system: between the gully pots and well WA, and between WA and sediment in the outlet pipe (location D, Paper III). This downstream increase may be attributed to two factors. First, TWP is usually found in finer size fractions (typically <200 µm; Klöckner et al., 2021a; Kreider et al., 2010), and fine particles are more mobile and less likely to be retained within stormwater systems. Second, coarser sediments (>500 µm) are more effectively retained upstream in system, in the sand traps of the gully pots and the well, which is supported by the observation that only 3–5% of sediments were >500 µm in the outlet pipe D (SI B.10, Paper III). This could result in a relative enrichment of TWP along the stormwater system.

4.1.3 TWP in the stormwater system: sampling of standing water volumes and runoff during rain

The grab samples of water collected from standing water volumes in the stormwater system (Paper III) showed no statistically significant difference between the three sampling occasions. Among the sampled locations, the second gully pot in the stormwater system (GP2) had higher concentrations than the first gully pot (GP1), the downstream connected well WA, and the standing water in Ditch 2 receiving water from the stormwater system (location D, Paper III). Overall, the TWP concentrations in the quiescent standing water volumes were similar (0.5–3 mg/L). The higher concentrations in GP2 were attributed to more sediment resuspension during sampling, due to a high sediment level in the sand trap.

Grab samples taken during rain on the 2022-11-08 yielded considerably higher TWP concentrations (Paper III). Specifically, the sample of outflowing water from well A (sample OA in Paper III) had 13 mg/L, and the grab sample of direct runoff from the road (location R) reached 21 mg/L. Even higher TWP concentrations were recorded during automatically sampled rain events using ISCO samplers (Paper IV): on 2023-04-29, the event mean concentration (EMC) from well WA was 93 mg/L, and on 2023-05-17, 64 mg/L. Based on the automated samples collected and the corresponding flow measurement at the outlet of well WA, the mass load of TWP emitted from well WA during the rains was estimated to be 204 g on 2023-04-29 and 3.9 g on 2023-05-17.

4.2 Sizes of TWP in different environmental compartments and detailed morpho-chemical characteristics in snow

4.2.1 Proportions of TWP in size fractions 1.6–20 µm

The concentrations of TWP in the size fraction 1.6–20 µm were specifically quantified separately from the TWP in size fraction 1.6–500 µm in the samples of snow, water and sediments (Paper I, III and IV). An overview of the relative proportions of TWP in the size fraction 1.6–20 µm can be seen in Figure 11.

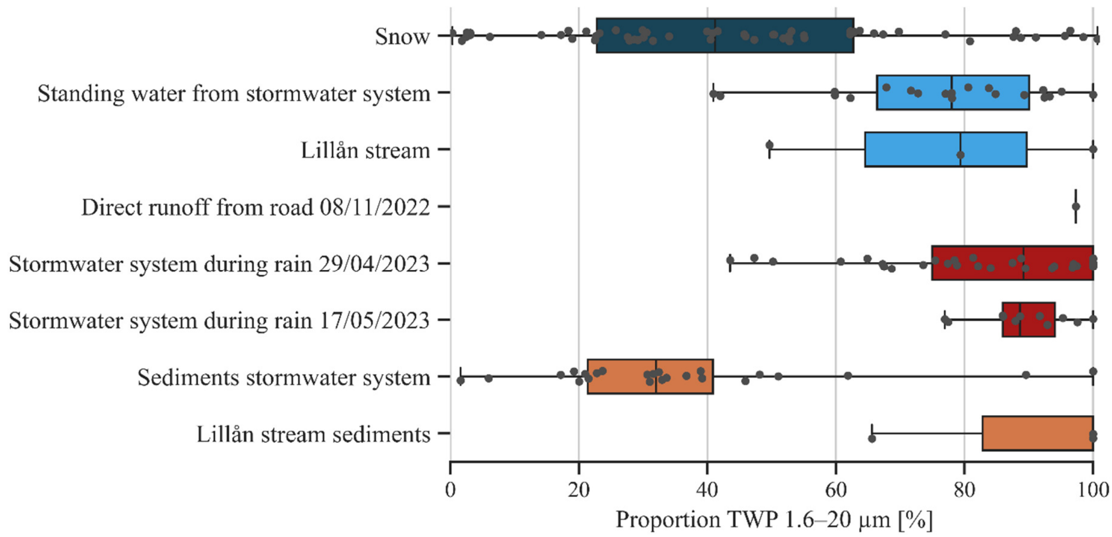

Figure 11: Boxplots of proportion of TWP in size fraction 1.6–20 µm. Median, 25th and 75th quartiles and whiskers indicating minimum and maximum values are shown, along with scatter plots of individual samples. The proportions have been capped to 100%. The results are grouped by sample type and coloured by study: Paper I (dark blue boxplots), Paper III (light blue and orange boxplots) and Paper IV (red boxplots).

The proportion of TWP in the fine fraction (1.6–20 µm) varied between the environmental compartments analysed. The highest median proportions of TWP 1.6–20 µm were observed in stormwater samples collected during rainfall events on 2023-04-29 and 2023-05-17, and these proportions were similar for the two events (median ± standard deviation = 89±17% and 89±7%, respectively). The grab samples from standing water along the stormwater system also showed a high median proportion of TWP 1.6–20 µm (78±17%). The snow samples exhibited a moderate proportion of TWP 1.6–20 µm (41±21%), but also had the largest variability between samples. Lillån stream samples showed particularly wide variation, 50–100% of TWP 1.6–20 µm. In contrast, sediment samples generally contained lower proportions of TWP 1.6–20 µm, with 32±23% in the stormwater system sediments, whereas the sediments from Lillån showed a relatively higher abundance (65–100%). The relatively abundant proportion of TWP in the finer size fraction <20 µm is in line with some more recent works on TWP size fractions, with reported modes at 34 µm for TWP from a road simulator (Kovochich et al., 2021) and <20 µm for TWP in tunnel road dust (Klöckner et al., 2021a).

The proportion of TWP 1.6–20 µm in the snow samples (Paper I) revealed some variability with sampling occasions, with the December 2022 samples generally containing a higher proportion of fine TWP than those collected in March 2023 (median values 52 and 28%, respectively). However, no consistent trend in the proportion of fine TWP was observed with road distance, nor with the presence of the kerbstone. This suggests that within 6 m from the road, there is no clear difference in deposition of TWP in finer and coarser size fraction, with meteorological factors and traffic conditions potentially influencing the particle size distribution of generated or deposited TWP in the near-road environment in complex interactions. Another cause could be the partial melting in March, that could play a stronger role in concentrating coarse particles. Transport processes likely vary between different size fractions, as was seen by Vijayan et al. (2022) for tyre and road wear particles of coarser size fractions in laboratory snow piles: the finer 50–100 µm was more mobile within the meltwater than size fractions 100– 300 µm and >300 µm. Another explanation for the larger variability in proportion of TWP 1.6–20 µm found in the snow samples could be the different pre-treatment steps that the size fractions 1.6–20 µm and <500 µm were subjected to, whereas the water samples, which showed less variability in proportion of fine TWP, were subjected to more similar sample retrieval and pre-treatment steps regardless of the size fraction analysed.

Compared to the other samples matrices, the grab samples of water (Paper III) were subjected to an additional pre-treatment step. This pre-treatment could have affected the proportion of TWP in fine size fraction 1.6–20 µm. Due to agglomeration of the solid material in the water sample prior to analysis due to sample age, the samples were subjected to an ultrasonication step to break up the agglomerations, as mentioned in chapter 3.3.3. To assess the effect of this additional step, a pre-treatment and aging test of an aggregated sample was performed (Paper III, SI C2). The test showed that ultrasonication increased the average proportion of fine TWP to 80%, compared to 60% without ultrasonication. This is likely due to the break-up of mineral encrustations previously reported on TWP (Järlskog et al., 2022b; Kovochich et al., 2021; Kreider et al., 2010).

The proportion of TWP 1.6–20 µm was different between samples of standing water and samples collected during rain (Paper III). The samples of direct runoff (R) and outflowing water from well WA (OA in Paper 3) showed the highest proportion of TWP 1.6–20 µm (over 97%), whereas standing water samples had lower proportions (42–93%). Similarly, high proportions of TWP 1.6–20 µm were found in the automatic stormwater samples collected from the gully pots and wells during the rain events on 2023-04-29 and 2023-05-17 (Paper IV). This could suggest that finer TWP may be more easily mobilised or re-suspended during the sampled rainfall events. Rain characteristics have been found to affect the size distribution of particles in runoff (Yan et al., 2024); however, more data on TWP size distribution in runoff are needed to assess this, as well as on the bio-chemical processes that could affect TWP sizes in stormwater systems over time (Li et al., 2024).

In sediments from the stormwater system, 20–60% of the SBR+BR mass was found in the fine size fraction, aligning with previous studies reporting 20–40% of TWP <20 µm in sedimentation basins (Klöckner et al., 2020). These proportions are still abundant, indicating that an important portion of TWP mass might be overlooked in the analysis of environmental sediment samples if the finer size fraction is removed in sample pre-treatment. The median proportion of TWP 1.6–20 µm in the sediments is lower than in other environmental compartments sampled, and could indicate that sedimentation in the stormwater system can retain coarser TWP, but a substantial fraction of fine particles remains suspended and may continue downstream. Similar to the observations in the snow samples, the sediment samples also showed a greater variability in the proportion of fine TWP. As for the snow, this variability might result partly from the different pre-treatment step that the size fraction 1.6–20 µm received compared to <500 µm: the finer sediment fraction was wet-sieved and filtered more similarly to the water samples, whereas sediments <500 µm were wet-sieved and dried in jars. Sediment samples also exhibit greater heterogeneity compared to other sample matrices, which may further explain the higher variability in the proportion of TWP 1.6–20 µm.

4.2.2 Morpho-chemical characteristics of solids and TWP in snow

In Paper I, four snow samples were analysed using SEM-EDX coupled to a machine learning algorithm for material classification of the particulate material. An overview of the results provided by SEM-EDX+ML analysis is provided here, and detailed results can be found in Paper I. From the SEM-EDX results for individual particles, the relative mass composition of different material categories in the snow could be estimated according to Paper I Table 2. The majority of particles found in the snow were mineral particles, with bitumen and TWP as second and third most abundant materials. This distribution aligns with previous studies investigating road dust composition: at Testsite E18 also employing SEM-EDX+ML (Järlskog et al., 2022a), in a road simulator under cold-climate driving conditions (Kupiainen et al., 2005), and in urban snow also utilising SEM-EDX+ML (Blomqvist et al., 2023). The urban snow samples however did show slightly higher TWP content in the 20–125 µm range (average 4%, Blomqvist et al., 2023), likely due to more intense tyre wear by urban driving conditions. Air samples collected near urban roads, however, showed different compositions, with substantially higher proportions of TWP (7–39%) and organic material (34–65%), and a lower mineral content (23–27%; Gao et al., 2022). The different compositions in snow compared to Gao et al. (2022) likely reflect regional variations in climate and traffic conditions. In cold environments, increased road wear can be expected from studded tyres and winter road maintenance, which may increase the relative abundance of mineral particles (Gustafsson et al., 2009).

The particle size, defined as equivalent circular diameter (ECD), and the aspect ratio of the identified TWP (size fraction 20–125 µm) and TBWP (size fraction 2–20 µm, see chapter 3.3.2 for definition) obtained from the SEM-EDX+ML results are shown in Figure 12. The average ECD was 29 µm and 4.5 µm respectively for TWP and TBWP. The aspect ratio was similar between TWP and TBWP (aspect ratio 1.55 and 1.58, respectively). The results are in line with what other studies employing SEM-EDX methodologies have assessed for TWP in air (Rausch et al., 2022) and particles from road simulators (Kovochich et al., 2021). Comparison between Figure 12 below and Figure 6 in Järlskog et al. (2022b), in which the authors analysed road dust from the same highway section also using SEM-EDX+ML, suggests that TWP and TBWP in snow present a more narrow range of aspect ratios, <4, compared to road dust, where some particles presented aspect ratios >4. Based on the two TWP subclasses firm-elastic and sub-elastic identified by Wilkinson et al. (2023), see chapter 2.3.2, the aspect ratios of TWP in snow suggest more similarity to the firm-elastic type, which is described as more rounded and cloud-like in appearance, and containing less mineral encrustations than the sub-elastic type. TWP in road dust was reported to have a less cloud-like appearance, compared to air samples (Järlskog et al., 2022b), which could indicate that TWP in snow are more pristine than samples from the road subjected to more aging and weathering, but it is likely that a combination of both firm-elastic and sub-elastic TWP were present in snow.

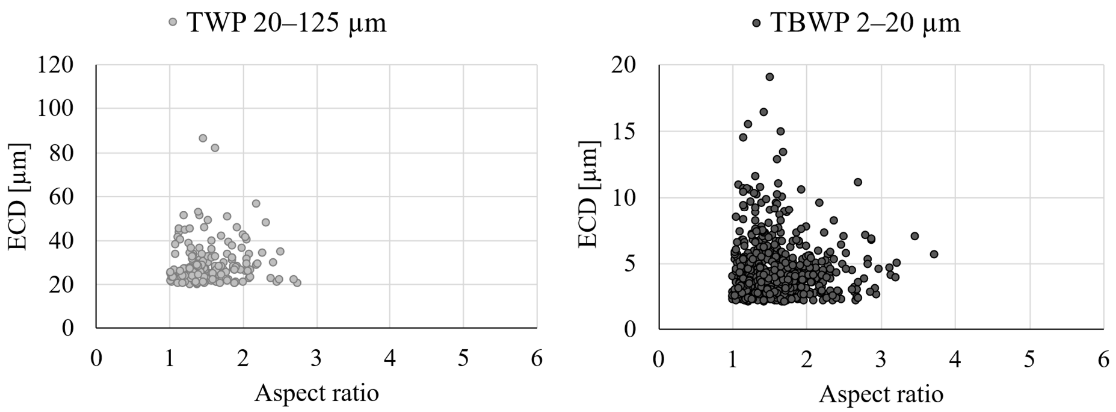

Figure 12: Aspect ratio vs ECD for TWP (20–125 µm) and TBWP (2–20 µm) identified in the snow samples analysed with SEM-EDX.

The elemental composition of TWP (20–125 µm) and TBWP (2–20 µm) consisted of primarily O, C and Si, consistent with previous studies (Järlskog et al., 2022b; Rausch et al., 2022). The rest of the elemental profile resembled that of the mineral and bitumen particles identified in the snow (Paper I SI D3), supporting the idea that the mineral encrustations embedded in TWP originate from the other road dust material, as was also seen in TWP from road dust (Järlskog et al., 2022b). Zinc was almost absent from the TWP and TBWP, aligning with earlier environmental findings (Järlskog et al., 2022b; Sommer et al., 2018) despite its known presence in TWP from road simulators (Kovochich et al., 2021).

4.3 Temporal variability of TWP during rain events

The results in this thesis (chapter 4.1 and 4.1.3, Paper III and Paper IV) point at a temporal variability and possible resuspension of TWP during rain events, where TWP concentrations in the gully pots and wells were much higher during rain events compared to TWP concentrations in the standing water at the same locations. The difference is expected as stormwater during rain would contain newly washed-off particles from the road as well as possible resuspension from the sediments in the stormwater system, whereas in quiescent water sedimentation would to some extent transfer TWP to the sediments. However, the large difference, with 16–42 times higher TWP concentrations during rain, highlights the dynamic and episodic nature of TWP transport in stormwater. Overall, grab sampling is a cost-effective sampling method with less risk for equipment theft compared to automatic sampling (Erickson et al., 2013), and has been employed in the monitoring of TWP and related compounds in water (Dröge and Tromp, 2019; Järlskog et al., 2022a; Klöckner et al., 2020; Rauert et al., 2022). The results in Paper IV show that for stormwater monitoring programmes, issues arise with grab sampling due to its limitations in capturing the temporal variability and peak concentrations of TWP during rain events, yielding lower TWP concentrations that are less relevant for understanding the environmental load. Automated sequential sampling allows more accurate characterisation of TWP transport dynamics and is generally preferable for understanding pollutant loads, even if some drawbacks exist for sampling time-sensitive pollutants and parameters (e.g. pH, DOC and faecal indicator bacteria; Gulliver and Anderson, 2008), and for samples where there is a risk of cross-contamination from the equipment (Ma et al., 2009).

Apart from the aforementioned difference between grab samples and automatic stormwater samples from Testsite E18, the temporal variability of TWP during rain events is also stressed by the TWP concentrations during the two rain events of 2023-04-29 and 2023-05-17 (Paper IV). For both rain events, TWP concentrations were higher at the beginning of the rain event, which is evidenced by the negative correlation between TWP concentration and time from the start of sampling. The strength of the correlation suggests that the rain on 2023-04-29 showed a larger decrease in TWP concentration with rain duration, especially in the gully pots GP1 and GP2 ($r = -0.92$), compared to the rain event on 2023-05-17 ($r = -0.67$). Linear regression highlighted that the decrease of TWP concentration with time from start of the sampling was significant, but the sampling date was not found to have a significant effect.

The rain event on 2023-04-29 yielded higher concentrations in the wells when comparing the EMCs of the two rain events. Same was seen in the highest TWP concentrations measured during the rains, with the rain on 2023-04-29 displaying higher TWP concentrations at the beginning of the rain, with six samples between 85 and 186 mg/L, compared to the max value of 85 mg/L on 2023-05-17. The reason for the difference is likely the short and high intensity profile of the rain on 2023-04-29. Variability in TWP release from the stormwater system between the rains was also demonstrated by the M(V) curves (Paper IV, SI 3.5): the slope of the normalised cumulative mass emission vs normalised cumulative runoff volume was larger than 45° for the rain event on 2023-04-29, thus meeting the definition of first flush by Geiger W (1987), but not on the 2023-05-17. A first-flush effect was however not confirmed for neither rain when applying the definition of 80% or more of TWP mass discharged with the first 30% of runoff volume (Bertrand-Krajewski et al., 1998). As only two rain events were captured in Paper IV, despite the extended sampling period of six months, further conclusions regarding TWP emissions from the stormwater system based on rain intensity, rain duration, antecedent dry days, traffic amounts and other meteorological effects cannot be drawn, and further studies are needed to highlight the factors affecting TWP wash-off from roads and release from stormwater system.

4.4 Correlations between TWP and other water quality parameters

In Paper I–IV, the relationship between TWP concentrations and other parameters and stormwater-related pollutants (Table 2) was assessed in the various environmental compartments by investigating the correlations between them. This chapter provides an overview of the correlations with TWP investigated in several of the papers, and additional details and correlation results can be found in the respective studies.

Correlations between TWP and solids concentrations/TSS in waterborne samples (snow and water) were assessed in Paper I, III and IV. The relationship between TWP and organic content (TOC, VSS and LOI) was also examined for snow, grab samples of water, stormwater during rain, and sediments (Paper I, III and IV). These results are summarised in Table 3.

Table 3: Correlations between TWP and solids concentrations or concentrations of organic material (see Table 2 for a list of laboratory analyses performed) in the environmental compartments investigated. The naming of the locations can be found in Figure 4 and Figure 5.

Paper	Sample type	Locations	TWP size fraction	Parameter	Correlation strength r	p-value	Correlation type
Paper I	Snow	Ditch 1, Ditch 2, kerbstone sections	<500 µm	Solids conc.	0.93	< 0.001	Spearman's rank order
Paper I	Snow	Ditch 1, Ditch 2, kerbstone sections	1.6–20 µm	Solids conc.	0.94	< 0.001	Spearman's rank order
Paper III	Grab water	GP1, GP2, WA, D, R, L	1.6–500 µm	TSS	0.87	< 0.001	Pearson
Paper III	Grab water	GP1, GP2, WA, D, R, L	1.6–20 µm	TSS	0.86	< 0.001	Pearson
Paper IV	Stormwater during rain	GP1, GP2, WA	1.6–500 µm	TSS	0.99	< 0.001	Spearman's rank order
Paper IV	Stormwater during rain	GP1, GP2, WA	1.6–500 µm	Turbidity	0.17	0.243	Spearman's rank order
Paper I	Snow	Ditch 1, Ditch 2, kerbstone sections	<500 µm	TOC	0.42	0.062	Spearman's rank order
Paper I	Snow	Ditch 1, Ditch 2, kerbstone sections	1.6–20 µm	TOC	0.40	0.083	Spearman's rank order
Paper III	Grab water	GP1, GP2, WA, D, R, L	1.6–500 µm	VSS	0.933	< 0.001	Pearson
Paper III	Grab water	GP1, GP2, WA, D, R, L	1.6–20 µm	VSS	0.927	< 0.001	Pearson
Paper IV	Stormwater during rain	GP1, GP2, WA	1.6–500 µm	VSS	0.98	< 0.001	Spearman's rank order
Paper III	Grab sediments	GP1, GP2, WA, D, L	1.6–500 µm	LOI	0.72	< 0.001	Pearson
Paper III	Grab sediments	GP1, GP2, WA, D, L	1.6–20 µm	LOI	0.91	< 0.001	Pearson

As can be seen in Table 3, the correlation between TWP concentrations and solids concentrations/TSS was very strong in all analysed samples, with *r*-value 0.87–0.99. The stormwater samples collected from the gully pots GP1 and GP2, and well WA during the rain events of 2023-04-29 and 2023-05-17 showed the strongest correlation with solids concentrations/TSS (Paper IV). The size fraction 1.6–500 µm of TWP showed slightly stronger correlation with TSS that the 1.6–20 µm did, but both correlations were strong. The correlations found in all environmental compartments indicate that the overall transport of TWP deposited on snow and mobilised in stormwater is very similar to the way in which generic particulate material would behave. This relation may only apply to specific locations and systems, and cannot yet be generalised from the results by Testsite E18, but there is some evidence that the correlation is not only site-specific, as similar relationship between TWP and TSS have been found in snowbanks ($r = 0.70$, $p < 0.0001$; Rødland et al., 2022b) and tunnel wash water (adjusted $r^2 = 0.88$, $p < 0.0001$; Rødland et al., 2022a). The possibility of exploiting TSS as an indicator parameter should therefore be explored further due to the high cost and extensive pre-treatment of TWP analyses, which currently makes them unsuitable for monitoring programs. Turbidity has been highlighted as a potential parameter for water quality monitoring (Parra et al., 2018), and since strong correlations have been reported with TSS (Al Ali et al., 2016; Rügner et al., 2013), a strong correlation with TWP was expected. However, this was not the case for the stormwater samples collected during rain, and a lack of correlation between TWP and turbidity was also reported in another study on road runoff (Lindfors et al., 2025). Further research is warranted, as the correlation result from Paper IV may have been affected by the relatively few samples analysed, and the time delay between sampling and turbidity measurement.

Organic material also showed strong correlations with TWP concentrations in standing water and water sampled during rain in the stormwater system at Testsite E18 (Paper III and IV, Table 3). Similarly, TWP in sediments from the stormwater system and from Lillån, correlated strongly with the organic material as measured through LOI (Paper III). Snow samples, on the other hand, did not show significant correlations between TWP and TOC. The variability in correlations between TWP and organic material may be influenced by the season during which sampling was carried out, as the snow samples were collected in winter, when biological activity is low, whereas the stormwater samples during rain were collected in spring, when biological activity such as pollen release is high.

The correlations between TWP concentrations and total metals were also assessed for several of the analysed matrices (Paper I, II and IV) and are summarised for selected metals in Table 4. As can be seen from Table 4, TWP concentrations in the snow and stormwater samples correlated strongly with all total metals shown in Table 4. Most metals in the snow were found in particulate form (Paper I SI F1.2) and similar transport pathways among particles are therefore a likely reason for the strong correlations. The fine fractions of TWP (1.6–20 µm) in snow correlated similarly to metals as the fraction <500 µm, possibly since this fine fraction constitutes a large proportion of TWP <500 µm, as discussed in chapter 4.2.1. The soil samples instead showed no clear relationships between TWP and metals, apart from Zn in Ditch 1, which correlated strongly with TWP, and Cu (Ditch 1) and V (Ditch 2), which correlated moderately. The differing results for soil compared to snow and stormwater may be due to the larger heterogeneity of the sample matrix, but most likely also reflects the different processes that TWP and metals are subjected to in soil compared to in the other environmental compartments.

Table 4: Correlations between TWP concentrations and selected total metals. Pearson's correlation was applied in Paper II. Spearmans' rank correlation was used for snow samples (March 2023. Paper I) and for stormwater sampled during rain on the 2023-04-29 (Paper IV).

Paper	Sample type	TWP size fraction [µm]	As	Ba	Co	Cr	Cu	Ni	Pb	V	Zn
Paper II	Soil Ditch 1	<500	-0.36	0.04	-0.09	-0.20	0.54**	-0.46*	0.13	0.33	0.80***
Paper II	Soil Ditch 2	<500	-0.15	0.17	0.35	0.41	-0.12	0.29	0.09	0.50*	0.11
Paper I	Snow	<500	0.93***	0.94***	0.96***	0.93***	0.81***	0.94***	0.94***	0.94***	0.95***
Paper I	Snow	1.6–20	0.87***	0.90***	0.88***	0.86***	0.78***	0.88***	0.88***	0.88***	0.90***
Paper IV	Stormwater 2023-04-29	1.6–500	0.91***	0.97***	0.97***	0.77***	0.73***	0.81***	0.89***	0.77***	0.96***

* $p < 0.1$
** $p < 0.05$
*** $p < 0.01$

4.5 Microplastic polymer concentrations measured in stormwater during rain

The abundance of other plastic polymers (size 10–1000 µm) in the highway stormwater system by Testsite E18 was also assessed, by analysing selected aggregated samples of stormwater from gully pots GP1 and GP2 and well WA from the rain events of the 2023-04-29 and 2023-05-17 (see chapter 3.3.3). The MP polymer types quantified and their concentrations in the stormwater samples varied between samples, and did not show any visible trend with time from start of sampling. The polymers PS, PMMA, PC, PET, PA6 (only detected in one sample at concentrations similar to blank sample), and PA66 were below detection limit. Polymers that were recurrently found in several samples were mainly PE, PP, SBR and PI. Acrylonitrile butadiene styrene, PB and PVC were detected in few samples. Microplastic PE and PP concentrations were 5.8–1300 µg/L and 2.4–552 µg/L, respectively. These concentrations are similar to quantified amounts in highway runoff from a sedimentation well (8.4–180 µg/L and up to 95 µg/L; Johansson et al., 2024, 2025). The polymers PE and PP are also among the most reported in stormwater and sediments from gully pots and stormwater outlets (Cho et al., 2023; Lange et al., 2022; Liu et al., 2019; Öborn et al., 2024b; Treilles et al., 2021).

For MP analysis, contamination from equipment and other sources is always a concern. The automatic ISCO samplers consist of several plastic parts, with bottles in PP and PE, and pump hose in PVC, meaning that plastic contamination could occur both from the equipment and from the laboratory environment. For this reason, an aggregated blank sample of milliQ water pumped through the sampler equipment was analysed too, showing detectable presences of ABS, PET and PP, but with PP value on average 17 times lower than in the samples and 1.7 times lower than in the sample with the least concentration.

Regarding the rubber polymers detected in the samples (PB, PI and SBR), they made up, on average, 41±24% of all MP quantified in the samples where they were detected. Hence, rubber polymers represent an important fraction of the quantified MP. However, the rubber polymers were not detected in all samples, even though tyre wear is expected to be the main source of MP in this road environment, and the only other likely MP sources are air deposition, littering and wear of plastic vehicle parts (Österlund et al., 2023). The quantified rubber polymers with the Pyr-GC/MS methodology of the commercial laboratory, are also a factor 1000 lower than the SBR+BR concentrations quantified with Pyr-GC/MS for rubber analysis (used in Paper I, II, III and IV, see chapter 3.3.1.2). Papers IV and V of the study were not designed specifically for comparison of the analytical methods. Therefore, the reason for the large discrepancy is difficult to assess due to the very different pre-treatment methodologies, size cut-off thresholds and polymer markers utilised, but is likely due to a combination of them. As previous results from Paper IV showed a very large proportion of TWP in size fraction 1.6–20 µm (median 89% for sequential stormwater samples during rain, section 4.2.1), a large amount of TWP <10 µm could at least partly explain the discrepancy. Additionally, as was seen in Paper III and chapter 3.3.3, the ultrasonication pre-treatment of the MP-samples in Paper V might have caused break-up of TWP into sizes <10 µm and caused additional loss of particles.

4.6 Implications for remediation strategies of TWP

The work carried out in this thesis deals with TWP as an emerging water pollutant spreading from the built environment through stormwater systems to recipient surface waters, where there is a risk of toxic effects on biota as TWP and their associated chemical compounds have shown toxic effects in laboratory studies (Boisseaux et al., 2024; Chen et al., 2023; Liu et al., 2022; Page et al., 2022; Tian et al., 2021b; Wik and Dave, 2006b). Hence, the thesis targets international environmental goals, as the UN Sustainable development goals 6 (Clean water and sanitation), 11 (Sustainable cities and communities), and 14 (Life below water) (United Nations, 2024).

Through the lens of sustainable development, defined as the equilibrium between the needs of present and of future generations (Brundtland, 1987), TWP emissions and spread represent a complex issue: TWP are a consequence of modern transportation, which is a necessary tool to provide goods and services, to develop human capital and knowledge and to provide basic needs of present and future generations. On the other hand, transportation in its modern form threatens the ecological preconditions for sustainable development, by potentially compromising the need to preserve ecosystems that can provide services to future generations (Millennium Ecosystem Assessment, 2005). Toxic effects of some TWP leachates appear so far to be specific to few sensitive species (Rauert et al., 2022), and some researchers state that toxic effects of TWP have not been assessed at environmentally relevant concentrations (Redondo-Hasselerharm et al., 2018). However, due to the still large knowledge gap regarding toxic effects and environmental concentrations in for example, soils, together with the high estimated emission of TWP (Kole et al., 2017; Polukarova et al., 2024), there is reason to apply the precautionary principle as defined by the Rio Declaration of 1992 (United Nations, 1992).

More research is needed both to fill the aforementioned knowledge gaps on spread and environmental toxicity of TWP, but also to assess effective solutions that can reduce the spread of TWP to sensitive environmental compartments. Moran et al. (2021) categorised potential management options for TWP in the following classes based on a scale from prevention to remediation: (1) elimination of TWP or removal of toxic ingredients, (2) reduction of wear debris formation, (3) reduction of wear debris emission, (4) collection of TWP, and (5) removal of TWP from runoff.

Key agents for the elimination of TWP or removal of toxic ingredients would be tyre and vehicle manufacturers (Moran et al., 2021), with legislation driving the actions. More research is needed to assess and improve the chemical compositions of tyres and the toxicity of TWP, especially as "eco-friendly" tyres on the market have been found to contain similar levels of toxic compounds as traditional tyres (Rødland et al., 2024). The papers in this thesis can inform future toxicological studies by providing environmentally relevant information about TWP characteristics and concentrations. Actions towards the reduction of TWP formation and emissions could be taken both by tyre and vehicle manufacturers, as well as by governments and the general population (Moran et al., 2021). Factors that can reduce TWP formation are for example reduction of traffic or modification of driving behaviours (limiting acceleration/deceleration and turning) and vehicle maintenance (Gehrke et al., 2023). Devices for collecting tyre wear debris are also being developed for installation on vehicles (Moran et al., 2021).

Moving towards the "remediation" end of the scale, collection or retention of TWP on the road could be strategies to limit TWP spread at the source, and potential solutions could be porous asphalt (Rasmussen et al., 2023) and street sweeping (Järlskog et al., 2020). However, these measures would not address the TWP fraction that would deposit further from the road. At the far end of the remediation spectrum defined by (Moran et al., 2021), are stormwater management practices, which can play a critical role in mitigating TWP transport into receiving waters. This is where the findings of this thesis are most directly applicable as it provides detailed insights into TWP concentrations, particle sizes, and mobility in stormwater systems and in the near-road environment.

As was seen in Paper I and by Rødland et al. (2022b), the high concentrations of TWP identified in snow highlight the risk for acute emissions during snowmelt. Additionally, the TWP contribution from winter tyres in Sweden is relatively large, as it is estimated to account for 44% of total TWP emission from PV (Polukarova et al., 2024). Thus, TWP emissions through snowmelt may represent an important pathway for TWP release in Sweden and further research is needed to assess this. In Sweden, the dumping of snow in watercourses is prohibited, as it is considered an environmentally hazardous material; however, exemptions can be granted by the County Administrative Boards (Swedish Agency for Marine and Water Management, 2017). Since snow, especially from roads in cold regions with prolonged pollutant accumulation, constitutes a highly polluted matrix, mitigation strategies such as centralised storage with treatment should be considered for areas with high traffic density.

In locations where roads are dewatered through stormwater systems, TWP from both snowmelt and ordinary runoff might spread further through the stormwater system, as observed in Paper III and IV. Several end-of-pipe treatment solutions have shown potential for reducing TWP in stormwater: rain gardens and bioretention cells (Carvalho et al., 2024; Johansson et al., 2024; Langet et al., 2021; Mengistu et al., 2022), as well as stormwater ponds and infiltration ponds (De Oliveira et al., 2024; Rasmussen et al., 2024; Ziajahromi et al., 2023). In contrast, systems such as gross pollutant traps are unlikely to retain TWP effectively (Lange et al., 2022, 2021; Monira et al., 2021). Solutions relying solely on sedimentation are not likely to remove TWP <30 µm from stormwater (Vogelsang et al., 2019). Findings from Paper I, III and IV support these considerations, as significant proportions of TWP were found to be finer than 20 µm. Evaluations of removal efficiencies of treatment technologies should therefore consider their effectiveness in capturing fine particles. It is also important to consider that, since TWP pollution originates from traffic, which is a diffuse source, it is not feasible to apply treatment solutions universally. Instead, the implementation of end-of-pipe solutions needs to target high priority areas, where elevated TWP concentrations and vulnerable recipients are identified. The findings presented in this thesis regarding TWP concentrations and transport processes can support municipalities and other stakeholders in the prioritisation of mitigation measures and assist governmental agencies in developing environmental guidelines.

5 CONCLUSIONS

In this thesis, the environmental concentrations of TWP in various near-road compartments along a Swedish highway (including snow, soil, stormwater, sediments, and receiving waters) have been assessed. The thesis has contributed to: (i) providing data on TWP concentrations across multiple environmental compartments, as well as concentrations of microplastic polymers in highway stormwater, thereby highlighting the risk of acute emissions due to the high TWP concentrations in snow and stormwater during rain; (ii) improving the understanding of environmental TWP size distributions, by including quantification of the fine fraction <20 µm (often not assessed in the literature due to analytical challenges), as well as describing the morpho-chemical properties of TWP in snow; (iii) assessing the transport behaviour of TWP in the near-road environment, highlighting the spread of TWP through stormwater systems, to deeper soil layers, and the increasing TWP abundance with road proximity in snow.

The main conclusions of the thesis are:

- TWP were consistently found and quantified across all sampled near-road matrices—roadside snow, soil, stormwater, sediments, and natural recipients—demonstrating their widespread presence in the environment adjacent to highways in Sweden (Papers I–IV).
- Concentrations of TWP in roadside snow were high (7.8–1300 mg/L), especially in areas closest to the road, with a decreasing concentration profile with road distance often described for other traffic-related pollutants.
- Concentrations of TWP in highway stormwater during rain events were substantial (9–170 mg/L), with the runoff from the wells showing transport of TWP across the system, as well as variable concentrations during rain (Papers IV–V).
- Analysis of microplastic polymers showed that PE, PP and rubber polymers were the most occurring MP in the highway runoff, consistent with other studies reporting MP concentrations in stormwater and sediments, and with the expected sources of MP in the catchment area. However, the large discrepancy between concentrations of rubber polymers quantified with two different methodologies (Paper IV and V) highlights the need for continued assessment and improvement of analytical methods.
- Morphological and chemical characterisation of solids and TWP in snow for both coarse (20–125 µm) and fine fractions (2–20 µm) confirmed the pre-dominant presence of minerals in cold-region road environments, followed by bitumen and TWP (Paper I). The morpho-chemical composition of TWP was in line with other studies in terms of mineral encrustations and elemental composition, except for the less elongated shape of TWP in snow compared to road dust from the same area (Paper I).
- Size-fractionated analyses revealed that TWP in the fine fraction (1.6–20 µm) represented a substantial proportion of the total TWP in the environmental compartments, especially in stormwater during rain (median 89%), underlining the need to account for these fractions in monitoring, risk assessment and remediation techniques (Papers I, III, IV).
- The occurrence and transport of TWP in stormwater systems were confirmed through their detection in stormwater wells, gully pots, and connected ditches. Concentrations in sediments did not decrease along the stormwater system, with substantial amounts measured in the water from the consecutive wells of the system during rainfall events, indicating that retention in the system is limited (Papers III–IV).
- TWP were also quantified in receiving waters downstream of the stormwater outlet, confirming that these particles can be transported beyond initial retention points into natural environments (Paper III).

- Correlation analyses showed strong associations between TWP concentrations and various water quality indicators, including TSS, VSS, LOI, and suspended solids, as well as metals such as Zn and Cu (Paper I, III and IV). These correlations indicate that TWP, solids and metals in the near-road environment are subjected to similar transport pathways in the form of particles, except for in soil where clear correlations were not found.

6 FUTURE RECOMMENDATIONS

Based on the results from this thesis, and the implications for remediation strategies discussed, some suggestions for future work can be given. TWP is a diffuse pollutant with traffic as emission sources which are geographically very extended. Therefore, the implementation of solutions to remove TWP from runoff is not likely to significantly reduce overall emissions into the environment, but may still be justified for protection of sensitive receiving waters. More research is therefore needed to clarify the risk posed by TWP and prioritise where action is needed, starting from assessment of long-term toxicity and fate of TWP at environmentally relevant conditions.

To enable robust risk assessments, the development and harmonisation of analytical techniques for TWP quantification are needed. Methods for detecting TWP in environmental samples need to be further refined to achieve standardised, comparable and cost-effective analyses, that can address both complex matrices and finer TWP size fractions. These efforts should also include the evaluation of sample pre-treatment methods and their potential impact on the results. Additionally, further assessment and validation of indicator parameters should be carried out as they could make TWP monitoring more accessible and practical for use by municipalities and other non-specialist stakeholders.

Knowledge of the concentrations of TWP in other environment compartments further from the road is also needed, specifically assessing the transport pathways and patterns of TWP to obtain a more systemic view of TWP mobility, that goes beyond estimates based on scarce empirical data. Even though the current work has helped to shed some light on the TWP transport dynamics of deposition within 6 m from the road and along a stormwater system, the need for broader understanding of TWP mobility under different meteorological and hydrological conditions, and at different locations, warrants further investigation.

End-of-pipe solutions can be used to remove TWP from runoff in areas where its concentrations are expected to be concerning. Further research is needed to evaluate the efficiency of stormwater treatment options such as bioretention cells, rain gardens, and infiltration ponds, especially for finer TWP fractions that have been found to be abundant. Also, based on the TWP concentrations found in melted snow, snow management strategies, such as centralised storage with treatment, also deserve greater attention in regions with significant traffic-pollution and winter precipitation. Further, modelling the transport of TWP through urban drainage networks could support targeted placement of these mitigation measures and optimise their performance, together with environmental guidelines.

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