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Atmospheric Deposition of Tire Wear in Switzerland

**Atmosphärischer Eintrag von Reifenabrieb
in der Schweiz**

**Dépôts atmosphériques provenant de l'abrasion
des pneus en Suisse**

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Research project TRU_20_00E_01
at the request of Federal Roads Office FEDRO
September 2026 | 1832

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List of abbreviations

BR	Butadiene rubber
CMTT	Cryogenically milled tire tread
DTV	Daily traffic volume
EF	Emission factor
FTIR	Fourier-transform infrared spectroscopy
LC	Critical level
LOD	Limit of detection
MP	Microplastics
MDR	Mass deposition rate
MFA	Mass flow analysis
NABEL	Swiss national air pollution monitoring network
NR	Natural rubber
PCA	Principal component analysis
PM ₁₀	Particulate matter with diameter larger than 10 µm
Py-GC-MS	Pyrolysis gas chromatography–mass spectrometry
SBR	Styrene-butadiene-rubber
SBS	Styrene-butadiene-styrene
SEM/EDX	Scanning electron microscopy with energy-dispersive X-ray spectroscopy
SPA	Sum of peak area
TRWP	Tire and road wear particles
TWP	Tire wear particles

Summary

Tire and road wear particles (TRWP) are created by friction between tires and the road surface while driving. Since friction between tires and the road is essential for driving, their formation cannot be completely prevented. TRWP consists of a mixture of tire tread material and road surface material, primarily asphalt. Asphalt, in turn, is a mixture of bitumen and mineral components.

The proportion of tire material in TRWP is referred to as tire wear particles (TWP), a class of particles that are increasingly recognized as a significant environmental pollutant. Besides rubber, TWP consist of several other substances, some of which are toxic. These include fillers, plasticizers, vulcanizing agents, and additives. Some of these substances can be leached into the environment, harming organisms and potentially entering our food chain.

Most of the mass of TWP formed on roads consists of particles with a diameter greater than 10 μm . Only a small fraction consists of smaller particles. TWP thus contributes to the mass concentration of inhalable fine particulate matter (PM₁₀). However, the contribution of tire wear to the PM₁₀ concentration is small. Even at heavily trafficked locations, the contribution of TWP to PM₁₀ is less than 10%.

Due to their polymeric nature, TWP are often categorized as microplastics (MP), although chemically they are elastomeric materials rather than thermoplastic polymers. Current estimates indicate that tire wear constitutes one of the largest sources of MP entering the environment in Switzerland and globally. Despite this relevance, considerable uncertainty remains regarding the distribution pathways of TWP in the environment, particularly concerning the atmospheric transport and deposition of tire wear in the environment at greater distances to the point of origin.

Most existing assessments of the environmental fate of tire wear are based on mass flow analyses that integrate emission estimates with assumptions about transport routes into environmental compartments such as soils, surface waters, and sediments. These analyses commonly assume that only a small fraction of emitted TWP becomes airborne, typically limited to particles smaller than 10 μm , and that consequently only about 1–5% of total tire wear is distributed via the atmosphere over longer distances. However, these assumptions represent a significant simplification and have rarely been experimentally verified due to a lack of reliable measurements of total atmospheric deposition of TWP. The main reason for this knowledge gap is the analytical complexity of identifying and quantifying TWP in environmental matrices.

The aim of this study was to contribute to a better quantitative understanding of the atmospheric deposition of tire wear and to determine the amount of TWP that is deposited into the environment via the atmosphere at different types of locations (urban, suburban, rural). To achieve this aim, the annual atmospheric deposition of tire wear was determined at six locations in Switzerland with different land uses. For this purpose, samples were collected every four weeks at these locations over the course of a

year using various passive samplers. Samples of both dry deposition (i.e. atmospheric deposition during dry weather conditions) and total or bulk deposition (i.e. the sum of dry deposition and wet deposition via precipitation) were collected. As part of a partner project supported by the Swiss Federal Office for the Environment (FOEN) to determine the atmospheric deposition of microplastics (excluding tire wear), separate samples of dry and wet deposition were collected simultaneously at the same locations. These samples were also analysed using image analysis methods.

The selected locations were a suburban traffic site (Dübendorf-Überlandstrasse, 20 m from the edge of a road with 19'000 vehicles per day), an urban background site (Zurich-Kaserne), a suburban site (Dübendorf-Empa), two rural sites north (Payerne-MeteoSuisse) and south of the Alps (Magadino-Cadenazzo), and a rural site at over 1'100 m above sea level (Chaumont). All sites except the suburban traffic site belong to the Swiss National Air Pollution Monitoring Network (NABEL).

A key focus of this study was the determination of tire wear in total atmospheric deposition. Since no generally applicable and satisfactory method for determining tire wear in environmental samples is available, a mass-based analytical method based on pyrolysis gas chromatography-mass spectrometry (Py-GC-MS) was developed and implemented within this study. This method relies on the thermal decomposition (pyrolysis) of the sample, followed by the separation of the pyrolysis products in the gas chromatograph and their subsequent detection in the mass spectrometer. Existing methods for determining tire wear with Py-GC-MS often rely on one or two pyrolysis products specific to the thermal decomposition of tires (specific markers). These methods are associated with considerable uncertainties.

The method developed here utilizes six characteristic pyrolysis products formed from the three main rubber components used in tires—natural rubber (NR), styrene-butadiene rubber (SBR), and butadiene rubber (BR). The selected markers include butadiene, isoprene, vinylcyclohexene, styrene, α -methylstyrene, and limonene. By summing the integrated peak areas of these six markers and determining their dependence on the mass of tire tread, a reliable approximation for the total mass of tire material is obtained. Method calibration was performed using a comprehensive set of cryogenically milled tire tread (CMTT) samples collected from a wide range of used passenger car and truck tires, as well as a reference mixture representative of tires sold on the U.S. market. The calibration demonstrated a strong linear relationship between the sum of peak areas of the six pyrolysis markers and the mass of tire tread material across all tested tire types. Although differences between passenger car and truck tires were observed, the overall variability could be encompassed by a calibration uncertainty ranging from -20 % to +33 %, which was considered a realistic representation of systematic uncertainty arising from tire composition heterogeneity. Additional sources of uncertainty, including incomplete recovery during sample preparation, sensitivity fluctuations of the Py-GC-MS system, and spatial representativeness of deposition sampling, were quantified and combined to derive expanded uncertainty estimates for the reported deposition rates.

In addition to determining total atmospheric deposition of TWP using Py-GC-MS, the collected dry deposition samples were analyzed using scanning electron microscopy

and energy-dispersive X-ray spectroscopy (SEM/EDX). This particle-based method identifies individual TRWP based on their morphology and elemental composition and allows the calculation of mass deposition rates by aggregating contributions from particles across the particle diameter range from 1 to 80 μm . The SEM/EDX method is well-established in the scientific literature. However, since the method relies on collecting samples directly on surfaces suitable for electron microscopy, it cannot be readily applied to total atmospheric deposition samples. Since the mass of TRWP consists of approximately half tire material and half road material, the mass deposition rate of TWP can be easily calculated from that of TRWP.

The results of the Py-GC-MS analyses show that TWP are consistently detectable in total atmospheric deposition at all investigated sites and throughout the entire year. Mean bulk mass deposition rates exhibit a clear spatial gradient related to the proximity to traffic. The highest annual mean mass deposition rate was observed at the suburban traffic site (approximately $700 \mu\text{g m}^{-2} \text{ day}^{-1}$), followed by the urban background and suburban sites (around $500\text{--}600 \mu\text{g m}^{-2} \text{ day}^{-1}$). Rural sites showed substantially lower mass deposition rates (approximately $150\text{--}250 \mu\text{g m}^{-2} \text{ day}^{-1}$), while the mountainous site exhibited the lowest values (around $140 \mu\text{g m}^{-2} \text{ day}^{-1}$). These patterns are to be expected. The relatively large TRWP have a relatively high settling velocity in the atmosphere, which favours deposition near their point of origin and limits widespread distribution. However, under certain meteorological conditions (especially turbulence), large-scale transport and deposition at remote locations occurs.

To place the site-specific measurements into a national context, the deposition rates were extrapolated to estimate the total annual mean atmospheric mass deposition of TWP across Switzerland. This extrapolation combined measured deposition rates with detailed land use and elevation data at hectare resolution. Settlement and non-settlement areas were distinguished, and different elevation classes were considered up to 2'000 m above sea level. The analysis yields an estimated total atmospheric deposition of approximately 2'410 tons of tire wear per year in Switzerland below 2'000 m above sea level. Contributions from high-alpine regions above 2'000 m above sea level are expected to be relatively small and were not explicitly addressed in this study. This estimate has significant implications for the current understanding of the environmental fate of tire wear. Depending on the assumed total annual generation of tire wear in Switzerland (8'800–10'100 tons per year based on recent emission estimates), the observed atmospheric deposition corresponds to approximately 24–27 % of total tire wear emissions. This fraction is substantially higher than typically assumed in available reports and the scientific literature. Atmospheric transport and deposition therefore represent a major pathway for the environmental distribution of tire wear particles in Switzerland and likely elsewhere.

Measurements at the suburban traffic site Dübendorf-Überlandstrasse show that the input of tire wear is increased in the immediate vicinity of busy roads. This increase is attributable to the contribution from local traffic directly at the site. Such local contributions at locations in the immediate vicinity of busy roads are additional and are not included in the above estimate for Switzerland.

Calculations based on the samples analysed using SEM/EDX, the atmospheric deposition of tire wear from dry deposition in Switzerland below 2'000 m a.s.l. amounts to 1'417 tons per year. The proportions of atmospheric deposition from dry and wet deposition depend on climatic conditions. In Switzerland, dry deposition contributes approximately 60% and wet deposition approximately 40% to the atmospheric deposition of tire wear into the environment.

A comparison with the results of the partner project on the atmospheric input of other forms of microplastics shows that the atmospheric deposition of TWP is about ten times higher than the atmospheric input of other forms of microplastics. In Switzerland, tire wear is therefore the dominant source of microplastics that enter the environment via the air.

It should be noted that this study also has important limitations, from which recommendations for future research directions can be derived. The number of sampling sites was limited due to the complexity and cost of the applied analytical methods, resulting in uncertainties of the national-scale estimates that cannot be fully quantified. Expanding the spatial coverage of measurements and including additional land use categories would improve the robustness of those results. Furthermore, there is a clear need for more cost-efficient, high-throughput analytical approaches suitable for denser measurements and for long-term monitoring. Preliminary investigations using optical image analysis show promise but require thorough verification against chemical methods. Further development of analytical methods and their application in well-planned monitoring would lead to refined exposure assessments and evidence-based strategies for reducing the environmental impact of tire wear.

Zusammenfassung

Reifen- und Strassenabriebpartikel (englisch tire and road wear particles, oder kurz TRWP) entstehen durch Reibungskräfte zwischen Reifen und Strasse während des Fahrens. Da die Reibung zwischen Reifen und Strasse für das Fahren unerlässlich ist lässt sich deren Entstehung nicht vollständig verhindern. TRWP bestehen aus einer Mischung aus Material der Reifenlaufflächen und Material der Fahrbahnoberflächen, hauptsächlich Asphalt. Asphalt ist wiederum eine Mischung aus Bitumen und mineralischen Bestandteilen.

Der Anteil Reifenmaterial in TRWP wird als Reifenabriebpartikel (englisch tire wear particles, oder kurz TWP) bezeichnet. TWP beziehungsweise die Laufflächen von Reifen bestehen neben vulkanisiertem natürlichem und synthetischem Gummi aus einer Reihe weiterer zum Teil toxischer Substanzen wie Füllstoffe, Weichmacher, Vulkanisationsmittel und Additive. Teilweise sind diese Substanzen wasserlöslich und können aus dem Reifenabrieb in der Umwelt ausgewaschen werden, dadurch Organismen schädigen und in unsere Nahrungsmittel gelangen. TWP sind eine Klasse von Partikeln, die zur Verschmutzung und Gefährdung der Umwelt beitragen. Jährlich entstehen auf den Strassen in der Schweiz etwa 8'800 bis 10'100 Tonnen TWP.

Der überwiegende Anteil der Masse der auf den Strassen gebildeten TRWP entfällt auf Partikel mit einem Durchmesser grösser als 10 µm, nur ein kleiner Anteil der Masse von TRWP entfällt auf kleinere Partikel. TRWP tragen damit zwar zur Massenkonzentration des einatembaren Feinstaubs (PM₁₀) bei, der Anteil von Reifenabrieb (TWP) an PM₁₀ ist jedoch an nicht direkt verkehrsbelasteten Standorten klein. Auch an stark verkehrsbelasteten Standorten ist der Anteil von TWP an PM₁₀ kleiner als 10 %.

Aufgrund ihrer polymeren Natur werden TWP häufig als Mikroplastik (MP) klassifiziert, obwohl es sich chemisch um Elastomere und nicht um thermoplastische Polymere handelt. Reifenabrieb ist in der Schweiz und weltweit eine der grössten Quellen von Mikroplastik in der Umwelt. Trotz der Bedeutung von Reifenabrieb als Schadstoff bestehen derzeit noch grosse Unsicherheiten hinsichtlich der in die Umwelt gelangenden Mengen. Insbesondere ist noch weitgehend unklar, welche Menge von Reifenabrieb durch den Wind verfrachtet und dadurch in grösserer Entfernung vom Entstehungsort in die Umwelt eingetragen wird.

Die meisten Studien zur Verteilung von Reifenabrieb in der Umwelt basieren auf Stoffflussanalysen, bei denen Emissionsabschätzungen mit Annahmen über den Transport und Verbleib von Reifenabrieb in Umweltkompartimenten wie Böden, Oberflächengewässer und Sedimenten kombiniert werden. Bei diesen Analysen wird häufig davon ausgegangen, dass nur ein kleiner Anteil der erzeugten TRWP in die Luft gelangt und über die Atmosphäre räumlich verteilt werden kann. Häufig wird angenommen, dass nur Partikel mit einem Durchmesser kleiner als 10 µm längere Zeit in der Luft verweilen können und dass folglich nur etwa 1–5 % des gesamten Reifenabriebs über die Atmosphäre über grössere Distanzen verteilt werden. Diese Annahmen stellen jedoch eine starke Vereinfachung dar, die bisher kaum durch Messungen überprüft wurden.

Hauptursache dafür sind vermutlich die messtechnischen Herausforderungen bei der Identifikation und Quantifizierung von TWP in Umweltproben.

Ziel dieser Studie war es, zu einem besseren quantitativen Verständnis über den atmosphärischen Eintrag von Reifenabrieb beizutragen und die Mengen an TWP zu bestimmen, die über die Atmosphäre an unterschiedlichen Standortsituationen (städtisch, vorstädtisch, ländlich) in die Umwelt eingetragen werden. Zu diesem Zweck wurde die jährliche atmosphärische Deposition von Reifenabrieb an sechs Standorten in der Schweiz bestimmt. An allen Standorten wurden während eines Jahres vierwöchentliche Proben mit verschiedenen passiven Sammlersystemen erhoben. Es wurden sowohl Proben der Trockendeposition, d.h. des atmosphärischen Eintrags bei trockener Witterung, als auch der Gesamtd deposition, d.h. die Summe von Nassdeposition (Eintrag durch Niederschlag) und Trockendeposition, gesammelt. In einem vom Bundesamt für Umwelt unterstützten Partnerprojekt zur Bestimmung des atmosphärischen Eintrags von Mikroplastik mit Ausnahme von Reifenabrieb wurden zeitgleich an denselben Standorten getrennte Proben von Trocken- und Nassdeposition gesammelt. Diese Proben wurden ebenfalls mit Methoden der Bildanalyse auf das Vorhandensein von Reifenabrieb untersucht.

Die ausgewählten Standorte waren ein vorstädtischer Verkehrsstandort (Dübendorf-Überlandstrasse, 20 m vom Rand einer Strasse mit 19'000 Fahrzeugen pro Tag), ein städtischer Hintergrundstandort (Zürich-Kaserne), ein vorstädtischer Standort (Dübendorf-Empa), zwei ländliche Standorte nördlich (Payerne-MeteoSuisse) und südlich der Alpen (Magadino-Cadenazzo) sowie ein ländlicher Standort auf über 1'100 m ü. M. (Chaumont). Alle Standorte ausser dem Verkehrsstandort Dübendorf-Überlandstrasse gehören zum Nationalen Beobachtungsnetz für Luftfremdstoffe (NABEL).

Ein Schwerpunkt dieser Studie war die Bestimmung von Reifenabrieb in der atmosphärischen Gesamtd deposition. Da keine allgemein einsetzbare und zufriedenstellende Methode für die Bestimmung von Reifenabrieb in Umweltproben verfügbar ist, wurde im Rahmen dieser Studie eine massenbasierte Analysemethode basierend auf der Pyrolyse-Gaschromatographie-Massenspektrometrie (Py-GC-MS) entwickelt und eingesetzt. Diese Methode basiert auf der thermischen Zersetzung (Pyrolyse) der Probe mit der anschliessenden Auftrennung der Pyrolyseprodukte im Gaschromatographen und schliesslich deren Erkennung im Massenspektrometer. Bestehende Methoden zur Bestimmung von Reifenabrieb mit Py-GC-MS basieren oft auf ein oder zwei Pyrolyseprodukten, die spezifisch für die thermische Zersetzung von Reifen sind (spezifische Marker). Diese Methoden sind mit erheblichen Unsicherheiten verbunden. Die hier entwickelte Methode nutzt sechs Pyrolyseprodukte die für die bei der Reifenherstellung verwendeten Gummiarten (Naturkautschuk, Styrol-Butadien-Kautschuk und Butadien-Kautschuk) charakteristisch sind. Anhand des Summensignals der sechs Pyrolyseprodukte lässt sich die Gesamtmasse des Reifenmaterials in einer Probe bestimmen. Die Kalibrierung der Analysemethode erfolgte mit mehr als zwanzig kryogen gemahlenen Proben von gebrauchten Pkw- und Lkw-Reifen. Es zeigte sich eine starke lineare Beziehung zwischen dem Summensignal der sechs Pyrolyseprodukte und der Reifenmasse. Trotz Unterschiede zwischen den individuellen Reifentypen kann die Reifenmasse mit einer Unsicherheit von etwa -20 % bis +33 % bestimmt werden. Weitere Faktoren, die einen Einfluss auf die Messunsicherheit haben sind Probenaufbereitung,

schwankende Sensitivität des Analysegerätes sowie die Repräsentativität des exakten Ortes der Probenahme. Der Einfluss dieser Faktoren wurden berücksichtigt, um die gesamte Messunsicherheit der eingesetzten Analysemethode zu bestimmen.

Ergänzend zur Bestimmung von TWP in der gesamten atmosphärischen Deposition mittels Py-GC-MS wurden die gesammelten Trockendepositionsproben mittels Rasterelektronenmikroskopie und energiedispersiver Röntgenspektroskopie (SEM/EDX) analysiert und die Menge Reifenabrieb bestimmt, die bei trockener Witterung durch Sedimentation von luftgetragenen Partikeln in die Umwelt eingetragen wird. SEM/EDX ist in der wissenschaftlichen Literatur bereits etabliert. Da die Methode darauf basiert, dass die Proben direkt auf Oberflächen gesammelt werden, welche sich für die Elektronenmikroskopie eignen, kann sie nicht ohne weiteres für die Proben der atmosphärischen Gesamtdosition eingesetzt werden. Mit SEM/EDX wurden einzelne TRWP im Grössenbereich von 1 bis 80 μm basierend auf deren charakteristischen Morphologie und Elementzusammensetzung identifiziert und deren Masse berechnet. Da TRWP je etwa zur Hälfte aus Reifenmaterial und Strassenmaterial bestehen, kann daraus die Masse von TWP abgeleitet werden.

Die Ergebnisse zeigen, dass TWP in allen atmosphärischen Depositionsproben von allen Standorten nachweisbar sind. Die Höhe des atmosphärischen Eintrags (Massendepositionsrates) hängt stark von der Nähe zum Verkehr ab: Die höchsten Werte wurden am Verkehrsstandort Dübendorf-Überlandstrasse ($\sim 700 \mu\text{g}/\text{m}^2/\text{Tag}$) gemessen, gefolgt von den städtischen und vorstädtischen Standorten Zürich-Kaserne und Dübendorf-Empa ($\sim 500\text{--}600 \mu\text{g}/\text{m}^2/\text{Tag}$). Tieferer Massendepositionsrates von TWP wurden an den beiden ländlichen Standorten ermittelt ($\sim 150\text{--}250 \mu\text{g}/\text{m}^2/\text{Tag}$), die niedrigsten Werte ergaben sich aus den Messungen der Proben vom ländlichen und erhöht gelegenen Standort Chaumont ($\sim 140 \mu\text{g}/\text{m}^2/\text{Tag}$). Die beobachtete Abhängigkeit des atmosphärischen Eintrags von Reifenabrieb von der Nähe zum Verkehr ist zu erwarten. Grössere Partikel haben eine höhere Sinkgeschwindigkeit und werden dadurch verstärkt in der näheren Umgebung des Entstehungsortes in Böden und Gewässer eingetragen. Unter bestimmten meteorologischen Bedingungen (z.B. bei Turbulenz) können jedoch auch grössere Partikel länger in der Luft verweilen und über weite Strecken transportiert werden.

Die Messungen an den verschiedenen Standorten wurden verwendet, um den gesamten jährlichen Eintrag von Reifenabrieb in der Schweiz abzuschätzen. Dabei wurde angenommen, dass die ermittelten Massendepositionsrates von TWP repräsentativ für andere Standorte in der Schweiz mit ähnlicher Landnutzung sind. Da keine Messungen an einem Standort im alpinen Raum vorliegen, wurde die Abschätzung für die Gebiete der Schweiz unterhalb von 2'000 m ü. M. durchgeführt. Der auf diese Weise ermittelte gesamte jährliche atmosphärische Eintrag von Reifenabrieb in der Schweiz (unterhalb von 2'000 m ü. M.) beträgt 2'410 Tonnen. Dies entspricht etwa 24–27 % des gesamten Reifenabriebs, der jährlich auf den Strassen in der Schweiz erzeugt wird. Dieser Anteil des atmosphärischen Eintrags an der gesamten erzeugten Menge TWP liegt deutlich über den in vielen Studien gemachten Annahmen. Die Ergebnisse dieser Studie zeigen, dass die Rolle der Atmosphäre bei der Verteilung von Reifenabrieb in der Umwelt wichtig ist.

Die Messungen am verkehrsbelasteten Standort Dübendorf-Überlandstrasse zeigen, dass in unmittelbarer Nähe von stark befahrenen Strassen der Eintrag von Reifenabrieb erhöht ist. Dieser erhöhte Eintrag ist auf einen Beitrag des lokalen Verkehrs direkt beim Standort zurückzuführen. Solche lokalen Beiträge an Orten in der unmittelbaren Nähe stark befahrener Strassen sind zusätzlich und sind in der zuvor erwähnten Abschätzung für die Schweiz nicht enthalten.

Die Hochrechnung der mit SEM/EDX analysierten Proben ergibt für die Gebiete der Schweiz unterhalb von 2'000 m ü. M. einen atmosphärischen Eintrag von Reifenabrieb durch Trockendeposition von 1'417 Tonnen pro Jahr. Die Aufteilung des Eintrags durch Trocken- und Nassdeposition hängt von den klimatischen Bedingungen ab. In der Schweiz tragen Trockendeposition etwa 60 % und Nassdeposition etwa 40 % zum atmosphärischen Eintrag von Reifenabrieb in die Umwelt bei.

Ein Vergleich mit den Resultaten des Partnerprojektes zum Eintrag der anderen Formen von Mikroplastik zeigt, dass der atmosphärische Eintrag von TWP etwa zehn Mal höher ist als der atmosphärische Eintrag der anderen Formen von Mikroplastik. In der Schweiz ist Reifenabrieb daher die dominierende Quelle von Mikroplastik, der über die Luft in die Umwelt gelangt.

Abschliessend ist festzuhalten, dass diese Studie auch wichtige Einschränkungen aufweist, aus denen sich zugleich Empfehlungen für die zukünftige Forschung auf diesem Gebiet ableiten lassen. Die Anzahl der Standorte an denen Proben gesammelt wurden war aufgrund der Komplexität und der Kosten der angewandten Analysemethoden begrenzt. Dies führt zu Unsicherheiten bei den Berechnungen des gesamten atmosphärischen Eintrags von Reifenabrieb in der Schweiz, die nicht vollständig quantifiziert werden können. Eine Ausweitung der räumlichen Abdeckung durch Einbeziehung einer grösseren Anzahl von Standorten würde die Unsicherheit reduzieren.

Für eine räumlich höhere Messdichte sowie für die Erfassung der zeitlichen Entwicklung des atmosphärischen Eintrags von Reifenabrieb im Rahmen eines Monitorings sind allerdings kostengünstigere und weniger arbeitsintensive Analysemethoden erforderlich. Derzeit sind solche Analysemethoden nicht verfügbar. Ein möglicher Ansatz für die einfache und kostengünstige Bestimmung von Reifenabrieb in Umweltproben basiert auf der Bildanalyse mittels Digitalmikroskopie. Vorläufige Untersuchungen mit dieser Methode sind vielversprechend, bedürfen jedoch noch einer gründlichen Überprüfung mithilfe chemischer Methoden. Die Weiterentwicklung solcher Analysemethoden und deren Anwendungen in einem sorgfältig geplanten Monitoringprogramm würden zu präziseren Expositionsabschätzungen beitragen und die Grundlage für evidenzbasierte Strategien zur Reduzierung der Umweltauswirkungen des Reifenabriebs schaffen.

Résumé

Les particules issues de l'usure des pneus et de la route (en anglais tire and road wear particles, ou TRWP) sont générées par les forces de frottement entre les pneus et la route. Le frottement entre les pneus et la route étant indispensable au transport routier, il est impossible d'empêcher totalement leur formation. Les TRWP sont constituées d'un mélange de matériaux provenant des bandes de roulement des pneus et de la surface de la route, principalement de l'asphalte. L'asphalte est lui-même un mélange de bitume et de composants minéraux.

La part de matériau provenant des pneus dans les TRWP est désignée sous le nom de particules d'usure des pneus (en anglais tire wear particles, ou TWP). Les TWP sont composées principalement de caoutchouc naturel et synthétique vulcanisé mais aussi d'une série d'autres substances dont certaines sont toxiques, telles que des plastifiants, des agents de vulcanisation et d'autres additifs. Certaines de ces substances sont solubles dans l'eau et peuvent être séparées des particules de pneus dans l'environnement, avec de potentiels impacts sur les organismes et se retrouver dans notre alimentation. Les TWP constituent une catégorie de particules qui contribuent à la pollution et à la dégradation de l'environnement. Chaque année, environ 8'800 à 10'100 tonnes de TWP sont générées sur les routes suisses.

La majeure partie de la masse des TRWP formées sur les routes est constituée de particules d'un diamètre supérieur à 10 µm; seule une petite partie de la masse des TRWP est constituée de particules plus petites. Si les TRWP contribuent donc à la concentration massique des particules fines inhalables (PM₁₀), la part de l'usure des pneus (TWP) dans les PM₁₀ reste toutefois faible sur les sites non directement exposés au trafic. Même sur les sites fortement exposés au trafic, la part des TWP dans les PM₁₀ est inférieure à 10 %.

En raison de leur composition à base de polymères, les TWP sont souvent classés comme microplastiques (MP), bien qu'il s'agisse chimiquement d'élastomères et non de polymères thermoplastiques. L'usure des pneus est l'une des principales sources de microplastiques dans l'environnement en Suisse et dans le monde. Malgré l'importance de l'usure des pneus en tant que polluant, de grandes incertitudes subsistent actuellement quant à la quantité de particules d'usure des pneus qui se retrouvent dans l'environnement. En particulier, on ignore encore largement quelles quantités de particules d'usure des pneus sont rejetées dans l'air, dispersées par le vent et finissent par être déposées dans l'environnement à une grande distance de leur lieu d'origine.

La plupart des études sur la répartition des particules d'usure des pneus dans l'environnement reposent sur des analyses de flux de matières, dans lesquelles les estimations des émissions sont combinées à des hypothèses concernant le transport et la persistance de ces particules dans les milieux environnementaux tels que les sols, les eaux de surface et les sédiments. Ces analyses partent souvent du principe que seule une petite partie des particules des TRWP produites est rejetée et dispersée dans l'air. On suppose souvent que seules les particules d'un diamètre inférieur à 10 µm peuvent

rester en suspension dans l'air pendant une longue période et que, par conséquent, seulement environ 1 à 5 % de l'ensemble des particules d'usure des pneus sont dispersés sur de longues distances par l'atmosphère. Ces hypothèses constituent toutefois une forte simplification qui n'a jusqu'à présent guère été vérifiée par des mesures. Cela s'explique principalement par les difficultés techniques liées à l'identification et à la quantification des TWP dans les échantillons prélevés dans l'environnement.

L'objectif de cette étude était de contribuer à une meilleure compréhension quantitative de l'apport atmosphérique de particules de pneus et de déterminer la quantité de TWP rejetée dans l'environnement par voie atmosphérique dans différents types de sites (urbains, périurbains, ruraux). Pour atteindre cet objectif, l'apport atmosphérique annuel (également appelé dépôts atmosphérique) de particules d'usure des pneus a été déterminé sur six sites en Suisse présentant différentes utilisations des sols. À cette fin, des échantillons ont été prélevés toutes les quatre semaines pendant un an à ces stations à l'aide de différents collecteurs passifs. Des échantillons de dépôt sec, c'est-à-dire les apports atmosphériques par temps sec, ainsi que de dépôt total, c'est-à-dire la somme des apports atmosphériques par précipitations (dépôt humide) et par dépôt sec, ont été collectés. Dans le cadre d'un projet partenaire soutenu par l'Office fédéral de l'environnement visant à déterminer les apports atmosphériques d'autres formes de microplastiques que les particules issues de l'usure des pneus, des échantillons distincts de dépôts secs et humides ont été prélevés simultanément sur les mêmes sites. Ces échantillons ont également été analysés à l'aide de méthodes d'analyse d'images.

Les sites sélectionnés étaient un site exposé au trafic en périphérie urbaine (Dübendorf-Überlandstrasse, à 20 m du bord d'une route où circulent 19'000 véhicules par jour), un site urbain (Zurich-Kaserne), un site en périphérie urbaine (Dübendorf-Empa), deux sites ruraux situés respectivement au nord (Payerne-MétéoSuisse) et au sud des Alpes (Magadino-Cadenazzo), ainsi qu'un site rural situé à plus de 1 100 m d'altitude (Chaumont). Tous ces sites, à l'exception du site exposé au trafic de Dübendorf-Überlandstrasse, font partie du Réseau national d'observation des polluants atmosphériques (NABEL).

L'un des axes principaux de cette étude était la détermination de l'usure des pneus dans le dépôt atmosphérique total. En l'absence de méthode universellement applicable et satisfaisante pour la détermination de l'usure des pneus dans les échantillons, une méthode d'analyse basée sur la masse, reposant sur la Pyrolyse-Chromatographie en phase Gazeuse-Spectrométrie de Masse (Py-GC-MS), a été développée et mise en œuvre dans le cadre de cette étude. Cette méthode repose sur la décomposition thermique (pyrolyse) de l'échantillon, suivie de la séparation des produits de pyrolyse dans le chromatographe en phase gazeuse, puis de leur identification dans le spectromètre de masse. Les méthodes existantes de détermination des TWP par Py-GC-MS reposent souvent sur un ou deux produits de pyrolyse spécifiques à la décomposition thermique des pneus (marqueurs spécifiques). Ces méthodes sont associées à des incertitudes considérables. La méthode développée ici utilise six produits de pyrolyse caractéristiques des types de caoutchouc utilisés dans la fabrication des pneus (caoutchouc naturel, caoutchouc styrène-butadiène et caoutchouc butadiène). Le signal total des six produits de pyrolyse permet de déterminer la masse totale du matériau des pneus dans un échantillon. L'étalonnage de la méthode d'analyse a été réalisé à partir de plus de

vingt échantillons cryo-broyés, provenant de pneus usagés de voitures et de poids lourds. Une forte relation linéaire a été mise en évidence entre le signal cumulé des six produits de pyrolyse et la masse des pneus. Malgré les différences entre les différents types de pneus, la masse des pneus peut être déterminée avec une incertitude comprise entre environ -20% et $+33\%$. D'autres facteurs influent sur l'incertitude de mesure, notamment la préparation des échantillons, la sensibilité variable de l'appareil d'analyse ainsi que la représentativité du lieu exact de prélèvement. L'influence de ces facteurs a été prise en compte pour déterminer l'incertitude de mesure totale de la méthode d'analyse utilisée.

En complément de la détermination du TWP dans l'ensemble des dépôts atmosphériques par Py-GC-MS, les échantillons de dépôts secs collectés ont été analysés par microscopie électronique à balayage et spectroscopie à rayons X à dispersion d'énergie (SEM/EDX) et la quantité de résidus de pneus rejetée dans l'environnement par sédimentation de particules en suspension dans l'air par temps sec a été déterminée. La technique SEM/EDX est déjà bien établie dans la littérature scientifique. Comme cette méthode repose sur le prélèvement direct des échantillons sur des surfaces adaptées à la microscopie électronique, elle ne peut pas être appliquée directement aux échantillons de dépôt atmosphérique total. À l'aide de la technique SEM/EDX, des particules individuelles de TRWP d'une taille comprise entre 1 et 80 μm ont été identifiées sur la base de leur morphologie caractéristique et de leur composition élémentaire, et leur masse a été calculée. Les TRWP étant composés pour moitié environ de matériaux provenant des pneus et pour moitié de matériaux routiers, il est possible d'en déduire la masse des TWP.

Les résultats montrent que les TWP sont détectables dans tous les échantillons et tous les sites. L'importance de l'apport atmosphérique (taux de dépôt massique) dépend fortement de la proximité du trafic : les valeurs les plus élevées ont été mesurées sur le site routier de Dübendorf-Überlandstrasse ($\sim 700\text{ }\mu\text{g}/\text{m}^2/\text{jour}$), suivi des sites urbains et périurbains de Zurich-Kaserne et Dübendorf-Empa ($\sim 500\text{--}600\text{ }\mu\text{g}/\text{m}^2/\text{jour}$). Des taux de dépôt massique de TWP plus faibles ont été déterminés sur les deux sites ruraux ($\sim 150\text{--}250\text{ }\mu\text{g}/\text{m}^2/\text{jour}$), les valeurs les plus basses ayant été obtenues à partir des mesures effectuées sur les échantillons prélevés sur le site rural et surélevé de Chaumont ($\sim 140\text{ }\mu\text{g}/\text{m}^2/\text{jour}$). Comme prévu, l'apport atmosphérique de particules issues de l'usure des pneus est corrélé avec la proximité du trafic. Les particules de plus grande taille ont une vitesse de chute plus élevée et sont donc davantage déposées dans les sols et les cours d'eau à proximité immédiate de leur lieu d'origine. Dans certaines conditions météorologiques (par exemple en cas de turbulence), les particules de plus grande taille peuvent toutefois rester plus longtemps en suspension dans l'air et être transportées sur de longues distances.

Les mesures effectuées sur les différents sites ont servi à estimer l'apport annuel total issu de l'usure des pneus en Suisse. On a supposé que les taux de dépôt massique de TWP déterminés étaient représentatifs d'autres sites en Suisse présentant une utilisation des sols similaire. En l'absence de mesures sur un site situé dans la région alpine, l'estimation a été réalisée pour les régions de Suisse situées en dessous de 2'000 m d'altitude. L'apport atmosphérique annuel total de particules de pneus en Suisse (en dessous de 2'000 m), ainsi déterminé, s'élève à 2'410 tonnes. Cela correspond à

environ 24 à 27 % de l'usure totale des pneus générée chaque année sur les routes suisses. Cette part des apports atmosphériques par rapport à la quantité totale de TWP produite est nettement supérieure aux hypothèses formulées dans de nombreuses études. Les résultats de cette étude montrent que l'atmosphère joue un rôle important dans la dispersion des particules d'usure des pneus dans l'environnement.

Les mesures effectuées sur le site très fréquenté de Dübendorf-Überlandstrasse montrent que, à proximité immédiate de routes très fréquentées, l'apport de poussière de pneus est accru. Cet apport accru est dû à la contribution du trafic local à proximité immédiate du site. Ces contributions locales, observées à proximité immédiate de routes très fréquentées, s'ajoutent aux autres sources et ne sont pas prises en compte dans l'estimation pour la Suisse mentionnée précédemment.

L'extrapolation des échantillons analysés par SEM/EDX donne, pour les régions de Suisse situées en dessous de 2'000 m, un apport atmosphérique de particules d'usure des pneus par dépôt sec de 1'417 tonnes par an. La répartition de cet apport entre dépôt sec et dépôt humide dépend des conditions climatiques. En Suisse, le dépôt sec contribue à hauteur d'environ 60 % et le dépôt humide à hauteur d'environ 40 % à l'apport atmosphérique de particules issues de l'usure des pneus dans l'environnement.

Une comparaison avec les résultats du projet partenaire portant sur l'apport des autres sortes de microplastiques montre que l'apport atmosphérique de TWP est environ dix fois supérieur à celui des autres formes de microplastiques. En Suisse, l'usure des pneus est donc la principale source de microplastiques qui se retrouvent dans l'atmosphère.

Il convient enfin de noter que cette étude présente également d'importantes limites, dont on peut toutefois tirer des recommandations pour les futures recherches dans ce domaine. Le nombre de sites sur lesquels des échantillons ont été prélevés était limité en raison de la complexité et du coût des méthodes d'analyse utilisées. Cela entraîne des incertitudes dans les calculs de l'apport atmosphérique total issu de l'usure des pneus en Suisse, qui ne peuvent être entièrement quantifiées. Une extension de la couverture géographique par l'intégration d'un plus grand nombre de sites permettrait de réduire cette incertitude. Toutefois, pour obtenir des mesures plus denses sur le plan géographique et pour suivre l'évolution dans le temps des apports atmosphériques de poussière de pneus (surveillance), il est nécessaire de disposer de méthodes d'analyse moins coûteuses et moins exigeantes en main-d'œuvre. À l'heure actuelle, de telles méthodes d'analyse ne sont pas disponibles. Une éventuelle méthode simple pourrait consister en une analyse d'images d'échantillons à l'aide d'un microscope numérique. Des études préliminaires sont prometteuses, mais doivent encore faire l'objet d'une vérification par des méthodes chimiques. Le perfectionnement des méthodes d'analyse et leur mise en œuvre dans le cadre d'un programme de surveillance bien conçu contribueraient à affiner les estimations d'exposition et à élaborer des stratégies fondées sur des données probantes visant à réduire l'impact environnemental des particules issues de l'usure des pneus.

1 Introduction

Frictional forces between the tires and the road surface cause abrasion on both the tire tread and the road surface during driving. This leads to the formation and release of tire and road wear particles (TRWP), consisting of a mixture of aggregated tire and road wear. The term tire wear particles (TWP) is often used to refer to the proportion of tire wear in TRWP. Tire wear and road wear each account for about 50% to the mass of TRWP.^[1] Tire wear consists of roughly 40-50% polymer by weight, i.e., natural rubber (NR, polyisoprene) and synthetic rubber, mainly styrene butadiene rubber (SBR) and butadiene rubber (BR).^[1] The content of synthetic rubber in tire wear ranges between 85% of total rubber in passenger cars and 20% of total rubber in tires for trucks. Therefore, TWP are generally classified as microplastics (MP)^[2,3], although rubber is not a plastic, but an elastic polymer material. An estimated amount of 3 to 6 million tons of tire wear are released into the environment annually, making tire wear an important source of microplastics.^[4,5] Besides rubber (natural and synthetic polymers), TWP contains additional compounds including fillers, softeners, vulcanization agents, and additives. Leaching of these compounds over time releases heavy metals and organic pollutants that are harmful to ecosystems and to human health.^[5-7]

In a recent study by Ospital et al. (2025), emission factors (EFs) for different vehicle categories have been reviewed and updated.^[8] In addition to those for other vehicle categories, average emission factors were determined for passenger cars, buses, and heavy-duty vehicles. These are specified as 87 mg/km, 374 mg/km and 801 mg/km, respectively. In the same study, the total annual TWP production on Swiss roads was calculated using a bottom-up method, i.e. by combining EFs and total traffic activity in Switzerland. This approach resulted in total annual TWP generation and emissions between 8'837 and 9'721 tons/year (reference year 2019). Steiner (2023) used a similar bottom-up method to calculate the annual TWP emissions in Switzerland and obtained a value of 10'100 tons/year (no reference year specified).^[9] The study by Sieber et al (2020) used a top-down method to estimate annual TWP emissions.^[10] By using tire sales statistics and average tire weight loss over its lifespan, the amount of tire rubber generated annually on Swiss roads was calculated to 10'600 tons/year (reference year 2018). With a rubber content in tire tread of 50%, this corresponds to annual TWP emissions of 21'200 tons/year. The same annual TWP emissions were calculated for Austria^[11] also by applying a top-down method, which means similar annual per capita emissions in Switzerland (2.5 kg/ year) and in Austria (2.4 kg/year). These per capita emissions appear to be higher than those obtained for bottom-up methods applied for Switzerland and other European countries, where per capita emissions of tire wear between 0.9 and 2.5 kg/year were found.^[8,12]

TRWP are predominantly particles with diameters larger than 10 μm , therefore most of the mass of TWP is associated with particles that are too large to penetrate deeply into the human respiratory tract. A literature review by Saladin et al. (2024) found for light-duty vehicles emission factors of TWP smaller than 10 μm (PM₁₀) of 2.7 mg/km, the TWP EFs of PM₁₀ reported in Ospital et al. (2025) are for passenger cars, buses and heavy-duty vehicles 6.7 mg/km, 30.2 mg/km and 42.1 mg/km, respectively. Under certain driving conditions and tire slip, ultrafine TWP (diameter below 0.1 μm) can be

formed and released through evaporation and condensation of specific compounds in the tire tread material.^[13] The generation of tire and road wear particles smaller than 10 µm mean that TWP also contribute to PM₁₀ in ambient air. The mass contribution of TWP to PM₁₀ is typically small and even at locations next to high traffic activities below 10 %.^[14]

The TWP generated and emitted on roads are released into the environment. Our understanding of the distribution of TWP in the environment is based primarily on mass flow analyses (MFA).^[7,10,11] MFA relies on data and assumptions about the fractions of TWP emitted that enter the various environmental compartments via different routes, such as direct runoff from roads, as effluents from sewer systems and by atmospheric transport and deposition. The amount of TWP transported through the atmosphere and deposited further away from roads is usually calculated on the assumption that only a small percentage of all emitted TWP, typically the fraction of TWP smaller than 10 µm, can become airborne. Accordingly, MFA studies predict that only a small proportion of the total TWP (about 1 to 5%) is transported over greater distances through the atmosphere and released into the environment.^[7,10,14–16]

A Swiss study on the distribution of tire wear in the environment should particularly be mentioned here.^[9] In this study by Steiner (2023) the input of tire wear into surface waters, runoff into roadside soils, and the amount of TWP retained in sewers was estimated. Although the study did not explicitly examine input through atmospheric deposition, some conclusions regarding airborne distribution can be drawn from the difference between the annually generated amount of tire wear and the sum of the estimated material flows (see section 4.5).

It is currently not possible to verify the estimations of atmospheric transport and deposition available in the literature, as there are hardly any measurements of total TWP in atmospheric samples from locations at some distance to roads. One reason for the few measurements of TRWP or TWP in environmental matrices is that the quantification of TRWP and/or the associated TWP in environmental matrices is analytically challenging. This is due to i) the variability in chemical composition of tires between different brands and ii) further organic and inorganic substances present in environmental matrices that interfere with the target analytes.^[16,17] Current analytical methods to quantify TRWP can be categorized into two classes: particle-based and mass-based quantification methods.^[18,19] Particle-based methods mainly rely on characteristic features of TRWP (e.g., color, morphology, and elemental composition), which are detected using optical or electron microscopy approaches.^[5,20,21] Other particle-based micro-spectroscopy approaches commonly used in microplastic particle analysis, such as Fourier-transform infrared (FTIR) and Raman micro-spectroscopy, are less suitable for analyzing TRWP because of their strong absorption of visible and IR light due to the presence of carbon black as a filler. Mass-based quantification methods such as pyrolysis gas chromatography-mass spectrometry (Py-GC-MS) rely on the detection of selected markers that are characteristic for tire wear. Ideal markers are specific to tire wear, are chemically stable over time, and remain within the tire material. Since TWP in the environment are present as a mixture of tires of different compositions, a suitable marker should also be representative for that mixture. Zinc (Zn), for example, was used as a marker, but because of the ubiquity of Zn in the environment, its suitability as a marker for TWP has been questioned.^[22] Specific organic compounds (additives such as oxidation retardants, accelerators, vulcanizing agents, and plasticizers) were also used as markers for the quantification of TWP in environmental samples.

However, the use of these markers is challenging as they leach from the tires or are chemically transformed over time, as for example documented for N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine (6-PPD) transforming to N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine-quinone (6-PPD-Q).^[18,23,24] The rubber materials used in tires, however, are chemically stable over the lifetime of a tire, and therefore the gaseous compounds formed during the pyrolysis of rubber (i.e., NR, SBR, and BR) and fragmented during ionization represent promising markers for TWP.^[25]

The aim of this study was to determine the amount of atmospheric deposition of TWP at locations in Switzerland with different proximity to roads and representing different land use situations, thereby closing an important knowledge gap regarding the role of atmospheric transport in the distribution of tire wear in the environment. To achieve this goal, it was necessary to develop an analytical method that allows the quantification of tire wear in total atmospheric deposition, i.e. tire wear that is removed from the atmosphere by precipitation (wet deposition) and by sedimentation under dry meteorological conditions (dry deposition). The sum of wet and dry depositions is often referred to as total or bulk deposition.

The analytical method developed and applied in this study is mass-based and relies on Py-GC-MS and the determination of markers that are formed during the pyrolysis of tire rubber. In parallel, dry atmospheric deposition of tire wear was measured using a particle-based method (scanning electron microscopy with energy-dispersive X-ray spectroscopy, SEM/EDX), a measurement technique that cannot readily be used for the analysis of wet atmospheric or bulk deposition but has already been used in several studies to determine tire wear in dry atmospheric deposition. Based on one-year measurements of wet and dry atmospheric deposition of tire wear at multiple sites in Switzerland, the total annual atmospheric input of tire wear into the environment in Switzerland is estimated.

In a parallel project, funded by the Swiss Federal Office for the Environment (FOEN), the atmospheric deposition of microplastics without TWP was determined.^[26,27] The sampling in both projects was synchronized, and the results of both projects are briefly compared to discuss the results on atmospheric TWP deposition in the broader context of the overall atmospheric input of microplastics.

2 Measurements

2.1 Collection of atmospheric deposition

Atmospheric deposition samples were collected for one year at five sites of the Swiss national air pollution monitoring network NABEL^[28] and an additional roadside site in Dübendorf (Dübendorf-Überlandstrasse, 20 m from the edge of a road with daily traffic volume of 19'000 vehicles per day). The sampling locations were selected to represent different types of land use and varying distances from roads and populated areas. Table 1 gives a short description of the selected sampling sites and Figure 1 shows a map of Switzerland with the location of the sampling sites. Aerial photographs of the sites and their surroundings are shown in Figure 2. Different analytical methods were applied to determine tire wear, with separate passive samplers being used for each method. Figure 3 shows a photograph of the sampling site Magadino-Cadenazzo and the passive samplers used. The different passive samplers are described in the following subsections.

2.1.1 Bulk deposition sampler

Bulk deposition, i.e. the sum of wet and dry deposition, was collected passively using cylindrical borosilicate glass containers (internal diameter 9.2 cm, volume of 1.5 L, Micasa in Dübendorf, Switzerland). The sample collectors were installed on poles 1.5 m above the ground, a free and unobstructed air flow to the collectors was ensured. Sampling took place on a four-weekly basis, the volume of the containers was sufficient to collect all precipitation. Design of the samplers and operation of bulk deposition sampling was in accordance with a method frequently used in monitoring networks for the determination of total atmospheric dust deposition.^[29] Bulk deposition sampling was conducted at all the sampling sites from mid of June 2024 to mid of June 2025 (see Table A 1). The collected bulk deposition samples were analysed for TWP using an analytical method based on pyrolysis gas chromatography-mass spectrometry as described in section 3.1.

Sampling sites		
Site name	Abbreviation	Site type
Dübendorf-Überlandstrasse	DUET	Suburban, traffic (19'000 vehicles/day)
Zürich-Kaserne	ZUE	Urban (background, park-like courtyard)
Dübendorf-Empa	DUE	Suburban
Magadino-Cadenazzo	MAG	Rural (south of Alps)
Payerne-MeteoSwiss	PAY	Rural (north of Alps)
Chaumont	CHA	Mountainous, 1'140 m asl.

Table 1: Sites selected for sampling four-weekly atmospheric deposition as part of this study.

2.1.2 Dry deposition sampler

Atmospheric deposition, i.e. airborne particulate matter that is removed from the atmosphere by gravity under dry weather conditions was sampled using Sigma-2 passive

samplers, which are designed to minimize air turbulence in the interior of the sampler while allowing accurate and consistent particle capture for particles with aerodynamic diameters ranging from $1\text{ }\mu\text{m}$ to $80\text{ }\mu\text{m}$.^[21,30] The samplers therefore capture inhalable fine (diameter between $1\text{ }\mu\text{m}$ and $2.5\text{ }\mu\text{m}$) and coarse (diameter between $2.5\text{ }\mu\text{m}$ and $10\text{ }\mu\text{m}$) particles as well as larger particles frequently referred to as super-coarse (diameter between $10\text{ }\mu\text{m}$ and $62.5\text{ }\mu\text{m}$) and giant particles (diameter $> 62.5\text{ }\mu\text{m}$).^[31] The particles adhere to a boron substrate (prepared by Particle Vision GmbH), which remained exposed for four-weekly periods, each. The Sigma-2 samplers were positioned at a height of 1.5–2.0 m above ground level.

Sampling with the Sigma-2 passive samplers was conducted from end of September/beginning of October 2023 to end of September/beginning of October 2024 on a four-weekly basis covering a complete year at five sites (Table A 1). Dry deposition sampling at DUET, the site in Duebendorf next to a busy road was started later (23.05.2024) and dry deposition samples are therefore only available for a shorter period.

2.1.3 Wet and dry deposition sampler

Wet and dry atmospheric deposition were collected separately using a dedicated passive sampler (Nesa Srl., Italy). A 2 L glass dish (Duran Crystallizing Dish, DWK Life Sciences, Germany) was placed at the (closed) bottom of the cylinder labelled "dry", to collect gravitationally depositing particles. The bottom end of the cylinder labelled "wet" has a funnel and a copper tube, which guides the precipitation to a custom-made filtration unit made of anodized aluminum installed at the end of the tube. The filtration device was fitted with a stainless-steel mesh (Haver & Boecker, Germany) of diameter 47 mm and mesh size $15\text{ }\mu\text{m}$ to collect particles $> 15\text{ }\mu\text{m}$ present in rainwater or snow (that melted over time).

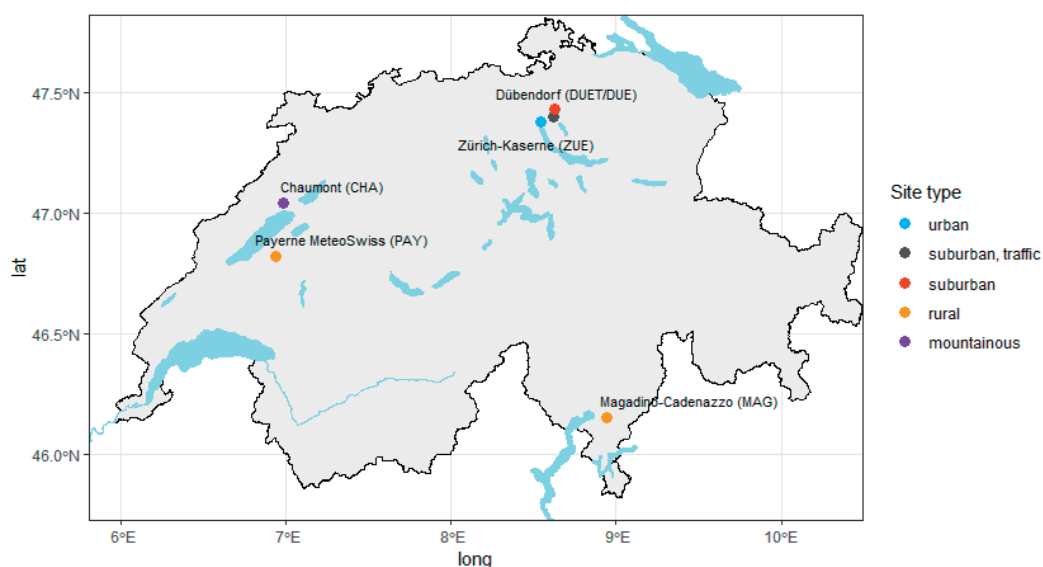


Figure 1: Map of Switzerland and the location of the sites where atmospheric deposition was collected (data source: Open-StreetMap).

These samples were primarily used for the partner project on atmospheric deposition of microplastics other than tire wear.^[26] However, in this project, these samples were additionally examined for tire wear using optical microscopy images in combination with image analysis techniques. The height of the wet and dry samplers were also about 1.5 meters above the ground. Wet and dry deposition sampling was conducted at all sites except DUET for the same four-weekly periods as bulk deposition sampling (mid of June 2024 to mid of June 2025, see Table A 1 in the Annex).



Figure 2: Aerial photographs (1 km x 1 km) of the sampling locations and their surroundings. The white arrows indicate the position of the sampling points (DUE: Dübendorf-Empa, DUET: Dübendorf-Überlandstrasse, ZUE: Zürich-Kaserne, PAY: Payerne-MeteoSwiss, MAG: Magadino-Cadenazzo, CHA: Chaumont). Source: <https://www.swisstopo.admin.ch/de/orthobilder-swissimage-10-cm>.

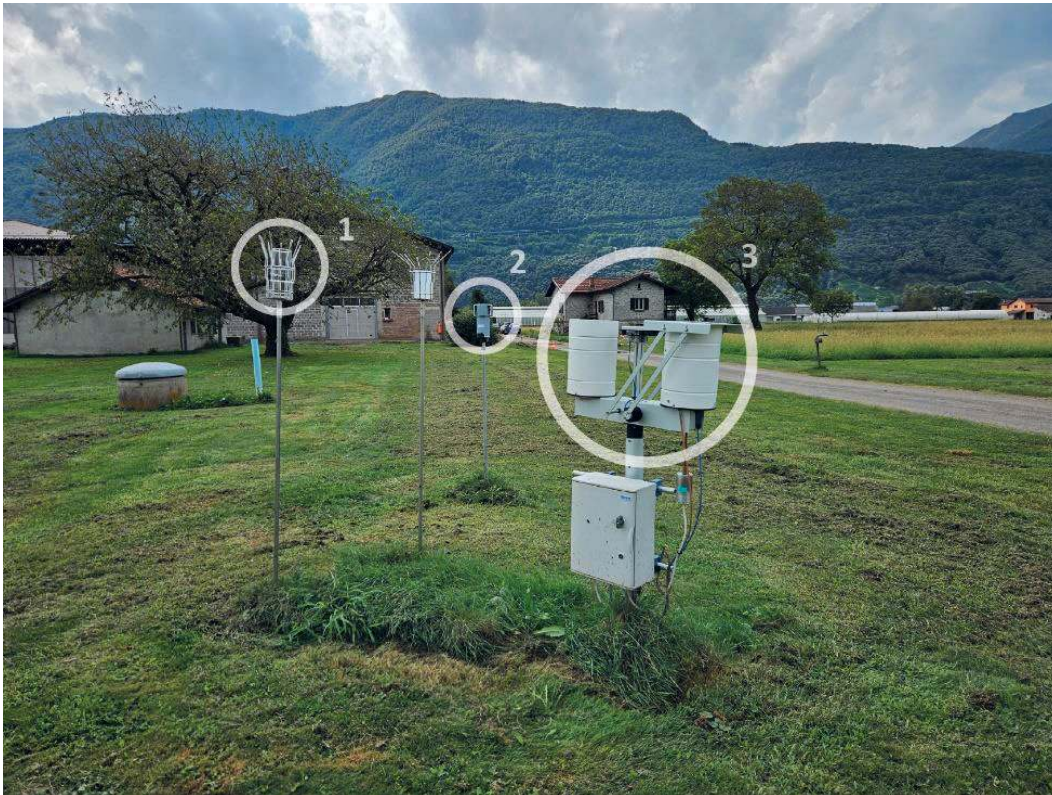


Figure 3: Photograph of the sampling site Magadino-Cadenazzo, a rural site south of the Alps. The passive samplers used in this study are also shown. 1: Glas flask for bulk (wet and dry) deposition sampling, 2: Sigma-2 sampler for dry deposition sampling, and 3: Wet & dry sampler for separate collection of wet and dry atmospheric deposition.

3 Analytical Methods

3.1 Pyrolysis-GC-MS

Thermoanalytical methods, such as pyrolysis gas chromatography-mass spectrometry and thermal extraction and desorption gas chromatography mass spectroscopy (TED GC-MS), have been established as leading techniques to quantify TWP in complex (environmental) matrices.^[32,33] In Py-GC-MS, TWP are thermally decomposed, and the volatile products of thermal rubber decomposition are chromatographically separated, allowing their identification and quantification after ionization using a mass spectrometer.^[34,35] To limit interferences between the target compounds and pyrolysis products from other organic matter and for signal enhancement or reduction due to the presence of inorganic materials, Py-GC-MS measurements require a sample-specific pre-treatment procedure.^[36] Furthermore, chromatography parameters need to be optimized to improve the signal-to-noise ratio and to prevent saturation of the mass detector of the GC-MS system.^[37,38] Various Py-GC-MS methods have been applied to quantify TWP by detecting pyrolysis products from rubber components.^[32,35,39,40] The main types of rubber used in tires are NR, SBR, and BR, with tires for passenger cars and light-duty vehicles containing more synthetic rubber (SBR and BR) than natural rubber, while NR dominates in tires for heavy-duty vehicles (e.g. trucks and buses).^[32] However, detailed quantitative information on the proportions of these main types of rubber in the various tire types are currently lacking. Nevertheless, the thermal decomposition of NR, SBR, and BR yields butadiene, isoprene, 4-vinylcyclohexene (VCH), limonene (or dipentene, DP), 4-phenylcyclohexene (PCH), styrene, and α -methylstyrene as the dominant pyrolysis products.^[35,41] However, since the proportions of NR, SBR, and BR are tire (brand) specific, the abundance of the respective pyrolysis products varies greatly with tire type and tire brand. Thus, the quantification of TWP in environmental samples based on measurement of a single pyrolytic compound is very risky and has recently been challenged.^[17,18] Among the available Py-GC-MS methods for determining TWP in environmental samples, those described in the technical specifications ISO/TS 20593:2017 and ISO/TS 21396:2017 are most widely used.^[42,43] However, both methods come with severe limitations as they both exclusively rely on VCH and limonene as markers for synthetic rubber (SB+SBR) and natural rubber, respectively. The proportions of synthetic and natural rubber differ significantly for different tire types, so that the relationship between VCH or limonene and the mass of tire material varies considerably for different tire types. This leads to very large measurement uncertainties with both methods.^[17]

In this project, we, therefore, developed a novel Py-GC-MS method that includes the signals of the six most abundant characteristic markers resulting from the thermal decomposition of NR, SBR, and BR. The linear relationship between the sum of the integrated signals of the six pyrolytic compounds and the mass of tire tread, established for a selection of different tires, allowed us to quantify TWP in complex (environmental) samples. We developed a targeted sample preprocessing to minimize possible interferences between the target compounds and the environmental matrix.

3.1.1 Materials and methods

3.1.1.1 Chemicals and materials

Cis-polybutadiene (butadiene rubber, BR) was purchased from Sigma-Aldrich (United States of America, USA). Polyisoprene (natural rubber, NR) and styrene-butadiene rubber (SBR-1500) were purchased from Polymer Source, Inc. (Dorval, Canada). Tetrahydrofuran (THF) and acetone were purchased from Merck (Buchs, Switzerland), and di-chloromethane (DCM) was purchased from Acros Organics/Thermo Scientific Chemicals (Zürich, Switzerland). All these chemicals were used as received without further purification. The sodium polytungstate (SPT) solution (2.0 g/mL) was prepared by dissolving the required amount of sodium metatungstate monohydrate (Thermo Scientific - Alfa Aesar) in deionized (DI) water and filtering the solution through a 0.45 µm cellulose acetate filter. Stainless steel pyrolysis Eco-cups (PY1-EC80F) were obtained from Frontier Lab (Fukushima, Japan). Cylindrical borosilicate glass containers (internal diameter 9.2 cm, volume of 1.5 L), used for collecting atmospheric deposition, were purchased from a house-hold goods store (Micasa in Dübendorf, Switzerland). These glass containers were heated at 600°C in a furnace to remove organic contaminants, followed by sonication in DI water (2×30 minutes). Subsequently, the borosilicate glass containers were carefully washed with acetone and ethanol to eliminate residual particles. Finally, the opening was covered with aluminum foil and stored at room temperature until the glass containers were deployed in the field.

3.1.1.2 Tire tread materials

Cryogenically milled Tire Tread (CMTT) samples, supplied by the U.S. Tire Manufacturers Association (USTMA), were used as received and served as reference material (denoted as US-CMTT). The US-CMTT is often used for studies addressing tire wear and tire-derived chemicals in the environment. It is a mixture of tire tread from new tires representative of those sold on the U.S. market. The US-CMTT comprises 41% of passenger car tires, 14% light truck tires, and 45% truck/bus tires.^[44] In addition, tread from used tires were collected from tire dealers in Zurich. Samples of 18 tires for passenger cars and light-duty vehicles and 5 tires for trucks and buses, including one re-treaded truck tire, were collected (Table A 2). These tire tread samples were sliced at a depth of approximately 2-5 mm from the surface using a stainless-steel knife blade, washed thoroughly with ethanol and nanopure DI water, and then dried overnight at 70°C. Afterwards, the isolated tire tread samples were cryo-milled in a Retsch M400 ball mill using 35 mL stainless steel containers and containing one 25 g stainless steel ball. The stainless-steel bucket was cooled in liquid nitrogen for 20 minutes before being used. The mill was operated twice at a frequency of 20 Hz for 60 seconds each. The resulting TWP powders were sieved through a 400 µm steel mesh (Zivipf, Treuchlingen, Germany), dried overnight at 70°C, and stored in an airtight container. The cryo-milled tire tread samples from used passenger cars and light-duty vehicle tires are referred to as cryo-milled car tire tread (CMCTT); those from truck and bus tires are referred to as cryo-milled truck tire tread (CMTTT).

3.1.2 Sample treatment

In contrast to the CMTT samples used for method development and calibration, the atmospheric deposition samples were complex mixtures of biogenic particles (insects, fungi, pollens, and plant debris) and particles from anthropogenic (e.g. road dust) and

natural sources (e.g., mineral dust). Furthermore, limonene may be introduced into the environment through domestic and industrial chemical applications, while aromatic hydrocarbons such as toluene and benzene are typically derived from vehicle exhaust emissions and petrochemical processes. During pyrolysis, these compounds can generate molecular fragments that overlap with the pyrolysis markers generated from the TWP. To limit matrix interferences with target analytes, the following treatment steps were applied:

3.1.2.1 Filtration and chemical digestion

One hundred milliliters (100 mL) of DI water was added to the glass flasks containing the collected atmospheric deposition samples. The walls of the glass flasks were thoroughly scrubbed with a glass rod and carefully rinsed. Then, the liquid sample was passed through a 400-micron stainless steel mesh to remove plant and insect debris, and the filtrate was collected in a 1000 mL Erlenmeyer flask. The flask was rinsed twice to prevent TWP loss. Moreover, the stainless-steel mesh was thoroughly rinsed with DI water. After adding 100 μ L of 5 M H_2SO_4 to the collected filtrate, the Erlenmeyer flask was placed in a water bath at 90°C until the volume of the liquid was reduced to approximately 50 mL. The liquid containing the TWP was neutralized by adding 2 mL of 0.1 N NaOH and placed again in a water bath at 90°C to reduce the volume to approximately 25 mL. The remaining liquid was transferred into 50 mL polypropylene centrifugation tubes. The walls of the Erlenmeyer flask were carefully rinsed with 25 mL of DI water, which was also added to the centrifugation tubes.

3.1.2.2 Density separation

The liquids in the centrifugation tubes were centrifuged (5000 rpm, 2 hours, in DI water), and the supernatant (40 mL) containing solids with a density < 1 g/cm³ (e.g., polyethylene and polypropylene) and digested water-soluble organic compounds was discarded. This step was repeated three times, with the discarded supernatant always replaced by DI water. The pellet remaining at the bottom of the centrifugation tube after the three centrifugation steps in DI water was resuspended in a SPT solution (2.0 g/mL, 50 mL). This suspension was centrifuged at 5000 rpm for 2 hours. The resulting supernatant, containing particles with a density < 2 g/cm³, was collected for subsequent filtration. TRWP suspend in the supernatant due to their density of approximately 1.8 g/cm³.^[1] The remaining pellet was resuspended in fresh SPT solution, centrifuged, and the supernatant was combined with the supernatant from the previous density separation step. The combined supernatants were filtered through a 1.6 μ m glass microfiber filter (Whatman Grade G/F, binder-free, 13 mm diameter) using an in-house-fabricated glass microfiltration unit.

3.1.2.3 Cleanup and thermal desorption

To remove sorbed organic compounds and soluble polymers, the glass fibre filters containing the TRWP were sequentially rinsed with 20 mL of THF, 20 mL of DCM, and 20 mL of acetone in the microfiltration unit. The air-dried filters were carefully transferred to pyrolysis cups and dried overnight at 70 °C. Afterward, the samples were heated to 100°C for 1 hour to remove any remaining volatile compounds before Py-GC-MS measurement. The efficiency of the applied solvent cleanup was evaluated based on spiking experiments using deuterated polystyrene and deuterated polyisoprene.

3.1.3 Analysis

After filling the CMTT materials into pyrolysis cups, each sample was weighed using a microbalance. The opening of the pyrolysis cups were covered with a glass fibre filter to prevent loss of material during sample handling. The collected and pre-treated environmental samples were also transferred to pyrolysis cups and treated in the same way.

A multi-shot pyrolyzer (EGA / Py-3030D) from Frontier Laboratories Ltd. (Koriyama, Fukushima, Japan) was used for pyrolysis. The pyrolysis temperature was set to 500 °C to prevent the formation of secondary degradation products from limonene and to minimize its loss.^[43] Furthermore, an efficient decomposition of NR, SBR and BR with minimal interference from carbon black and other fillers can be expected.^[45] The instrument was operated in single shot mode. The temperature of the transfer line was maintained at 320 °C. A Thermo Scientific TRACE 1310 gas chromatograph (GC, Thermo Fisher Scientific, USA) equipped with an Agilent DB-5MS non-polar phenyl arylene polymer column (60m x 0.25 mm, 0.25 µm film thickness) was used in combination with helium as mobile phase with a purge flow of 10 mL/min and injection temperature set at 300 °C. The GC separation was performed at split ratio of 400:1. A Thermo Scientific ISQ 7000 single quadrupole mass spectrometer (MS) with an interface temperature of 300 °C was used to record the mass spectra of eluents. The transfer line temperature and ion source temperature were also maintained at 300 °C. All the mass spectra were acquired using electron ionization (EI) of eluents at an electron energy of 70 eV and an ion dwell time of 15 ms. A synchronous scan and total ion count (TIC) monitoring mode were used with a mass spectrometer scan range of 50–550 m/z to record the spectra of all eluents during the measurements.

Each pyrolysis cup was heated twice at 500 °C, but no signal of the tire rubber markers was observed the second time. This shows that the components of tire rubber that can be thermally decomposed at this temperature were completely decomposed during the first single shot measurement. Furthermore, the GC oven temperature ramping was carefully optimized to achieve sufficient separation between the pyrolysis products of tire rubbers and high-molecular-weight molecules.

All the data were processed using Thermo Scientific™ Dionex™ Chromeleon 7.2.10 software using the Cobra MS detection algorithm. The detection window was set to 0.07–0.2 min for the retention time of the specific individual markers. The component-matching algorithm based on retention time and peak height as implemented in Chromeleon was applied. The extracted ion chromatogram (EIC) for the tire-specific markers butadiene, isoprene, VCH, α-methylstyrene, styrene, and limonene were obtained by selecting one quantifier and two qualifier ions related to each individual marker (Table A 3) in the processing method at their specific retention time. The peak areas of individual markers were calculated using their quantifier marker ions, while two qualifier ions were monitored to confirm the chemical identity of those markers.

3.1.4 Identification of tire-specific markers

For the Py-GC-MS method development, the CMTT samples were individually processed as described in section 3.1.1 and analyzed by Py-GC-MS. In all chromatograms, we consistently observed six distinct peaks which represented the formation of the following pyrolysis products (Figure 4, blue labels): butadiene, isoprene, VCH, α-methylstyrene, styrene, and limonene. The quantifier ions of butadiene, isoprene, styrene, and α-methylstyrene corresponded to their molecular ion (M^+). Conversely, VCH and

limonene were fragmented into their main building blocks, butadiene and isoprene. Hence, we chose the substructure fragment ions ($m/z = 54$ and $m/z = 68$) as a quantifier for VCH and limonene at their respective elution times of 9.57 min and 13.57 min (Table A 3). The Py GC-MS analysis of reference polymers (SBR-1500, BR, and NR), representing most important polymers used in tire manufacturing, enabled assigning the different pyrolysis products to the respective rubber types (Figure A 1). Butadiene and VCH were identified as characteristic pyrolysis products of butadiene rubber (BR) and styrene-butadiene rubber (SBR). Styrene and α -methylstyrene were primarily generated from SBR. It should be noted that α -methylstyrene is also produced during the thermal decomposition of styrene-butadiene-styrene (SBS), a compound found in polymer-modified bitumen, which is also used in road asphalt.^[40] In Switzerland, however, the amount of SBS used in road construction is negligible (see Section A.4). Limonene and isoprene were pyrolytically formed from NR.

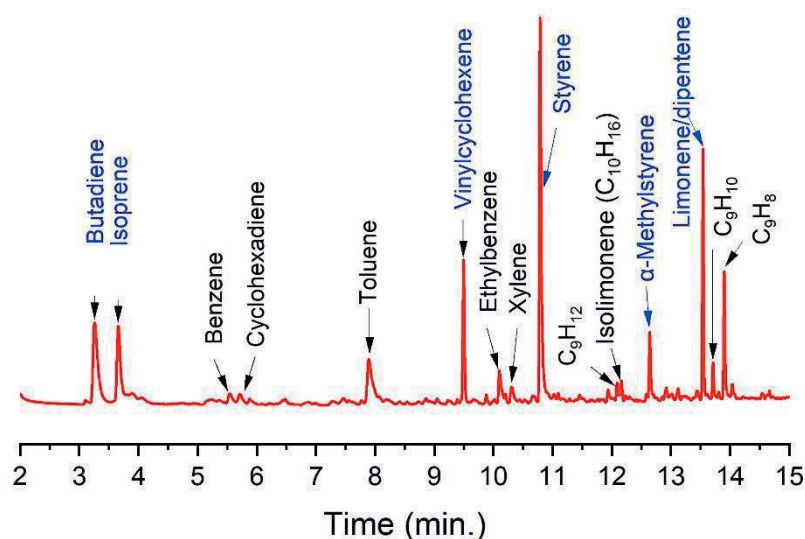


Figure 4: Typical chromatogram of a cryogenically milled tire tread sample, illustrating the distribution of pyrolysis products generated at 500 °C.

These identified markers agree with the NIST/EPA/NIH Mass Spectral Library (NIST 08 program) and have been used as markers for the quantification of TWP in road dust.^[46] In addition to the identified pyrolysis products associated with different rubber types, we detected signals from common tire additives, including benzothiazole, 2-mercaptobenzothiazole, 6-PPD, aniline, toluene, and long-chain saturated fatty acids (C_{16} and C_{18}) (Figure A 2). These additives, however, were not consistently detected in all the CMCTT and CMTTT samples, possibly as these substances can leach from the tires over time.^[47] These compounds were therefore not suitable for the assessment of TWP and not further considered in this study. Analysis of the different CMTT samples used in this study demonstrated that the shares of the six characteristic pyrolysis products varied between tires for passenger cars and tires for trucks but also differed within passenger car and truck tire categories (Figure 5). Styrene was often the dominant pyrolysis product in passenger car tires, whereas isoprene and limonene were the dominant pyrolysis products in truck and bus tires. This can be explained by the dominance of synthetic rubber in the rubber mixture of passenger car tires, whereas the proportion

of NR is higher in truck tires. The US-CMTT samples had isoprene and limonene as the dominant markers, which is a confirmation of the high proportion of truck and bus tires in this mixture.^[44]



Figure 5: Relative contribution of the selected six markers (individual signal intensity relative to the sum of the intensity of all six peaks) for all cryo-milled tire tread samples. Cryo-milled tire tread (CMCTT)-1 to 18 are samples from old tires for passenger cars, and CMTT-1 to 5 are samples from old tires for trucks and buses. US-CMTT is a mixture of samples from new tires that are representative of those sold on the US market.

Similar to Figure 5, the differences in the signals of the six pyrolysis markers in the different tire samples can be investigated using principal component analysis (PCA). Principal component analysis of the composition data of the six pyrolysis markers measured in all CMTT samples resulted in two first principal components (PC1 and PC2) that explain 92.2% of the variance in the data. PC1 (84.6% explained variance) has high negative loadings for styrene and high positive loadings for limonene and isoprene. Projection of the measurements of CMTT samples onto the plane spanned by the first two principal components allows a separation of the CMTT samples according to the relative signals of these markers for synthetic and natural rubber and therefore a (partial) separation of CMTT from tires for cars and trucks/buses (Figure A 9). PC2 explains 7.6% of the variance with high loadings for butadiene and α -methylstyrene, two markers indicative for synthetic rubber.

3.1.5 Method calibration

The relationship between the integrated signals of the selected quantifier ions related to the characteristic pyrolysis products and the mass of the tire tread material was determined by Py-GC-MS analysis of different amounts (15 μg to 430 μg) of various CMTT samples. The maximum quantification amount of CMTT at the 400:1 split ratio was approximately 450 μg . Above this upper limit, saturation of the detector and peak tailing in the chromatograms were observed.

The integrated peak areas of the quantifier ions from the six tire-specific pyrolysis products were linearly correlated to the mass of the individual CMTT samples (Figure A 3). However, the slope of the linear regression between the integrated peak areas of the individual quantifier ions from the six characteristic pyrolysis products and the mass of CMTT varied between individual tires. This can best be seen when projecting the data from all CMTT samples for the individual markers onto the same diagram (Figure 6).

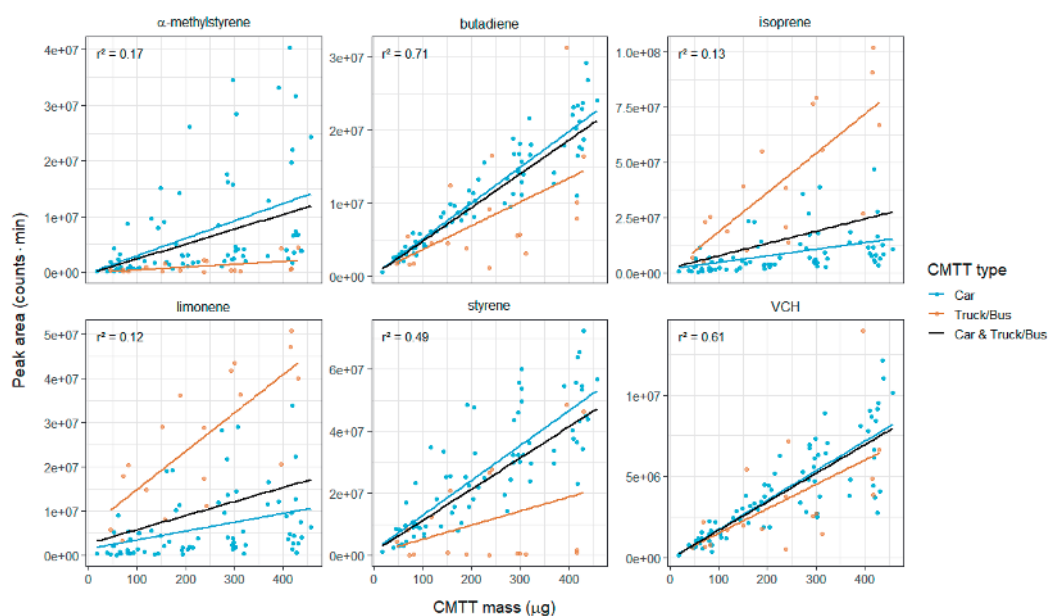


Figure 6: Measured peak area for the individual pyrolysis markers versus the mass of the cryo-milled tire tread (CMTT) samples from used tires. The CMTT samples from passenger cars and truck/buses were grouped according to the vehicle category. The linear regression lines are shown for both types of CMTT and for all CMTT samples together. The coefficients of determination (r^2) refer to the relationship between the peak area and all CMTT (Car & Truck/Bus) samples. The r^2 values separately for cars and trucks/buses are listed in Table A 4 in the Annex.

The correlations between the integrated signals of the quantifier ions of the characteristic pyrolysis products and CMTT mass, as expressed by the coefficients of determination (r^2), ranges from 0.12 for limonene to 0.71 for butadiene. For most markers, the dependence of the peak area on CMTT mass is different for passenger car tires than for tires for trucks and busses, reflecting the different compositions of the respective tires. Butadiene ($r^2=0.86$) and VCH ($r^2=0.70$) showed the highest coefficients of determination when CMTT from passenger car tires (CMCTT) were considered separately. For the CMTT samples from truck/bus tires (CMTTT), the highest coefficients of determination were found for isoprene ($r^2=0.57$) and limonene ($r^2=0.56$) (Table A 4). A clear improvement of the r^2 values was obtained when using the sum of the integrated peak areas from all six markers (denoted as sum of peak areas, SPA) instead of the peak areas of individual markers (Figure A 4). The linear regressions of SPA against CMTT mass were calculated for US-CMTT, each vehicle category, and for all CMTT from car and truck/bus tires together. The intercepts of all these linear regressions were not significantly different from zero. Therefore, linear regressions were eventually calculated with the intercepts set to zero (Figure 7). The slope for the truck and bus tires was larger than that for the passenger car tires. The higher total response of the sum of the

six pyrolysis markers per mass of truck/bus tire material compared to CMTT from passenger car tires is due to the higher contributions from isoprene and limonene (Figure 6), which results from the higher NR content in tires for trucks and busses. A similar slope was also observed for the US-CMTT, which represents a mixture of tires with a high fraction of truck/bus tires.^[44] Linear regression of SPA against the mass of all CMTT samples from car and truck/bus tires used in this study yields a slope of 297'311 counts·min per μg of CMTT (standard error of 8'324 μg^{-1} , $r^2=0.93$). This slope can be used to quantify tire tread material based on Py-GC-MS results and will be used as the reference throughout this study. The slope can be interpreted as the calibration constant that converts the SPA obtained for CMTT samples and for atmospheric deposition samples into the mass of tire tread material.

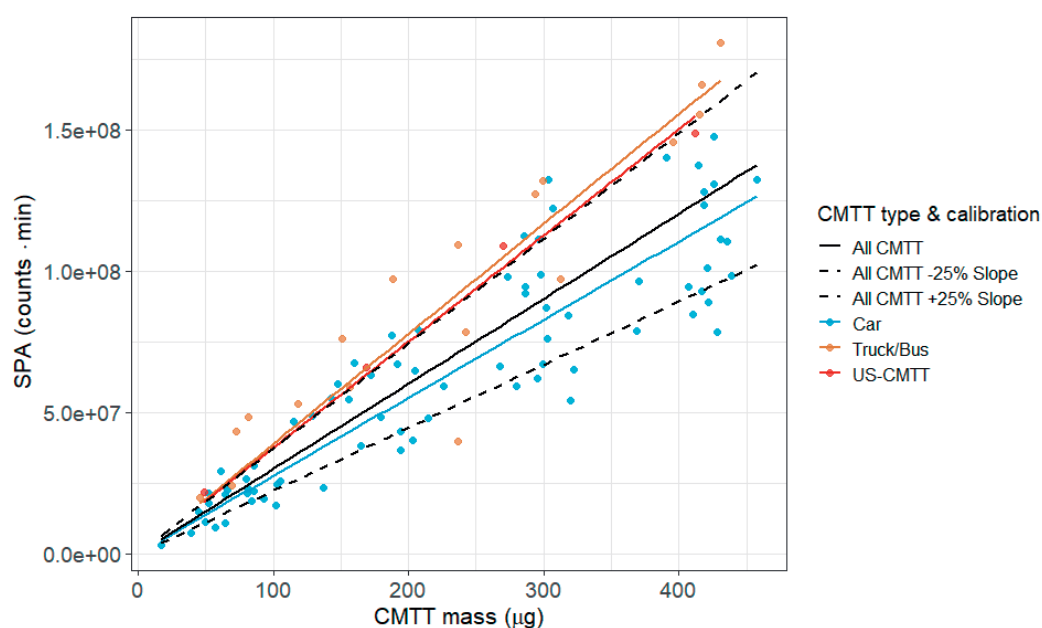


Figure 7: Sum of the peak areas (SPA) of the six pyrolysis markers versus the mass of cryo-milled tire tread (CMTT) samples from old tires for passenger cars and for trucks and buses, as well as for a mixture of new tires (US-CMTT). Linear regressions for the different CMTT types and for all measured CMTT samples are also shown (intercepts forced to zero). The estimated slope of the linear regression on all data $\pm 25\%$ covers most of the observed variation of all measured values.

It is important to note that TWP in the environment are always a mixture of tire tread from various tires, and that this mixture of tires is unknown. We, therefore, assume that the SPA obtained from analysis of CMTT from the chosen 23 tire types was representative for the tires available on the market and that the spread of the SPA in Figure 7 covers the variability of all different tires. Consequently, the SPA for a certain mass of TWP in environmental samples is expected to lie within the area defined by the points in Figure 7. We therefore consider the calibration constant (slope), determined from the regression analysis described above, as the best estimate for converting a SPA measurement into TWP in environmental samples.

Despite the varying rubber mixtures of the different tire types, a $\pm 25\%$ range of the above-mentioned slope (calibration constant) encompasses the variation observed in the individual CMTT samples. Assuming that the true conversion factor is within the above mentioned $\pm 25\%$, the percentage systematic uncertainty or systematic error for the calculated TWP mass lies between -20% ($1 - \frac{1}{1.25}$) and $+33\%$ ($1 + \frac{1}{0.75}$). Applying

the obtained calibration constant to an environmental sample that is dominated by truck tires, e.g. like the mixture of the US-CMTT, will result in a systematic overestimation of TWP mass. On the other hand, the mass of TWP in environmental samples that are dominated by passenger car tires are likely to be underestimated. Considerations regarding the total measurement uncertainty of the analytical method can be found in section 3.1.8.

3.1.6 Data processing of the Py-GC-MS method

The response of Py-GC-MS instruments varies over time, partly due to differences in the detector response. Linear deuterated polymers are therefore often used as internal standards to correct for these variations. However, their degradation kinetics in pyrolysis differs from those of cross-linked tire rubber, and thus deuterated polymers are not suitable for correcting signal fluctuations that occur when analyzing multiple series of samples as they only contain C-C backbones while the tire rubber degradation involves the fragmentations of the C-C (~ 346 kJ/mol), C-S (~ 272 kJ/mol), and S-S (~ 226 kJ/mol) bonds distributed in a complex crosslinked network. Due to the lack of deuterated tire materials we used one of the CMTT samples (CMCTT-6) during the measurement as an external calibrant as an alternative to the deuterated polymers. Usually, five samples with different and known masses of CMCTT-6 were therefore measured as external controls with each series of environmental samples. The peak areas of the six pyrolysis markers measured in CMCTT-6 from the data set used for method calibration served as the reference for each measurement of external controls. All subsequent measurements of atmospheric deposition samples were corrected for possible changes in sensitivity of the Py-GC-MS based on the measurements of the external controls, i.e. CMCTT-6.¹ Specifically, the linear regression parameters (intercept and slope) were calculated for each pyrolysis marker determined in CMCTT-6 from both the method calibration dataset and the external control measurements. The intercepts for all six markers for both the calibration measurements and the external control measurements used in this study did not deviate from zero (95% confidence level). VCH was an exception in the calibration dataset, where a small intercept deviated from zero, as the measured peak areas for the different masses of CMCTT-6 lay on a straight line ($r^2=1.00$). However, no offset correction was generally needed and the correction for variation in the sensitivity of the Py-GC-MS was derived from linear regressions with the intercept forced to zero. Our sensitivity correction essentially relates the slope obtained for the external control (CMCTT-6) to the slope obtained for the same material (CMCTT-6) from the method calibration measurements (Figure A 5 and Figure A 6), with the ratio between the two slopes defined as the sensitivity factor (SF). Our data (SPA) obtained from the individual measurements of environmental samples were thus corrected for variable sensitivities by multiplication with the SF.

The steps of the data processing from the determined peak area of individual pyrolysis markers to the atmospheric mass deposition rate are described in detail in the Annex, section A.7.

3.1.7 Recovery of TWP

The recovery of TWP was determined by analyzing $n=12$ test samples. These test samples consisted of a mixture of a known amount of CMTT, here CMCTT-5 and sand that

¹ For Series 10 and 11 of atmospheric deposition samples (see Table A 1), CMCTT-5 was used as external control.

was muffled at 600°C for 12h to remove any potential traces of tire wear before use. The test samples were treated and analysed the same way as the atmospheric deposition samples. The recovery r was calculated as the ratio of the measured or recovered amount of CMTT and the known amount of CMTT present in the samples. The average recovery was determined to $r = 68.5 \%$, with a standard deviation of $\sigma_r = 11.9 \%$. For the quantification of TWP in environmental samples, a correction for incomplete recovery is therefore applied (Section A.7).

3.1.8 Measurement uncertainty

As described in section 3.1.5, the calibration constant determined in this study, which is used to convert the measured SPA into mass of TWP, may be systematically incorrect by between -20% and +33%. This range of potential systematic error is expected to cover all plausible calibration constants and can therefore be approximated by a bias u_b ranging from -10.0% to 16.5%.

Other main components of uncertainty are the sensitivity correction with a standard uncertainty $u_s = 8.1 \%$ (Sections 3.1.6 and A.7) and the incomplete recovery with $s_r = u_r = 11.9 \%$ (Section 3.1.7). For the quantification of the mass deposition rate of TWP with a passive sampling setup as used in this study, the choice of the exact sampling position influences the amount of TWP collected. Therefore, an uncertainty component covering the representativeness of the sampling site should be considered. Air quality monitoring networks use similar passive sampling containers for the measurement of the atmospheric deposition of total dust.^[29] Based on measurements of total dust deposition within the Swiss National Air Pollution Monitoring Network, the standard deviation from duplicate side-by-side measurements was $u_d = 22.8 \%$.^[26] Here, u_d represents the difference between measurements taken at locations that differ only slightly (a few meters). We assume that similar differences apply to atmospheric deposition of TWP collected in this study and that u_d describes the influence of the exact position of the sampling container on the result. When other sampling methods are used, or when the analytical method is applied to other environmental samples (e.g. active sampling of airborne TWP), the uncertainty component u_d should be adapted accordingly. For applications where only the amount of TWP in a collected sample is of interest, the uncertainty component u_d might not even be needed. The uncertainty components u_r , u_s and u_d can be considered as random, the combined expanded uncertainty for the quantification of TWP mass and mass deposition rate from the mean of n samples can then be expressed as^[48]

$$U = 2 \cdot \sqrt{\frac{u_s^2 + u_r^2 + u_d^2}{n} + u_b^2} \quad (1)$$

The coverage factor of 2 leads to a confidence level of 95%. For a single sample ($n=1$), the ranges of the combined expanded uncertainties for the quantification of TWP (i.e. $u_d = 0 \%$) and for the mass deposition rate of TWP are [-35 % to 44 %] and [-58 % to 63 %], respectively. The combined expanded uncertainties decrease when the mean TWP mass or mass deposition rate is calculated from an increasing number of samples and converges with increasing n towards the systematic error of the calibration constant used.

3.1.9 Limit of detection

The limit of detection (LOD) and limit of quantification (LOQ) of the applied method was determined based on $n=7$ procedural blanks, i.e. sampling containers that were not exposed to atmospheric deposition but otherwise treated in the same way as the environmental samples. The sampling vessels contained neither the target analytes nor the environmental matrix consisting, e.g. of mineral dust and biological particles. The signals of the six pyrolysis markers from the blank samples were processed analogously to those from the environmental samples; CMCTT-5 served as an external control. The mean mass of CMTT determined in the blank samples was $\mu_B = 0.08 \mu\text{g}$, the standard deviation was $\sigma_B = 0.02 \mu\text{g}$. We interpret the signals from the blank measurements as noise as they did not match the qualifier ions. However, there is a small contribution to the signal of the quantifier ions of the pyrolysis markers.

At low masses of CMTT, the measurement uncertainty is probably dominated by the correction for changes of the sensitivity of the Py-GC-MS instrument. The associated standard uncertainty u_s was 8.1 % and the measurements of blank values and external controls with the lowest CMTT mass indicate that this relative uncertainty is also applicable at very low amounts of CMTT (Figure A 8) and σ_B is also a reasonable representation of the standard deviation of measurements at masses of CMTT in the range of the detection limit, denoted as σ_D , therefore $\sigma_D = 0.02 \mu\text{g}$. We calculated the limit of detection (LOD) of the analytical method following an approach described by Currie (1968), which was also applied to an analytical chain for atmospheric microplastics.^[26,49] In this approach, a critical level L_C is defined which is the threshold above which a measured signal is very likely due to the presence of the quantity being measured, here tire tread material. L_C is therefore the threshold at which false positive decisions are minimized, it is $L_C = \mu_B + 2 \cdot \sigma_B = 0.12 \mu\text{g}$. The limit of detection is then the threshold at which the risk of false negative decisions is acceptable (here the 95% confidence level), the LOD can then be specified as

$$LOD = L_C + 2 \cdot \sigma_D = 0.16 \mu\text{g} \quad (2)$$

3.2 SEM/EDX

For the identification and quantification of the dry atmospheric deposition of TRWP, there is a measurement method based on scanning electron microscopy combined with energy dispersive X-ray analysis (SEM/EDX) that has already been used in several studies.^[21,50] It was planned in this project that measurements of dry deposition using this established method should complement the measurement of bulk atmospheric deposition using the method developed in this project for checking the plausibility of the results obtained with the latter method. Since the determination of TRWP and the calculation of TWP using the SEM/EDX method are described in detail in the literature, only a brief summary follows here.

3.2.1 Analysis of TRWP in dry deposition

Airborne particulate matter in the size range between $1 \mu\text{m}$ and $80 \mu\text{m}$ were collected using Sigma-2 passive samplers (VDI2119:2013)^[30] and subsequently analyzed by Automated SEM/EDX (Scanning Electron Microscopy with energy dispersive x-ray

spectroscopy) Single Particle Analysis (ASPA MC2®) using a Zeiss Gemini 300 Field Emission Gun (FEG)-SEM equipped with an Oxford X-MAX EDS detector with an 80 mm² window, a high efficiency 4 quadrant backscatter electron (BSE) detector and the particle analysis software AZtecFeature (Oxford Instruments). TRWP classification and quantification were performed with a machine learning (ML) algorithm, which makes use of all the available elemental and morpho-textural descriptors (in total 67, 43 determined by the AZtecFeature software and 24 extracted by the ML-algorithm developed by Particle Vision and described in detail elsewhere).^[21]

In brief, the ML-algorithm is based on a “random forest” model trained with >100'000 particles from air samples collected in contrasting environments (urban, rural, industrial, road canyons, railways, etc.). The prediction accuracy that is expected from the ML-classifier and further details on the ML-model are shown in Rausch et al. (2022).^[21] Cross-validations of the element profiles for the different particle categories typically found in the particle size fraction between 2.5 µm and 10 µm confirmed a good agreement between the source profiles obtained by SEM/EDX single particle analysis and those from “bulk-PMF” (i.e. Positive Matrix Factorization using element concentrations derived from inductively coupled plasma mass spectrometry measurements and data using the thermal-optical transmission method for carbon concentrations).^[51]

3.2.2 Adaptation of the methodology

After initial analytical tests aimed at ensuring an optimal adaptation of the previously developed SEM/EDX based methodology for the quantification of TRWP specifically for dry TRWP mass deposition covering the full particle size range from 1 µm and 80 µm, it was decided to proceed with the analysis of two fractions separately (i.e. particles with diameters below and above 20 µm). The total particle counts per fraction over the full year was for all sites larger than 1000. The reason to perform two separate analyses per sample and aggregate them later was to ensure that a balanced and representative number of TRWP was measured for the entire size spectrum. A single analysis would result in a much higher number of particles of smaller size (i.e. < 20 µm) in a defined area. Consequently, the mass deposition of larger particles (>20 µm) would be underrepresented, as observed in the initial tests.

3.2.3 Calculation of TRWP mass deposition rates

After classification in the TRWP category following the above-described ML-based methodology, particle mass deposition rate (g/cm²/s) for each TRWP was calculated according to VDI 2119:2013.^[30] The SEM/EDX single particle analysis on passively collected Sigma-2 samples provides particle sizes in the 1-80 µm range as equivalent circular diameters (ECD). Subsequently, based on the ECD values obtained the volume of each particle is calculated assuming a spherical shape. A volume form factor of 0.75 (as proposed in VDI2119:2013) is subsequently applied to account for the fact that the particles are not spherical. As next, the mass of each particle is calculated by assigning a particle density based on its elemental composition and particle classification (specifically 1.8 g/cm³ for TRWP, in this study). Finally, the mass deposition values of each particle in a given area of the analyzed sample and for the specific collection time are aggregated to obtain the total TRWP dry mass deposition rate in the specific sample. For more details on the calculation of mass deposition rates based on microscopy single particle analysis and on estimated detection limits of the ASPA MC2® method we refer to the literature.^[21,30]

3.2.4 Measurement uncertainty

The main sources of uncertainty of the mass deposition rate calculations obtained from SEM/EDX single particle analysis arise from the counting statistics of the particles detected and from assumptions regarding particle density and morphology. The number of TRWP detected in samples can be low. The uncertainty of the mass of TRWP in the samples is calculated from the number counts and corresponding particle sizes assuming Poisson statistics and using confidence intervals derived from Chi-square distributions.

In addition to counting statistics, further uncertainties arise from assumptions regarding particle density and morphology. Density and shape uncertainties were therefore evaluated using sensitivity analyses based on literature density ranges and alternative volume approximations. In this study, we calculated the density uncertainties using the end member values for TRWP reported in the literature, i.e. instead of 1.8 g/cm³, densities of 2.1 g/cm³ and 1.5 g/cm³ were applied.^[22,52] The resulting TRWP mass deposition rates deviate up to $\pm 16\%$.

An ellipsoidal approximation for describing the geometry of TRWP has also been applied by other authors in SEM/EDX single particle analysis.^[50] The calculated particle volume was compared to the results assuming spherical shapes and a volume form factor of 0.75 as used in this study and elsewhere.^[53,54] It was found that the corrected sphere approximation leads to a slightly higher volume (median of 6 %).

The resulting uncertainties for counting statistics, density and morphology were combined assuming independence of the individual contributions and calculated as the root sum of squares.

The corresponding particle-specific relative uncertainty was converted into an absolute uncertainty of the reported mass deposition rates. The resulting combined uncertainties are shown as error bars in Figure 12 to Figure 14.

These analytical uncertainties are treated separately from the overall methodological uncertainty associated with the Sigma-2 sampling approach. The total uncertainty of particle sampling using the Sigma-2 passive sampler have been derived from parallel measurements and reported to be below 25%.^[30] This uncertainty due to sampling is independent of the particle-specific uncertainties derived in this study and is not included in the uncertainties presented.

3.3 Image analysis

The two analytical methods (Py-GC-MS and SEM/EDX) used in this study and described in sections 3.1 and 3.2 are suitable for determining and quantifying tire wear in atmospheric samples, although the SEM/EDX method has only been established for dry deposition samples. However, both methods are labour intensive, require sophisticated instrumentation and well-trained research staff. These methods are, therefore, less suitable for monitoring tire wear in the environment, which requires less demanding methods with a higher sample throughput.

A simple and cost-efficient method for determining TRWP is based on the analysis of optical microscopy images.^[20] On such images, TRWP can be identified based on their characteristic shape and colour. However, one limitation of optical methods is that only TRWP larger than approximately 15 μm can be detected and the contribution of

particles smaller than the detection limit must be estimated based on typical size distributions.

In this project, two approaches for the optical identification of TRWP were developed and evaluated based on a selection of wet and dry atmospheric deposition samples. These additional samples were available from a partner project, which focused on MP contents in atmospheric deposition (section 2.1.3). For that purpose, atmospheric deposition samples underwent a series of process steps to isolate MP from the matrix and were eventually filtered on Anodisc filters.^[26] The processing pipeline is expected to also preserve TRWP and therefore, the same samples (Anodisc filters) were used for the detection of TRWP based on optical microscopy (Note, that several other sample preparation pipelines include a density separation step. Such a step would, however, most likely result in a substantial loss of TRWP if the density of the fluids were not properly adjusted to values around 2 g/cm³). The whole Anodic filter was imaged with an automated optical microscope (VHX-7000, Keyence, Japan) at 50x magnification and using autofocus routines. Stitched images were stored for later processing.

The first image analysis approach relied on detection of all particles on a filter and subsequent identification of TRWP based on thresholds in the HSV colour space. The second approach was based on YOLO11, a collection of pre-trained AI models for computer vision tasks such as object detection.^[55] Both approaches provided the contours of all TRWP on Anodic filters and based on these data, the size, volume and mass of all TRWP were calculated.

It is important to note that the results obtained with both approaches were not validated in this study due to time and personnel constraints. Validation of these methods is, however, essential and means that random samples of identified TRWP must be confirmed or rejected by an independent method, and the rates of correctly identified TRWP and the rates of false-positive and false-negative classifications must be determined. This can be done by chemical characterization of individual particles that were identified as TRWP, for example by attenuated total reflection Fourier transform IR (ATR-FTIR) spectroscopy or by element analysis using micro-X-ray-fluorescence (μ XRF). Since the work on the detection and quantification of TRWP using image analysis is not yet complete, it is only briefly and exemplarily presented in the Annex (Section A.9). Image analysis as an alternative analytical approach should nevertheless be mentioned here, as we will discuss its potential in the outlook (Section 4.6).

4 Results

4.1 Total mass deposition rates of TWP

Figure 8 shows the mass of TWP determined in the 13 series of four weekly samples of total atmospheric deposition, i.e. series 10 to series 22 spanning the time from June 2024 until June 2025 (see Table A 1 for the exact sampling times). The measured mass of TWP is generally well above the limit of detection and within the range of calibration (Figure 8).

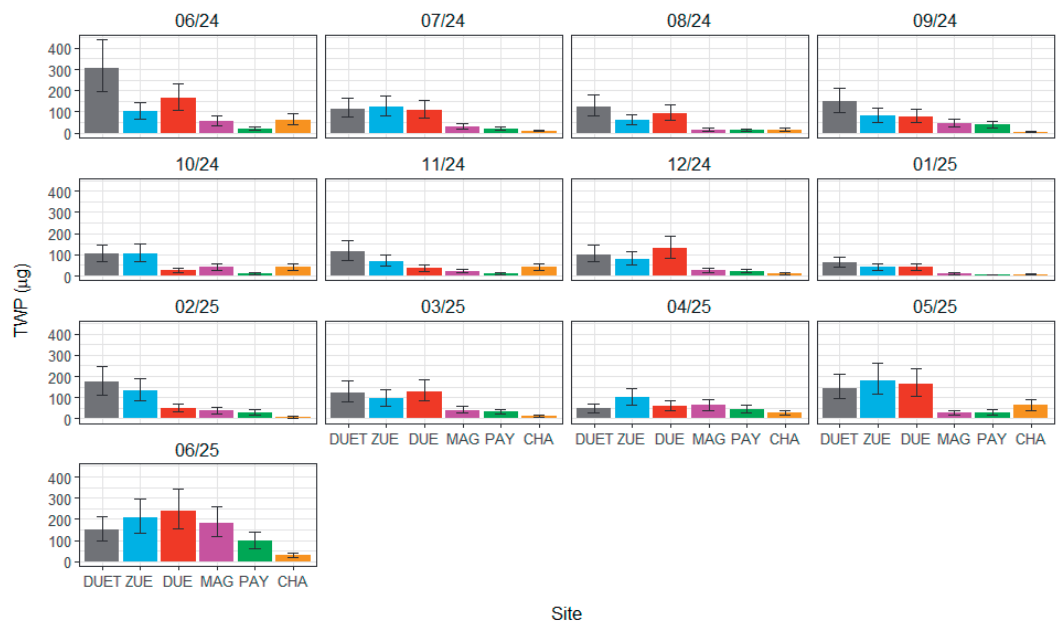


Figure 8: Mass of tire wear particles (TWP) determined in total atmospheric deposition samples collected at six sites across Switzerland from June 2024 to June 2025 (Series 10 to 22, see Annex 5.1 for exact dates). DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous. The title of each plot indicates the month and the year of bulk deposition sampling (MM/YY). The error bars indicate the combined extended uncertainty for the mean of each set of single analysed samples and range from -35 % to 44 % (section 3.1.8).

Based on the TWP masses shown in Figure 8 and using the equations given in Annex 5.6, the atmospheric mass deposition rates (MDR) of TWP were calculated. The average of all measurements taken at each site over the course of one year (Series 10 to 22) yields the average TWP bulk mass deposition rates (sum of wet and dry deposition) shown in Figure 9 and listed in Table 2. The measured values of the individual four weekly samples and their distributions are shown in Figure A 13. There is a clear dependence of the TWP mass deposition rate on the sampling location. MDR are highest at the traffic site and higher at the sites in densely populated areas than at the rural and mountainous sites. This dependence on the proximity to the locations where TWP are generated can be explained by the large size of TRWP and the resulting high settling velocity.

We expect that the values reported for the bulk mass deposition of TWP at the individual sample sites (Table 2) can be transferred to other locations in Switzerland with similar land use characteristics. The deposition rates determined in this study improved our understanding of the environmental exposure of TWP and for the assessment of the accumulation of TWP in the environment due to the atmospheric transport and input.

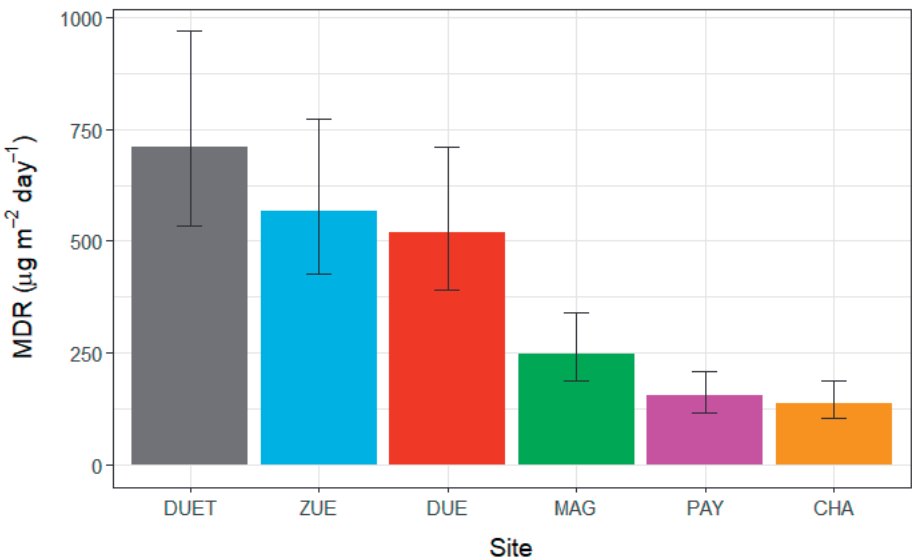


Figure 9: Mean bulk mass deposition rates of tire wear particles (TWP) measured by pyrolysis gas chromatography-mass spectrometry for the entire measurement campaign, i.e. for 13 four-weekly samples from each site from June 2024 to June 2025 (DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous). The error bars indicate the combined extended uncertainty for the mean of each set of thirteen analysed samples and range from -25 % to +36 % (section 3.1.8).

To put the obtained MDR for TWP into a broader context, the values can be compared with the MDR obtained for all types of atmospheric particles and dust as measured in the NABEL network. In NABEL, the Bergerhoff method [28] is applied to determine the mass of dry residues of atmospheric bulk deposition. In this method, the atmospheric bulk deposition is collected in polypropylene cylinders with a diameter of 10.5 cm, like the passive collectors for TWP used in this study. After sampling, the contents of the collection vessels are dried, and the mass and the mass deposition rate of the dry residues are determined and referred to as the mass and mass deposition rate of dust. Table 2 lists the annual dust deposition rates determined for the same or similar sampling sites and for the same time period as the TWP sampling campaign. The sampling sites ZUE, MAG and PAY were identical, for the suburban and the mountainous site types, the reported dust deposition sampling was done in Basel-Binningen (suburban) and Rigi-Seebodenalp (mountainous site at 1050 m asl.). NABEL has no suburban traffic site and therefore no dust deposition rate is available for Dübendorf-Überlandstrasse. A comparison of the mass deposition rates of TWP and of dust particles suggests that in urban and suburban areas, approximately 1.3% of the mass of atmospheric dust deposition consists of tire wear. In rural areas, the proportion of tire wear in atmospheric dust deposition is lower (approximately between 0.3% and 0.4%).

Mean annual bulk TWP and dust deposition rates			
Site	Site type	TWP deposition rate ($\mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$)	Dust deposition rate ($\mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$)
DUET	Suburban, traffic	711	n.d.
ZUE	Urban, background	568	43237
DUE	Suburban	520	38533
MAG	Rural (south of Alps)	250	86310
PAY	Rural (north of Alps)	154	59192
CHA	Mountainous, 1140 m asl.	139	31590

Table 2: Mean atmospheric mass deposition rate of tire wear particles (TWP) for each sampling site representing different land-use situations, i.e. different proximity to road traffic. For comparison, the bulk deposition of all types of atmospheric particles as measured in the NABEL network is shown (see text). The expanded uncertainty (95% confidence) for the annual bulk deposition rates of TWP range from -25 % to +36 %.

The determined TWP mass deposition rate did not show a clear dependence on the time of the year, the measured bulk deposition rates of TWP were similar in the cold season to the values in the warm season (Figure 10).

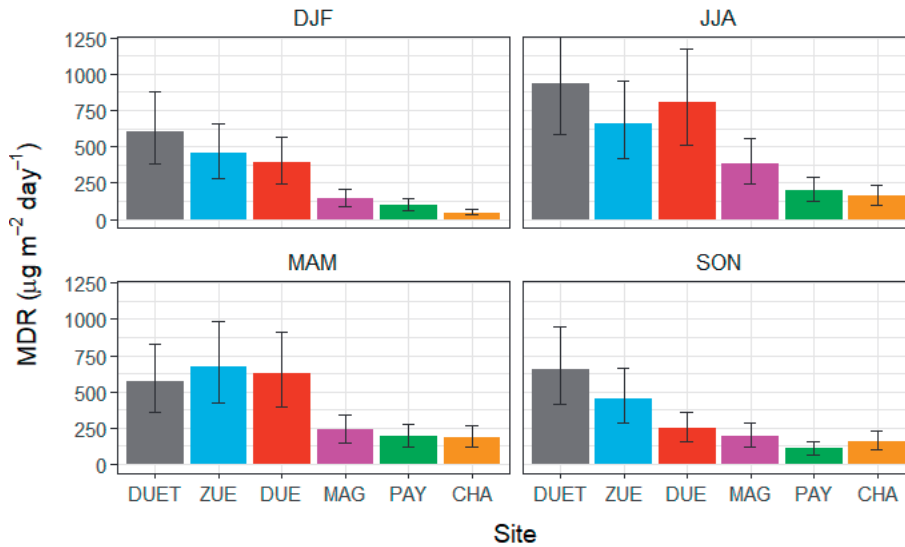


Figure 10: Mean bulk mass deposition rate of tire wear particles (TWP) for different seasons measured by pyrolysis gas chromatography-mass spectrometry (DJF: December, January, February; MAM: March, April, May; JJA: June, July, August; SON: September, October, November). DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous. The error bars indicate the combined extended uncertainty for the mean of each set of three analysed samples and range from -37 % to +45 % (section 3.1.8).

As mentioned in section 3.1.4, principal component analysis of the measurements of CMTT facilitates the comparison of the relative contributions of the six pyrolysis markers in atmospheric deposition and in the different types of CMTT. Figure 11 shows such a comparison, i.e. the measurements of the atmospheric deposition samples shown in the coordinate system of the first two principal components. The projections of the measurements of atmospheric deposition at the different sites show a considerable variation, however lean more towards styrene than towards limonene and isoprene, the

pyrolysis markers indicative for synthetic (SBR) and natural rubber (NR), respectively. This is a clear indication that the mixture of different tires in the collected atmospheric deposition is more similar to tires for passenger cars than to those for trucks and buses. The fact that the CMTT samples in Figure 11 cover the entire area occupied by the environmental samples indicates that the selection of CMTT samples considered in this study is representative for TWP in atmospheric deposition supporting the chosen range of systematic error of the applied calibration constant.

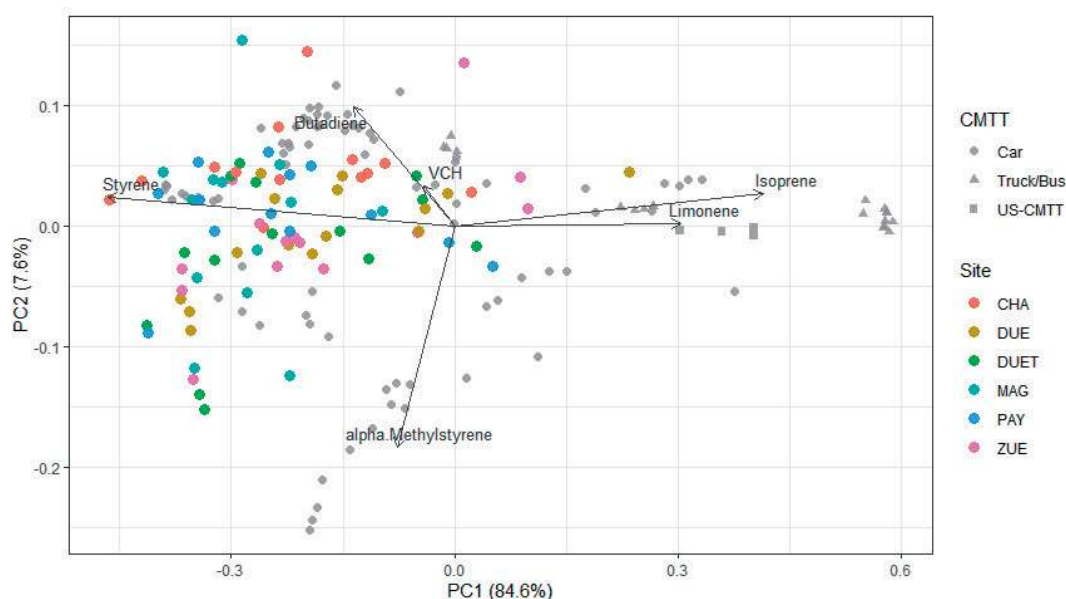


Figure 11: Biplot of the measured pyrolysis markers in the CMTT samples grouped by vehicle category (grey points) and in the four-weekly atmospheric deposition samples grouped by site (coloured dots). The principal components PC1 and PC2 have been calculated from the CMTT data set only. The points are the scores of the observations in the coordinates of the principal components, the arrows indicate the loadings of the original features, i.e. the six pyrolysis markers. More information on the data analysis can be found in the Annex A.8.

4.2 Dry mass deposition rates of TWP

Figure 12 shows the dry mass deposition rate of TRWP (i.e. the particle mixtures consisting of rubber and minerals/metals encrusted from the road surface) derived from SEM/EDX measurements and determined for the 13 series of four-weekly samples collected over one year. Series 1 to series 13 span the time window from the end of September/beginning of October 2023 to the end of September/October 2024 (Table A 1 for the exact sampling times).



Figure 12: Mass of tire and road wear particles (TRWP) determined in dry atmospheric deposition samples collected at six sites across Switzerland from October 2023 to October 2024 (Series 1 to 14, see Annex 5.1 for exact dates). DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous. The title of each plot indicates the month and the year of dry deposition sampling (MM/YY), the error bars indicate the extended uncertainty for each of the analysed samples (section 3.2.4).

Figure 13 shows the same data as shown in Figure 12 except that the data is aggregated to seasonal mean dry mass deposition rates of TRWP.

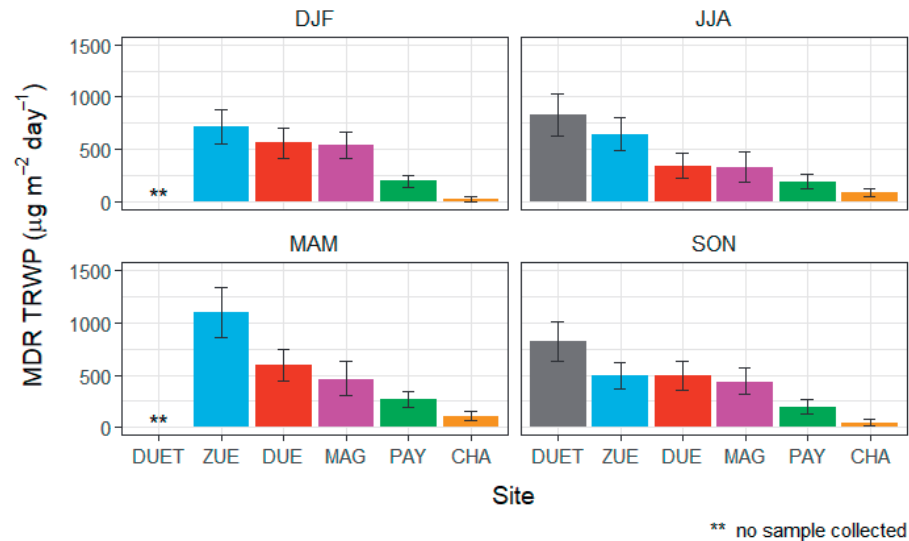


Figure 13: Mean dry mass deposition rates of tire and road wear particles (TRWP) for different seasons as collected with Sigma-2 passive samplers and analysed by SEM/EDX. (DJF: December, January, February; MAM: March, April, May; JJA: June, July, August; SON: September, October, November). DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous. Note that measurements at DUET are only available from June 2024 to October 2024. The error bars indicate the extended uncertainty for the seasonal means (section 3.2.4).

The results in Figure 13 show that the dry mass deposition rates of TRWP do not show a distinct seasonal dependence, similar to the bulk mass deposition rates of TWP as shown in Figure 10. Also similar to the measured bulk mass deposition rates presented in section 4.1, the yearly average value of each site indicates a clear dependence on the site type with the highest TRWP mass deposition rate at DUET (826 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$), followed by ZUE (713 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$), DUE (485 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$), MAG (436 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$), PAY (179 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) and finally with the lowest mean annual value at CHA (63 $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) (Figure 14 and Table 3). This dependence mirrors the proximity of the road network to the respective sites, i.e. the larger the distance to highly trafficked roads, the lower the measured TRWP mass deposition. The values of the individual four weekly samples measured at the different sampling locations are shown in Figure A 14.

The SEM/EDX measurements suggest that the dry mass deposition rates of TRWP are at the suburban traffic site DUET ten times higher than at the mountainous site (CHA). The differences in dry TRWP mass deposition rates are smaller between stations not directly affected by local traffic. Interestingly, the obtained dry TRWP mass deposition rate at the rural site south of the Alps (MAG) is surprisingly high and almost similar to those at the suburban site DUE. This was not observed to this extent in the measurements of bulk TWP mass deposition using Py-GC-MS.

The annual mean dry mass deposition rate of TRWP at the suburban traffic site DUET needs to be interpreted with care because the measurements at this site only took place during the five months from June 2024 to October 2024.

Mean annual dry TRWP mass deposition rates			
Site	Site type	TRWP deposition rate ($\mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$)	TWP deposition rate ($\mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$)
DUET*	Suburban, traffic	826	413
ZUE	Urban, background	713	356
DUE	Suburban	485	242
MAG	Rural (south of Alps)	436	218
PAY	Rural (north of Alps)	179	90
CHA	Mountainous, 1140 m asl.	63	31

Table 3: Mean atmospheric dry mass deposition rate of tire and road wear particles (TRWP) and for tire wear particles (TWP) for each sampling site representing different land use situations, i.e. different proximity to road traffic.
 * The mean values for DUET correspond to an average of 5 series (June 2024 – October 2024) and are therefore not annual means.

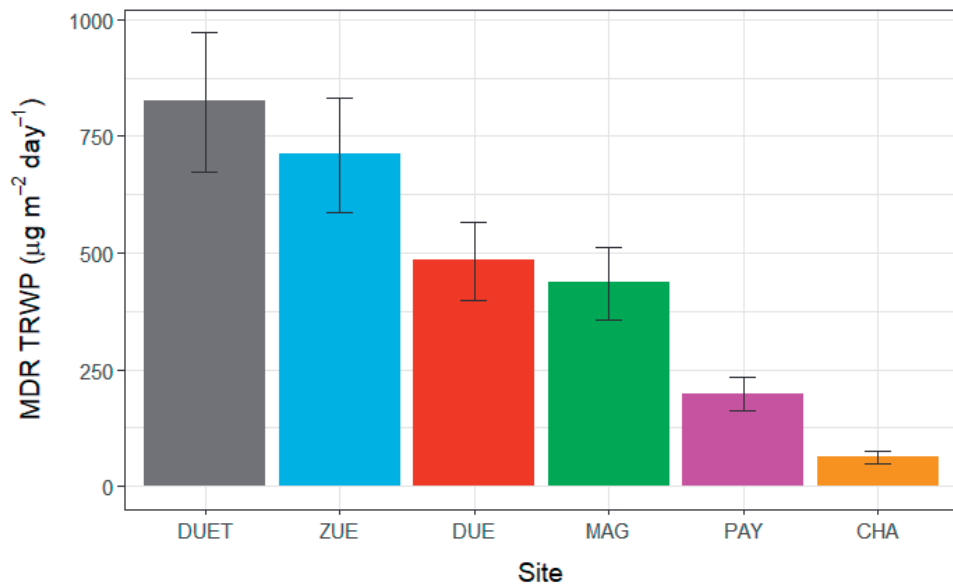


Figure 14: Mean dry mass deposition rates of tire and road wear particles (TRWP) measured by SEM/EDX for the entire measurement campaign, i.e. for 13 four-weekly samples from each site from October 2023 to October 2024 (DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-MeteoSwiss, rural; CHA: Chaumont, mountainous). The error bars indicate the extended uncertainty for the annual means (section 3.2.4).

In Figure 15 the TRWP mass deposition rates of the smaller (1-20 µm) and larger particle size fractions (20-80 µm) are presented as the mean annual percentage for the respective sites. The relative proportion of smaller TRWP (1-20 µm) is the highest in the two most remote sites and very similar among them (CHA: 39 % and PAY: 41 %). This is to be expected, since smaller particles have a lower settling velocity and can therefore be transported further away from their point of origin. Conversely, the relative proportion of larger TRWP (20-80 µm) is therefore highest at the location closest to road traffic (DUET: 90%). The urban and suburban sites show very similar proportions of smaller and larger TRWP (DUE: 21 % vs. 79%; ZUE: 25 % vs. 75 %). The result at MAG lies between PAY and DUE with a 35 % of smaller and 65 % of larger TRWP on the annual average.

The large contribution of TRWP in the size range of 1-20 µm at most of the sites indicates that it is important to include this size fraction in mass deposition studies. Neglecting this fraction would result in an underestimation of the mass deposition rates by approximately 20 - 40 %.

Additionally, even in remote areas a large proportion (approx. 60%) of TRWP are found in the size range 20-80 µm. This implies that those particles can be transported over long distances. The low traffic volume on smaller roads certainly also contributes to the atmospheric deposition of TRWP in rural areas.

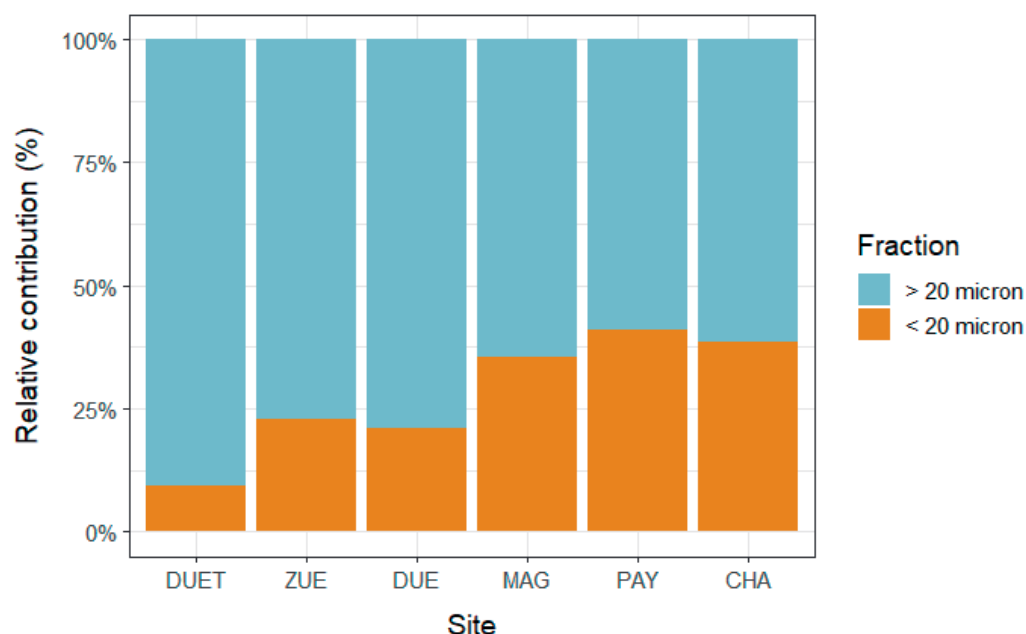


Figure 15: Proportion of larger tire and road wear particles (TRWP, 20–80 μm) compared to the smaller TRWP (1–20 μm) for the annual average dry mass deposition rates determined by SEM/EDX at each site (DUET: Dübendorf-Überlandstrasse, traffic; ZUE: Zurich-Kaserne, urban background; DUE Dübendorf-Empa, suburban; MAG: Magadino-Cadenazzo, rural; PAY: Payerne-Meteo-Swiss, rural; CHA: Chaumont, mountainous).

4.3 Comparison of dry and bulk deposition

For the comparison of the results presented in the previous sections 4.1 and 4.2 it should be noted that the measurement times only partially overlapped. In addition, the comparison of the annual dry and total mass deposition rates of TWP may be influenced by the different measurement methods used. However, at four out of six measurement sites, the ratio of dry to the total mass deposition rate of TWP was between 47 and 63% (derived from values in Table 2 and Table 3). This suggests that the atmospheric input of tire wear into the environment occurs in comparable proportions through dry deposition and through precipitation. In CHA and MAG, the ratio of dry to the total mass deposition rate of TWP were 23% and 87%, respectively. This deviation can hardly be explained by climatic differences but is more likely due the different sampling times and the different methods used.

4.4 Comparison with literature

In a recent study, SEM/EDX was applied to determine the dry MDR of TRWP collected at increasing distances to a highway in Germany (DTV $\approx$ 100'000–120'000).^[50] The dry MDR of TRWP was at a distance of 20 m from the highway 4'781 $\mu\text{g m}^{-2} \text{ day}^{-1}$ (calculated using the data provided in the study). Assuming that 50% of the TRWP mass is tire wear, the dry MDR at a 20 m distance from the German highway reduced to 2'391 $\mu\text{g m}^{-2} \text{ day}^{-1}$, which is almost three times higher than the dry MDR determined in this study for DUET, the suburban site located 20 m from a road with a DTV of 19'000. This

appears to be consistent with the results of our study, when the significantly higher traffic density is considered.

Another study determined the bulk MDR of tire wear particles at two urban background sites in Oslo and Trondheim (Norway) using Py-GC-MS.^[56] The mean MDR of TWP in Oslo and Trondheim were $557 \mu\text{g m}^{-2} \text{ day}^{-1}$ and $191 \mu\text{g m}^{-2} \text{ day}^{-1}$, respectively. The Py-GC-MS method applied in this Norwegian study is not fully described and likely different from the approach developed in this study. Nevertheless, the bulk MDR of TWP in Oslo is very similar to the values at ZUE and DUE, and the bulk MDR of TWP in Trondheim is comparable to those at the rural sites MAG and PAY.

Comparisons with data from literature are currently difficult because there is no widely accepted reference method and various measurement methods are used. However, based on the comparison with the two studies mentioned, we conclude that the TWP inputs into the environment determined in this study are plausible and correct within the stated uncertainties.

4.5 Annual deposition of TWP in Switzerland

The total annual deposition of tire wear in Switzerland was estimated based on the assumption that the measurement sites are representative for other locations of comparable land-use type. For this, the Swiss land use statistics with a spatial resolution of one hectare and 17 land use types (NOSAO4, Swiss Federal Office for Statistics) was used and combined with a height model in hectare resolution. For each grid cell, the land-use type and elevation were linked, and elevation was grouped into three classes, below 1000 m above sea level (m asl), between 1000 and 2000 m asl, and above 2000 m asl (Figure A 13). The 17 detailed land-use categories were then aggregated into two main classes – settlement and non-settlement – to quantify their area within each elevation zone.

The aggregation into these two main categories was due to the relatively small number of measuring stations that could be considered in this study. It is known that atmospheric dry deposition of particles depends on the surface type.^[57] The downward movement of particles in air occurs under the influence of gravity, buoyancy and surface winds that depend on the characteristics of the surfaces referred to as the surface roughness. The scientific literature further indicates that airborne particles can in vegetated areas such as forests be trapped by plant surfaces and reach the soil by through-fall and litter fall.^[58] This process occurs in addition to the direct dry settling of atmospheric particles that is measured in this study using passive samplers.

Keeping these limitations in mind, the atmospheric deposition measurements from the monitoring sites except DUET were assigned to land-use and elevation classes by assuming that settlement areas below 1000 m asl receive depositions as observed at the urban and suburban sites (ZUE and DUE) and non-settlement areas receive depositions as determined at the rural sites MAG and PAY. Settlement and non-settlement areas in the mid-elevation zone (1000–2000 m asl) are further assumed to be represented by the conditions measured at the mountainous station CHA. High-elevation areas (above 2000 m asl) were not included in this extrapolation as we did not measure MP at any site representative of such an area. For each hectare sized grid cell, the corresponding measured dry and bulk TWP mass deposition rates were multiplied by its

area and scaled to annual totals. Summing over all grid cells provided the total annual atmospheric dry and bulk TWP mass depositions for Switzerland (Table 4).

Annual dry and bulk TWP deposition estimation for Switzerland				
Height Class (m asl)	Land Use	Area (km ²)	Dry deposition (tons·year ⁻¹)	Bulk deposition (tons·year ⁻¹)
<1000	Settlement	2944	357	584
<1000	Non-Settlement	16403	921	1209
1000-2000	Settlement	315	4	16
1000-2000	Non-Settlement	11885	135	601
>2000	Settlement	13	n.d.	n.d.
>2000	Non-Settlement	9731	n.d.	n.d.
Total		41291	1417	2410

Table 4: Total estimated annual atmospheric dry deposition (SEM/EDX) and bulk deposition (Py-GC-MS) of tire wear in areas with different land uses at different altitudes in Switzerland.

We find that 2'410 tons of tire wear was deposited in Switzerland per year in areas with an altitude below 2000 m asl. The measurements from DUET show increased deposition along heavily trafficked roads, indicating that a proportion of the TWP produced on roads enters the environment in the immediate vicinity of the road via the air. The measurements from DUET were not used for the estimation of annual atmospheric TWP depositions in Switzerland. Therefore, the additional contribution of locally produced TWP to the atmospheric input into soils next to roads is not included in the annual deposition estimation for Switzerland.

Note that the atmospheric deposition of TWP in Swiss regions with an elevation above 2'000 m asl will only have a small contribution to the total annual mass deposition. Assuming that the deposition rate in these alpine areas is like the MDR in remote locations in Norway ($23.0 \mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ [56]), then an additional 82 tons of TWP per year would be deposited in Switzerland through the atmosphere.

The measurements with SEM/EDX, an analytical method that differs fundamentally from the Py-GC-MS method, yielded with the identical calculations a mean annual mass deposition of 1'417 tons of tire wear per year. These results suggest that approximately 60 percent of the atmospheric deposition of tire wear occurs through dry deposition, while approximately 40 percent is released into the environment through wet deposition. Unfortunately, a comprehensive assessment of the uncertainty or error of these estimates is not possible. While the measurement uncertainties of the analytical methods are known, the model error, i.e. the error arising from the assumption that the measured deposition rates are representative of settled and non-settled areas at different elevations, cannot be quantified. However, the estimate of annual TWP deposition in Switzerland is based on the analysis of thirteen samples per measurement site. Therefore, the random uncertainties of the analytical methods contribute only slightly to the uncertainties of the nationwide estimates. For the Py-GC-MS method, the contribution of the analytical method to the uncertainty of the estimated total annual TWP deposition in Switzerland is mainly determined by the systematic error, which ranges between -20% and +33% (see section 3.1.8).

To investigate the variability of atmospheric TWP deposition between similar sampling sites and to perform a rigorous uncertainty assessment for the estimation of atmospheric TWP deposition in Switzerland, would require measurements from a larger number of sampling sites. Measurements at a larger number of sampling sites should also cover different surface types. However, covering more sampling sites would be challenging with the sophisticated measurement methods used in this study. The detection of TRWP on optical microscopy images may provide an alternative approach offering higher sample throughput. Within this project, initial approaches for the identification and mass estimation of TRWP filtered on Anodisc filters based on image analysis tools have been developed and evaluated. Although the results are promising, further efforts into method validations are required to make the image analysis approach fit for purpose.

The mass deposition rates obtained in this study and the associated estimate of the amounts of TWP released into the environment through the atmosphere suggest that a larger fraction of the tire wear produced than generally assumed is distributed through the air. As mentioned in section 1, mass flow analyses predict that only about 1 to 5 % of tire wear is transported through the air and released into the environment further away from the roads where it originated.^[7,10,15,16] Depending on the total amount of tire wear generated annually in Switzerland, which ranges between 8'837 and 10'100 tons/year, the estimated annual atmospheric deposition of TWP of 2'410 tons/year corresponds to 24-27 % of the total amount.

In the study by Steiner (2023)^[9], the annual amount of tire wear generated in Switzerland was modelled and the amount of the tire wear that enters surface waters, roadside soils through runoff and the amount of tire wear that is retained in sewer systems were estimated. Zinc was used as a marker to determine tire wear in road runoff, which means that the quantitative conclusions drawn from these measurements are likely subject to considerable uncertainties for the reasons mentioned in section 3.1. However, the amount of tire wear released into the air was calculated from a mass balance, i.e. from the difference of the total amount of tire wear generated and the sum of the amounts of tire wear that enters surface waters, that is retained in various sewer systems, and the amount that enters roadside soils via runoff. According to Steiner (2023) 3'434 tons or 34 % of the total annual tire wear generated in Switzerland are getting airborne and enter the environment via the air. However, the study by Steiner (2023) did not differentiate between TWP deposited on surfaces next to the road and TWP that is carried by the wind and deposited at a greater distance from the point of origin. It is therefore not possible to directly compare the amount of airborne TWP reported by Steiner (2023) with the annual deposition of TWP estimated in this study.

Tire wear particles are environmentally persistent and widespread anthropogenic particulate contaminants of concern. TWP are often classified as microplastics, however, MP and TWP are typically quantified separately, because they require different analytics. The atmospheric deposition of MP other than TWP (for simplicity thereafter denoted as MP*) was quantified in a partner project funded by the Swiss Federal Office for the Environment (FOEN).^[26,27] Atmospheric deposition was collected at the same measurement sites (except DUET) and in parallel to this study. Particle-based focal plane array micro-Fourier transform infrared spectroscopy (FPA- μ -FTIR) was used as the analytical method. The mass deposition rates for MP* in the particle size range between 20 μm and 215 μm was found to be at the five considered sampling sites between 13 $\mu\text{g m}^{-2} \text{ day}^{-1}$ at Magadino-Cadenazzo and 53 $\mu\text{g m}^{-2} \text{ day}^{-1}$ at Zurich-Kaserne.^[27]

Because TWP in this study were measured across a 0 μm – 400 μm size range, the mass deposition rates for MP* were extrapolated to the same wider size range to ensure data comparability. It can then be concluded that atmospheric mass deposition rates of TWP were approximately ten times higher than those of MP*, i.e. microplastics other than TWP. Consequently, tire wear dominates the atmospheric input of microplastics into the Swiss environment.

4.6 Conclusions

This study provides a comprehensive, year-long assessment of the atmospheric deposition of tire wear at urban, suburban, rural and mountainous sites in Switzerland. The key results are the annual deposition rates of tire wear at these different types of sites, which help to improve exposure assessments and serve as a baseline for future studies. The analysis of TWP in environmental samples is labor-intensive and this study was therefore limited to a small number of study sites. Based on measurements from five sites, it was estimated that 2'410 tons/year of TWP enter the environment in Switzerland below 2000 m asl through atmospheric deposition. The atmospheric deposition rates of tire wear and the annual estimate for Switzerland as determined in this study show that atmospheric transport and deposition is an important pathway for the distribution of tire wear in the environment.

Future studies are recommended to increase the number of measurement sites in order to reduce the uncertainty in estimating the atmospheric deposition of TWP in a larger area such as the whole of Switzerland. Including several collection sites of the same type would also help to examine the representativeness of the sites selected.

Furthermore, it is recommended that the temporal development of the atmospheric input of TWP into the environment is quantified periodically, for example as part of regularly scheduled and targeted monitoring programs. In addition, the amount of tire wear in other environmental compartments in Switzerland such as soils and surface waters should be quantified. Currently, this information is largely lacking hampering exposure assessments.

An environmental monitoring program for TWP requires the continuous analysis of a large number of samples and should therefore be based on a cost-efficient analytical method. The analytical methods used in this study do not meet this requirement. While the Py-GC-MS based method is state of the art regarding the quantification of TWP in environmental samples, it is labor-intensive and requires dedicated analytical facilities and well-trained scientific staff.

The SEM/EDX-based method might be more cost-effective, but in its applied form, it is only suitable for determining dry atmospheric deposition, as the established workflow requires direct deposition of samples on polished boron crystals. Analysing TWP in wet or bulk atmospheric deposition, as well as in other environmental matrices, requires additional sample preparation, which also makes SEM/EDX less suitable for high-throughput analyses.

A cost-effective and suitable method for the environmental monitoring of TWP is currently not available. Within the scope of this study, the possibilities of optical methods for the identification and quantification of TRWP and the calculation of the mass of TWP in atmospheric deposition samples were investigated. This work was carried out as a supplement and financed through in-kind contributions. Although it was not

possible to validate the approach, initial results look very promising. However, the results obtained with an optical identification method should be regularly verified by using a method that allows for the unambiguous identification of TRWP based on its chemical composition, e.g. SEM/EDX and Py-GC-MS.

Despite the limitations described above regarding its suitability for routine analysis in a monitoring programme, it should be emphasized once again that, within the scope of this project, it was necessary to develop a reliable method for determining tire wear in environmental samples. We believe that this has been achieved with the Py-GC-MS method described in this report, and that this method represents an important contribution to the analysis of tire wear in the environment.

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Annex

A.1 Sampling protocol

Schedule of atmospheric deposition sampling							
Series	mtb/yr	DUET	ZUE	DUE	MAG	PAY	CHA
1	10/23		11.10.23	28.09.23	04.10.23	03.10.23	27.09.23
2	11/23		08.11.23	09.11.23	01.11.23	31.10.23	25.10.23
3	12/23		06.12.23	07.12.23	29.11.23	28.11.23	22.11.23
4	01/24		03.01.24	04.01.24	27.12.23	26.12.23	20.12.23
5	02/24		31.01.24	01.02.24	24.01.24	23.01.24	17.01.24
6	03/24		28.02.24	29.02.24	21.02.24	20.02.24	14.02.24
7	04/24		27.03.24	28.03.24	20.03.24	19.03.24	13.03.24
8	05/24		24.04.24	25.04.24	17.04.24	16.04.24	10.04.24
9	06/24	23.05.24	22.05.24	23.05.24	15.05.24	14.05.24	08.05.24
10	06/24	20.06.24	19.06.24	20.06.24	12.06.24	11.06.24	05.06.24
11	07/24	18.07.24	17.07.24	18.07.24	10.07.24	09.07.24	03.07.24
12	08/24	15.08.24	14.08.24	15.08.24	07.08.24	06.08.24	31.07.24
13	09/24	12.09.24	11.09.24	12.09.24	04.09.24	03.09.24	28.08.24
14	10/24	10.10.24	09.10.24	10.10.24	02.10.24	01.10.24	25.09.24
15	11/24	07.11.24	06.11.24	07.11.24	30.10.24	29.10.24	23.10.24
16	12/24	05.12.24	04.12.24	05.12.24	27.11.24	26.11.24	20.11.24
17	01/25	02.01.25	01.01.25	02.01.25	25.12.24	24.12.24	18.12.24
18	02/25	30.01.25	29.01.25	30.01.25	22.01.25	21.01.25	15.01.25
19	03/25	27.02.25	26.02.25	27.02.25	19.02.25	18.02.25	12.02.25
20	04/25	27.03.25	26.03.25	27.03.25	19.03.25	18.03.25	12.03.25
21	05/25	24.04.25	23.04.25	24.04.25	16.04.25	15.04.25	09.04.25
22	06/25	22.05.25	21.05.25	22.05.25	14.05.25	13.05.25	07.05.25
end		19.06.25	18.06.25	19.06.25	11.06.25	10.06.25	04.06.25

Table A 1: Dates of the start of the various series and the date of the end of atmospheric deposition sampling at the different locations. Note that sampling for one series ended on the same day that the following series were started (constant sampling duration of 28 days). Sampling at the various locations could not be synchronized due to logistical reasons. The series were assigned to the month and year (column mth/yr) that showed the greatest overlap at most locations. Also note that series 9 and 10 were both assigned to 06/24 (June 2024), the values for 06/24 in the figures are calculated as the mean of series 9 and 10. The fields marked in grey indicate sampling series, sites and dates when only dry deposition sampling and SEM/EDX analysis was conducted, the fields marked in green indicate sampling series, sites and dates when only bulk deposition sampling and Py-GC-MS analysis was conducted, and the fields marked in white indicate sampling series, sites and dates when all three sampling and analytical methods were performed. An exception is wet and dry deposition sampling and optical microscopy that was not conducted in DUET as indicated by the red frame.

A.2 Reference tire samples

Tire samples				
Sample ID	Manufacturer	Size	Season	Vehicle category
CMCTT-1	Orium	195/65 R15 95V	All season	Car/LDV
CMCTT-2	Bridgestone	215/45 R17	Winter	Car/LDV
CMCTT-3	Atlas	195/55 R16 87H	Winter	Car/LDV
CMCTT-4	Zeetex	205/55 R16	Winter	Car/LDV
CMCTT-5	Goodyear	225/50 R17 94H	Winter	Car/LDV
CMCTT-6	Eskimo Sava	175/65 R14	Winter	Car/LDV
CMCTT-7	Azenis	225/40Z R18 (92Y)	Summer	Car/LDV
CMCTT-8	Bridgestone	175/65 R14 82T	Summer	Car/LDV
CMCTT-9	StarMaxx	205/55 R16 91H	Winter	Car/LDV
CMCTT-10	Dunlop	205/55 R16 91V	Summer	Car/LDV
CMCTT-11	Continental	195/65 R15 T	Winter	Car/LDV
CMCTT-12	Continental	185/65 R15 T	Winter	Car/LDV
CMCTT-14	Star Performer	235/35Z R19	Winter	Car/LDV
CMCTT-14	Bridgestone	175/65 R14 82T	Summer	Car/LDV
CMCTT-15	Michelin	195/70 R15C	Winter	Car/LDV
CMCTT-16	Pirelli	205/55 R16	Winter	Car/LDV
CMCTT-17	Platin	205/45 R16	Summer	Car/LDV
CMCTT-18	Michelin	185/65 R15	Winter	Car/LDV
CMTTT-1	Bridgestone	385/65R 22.5	All season	Truck/Bus
CMTTT-2	Continental	275/70 R22.5	All season	Truck/Bus
CMTTT-3	Bridgestone	315/80 R22.5	Summer	Truck/Bus
CMTTT-4	Continental Re-thread	275/70R 22.5	Not known	Truck/Bus
CMTTT-5	Continental	315/80R 22.5	All season	Truck/Bus

Table A 2: Details of the used tire tread samples that were collected from local tire dealers, cryo-milled and used as reference material.

A.3 Tire-specific pyrolysis products

Pyrolysis products considered in this study

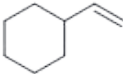
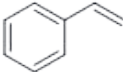
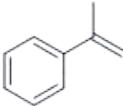
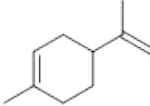
Pyrolysis Product	Structure	Quantifier ion (m/z)	Qualifier ion (m/z)	Retention time (min)	Tire rubber
Butadiene		54	53, 50	3.26	BR, SBR
Isoprene		68	53, 67	3.98	NR
Vinylcyclohexene (VCH)		54	79, 67	9.57	BR, SBR
Styrene		104	78, 103	10.84	SBR
α -Methylstyrene		118	103, 78	12.75	SBR
Limonene (Dipentene)		68	136, 107	13.57	NR

Table A 3: Identified tire-specific pyrolysis including retention time, quantifier and qualifier ion fragments.

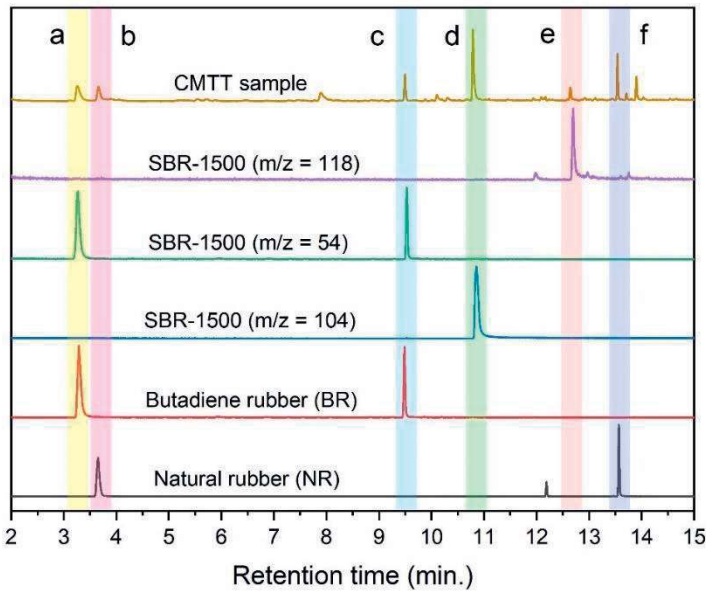


Figure A 1: Comparison of the characteristic pyrolysis markers (a = butadiene, b = isoprene, c = 4-vinylcyclohexene, d = styrene, e = α -methylstyrene, and f = limonene) observed in pure polymer standards with their corresponding pyrolysis products in a CMTT sample at 500 °C, indicating the origin of each marker in tire tread samples.

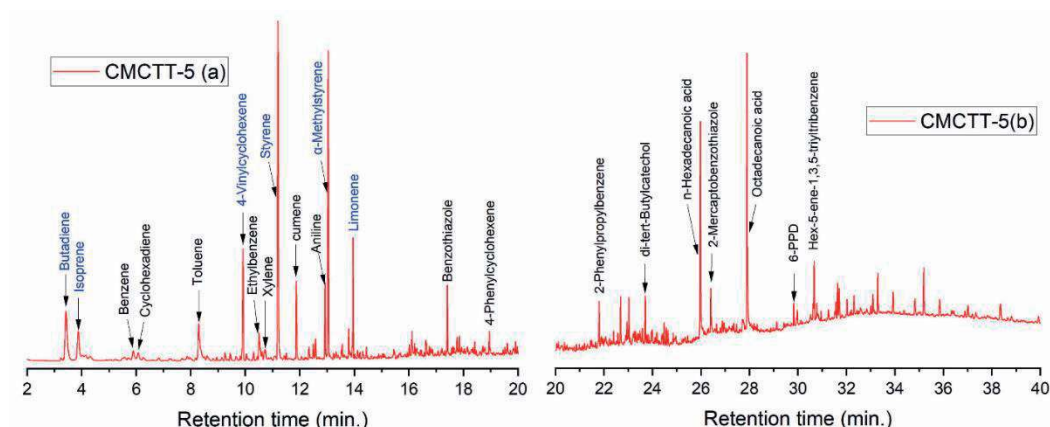


Figure A 2: Full chromatogram of the CMCTT-5 sample at a pyrolysis temperature of 500°C split in part (a) and (b) at a retention time of 20 min. The most prominent pyrolysis products are labelled.

A.4 SBS in road wear

One of the pyrolysis markers used (α -methylstyrene) is also formed during the thermal decomposition of styrene-butadiene-styrene (SBS), a compound found in polymer-modified bitumen and used in road asphalt.^[40] Simple calculations suggest that the amount of SBS used in road construction in Switzerland is small and its contribution is negligible.

In Switzerland, bitumen makes up 1% of the total mass of the road material.^[59] There is no literature available on the proportion of polymer bitumen in the total bitumen used in road construction in Switzerland. According to expert judgement about 20–40% of the bitumen used is polymer modified. Therefore, the road material consists of less than 0.4% polymer-modified bitumen. The SBS content of polymer-modified bitumen is about 5%.^[60] This means that, on average, road material and road wear contain less than 0.02% SBS.

Tire and road wear material each contribute about 50% to the mass of tire and road wear particles (TRWP).^[1] The tire tread material contains an average of about 11% styrene-butadiene-rubber (SBR), the polymer that also forms α -methylstyrene during pyrolysis. However, in an average TRWP, the SBR content in tire wear exceeds the SBS content in road wear by more than a factor of 100.

A.5 Pyrolysis marker signal vs. mass of CMTT

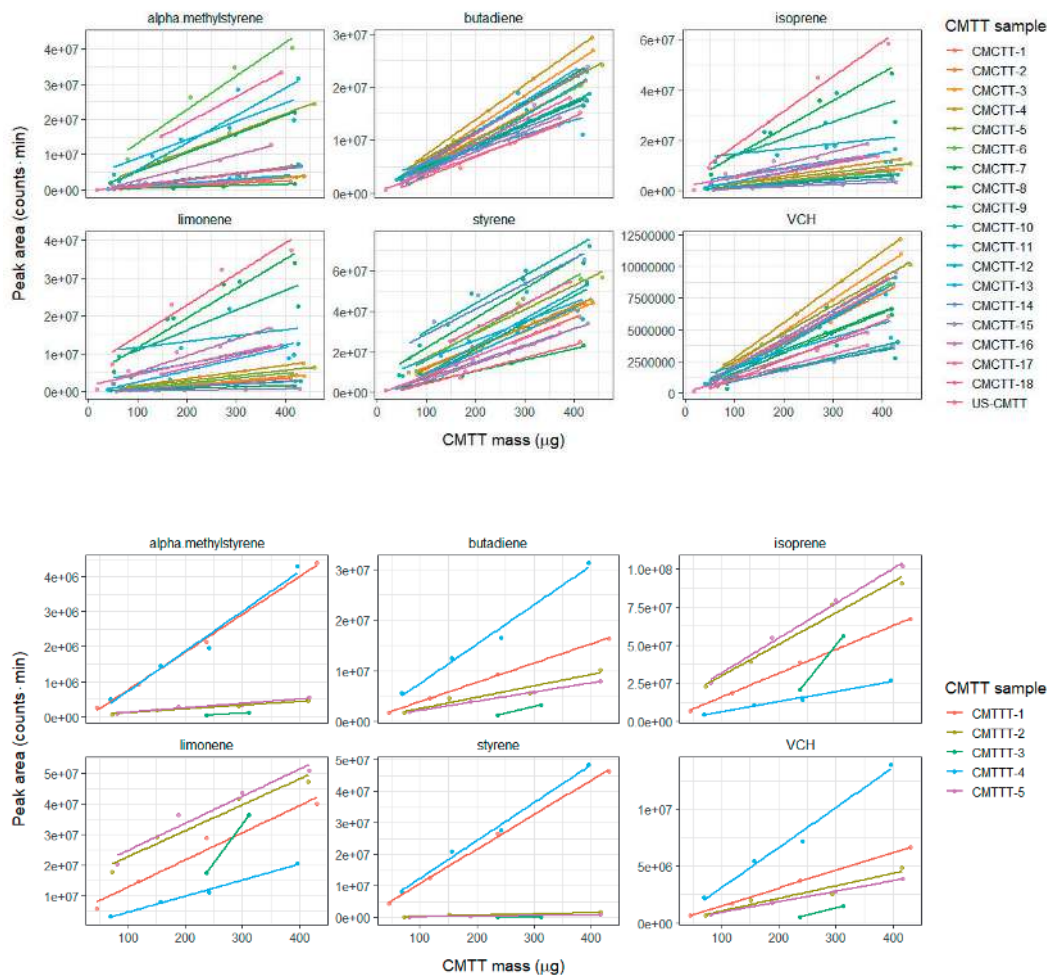


Figure A 3: Dependence of the measured peak areas for the six pyrolysis markers on the mass of cryo-milled tire tread (CMTT) from passenger car tires (CMCTT) and the US-CMTT mixture (top) and from truck tires (CMTTT, bottom). Note that for CMTTT-3 only measurements for two sample masses are available. The relationship between the signals of the pyrolysis markers and CMTT mass is highly linear, all coefficients of determination are $r^2 > 0.90$ with few exceptions, e.g. when the signal was low as for limonene for some of the passenger car tires.

Coefficient of determination (r^2) for Marker vs. CMTT mass per vehicle category

Marker	Car	Truck/Bus	All
VCH	0.70	0.34	0.61
α -methylstyrene	0.20	0.23	0.17
butadiene	0.86	0.31	0.71
isoprene	0.19	0.57	0.13
limonene	0.13	0.56	0.12
styrene	0.67	0.13	0.49

Table A 4: Coefficients of determination (r^2) expressing the degree of correlation between the six pyrolysis markers and the CMTT samples grouped by vehicle category.

A.6 Sum of peak areas (SPA) vs. mass of CMTT

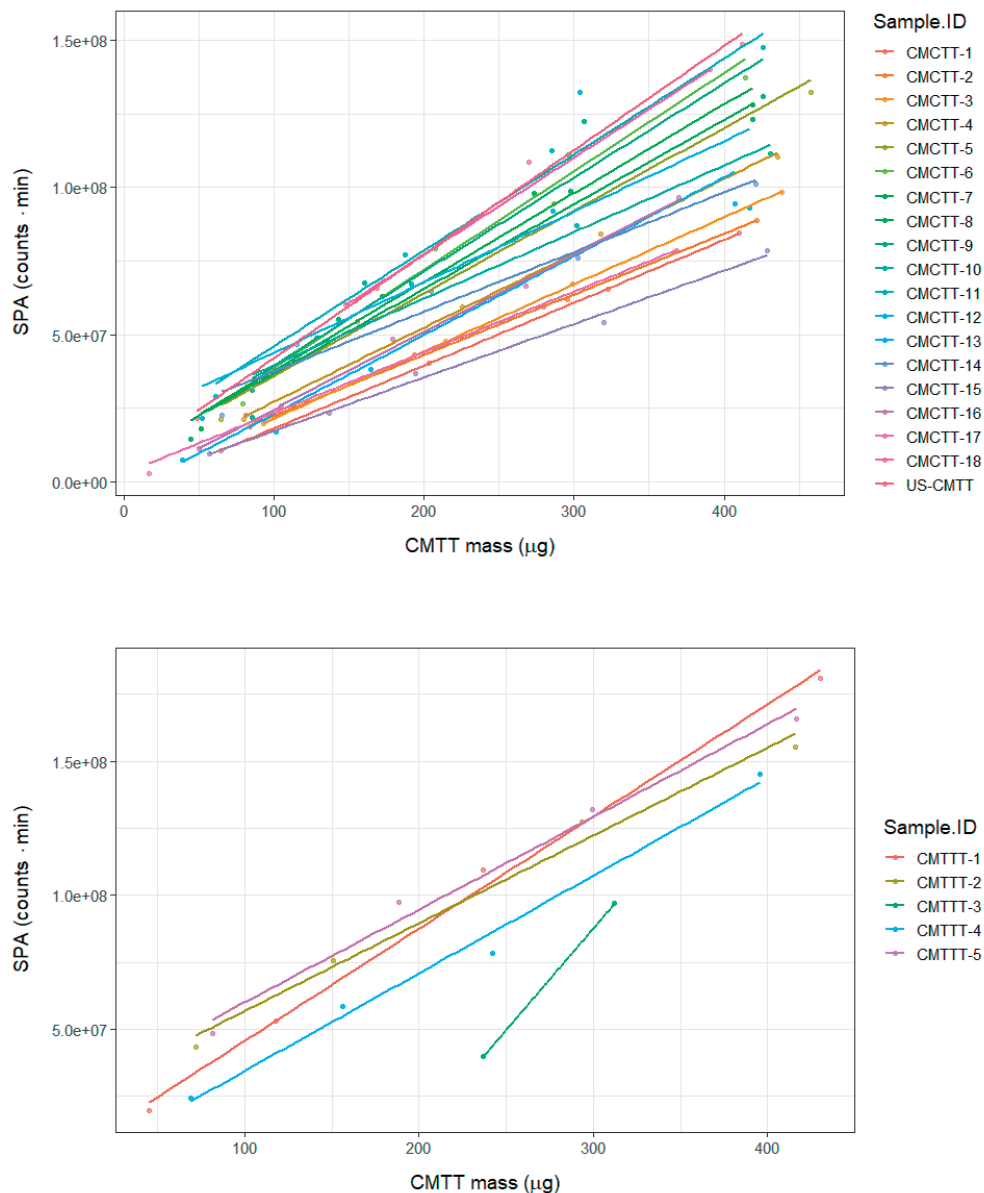


Figure A 4: Dependence of the sum of peak areas of all six pyrolysis markers (SPA) on the mass of cryo-milled tire tread (CMTT) from passenger car tires (CMCTT) and the US-CMTT mixture (top) and from truck tires (CMTTT, bottom). Note that for CMTTT-3 only measurements for two sample masses are available. The relationship between SPA CMTT mass is highly linear, all coefficients of determination are $r^2 > 0.90$ with the only exception of CMCTT-12 $r^2 = 0.66$.

A.7 Data processing of Py-GC-MS method

Py-GC-MS sensitivity correction

The peak area of each pyrolysis marker for every measurement of a CMTT sample of mass x (in μg) is denoted as $PA_{i,c}(x)$ and $PA_{i,s}(x)$, depending on whether the measurement stems from method calibration or from external controls carried out in conjunction with the measurement of atmospheric deposition samples. The index i denotes the pyrolysis marker, i.e., butadiene, isoprene, VCH, styrene, a-methylstyrene, and limonene ($i = 1, \dots, 6$). Figure A5 and Figure A6 show the peak areas $PA_{i,c}(x)$ and $PA_{i,s}(x)$ for CMCTT-5 and CMCTT-6 measured during method calibration ($PA_{i,c}(x)$) and the external controls ($PA_{i,s}(x)$) that are always measured together with the atmospheric deposition samples.

For all measurements of CMTT samples, linear regressions of $PA_{i,c}(x)$ and $PA_{i,s}(x)$ against the mass of CMTT x were calculated, $a'_{i,c}$ and $a'_{i,s}$ are the corresponding slopes of the regression lines describing the sensitivity of the measurement (change in peak area per μg of CMTT), and $b_{i,c}$ and $b_{i,s}$ are the intercepts representing possible offsets (peak area in the absence of CMTT).

$$PA_{i,c}(x) = a'_{i,c} \cdot x + b_{i,c} \quad (3)$$

$$PA_{i,s}(x) = a'_{i,s} \cdot x + b_{i,s}$$

All $b_{i,c}$ and $b_{i,s}$ obtained in this study did not deviate from zero (95% confidence). VCH was an exception in the calibration measurements, where a small intercept deviated from zero, as the measured peak areas for the different masses of CMCTT-6 lay on a straight line ($r^2=1.00$), see Figure A6. Table A5 shows the values for $a'_{i,c}$ and $b_{i,c}$, including the corresponding p values and coefficients of determination (r^2) for the measurement of CMCTT-5 and CMCTT-6 in the method calibration dataset. Since the intercepts $b_{i,c}$ and $b_{i,s}$ were generally negligible, the slopes were recalculated ($a_{i,c}$ and $a_{i,s}$), this time with the intercepts set to zero, i.e.

$$PA_{i,c}(x) = a_{i,c} \cdot x \quad (4)$$

$$PA_{i,s}(x) = a_{i,s} \cdot x$$

Changes in the sensitivity of the Py-GC-MS at the measurements of atmospheric deposition samples compared to the state during the calibration measurements can be compensated by a correction factor $c_{i,s} = a_{i,c}/a_{i,s}$, and the projection of the Py-GC-MS sensitivity behavior onto the sensitivity behavior during the calibration measurement can be written as follows

$$PA_{i,s}^p(x) = PA_{i,s} \cdot c_{i,s} \quad (5)$$

with $PA_{i,s}^p(x)$ the projected (or corrected) response (or peak area) for pyrolysis marker i when a mass x of CMTT sample s is analysed.

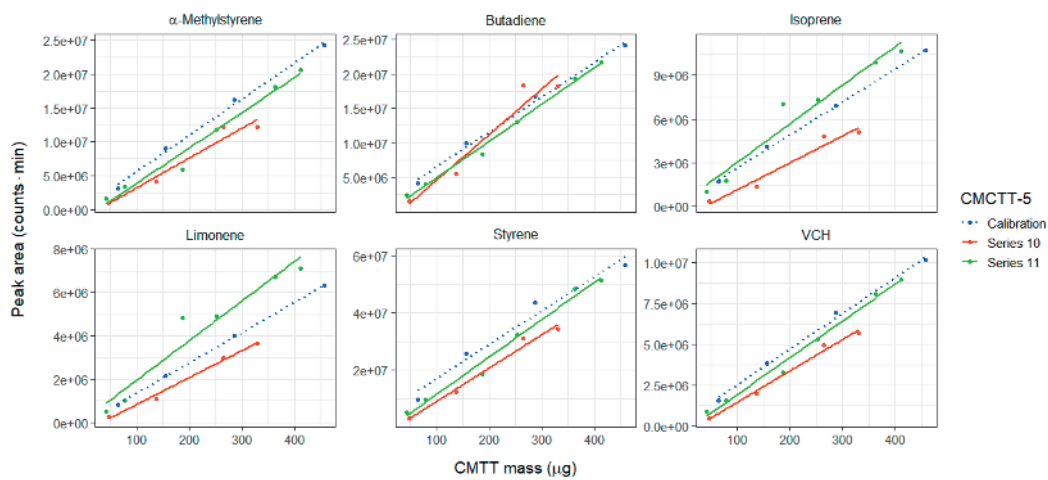


Figure A 5: Total peak area of the six pyrolysis markers against the mass of cryo-milled car tire tread (CMCTT)-5 in the method calibration dataset and in the external controls measured along the atmospheric deposition samples (series 10 and 11).

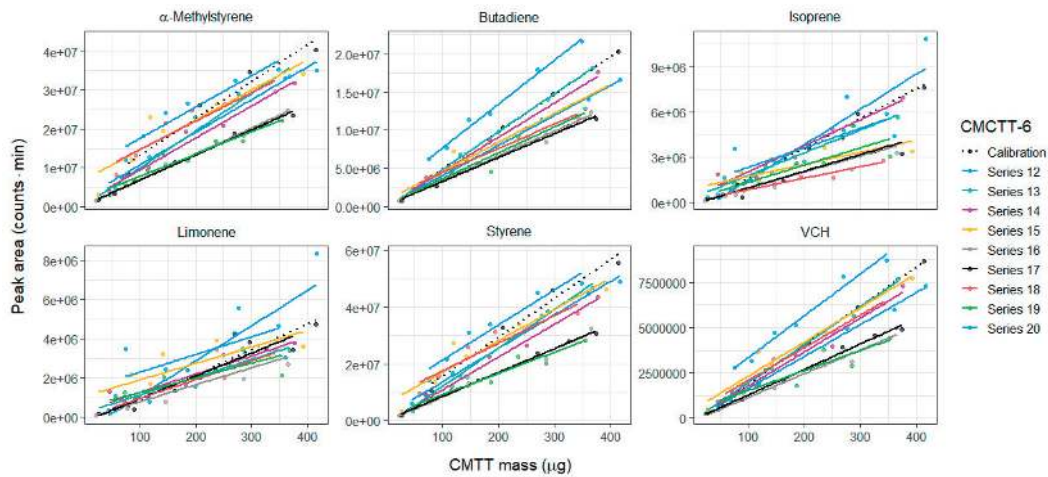


Figure A 6: Total peak area of the six pyrolysis markers against the mass of cryo-milled car tire tread (CMCTT)-6 in the method calibration dataset and in the external controls measured along the atmospheric deposition samples (series 12 to 20).

Linear regression analysis of calibration measurements						
Marker	CMTT	$a'_{i,c}$	p-value $a'_{i,c}$	$b_{i,c}$	p-value $b_{i,c}$	r^2
Buta e	CMCTT-5	50373	0.003	1571540	0.198	0.993
Isoprene	CMCTT-5	22645	0.001	391871	0.116	0.999
VCH	CMCTT-5	21861	0.002	340324	0.366	0.996
Styrene	CMCTT-5	118654	0.016	5302825	0.345	0.968
α -Methylstyrene	CMCTT-5	53274	0.002	312253	0.698	0.996
Limonene	CMCTT-5	13948	0.000	-38836	0.333	1.000
Butadiene	CMCTT-6	49097	0.000	42490	0.837	1.000
Isoprene	CMCTT-6	19781	0.004	-338229	0.432	0.992
VCH	CMCTT-6	21550	0.000	-263429	0.002	1.000
Styrene	CMCTT-6	136072	0.014	2360051	0.656	0.972
α -Methylstyrene	CMCTT-6	95237	0.026	3705556	0.485	0.948
Limonene	CMCTT-6	12953	0.012	-438368	0.392	0.975

Table A 5: Slope $a'_{i,c}$, intercept $b_{i,c}$, corresponding p-values, and coefficient of determination (r^2) for the linear regressions of measurements used for method calibration.

Measurement of atmospheric deposition

The measured peak areas of the six pyrolysis markers in atmospheric deposition samples ($PA_{i,AD,S}$ with $i = 1, \dots, 6$) were corrected based on the external controls that were measured with each series of atmospheric deposition samples. To account for sensitivity changes of the Py-GC-MS, the measured peak areas were multiplied with the corresponding correction factors $c_{i,S}$ (see equation 5), i.e.

$$PA'_{i,AD,S} = PA_{i,AD,S} \cdot c_{i,S} \quad (6)$$

If the measurement of external controls and application of equation 3 were to result in a significant intercept, an offset correction could be performed by using the estimated parameters from equation 3. However, this was not necessary in this study. The sum of peak areas is then calculated as

$$SPA_{AD,S} = \sum_{i=1}^6 PA'_{i,AD,S} \quad (7)$$

The sum of peak areas in individual atmospheric deposition samples can be converted to the mass of TWP (in μg) by

$$TWP_{AD,S} = \frac{SPA_{AD,S}}{r \cdot c_f} \quad (8)$$

with $c_f = 297311 \mu\text{g}^{-1}$ the calibration constant derived from the analysis of all available CMTT samples (CMCTT-1 to CMCTT-18 and CMTTT-1 to CMTTT-5) as explained

in the main text, and $r = 0.685$ the mean recovery of tire wear for the applied analytical method. Finally, the mass deposition rate MDR (in $\mu\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$) is calculated as

$$MDR_{AD,s} = \frac{TWP_{AD,s}}{A \cdot t} \quad (9)$$

where A is the area of the inlet of the glass container used for sample collection (0.00866 m^2), and t is the sampling time (28 days).

Uncertainty of sensitivity correction

The correction for changes of the Py-GC-MS sensitivity over time is not perfect. The calculated slopes ($a_{i,s}$) used for the correction have uncertainties which are, however, available as the standard errors of the regression analyses. These standard errors can be regarded as random and independent and are combined when the responses of all six pyrolysis markers are summed. The sensitivity corrected sum of peak areas from CMTT samples measured as external controls can be written as

$$SPA_s(x) = \sum_{i=1}^6 PA_{i,s}^p(x) \quad (10)$$

The corresponding absolute standard uncertainty $u_s(x)$ of $SPA_s(x)$ for the correction of a measurement of a CMTT sample s can be calculated as

$$u_s(x) = \sqrt{\sum_i u_{i,s}^2(x)} \quad (11)$$

with $u_{i,s}(x)$ the absolute standard uncertainty for pyrolysis marker i . Note that $u_s(x)/x$ is constant and the standard uncertainty for $SPA_s(x)$ can also be expressed as a relative standard uncertainty σ_s . We measured thirteen series of CMCTT-6 in total, each on different days spanning a one-year period and each consisting of typically five different and known sample masses. The measurements of external controls allow the estimation of $u_s(x)$. Figure A 7 shows the sensitivity corrected sum of peak areas for all measurements of CMCTT-6 including the expanded uncertainty due to sensitivity correction, i.e. $\pm 2 \cdot u_s(x)$. The range of the relative standard uncertainties for all thirteen series was $u_s = 4.6 - 8.1 \%$, we consider the upper value of 8.1% as a conservative and realistic estimate for the uncertainty introduced by sensitivity correction.

The measured SPAs(x) of measurements can be converted to the corresponding CMTT mass using the relationship (or slope) of the measured sum of peak areas and CMTT mass in the calibration data set as the reference and proceed as outlined in sections 5.6.1 and 5.6.2.

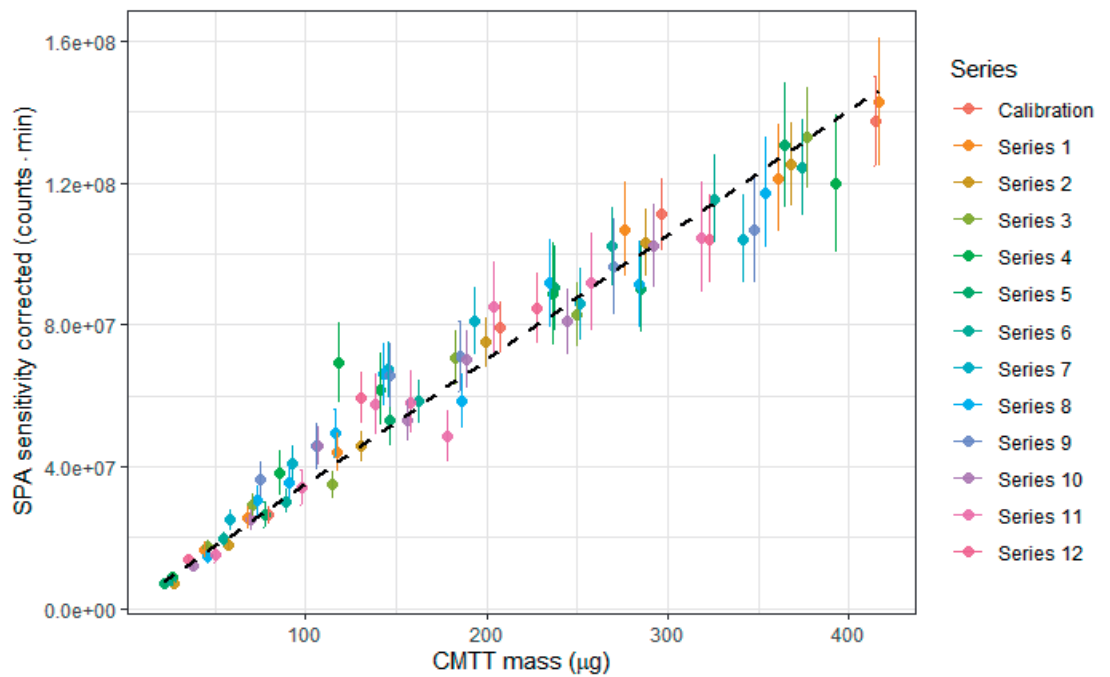


Figure A 7: Sensitivity corrected sum of peak areas of the six pyrolysis markers for the analysis of CMTT samples of known and variable mass (all CMCTT-6). The measurements shown are those of the calibration data set and the measurements of external controls, which were carried out in twelve series over a period of one year. The broken line indicates the linear regression of the measurements in the calibration data set, i.e. the measurements that served as the reference for the applied sensitivity correction. The error bars indicate the uncertainty of the sensitivity correction at 95% confidence level. Most measurements overlap with the reference line, indicating a realistic estimation of $us(x)$, the absolute standard uncertainty associated with the sensitivity correction. The reference line can be used to convert the sensitivity corrected SPA into CMTT mass.

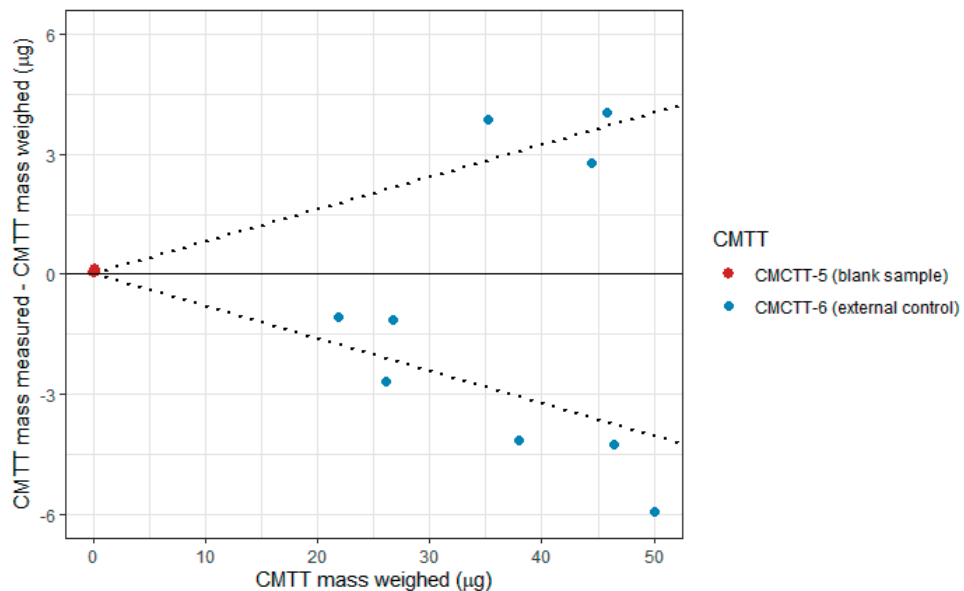


Figure A 8: Difference of CMTT mass derived from sensitivity corrected SPA and the (true) weighed mass of CMTT for the available measurements of blanks and CMTT at low masses (denoted as residuals). The measurements of the seven blank samples (red dots) appear at this scale of the y-axis on top of each other. The CMTT measurements of CMCTT-6 are those shown in Figure S12 with CMTT masses below 50 mg. The dotted lines indicate the relative uncertainty of us of $\pm 8.1\%$.

A.8 PCA of CMTT sample data

Principal component analysis (PCA) was done using the *prcomp* function in R.^[61] The input data (measurement of the six pyrolysis markers in CMTT samples and in atmospheric deposition samples) were normalised to one (i.e. converted into compositional data) and centered for performing PCA.

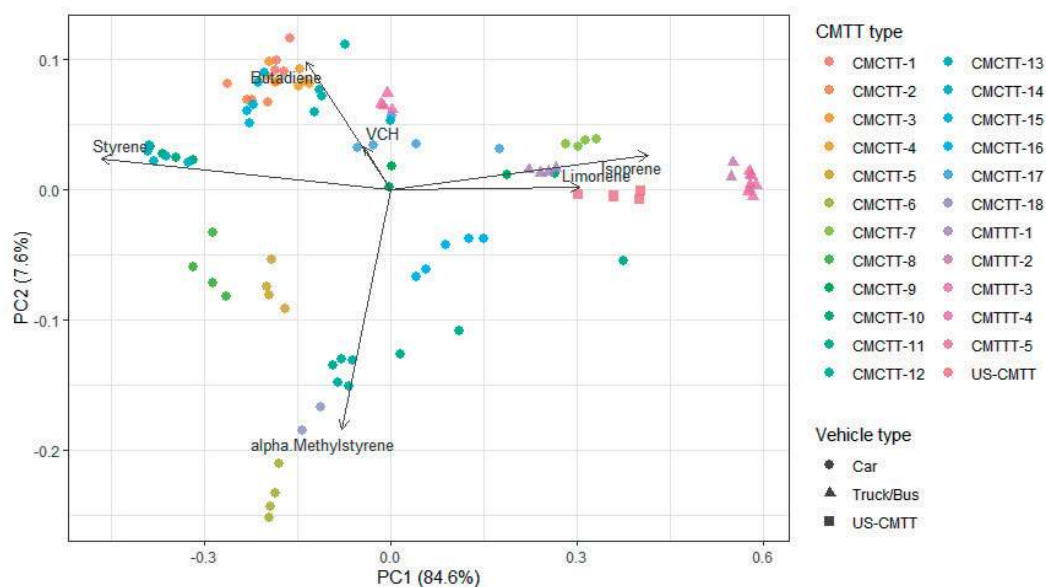


Figure A 9: Biplot showing the scores of the measured CMTT samples in the plane spanned by the first two principal components (PC1 and PC2).

A.9 TRWP detection using image analysis

Method 1 (HSV color threshold)

Figure A 10 shows the different steps of the applied method based on thresholds in the HSV colour space. After image-preprocessing including removal of background noise, a binary image (white particles on black background) was generated. The binary image allowed the calculation of the contours of all particles, and their characteristics (e.g. size). The contours of the identified particles were transferred to the optical image and the mean colour of each individual particle in the HSV colour space was determined. Finally, the mean colour of each particle was evaluated against the lower and upper thresholds defined for the particles of interest, here TRWP.

The results are promising, but in addition to method validation, there are still open issues that need to be addressed. In particular, the current implementation cannot deal with aggregated particles, as the mean colour of an aggregate can be meaningless. Therefore, an image processing step for particle disaggregation should be included.

The optical images of wet and dry atmospheric deposition samples from ZUE, DUE, PAY, MAG and CHA from series 10 to 14 (see Table A 1) were analysed using method 1. The determined TRWP mass deposition rates showed decreasing values from the urban to the rural sites. However, the corresponding TWP mass deposition rates,

assuming a mass fraction of 50%/50% tire tread and road wear, were lower than the values determined with Py-GC-MS and SEM/EDX.

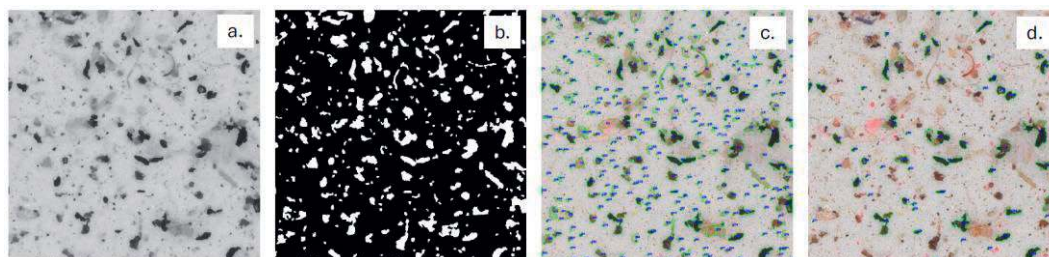


Figure A 10: Illustration of the different steps of TRWP detection based on thresholds in the HSV colour space. Section of the full image of an atmospheric deposition sample after image preprocessing (a.), the corresponding binary image (b.), contours (green) and labels of all identified particles (c.), and finally the contours of all particles identified as TRWP.

Method 2 (YOLO11)

The second implemented and applied method is based on the AI model YOLO11, which can be used for object recognition.^[55] For the detection of TRWP, a YOLO11 model was trained using optical images of particles in atmospheric deposition samples, which experts classified as TRWP or not based on their shape and colour. This is certainly a clear weakness in the way we applied this method. In future applications, the model should be trained with a dataset of particles that have been classified as TRWP or not TRWP using a specific method. In principle, any method that can characterize individual TRWP would be suitable both for generating a training dataset and, in a later step, for validating the results obtained with a YOLO11 model. However, a method that can be applied to the same filter samples used for optical image analysis is of great advantage. Such a method includes attenuated total reflection Fourier transform IR (ATR-FTIR) spectroscopy, element analysis using micro-X-ray-fluorescence (μ XRF) or laser induced breakdown spectroscopy (LIBS).

Figure A 11 shows a section of an optical image of a dry atmospheric deposition sample from the suburban site in Duebendorf (DUE), the particles identified as TRWP are marked. All samples of wet and dry atmospheric deposition collected and processed within the partner project on microplastics excluding tire wear were analysed for TRWP using the YOLO11 model. The results were similar to those of the HSV colour threshold approach (method 1). Again, the determined TWP mass deposition rates were lower than those from Py-GC-MS and SEM/EDX.

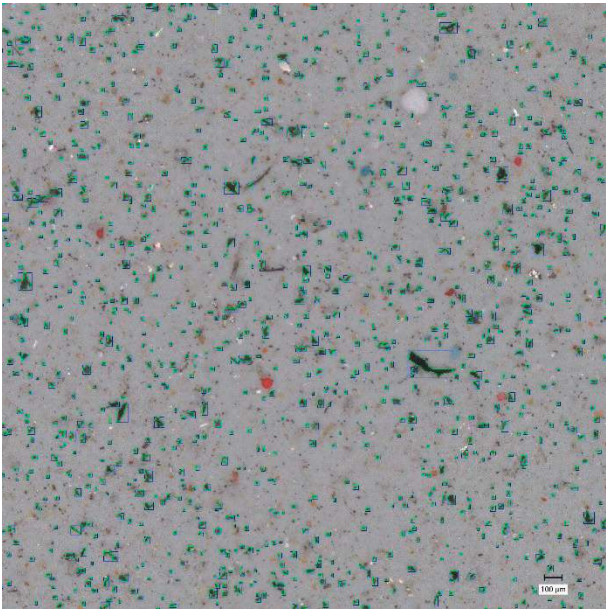


Figure A 11: Section of an optical image (50x magnification, digital microscope VHX-7000, Keyence, Japan) of the dry atmospheric deposition sampled from 15.08.24 to 12.09.24 at the suburban site in Duebendorf (DUE) and filtered onto an AlO₂-membrane filter (Whatman Anodisc, Cytiva, Germany). The particles classified as TRWP by the YOLO11 model are shown by the green contours and the blue bounding areas. The round red particle in the upper half of the image can be used as a size scale. It has a diameter of 50 µm.

A.10 Comparison of Py-GC-MS and SEM/EDX

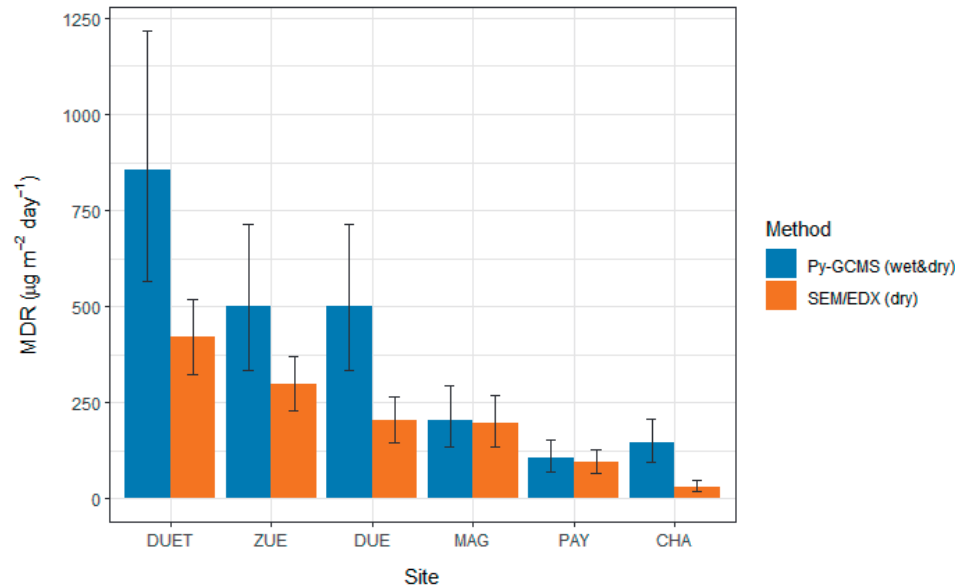


Figure A 12: Comparison of the mean TWP-MDR for the times when parallel measurements are available (Series 10 to Series 14 for most sites, see Table A 1). The measured values were determined by the Py-GC-MS method and by SEM/EDX. Note that the samples analysed with Py-GC-MS are bulk deposition samples, i.e. the sum of wet and dry deposition, the samples analysed by SEM/EDX are dry deposition only. The agreement between the values determined using the different methods is reasonable and plausible: Both methods show the same order of measurement sites with respect to the values of the mass deposition rates of TWP, the bulk atmospheric deposition as measured with Py-GC-MS is larger than dry deposition determined using SEM/EDX. Note that the values given for dry deposition were calculated from the measured deposition rates of TRWP, assuming a mass fraction of TWP to TRWP of 50 %.

A.11 Mass deposition rates

The determined bulk and dry mass deposition rates of tire wear particles and in all collected four weekly samples are shown as box and whisker plots in Figure A 13 and Figure A 14.

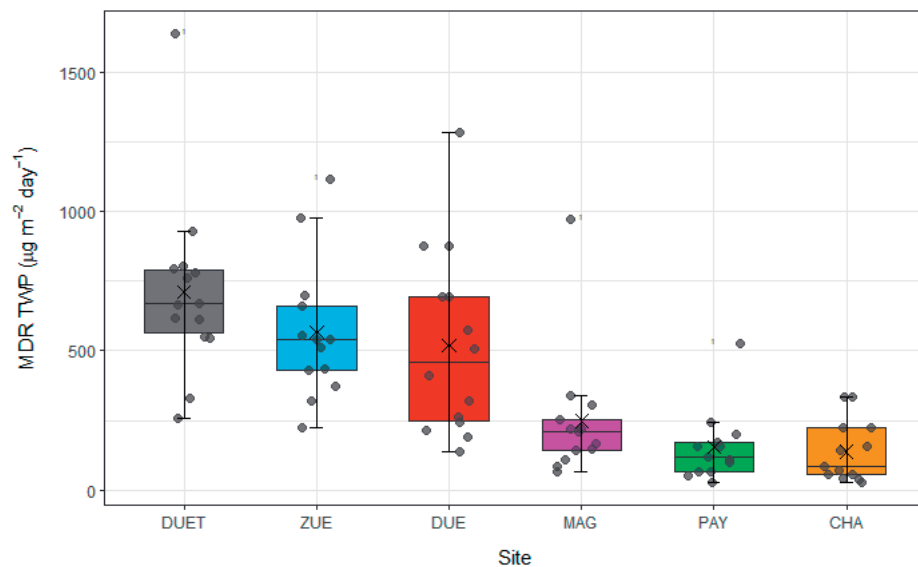


Figure A 13: Box and whisker plot of the atmospheric bulk mass deposition rates of TWP determined from the analysis of the four weekly bulk deposition samples collected at the considered measurement sites. The figure shows the distribution of the measured data (grey dots). The width of the boxes corresponds to the first and third quartiles, and the horizontal lines in the boxes to the medians. The mean of the determined mass deposition rates of TWP at each site are also shown (crosses). The whiskers extend to the lowest and highest measured value that lie within 1.5 of the difference between the third and the first quartiles.

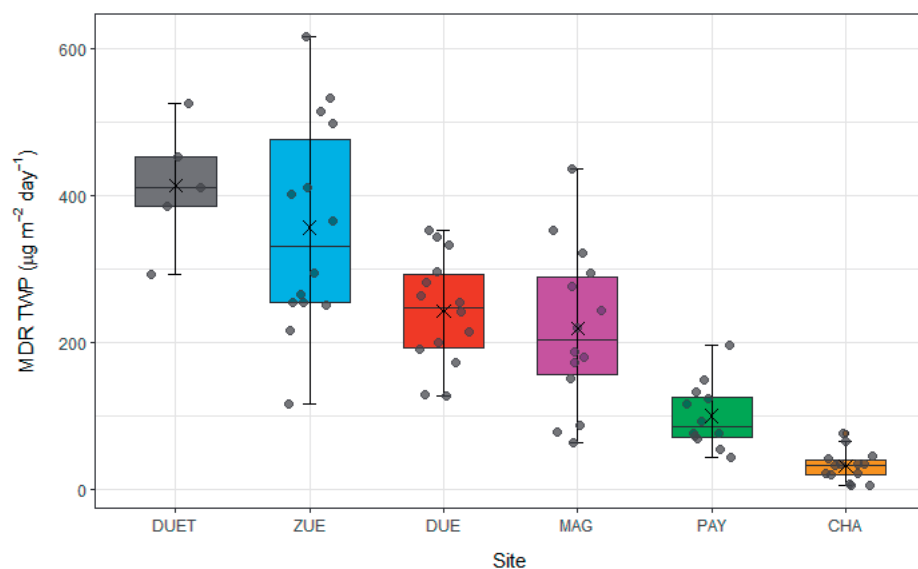


Figure A 14: Box and whisker plot of the atmospheric dry mass deposition rates of TWP determined from the analysis of the four weekly dry deposition samples collected at the considered measurement sites. The figure shows the distribution of the measured data (grey dots). The values given were calculated from the measured mass deposition rates of TRWP, assuming a mass fraction of TWP to TRWP of 50%. The width of the boxes corresponds to the first and third quartiles, and the horizontal lines in the boxes to the medians. The mean of the determined mass deposition rates of TWP at each site are also shown (crosses). The whiskers extend to the lowest and highest measured value that lie within 1.5 of the difference between the third and the first quartiles.

A.12 Land-use data

The Swiss land-use statistics with a spatial resolution of one hectare and 17 land-use types (NOSAO4, Swiss Federal Office for Statistics) was used and combined with a height model in hectare resolution to calculate the total area of different land use types in three altitude classes (< 1000 m a.s.l., 1000–2000 m a.s.l., > 2000 m a.s.l.). Figure A 13 shows the combination of the original 17 land-use categories with the height model. The data shown in Figure A 13 was aggregated into two main categories (settlement and non-settlement) for the calculation of the mean annual mass deposition of TWP in Swiss areas below 2000 m a.s.l. (see Table 4 in the main text).

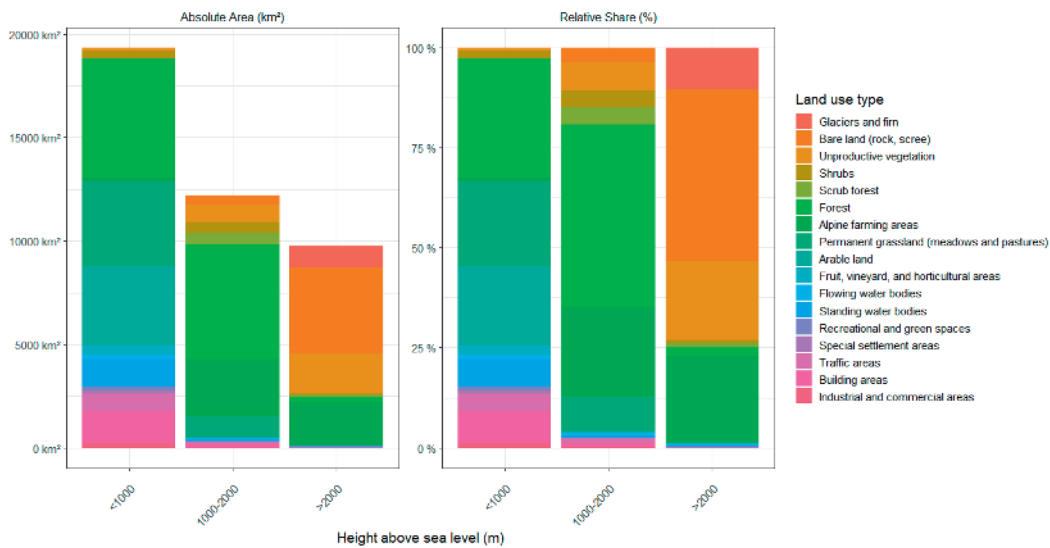


Figure A 15: Absolute area and relative share of areas of seventeen land use types in three altitude classes (< 1000 m a.s.l., 1000–2000 m a.s.l., > 2000 m a.s.l.) in Switzerland (data source: Federal Office for Statistics).

Use of data

The data will be provided by the authors upon request.

Project conclusion



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FORSCHUNG IM STRASSENWESEN DES UVEK

Version vom 09.10.2013

Formular Nr. 3: Projektabschluss

erstellt / geändert am: 06.08.2026

Grunddaten

Projekt-Nr.: TRU_20_00E_01

Projekttitel: Atmosphärischer Eintrag von Reifenabrieb in der Schweiz

Enddatum: 31.10.2026

Texte

Zusammenfassung der Projektergebnisse:

Reifenabrieb ist die Hauptquelle von Mikroplastik in der Umwelt. Ein grosser Anteil des gebildeten Reifenabriebs wird über die Abwasserbehandlungsanlagen zurückgehalten, ein wesentlicher Anteil des Reifenabriebs gelangt jedoch in die Umwelt. Reifenabrieb wird dabei nicht nur im strassennahen Raum in die Umwelt eingetragen, sondern kann auch über die Luft grossräumig verteilt werden. Die Menge Reifenabrieb, die über die Luft an strassenfernen Orten in die Umwelt eingetragen wird, wurde bisher kaum untersucht und war daher weitgehend unbekannt. Dadurch fehlten wesentliche Grundlagen für die Beurteilung der Belastung der Umwelt durch Reifenabrieb. In diesem Projekt wurden die Einträge von Reifenabrieb an sechs Standorten in der Schweiz während eines Jahres bestimmt. Die ausgewählten Standorte repräsentieren dabei Gebiete unterschiedlicher Landnutzung (verkehrsnahe, städtisch, vorstädtisch und ländlich). An allen Standorten ist Reifenabrieb im atmosphärischen Eintrag nachweisbar. Die Menge von Reifenabrieb, die über die Atmosphäre eingetragen wird hängt dabei stark von der Nähe zu Strassen ab: Die höchsten Eintragsraten wurden am verkehrsnahen Standort ($\sim 700 \mu\text{g}/\text{m}^2/\text{Tag}$) gemessen, gefolgt von den städtischen und vorstädtischen Standorten ($\sim 500\text{--}600 \mu\text{g}/\text{m}^2/\text{Tag}$). Tiefere Eintragsraten von Reifenabrieb wurden an zwei ländlichen Standorten ermittelt ($\sim 150\text{--}250 \mu\text{g}/\text{m}^2/\text{Tag}$), die niedrigsten Werte ergaben sich aus den Messungen an einem erhöht gelegenen ($1'140 \text{ m ü.M.}$) ländlichen Standort ($\sim 140 \mu\text{g}/\text{m}^2/\text{Tag}$).

Die Resultate von den verschiedenen Standorten wurden verwendet, um den gesamten jährlichen Eintrag von Reifenabrieb in der Schweiz abzuschätzen. Dabei wurde angenommen, dass die Einträge von Reifenabrieb an den gewählten Standorten repräsentativ für andere Standorte in der Schweiz mit ähnlicher Landnutzung sind. Da keine Messungen an einem Standort im alpinen Raum durchgeführt wurden, beschränkte sich die Abschätzung für die Gebiete der Schweiz unterhalb von $2'000 \text{ m ü.M.}$ Der auf diese Weise ermittelte gesamte jährliche atmosphärische Eintrag von Reifenabrieb in der Schweiz (unterhalb von $2'000 \text{ m ü.M.}$) beträgt $2'410 \text{ Tonnen}$. Dies entspricht etwa $24\text{--}27 \%$ des gesamten Reifenabriebs, der jährlich auf den Strassen in der Schweiz erzeugt wird ($8'800 \text{ bis } 10'100 \text{ Tonnen pro Jahr}$). Dieser Anteil des atmosphärischen Eintrags an der gesamten erzeugten Menge Reifenabrieb belegt die wichtige Rolle der Atmosphäre bei der Verteilung von Reifenabrieb in der Umwelt.



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Zielerreichung:

Die Ziele des Projektes wurden erreicht. Dieses Projekt liefert eine umfassende, einjährige Bewertung des atmosphärischen Eintrags von Reifenabrieb an städtischen, vorstädtischen, ländlichen und gebirgigen Standorten in der Schweiz. Die wichtigsten Ergebnisse sind die jährlichen Eintragsraten von Reifenabrieb an diesen verschiedenen Standorttypen. Sie tragen zu einer verbesserten Bewertung der Exposition gegenüber Reifenabrieb bei und können als Grundlage für zukünftige Studien dienen.

Zeitgleich wurde ein vom BAFU gefördertes Projekt durchgeführt, in dem an den gleichen Stationen der atmosphärische Eintrag von Mikroplastik ohne Reifenabrieb bestimmt wurde. Durch Kombination der Resultate beider Projekte lässt sich zeigen, dass Reifenabrieb in der Schweiz den Eintrag von Mikroplastik in die Umwelt über die Luft dominiert.

Folgerungen und Empfehlungen:

Die Analyse von Reifenabrieb in Umweltproben ist arbeitsintensiv. Deshalb beschränkte sich diese Studie auf eine kleine Anzahl von Untersuchungsstandorten. Zukünftige Studien sollten die Anzahl der Messstandorte erhöhen, um die Unsicherheit bei der Schätzung des atmosphärischen Eintrags von Reifenabrieb in einem größeren Gebiet wie der gesamten Schweiz, zu verringern. Darüber hinaus wird empfohlen, die zeitliche Entwicklung des atmosphärischen Eintrags von Reifenabrieb in die Umwelt zu verfolgen, beispielsweise im Rahmen eines regelmässigen und zielgerichteten Monitorings. Zusätzlich sollte der Reifenabrieb in anderen Umweltkompartimenten der Schweiz, wie Böden und Oberflächengewässern, bestimmt werden. Derzeit fehlen diese Informationen weitgehend, was Expositionsbewertungen erschwert. Ein Monitoring von Reifenabrieb erfordert die Analyse einer grösseren Anzahl Proben von unterschiedlicher Art und setzt daher eine flexible kosteneffiziente Analysemethode voraus. Die in diesem Projekt eingesetzten Methoden sind aufwändig und nur bedingt geeignet. Es sollten Anstrengungen unternommen werden, um eine Analysemethode zu entwickeln, die sich für ein Umweltmonitoring eignet.

Publikationen:

Rahman, Md Atiur, Kaegi, Ralf, Hueglin, Christoph (2026). Development and application of an analytical method for the quantification of tire wear in atmospheric deposition. Atmospheric Measurement Techniques, eingereicht.

Hueglin, Christoph; Jaramillo-Vogel, David; Rahman, Md Atiur; Rausch, Juanita; Kaegi, Ralf (2026) Atmospheric Deposition of tire wear in Switzerland, Atmospheric Chemistry and Physics, in Vorbereitung.

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Name: Hüglin

Vorname: Christoph

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Unterschrift des Projektleiters/der Projektleiterin:



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Beurteilung der Begleitkommission:

Beurteilung:

Das vorliegende Projekt liefert erstmals belastbare experimentelle Daten zu atmosphärischer Deposition von Reifenabrieb in der Schweiz über Zeit und Raum. Die Befunde erlauben es, bisherige Modellabschätzungen zu Verfrachtung und Einträgen über die Atmosphäre besser einzuordnen. Beispielsweise konnte ein vermuteter Gradient in den Depositionsraten von urbanen zu ruralen Standorten bestätigt werden. Nichts desto trotz wurde aber auch in ländlichen und erhöht gelegenen Standorten noch gut 100 µg Reifenabrieb pro m² und Tag deponiert. Mit rund 25% lag der über diese Messungen abgeschätzte Anteil des gesamten jährlich produzierten Reifenabriebs deutlich höher als bisherige, überwiegend auf Modellen beruhenden Werte. Ein Vergleich mit der atmosphärischen Deposition von anderen Formen von Mikroplastik ergab zudem eine mengenmässige Dominanz von Reifenabrieb. Insgesamt tragen die in diesem Projekt gewonnen Erkenntnisse deshalb massgeblich zu einem verbesserten Verständnis des Vorkommens und Verhaltens von Reifenabrieb in der Atmosphäre im Speziellen und der Massenbilanzierung in der Umwelt im Allgemeinen bei.

Umsetzung:

Die analytischen Herausforderungen zur Quantifizierung von Reifenabrieb sind beträchtlich und nahmen einen grossen Teil der im Projekt verfügbaren Ressourcen in Anspruch. Der investierte Aufwand hat sich aber gelohnt, denn erst mit der Integration und gemeinsamen Auswertung einer grösseren Anzahl von molekularen Markern, dem Einbezug von Qualifier-Ionen in die Pyrolyse-GC/MS und der Bestimmung von Messunsicherheiten über die gesamte analytische Kette konnte eine reproduzierbare und robuste analytische Methode entwickelt werden. Dies wurde auch durch die orthogonale Bestätigung der damit erhaltenen Resultate durch SEM/EDX untermauert. Das Projekt war von Anbeginn weg solide geplant, wurde systematisch abgearbeitet und hat schliesslich seine Ziele vollumfänglich erreicht.

weitergehender Forschungsbedarf:

Das in diesem Projekt entwickelte und angewendete analytische Vorgehen ist relativ anspruchsvoll und für den routinemässigen Einsatz noch nicht geeignet. Die Pyrolyse-GC/MS erwies sich im täglichen Betrieb noch als zu aufwändig in der Wartung und anfällig für Störungen. Ein Teil der aufgetretenen Probleme mag dem etwas in diese Jahre gekommenen Gerät geschuldet worden sein, welches nicht mehr ganz dem neuesten Stand der Technik entsprach. Alternative analytische Vorgehensweisen wie etwa über bildgebende Verfahren wären hier vielleicht vielversprechender und zu prüfen.

Einfluss auf Normenwerk:

Die in diesem Projekt erhobenen Daten können als Basis für künftige umfangreichere und systematischere Monitoring-Studien dienen, allerdings bleibt eine entsprechende analytische Referenzmethode basierend auf (oder auch ergänzend zu) den im vorliegenden Projekt angewandten noch zu entwickeln.

Der Präsident/die Präsidentin der Begleitkommission:

Name: Bucheli

Vorname: Thomas

Amt, Firma, Institut: Agroscope

Unterschrift des Präsidenten/der Präsidentin der Begleitkommission:



Bucheli Thomas 0GJ00G
12.08.2026

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