Research and Innovation action

NUMBER — 955387 — LEON-T

LEON-T

Low particle Emissions and lOw Noise Tyres

Tyre Wear Particles and Microplastics Dispersion in the Environment

Deliverable No.	3.5	
Deliverable Title	Tyre Wear Particles and Microplastics Dispersion in the Environment	
Dissemination	PU	
Written by	Joris T.K. Quik, Anne Hids, Joris Meesters (RIVM) Peter Tromp, Luke Parker (TNO)	4/12/2024
Checked by	Barouch Giechaskiel	19/11/2024
Approved by	Juan J García (IDIADA)	6/12/2025
Issue date	6/12/2024	

This Project has received funding from the European Union´s Horizon 2020 research and innovation programme under grant agreement Nº 955397.
The content of this report reflects only the author´s view CINEA is not reponsible for any use that may be made of the information it contains.

Revision history

REVISION	DATE	DESCRIPTION	AUTHOR (ORGANIZATION)
0	17/11/2024	First draft	Joris Quik (RIVM)
1	19/11/2024	Comments	Barouch Giechaskiel (JRC)
2	04/12/2024	Revision and update model output	Joris Quik (RIVM)
3	06/12/2024	Revisions based on input Juan J. Garcia	Joris Quik (RIVM)

Contents

ABBREVIATIONS AND GLOSSARY

A2 / A27 / A28	Highways in The Netherlands
ABS	Acrylonitrile butadiene styrene
ADAC	Allgemeiner Deutcher Automobil-Club: German automobile association
Attachment efficiency	The fraction of collisions between particles that stick together to form an aggregate or agglomerate.
DAB	Dicht Asfalt Bton – Dense Asphalt Concrete
Degradation rate constants	The rate constant (s^{-1}) that describes the degradation half life of microplastics. This is the transformation of microplastics to molecules, e.g. mineralized to CO_2 and other elements.
DPMFA	Dynamic Probabilistic Material Flow Analysis
EPS	Expanded polystyrene
Fragmentation rate constants	The rate constant (s^{-1}) that describes the break-up of larger particles to smaller particles.
HDPE	High density polyethylene
LDPE	Low density polyethylene
MFA	Material Flow Analysis model which follows material flows in anthropogenic compartments.
NMPs	Nano- and MicroPlastics
NR	Natural rubber
PA	Polyamide
PC	Polycarbonate
PET	Polyethylene terephthalate
PM	Particulate matter
PM1	Particulate matter with an upper limit of 1 µm
PM10	Particulate matter with an upper limit of 10 µm
PM2.5	Particulate matter with an upper limit of 2.5 µm
PMMA	Polymethyl methacrylate
polymer particle density	The mass per volume of a specific polymer particle.
polymer particle radius	The spherical radius of a polymer particle or the spherical equivalent.
PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinyl chloride
R	A Language and Environment for Statistical Computing
Runoff	Flow of water across the soil surface to surface water
SBR	Styrene-butadiene rubber
SimpleBox	Mutlimedia fate model
TC	Transfer Coefficient used in Material Flow Analysis models

TRWPs	Tyre and road wear particles consisting of both the tyre and road wear material
TSP	Total Suspended Particles
TWPs	Tyre wear particles consisting of only the Tyre wear material.
ZOAB	Zeer Open Asfalt Beton – Very Open Asphalt Concrete

1 Introduction

1.1 Nano and microplastics in the environment

The release of nano and microplastics (NMPS) into the environment is of increasing concern as a growing volume of microplastics is found in the environment, including the sea, food, drinking water, plant life and terrestrial ecosystems (Thompson et al., 2024). Once in the environment, microplastics degrade very poorly and slowly, so that they tend to accumulate. Moreover, NMPs are able to reach pristine environments such mountain tops, polar regions, and ocean water as they disperse through the atmosphere, troposphere, surface water bodies and groundwater (Bergmann et al., 2019; Materic et al., 2021; Zhang et al., 2020). Environmental pollution with plastics will keep on increasing if no measures against their release and presence in the environment are taken (Lau et al., 2020). Tackling plastic pollution as such is a challenge, not to be taken lightly (Borrelle et al., 2020). Specifically reducing the unintentional release of NMPs to the environment requires additional effort. This is part of the European Green Deal and circular economy action plan which aim to reduce microplastics emission to the environment by 30% in 2030, see Microplastics (europa.eu).

1.2 Tyre wear release

Tyre wear is considered a major source of microplastics to the natural environment (Quik et al., 2024; Schwarz et al., 2023; Verschoor and de Valk, 2018). A recent estimate calculated almost 10fold higher release of microplastics from tyre wear compared to other sources of microplastics (Figure 1). This is in line with previous studies for Switzerland (Kawecki et al., 2021; Sieber et al., 2020) as compared to each other in Rutgers et al. (2022).

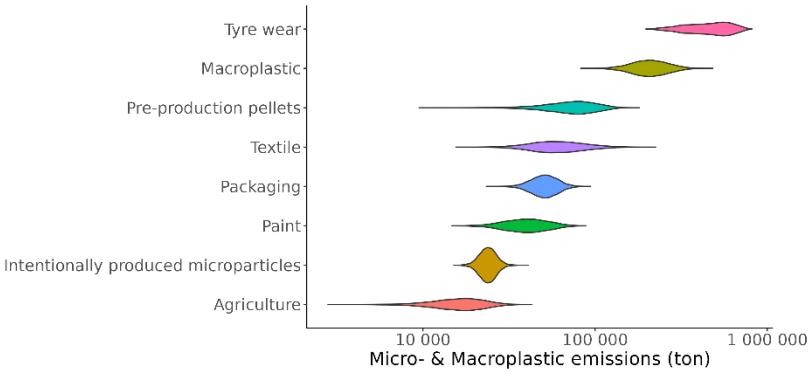

Figure 1 Plastic emissions to the environment, where non-microplastics emissions are also included as macroplastic for EU-27. Source: RIVM report 2024-0106 (Quik et al., 2024).

In another study estimating microplastics release at global level, the income region highly affects the degree to which tyre wear or other microplastics sources contribute to total microplastics release (Schwarz et al., 2023). Middle income countries are estimated to have a higher emission of tyre wear compared to low income or high income countries. This means that in countries where less mileage is driven or cleaning and maintenance practices are well established other microplastics sources contribute more to environmental release.

1.3 Aim and scope

The aim of this research is to estimate the concentration of rubber from tyre wear in the environment using model estimates. This is based on material flow analysis for estimating the emission and multimedia mass balance modelling to estimate the environmental fate and transport. The model performance is further discussed by comparing model results to field measurements of TRWPs. Furthermore, the environmental release, fate and transport of other microplastics sources are assessed for comparison to Tyre Wear Rubber.
This work thus presents results on the status of release of Tyre Wear Rubber to the environment based on an emission scenario going from 1950 to 2050 based on current policies. As policy measures are being developed, these modelling approaches can be further applied to assess their effectiveness. This is part of LEON-T Deliverable 6.2.

2 Methods

2.1 Overview

Modelling the environmental fate of tyre wear particles and other microplastics is done in three steps (Figure 2).

First, the amount of tyre wear is estimated using a commonly applied approach based on emission factors and mileage per vehicle category following the approach taken for the national emissions registry of the Netherlands ((Geilenkirchen et al., 2023; RWS, 2022)). The emission factors are scaled to include variability between tyres based on recent wear measurements of the ADAC (ADAC, 2022).

Second, the emission to different environmental compartments, such as road side soil, other soils, water and air is estimated using a dynamic probabilistic material flow analysis (DPMFA) model for microplastics (Quik et al., 2024). This includes for instance the effects of waste management practices and the effect of road cleaning and porous asphalt.

Third, the transport, degradation, agglomeration and fragmentation of Tyre and Road Wear Particles is modelled using SimpleBox4Plastics (Quik et al., 2023). The novel implementation of SimpleBox in R is applied for this first time as well as the newly measured degradation rates based on UV and microbiological processes (van Os et al., 2024).

This results in the mass of micro- and macroplastics emitted to each environmental compartment per year per source. These can be used to calculate concentrations or fate factors while include the uncertainty and variability of different input variables. Further methodological details are provided below and in the supplementary information.

Finally the calculated concentrations are compared to the measurements collected in a field measurement campaign as previously described in Tromp et al. (2023).

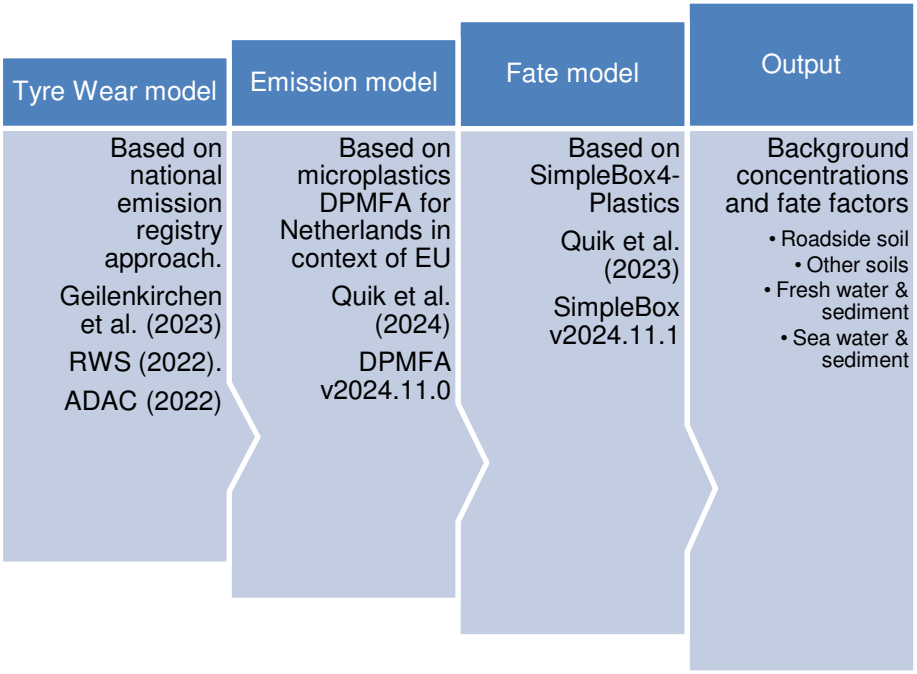

Figure 2 Modelling approach to estimate environmental transport and fate of Tyre Wear particles and other microplastics.

2.2 Tyre wear rubber release

Tyre wear rubber consists of natural rubber (NR) and styrene butadiene rubber (SBR). No distinction is made between these two types of rubber for the DPMFA model. There are different approaches for estimating tyre wear, for example by using tyre sales or emission factors. The latter was chosen for it is the standard method used by the National Emissions Registry (Geilenkirchen et al., 2023).
Geilenkirchen et al. reports emission factors for three road types (urban, rural and highway) and 6 vehicle types (passenger cars, motorcycles, mopeds, delivery vans, lorries and busses). Because just one average emission factor was reported for each of these categories, the emission factors were recalculated using the 57% - 166%

variability found by the ADAC (2022) in their wear tests of passenger car tyres. This resulted in new high and low emission factors for each category (SI 6.1.3).

Total Tyre wear release was calculated by multiplying the total driven kilometers per vehicle type and road type in the Netherlands (RWS, 2022) with the respective emission factors. This resulted in a high and a low estimate of TWP mass released each year. As data for the number of driven kilometers in the EU per road and vehicle type is not available, the resulting data for the Netherlands was scaled to the EU by using the ratio of population size between the Netherlands and the EU (SI 6.1.3). This is a large oversimplification of the actual driven kilometers per vehicle type in the EU. Travel distances per vehicle category in all EU member states are not readily available.

For the Netherlands and the EU, tyre wear release was calculated for the years 1990, 1995, 2000, 2005, 2010, 2015, 2019 and 2020. Based on the assumption that there is a linear increase of production of polymers from 1950 up to the earliest data point for which data is collected, e.g. 1990 for Tyre Wear. The future consumption of polymers is based on the plastic use polymer projections, which provides baseline plastic use projections to 2060 (OECD, 2022).

2.3 Dynamic Probabilistic Material Flow Model

Emissions of micro and macroplastics to the natural environment were estimated using a Dynamic Probabilistic Material Flow Analysis (DPMFA) model (Quik et al., 2024). Material flow analyses are used to predict the flows of a material through a system. In this case, the analysis also included the changes of the material flow over time and uncertainty about the data. The DPMFA model used in this research was based on the model by (Kawecki et al., 2021), and further developed by (Quik et al., 2024). Quik et al. added data on more polymers to the model, and specifically collected data for the Netherlands and the EU. Micro- and macroplastic emissions to environmental compartments were estimated for 7 sources of plastic emissions:

1. pre-production pellets,
2. tyre wear,
3. agriculture,
4. textile,
5. paint and coatings,
6. intentionally produced microplastics and
7. packaging

The model consideres 15 different polymers (PP, LDPE, HDPE, PVC, PA, PET, PUR, PC, PS, EPS, ABS, PMMA, acryl, rubber and other unspecified polymers).

Input data for the model includes plastic consumption in tonnes per source, year and region, transfer coefficients from one compartment to the next (e.g. Tyre Wear to Road pores, runoff or air) per polymer and scale (NL or EU), and lifetimes for plastic product that are used for more than 1 year.

For this research, a few adjustments were made to the input data and the model. Firstly, the way sewage sludge is used in the EU was adjusted. In Quik et al. (2024), all sewage sludge was assumed to be incinerated. While this is true for the Netherlands, sewage

sludge has other applications in other EU countries. Examples are application to agricultural soil, use in compost and disposal in landfills. Basically a flow of microplastics from sewage sludge to agricultural soil was added, see details in SI 6.1.1.

The previous analysis for the EU included the Netherlands (Quik et al., 2024). However, in order to link this model to the SimpleBox which consists of nested scales for NL (regional scale) within EU (continental scale), emission data for the EU and the Netherlands needed to be entered separately. Therefore, the DPMFA model was slightly adjusted by subtracting NL input data from the EU input data, and running the DPMFA model separately for the Netherlands and the EU. Details of the adjustments can be found in SI 6.1.2.

DPMFA model version 2024.11.0 is used and made available via https://github.com/rivm-syso/DPMFA_NL_EU.

2.3.1 Tyre Wear (NR and SBR)

The input data on microplastic emissions from tyre wear for the Netherlands and EU is mainly based on Quik et al. (2024) with some adjustments based on data on the fraction of Natural Rubber in Tyre Wear. This was done in order to separately model the fate of Natural Rubber and Styrene Butadiene Rubber.

2.3.1.1 Transfer coefficients

The distribution between TWP emissions on urban roads, rural roads and highways was made based on data for the Netherlands (Quik et al., 2024) and extrapolated to EU based on population. The vehicle and road type distribution was thus assumed to be the same for the EU as for the Netherlands (RWS, 2022). While in the Netherlands nearly all highways are made of open asphalt (ZOAB), the opposite is true for Europe. Therefore, 5% of the emissions were assumed to occur on open asphalt roads, whereas 95% of the emissions are assumed to occur on dense asphalt (DAB) highways (Figure 2).

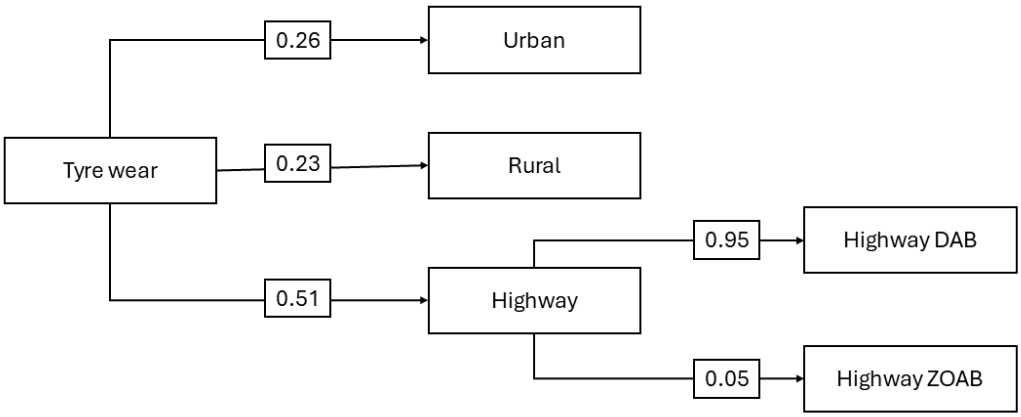

Figure 3 Distribution of TWP emissions between urban roads, rural roads and highways in the EU. Highways are divided into two categories: DAB highways and ZOAB highways.

The distinction between different types of roads is made because it is believed TWP emissions follow a different route through the system based on where the emissions take place.

It is assumed that 5-10% of Tyre Wear is PM10, and thus emitted to air (Geilenkirchen et al., 2023; Hoeke et al., 2024; Verschoor et al., 2016). This transfer coefficient is thought to be the same across road types. Other emissions are transferred to either road side soil (Verschoor et al., 2016), road runoff or road cleaning (Sieber et al., 2020; Verschoor et al., 2016) (Figure 3 and Figure 4). The TC for cleaning on ZOAB highways is much higher, because TWPs are trapped in the open pores of the asphalt making cleaning more effective. On rural roads and DAB highways, cleaning is assumed to be 1-2% effective (Sieber et al., 2020). The effectiveness of cleaning on urban roads is assumed to be higher (between 2% and 10%), because of a higher frequency of cleaning by municipalities.

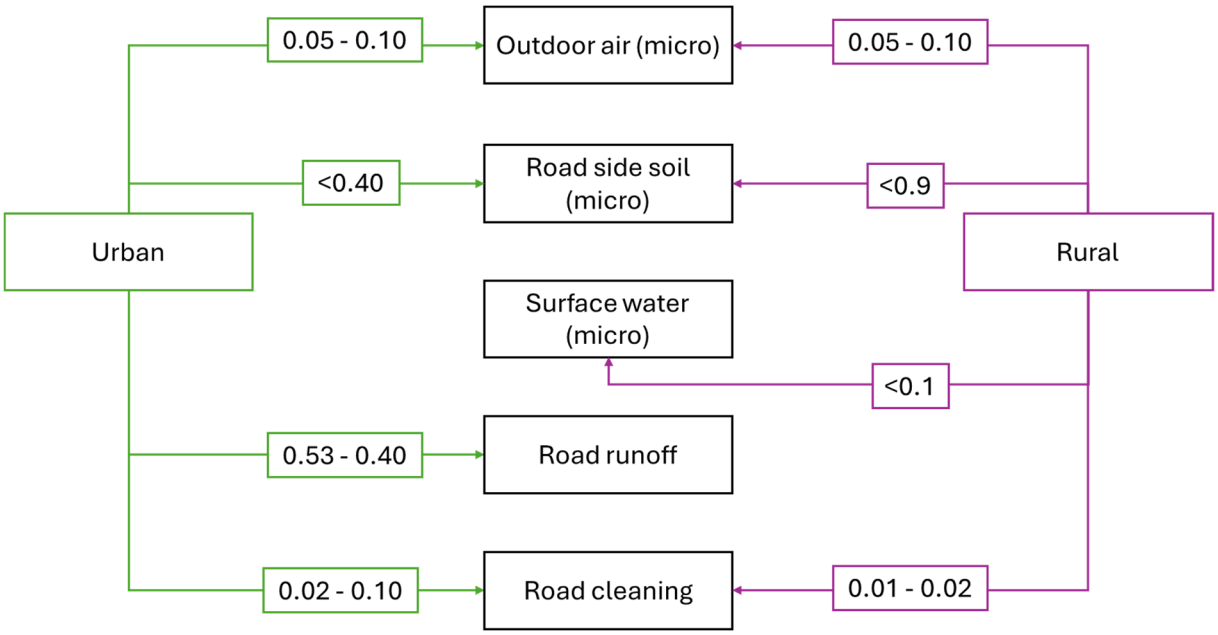

Figure 4 Transfer coefficients from urban roads and rural roads to subsequent compartments. TC's always sum up to 1 using a scaling procedure.

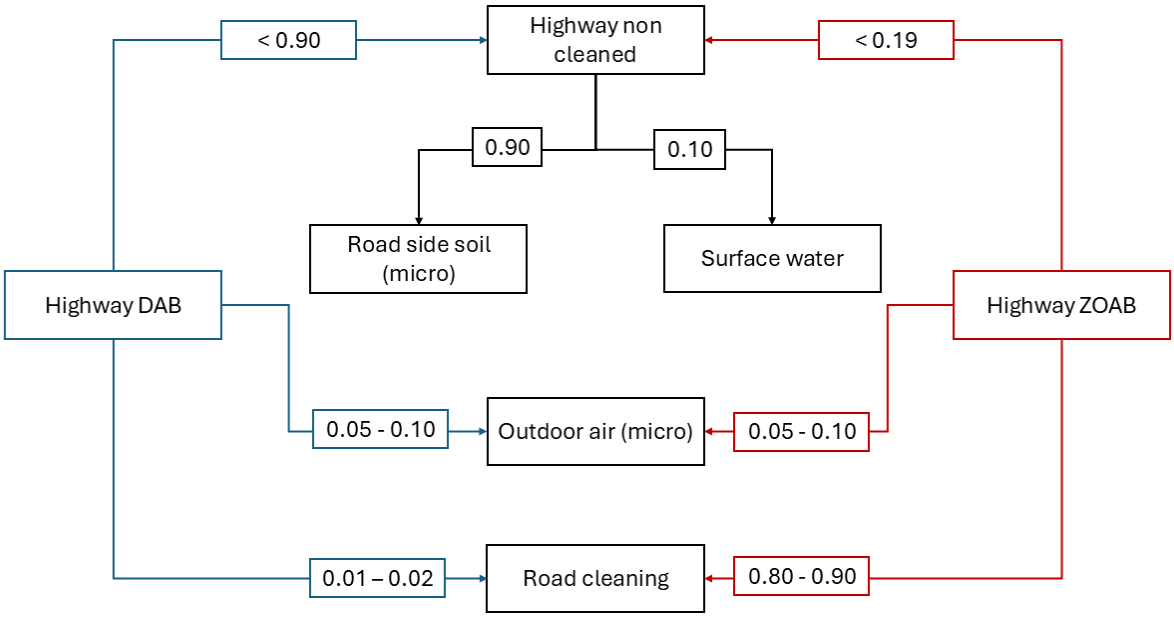

Figure 5 Transfer coefficients from DAB and ZOAB highways to subsequent compartments.

5% of cleaned TWPs are assumed to end up in wastewater, while the rest is incinerated (Sieber et al., 2020) (Figure 5). TWPs in road runoff are expected to go to one of four compartments: road side soil, surface water, stormwater or wastewater (Hoeke et al., 2024) (Figure 6). The applied TC's are for the Netherlands, but also applied to EU. From wastewater, TWPs can be emitted to the environment via two routes: to agricultural soil through sludge application (Eurostat, 2024) and to sub-surface soil through leakage from the sewage system (Kawecki et al., 2021; Rutsch et al., 2006).

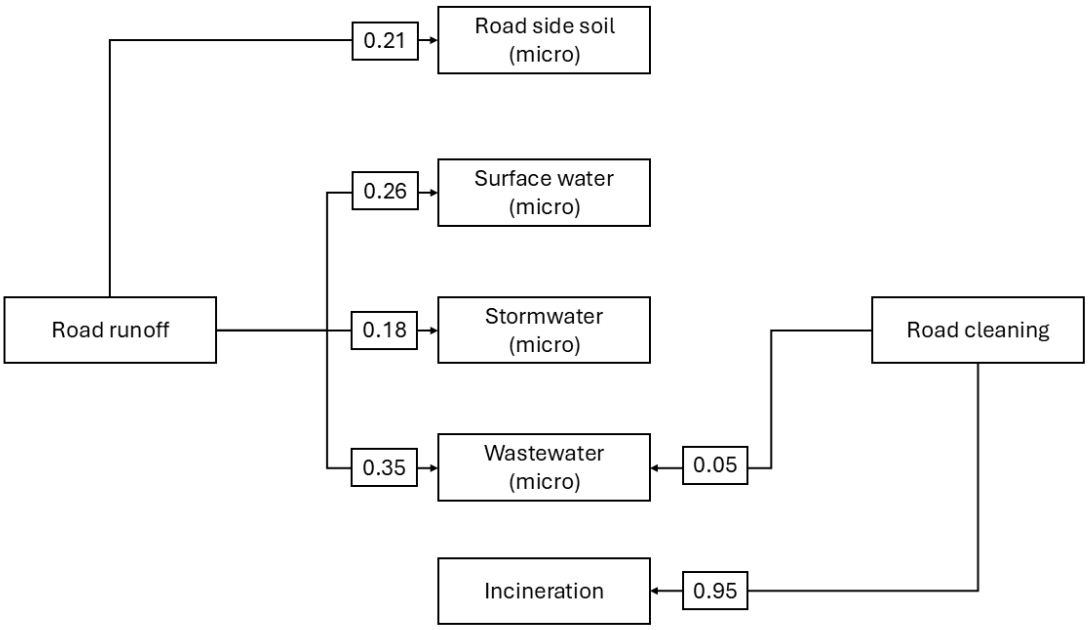

Figure 6. Transfer coefficients from road runoff and road cleaning to consequent compartments.

Overall, the uncertainty and variability of transfer coefficients is taken into account. First the uncertainty is estimated based on reliability of the data (Quik et al., 2024). This is done based on estimating a spread around the transfer coefficient, resulting in a triangular distribution. Second, variability (and uncertainty) is included by setting a range between two transfer coefficients, this results in a trapezoidal distribution.

2.3.2 Other Microplastic sources

Other microplastic sources in this analysis include Pre-production pellets, Paints and coatings, Textiles, Agriculture, Intentionally produced polymer microplastics and packaging. These are assumed to be some of the largest sources of microplastics to the environment. The inflow of plastics from these sources are related to consumption or use of polymers for these product categories at NL and EU scale. The data and modelling approach is applied as described in Quik et al. (2024).

In summary, the DPMFA model follows 12 polymers and a category of non-specified polymers, called Others. This is based on the combination of several MFA studies reporting most of the original data for the EU or Switzerland (Kawecki et al., 2021, 2018; Liu and Nowack, 2022). The production and manufacturing of polymer based products is decoupled from the actual consumption of these products in order to simplify the MFA as show in Figure 7. A detailed account of the model and further data sources is given in Quik et al. (2024).

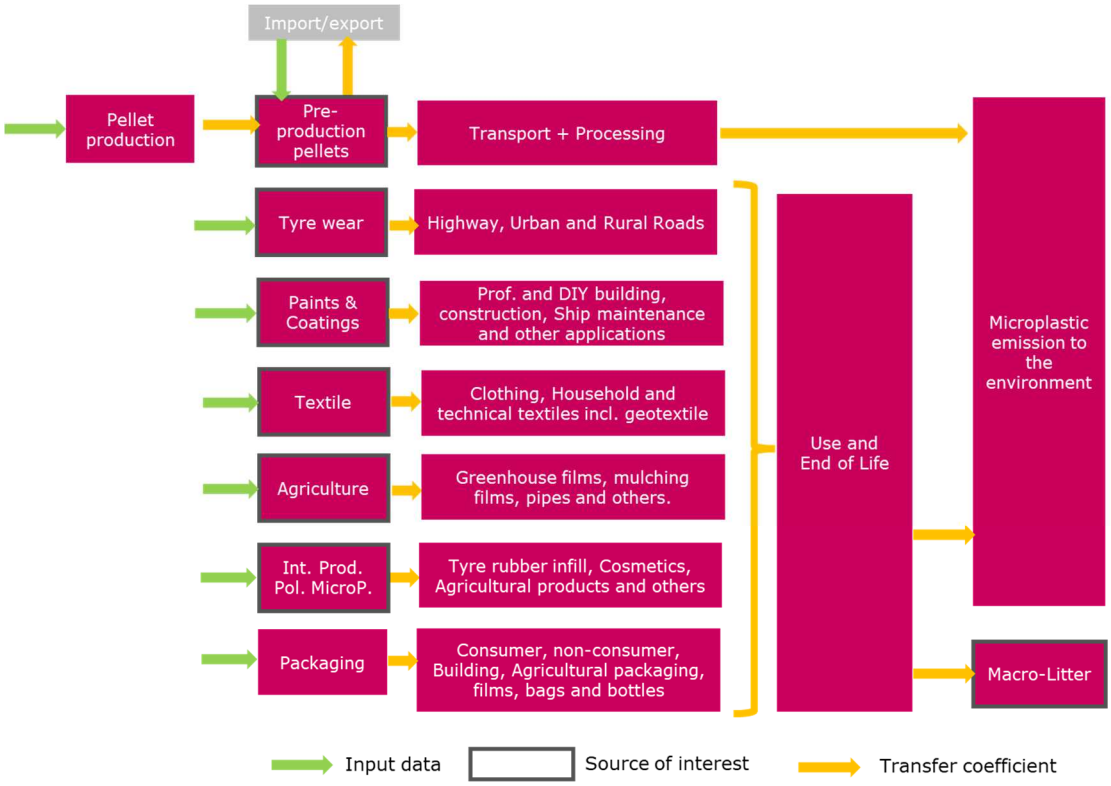

Figure 7. Overview of microplastic sources of interest and link to environmental compartments, soil, air and water as applied in Quik et al. (2024).

2.4 Environmental fate modelling

2.4.1 Model description

The screening level multimedia fate model adapted for use with microplastics is applied: SimpleBox4plastics (version 2024.11.1). This is a multimedia environmental fate model aimed at screening level assessments. The model is based on previous versions of SimpleBox (Hollander et al., 2016; Meesters et al., 2014). In brief, SimpleBox can estimate environmental distribution of mass and concentrations for substances in the form of unbound molecules (conventional substances) and in the form of particles. Capabilities of the model include calculating concentrations in a steady state and dynamically in time. This can be done both deterministically or probabilistically. The main change from the previous version of SimpleBox4plastics (Quik et al., 2023) is the transfer of the model from Excel to R making dynamic probabilistic analysis possible and improving the interoperability of the model.

SimpleBox4Plastics is applied providing environmental concentrations in seawater, fresh water, lake water, natural soil, agricultural soil, other soil, air, freshwater sediment and marine sediment. The model is used for estimating Fate Factors and the distribution of tyre wear plastics at steady state using a probabilistic input for the specified particle properties (see details below). The concentrations are estimated using the same probabilistic input, but analysed dynamically using the emission estimates from the DPMFA model.

2.4.2 Model parameterization

2.4.2.1 Landscape

The landscape scenario is adjusted for this study. The regional area of the model is adjusted from the catchment of the river Rhine to the whole of the Netherlands, because the DPMFA model outcomes are for the EU (continental scale in SimpleBox) and NL (regional scale in SimpleBox). For details see SI 6.2.1.

2.4.2.2 Input parameters

To model the mass and concentration of if TWPs and microplastics from other sources the relevant parameters were collected and probability distributions defined. This is done for: attachment efficiencies, degradation rate constants, fragmentation rate constants, polymer particle radius and polymer particle densities. Further details given in SI 6.2.2 and 6.2.3.

For Tyre and Road Wear Particles (TRWP) we assume emission of the attached (P) species which consists of the polymer attached to an inorganic particle, in this case resembling road wear. This TRWP particle due to fragmentation can fall apart in Tyre Wear Particle, which is resembled by the Solid (S) species in SimpleBox4plastics. For all other microplastics we consider the emission to be of the Solid species (100% polymer).

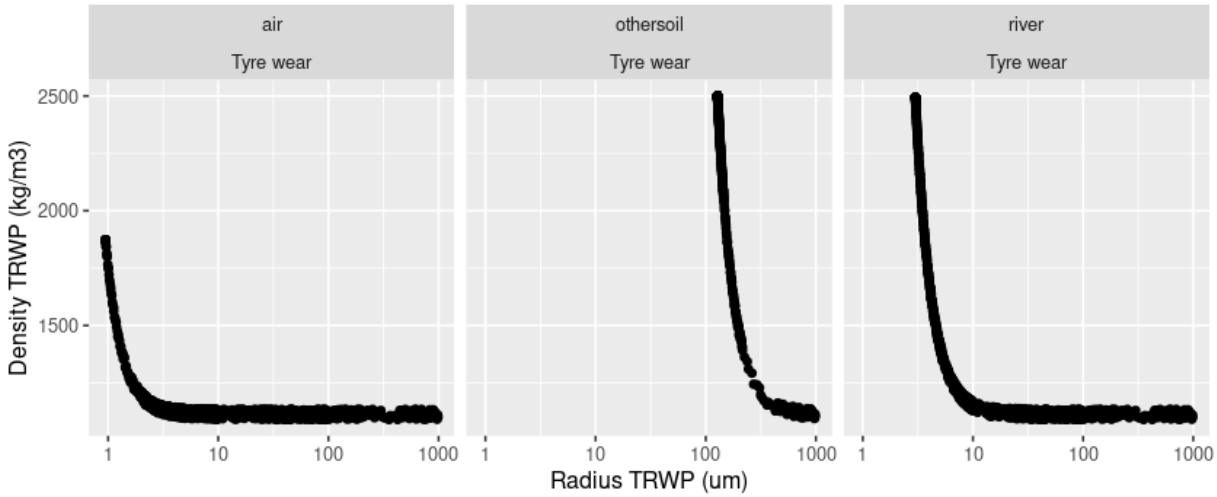

Figure 8 Size and Density relationship of the TRWP particles as made up of an inorganic particle combined with a NR/SBR particle based on size a measured size distribution between 0.5 and 1000 µm (see Table S15). NOTE: othersoil here is the SimpleBox compartment name for the soil which includes road side soil.

As the particle size and density is calculated from the density of the inorganic road wear particle and the size of the tyre wear particle, there is a clear relationship between the size and density (Figure 8). A constant radius of the road wear particle is assumed, which differs for air (0.9 µm), soil (128 µm) and water (3 µm). The model assumes spherical particles composed of the mass of these two particles (Road wear and Tyre

wear). As can be seen in Figure 8, the minimum radius of TRWP is increased due to the radius of the road wear particle. Although the reality is more complex, e.g. TRWP shape is not spherical, it is known that inorganics fraction changes with TRWP size (Tromp and Esveld, 2023). However, this is not taken into account. In reality for instance a larger fraction of Road Wear is expected in larger particles which does not correspond to the decrease in density shown here and the size of TRWP going to soil is relatively large compared to measurements.

Distribution between NR and SBR
Tyre wear consist of two types of rubber: natural rubber (NR) and styrene butadiene rubber (SBR). This distinction was made by distributing the outcomes of the DPMFA model for tyre wear between NR and SBR with an average of 17.8% NR based on characterisation of TRWP samples (Tromp and Esveld, 2023). Details provided in SI 6.2.4.

Degradation rate constants
Degradation rate constants are thought to follow separate triangular distributions for water and soil/sediment. The distributions used are based on (Chamas et al., 2020) for polymer other than NR or SBR. NR and SBR degradation rates are derived from abiotic and biological degradation data (Parker et al., 2024). For more information on the distributions used, see Table S9 and Table S10. The degradation is the process which causes polymers to be transformed into a non-polymer form, e.g. mineralisation. This is a removal process.

Fragmentation rate constants
Fragmentation is the process where a polymer particle falls apart in smaller or other polymer particles. For TRWP we assume that they fragment from TRWP to TWP particles, the rate constant was based on UV degradation experiments (Parker et al., 2024), see Table S12. In addition, small pieces (fragments) break off from the original TWP particles. Static light scattering (SLS) analysis on UV aged TWP show the formation of a bimodal distribution, where the original unimodal distribution shifts to a smaller particle size and an additional distribution of the detached fragments is formed (around 10 µm). From this a particle size reduction rate of -0.03 (0.02 – 0.05) µm day^{-1} was calculated (Parker et al., 2024).

For other polymers we also assume that fragmentation will only result in falling apart of attached species to the original emitted microplastic particles, see Table S11. The incorporation of fragmentation as a process affecting the overall size distribution of microplastics in the environment is thus not considered and is part of ongoing research and development of SimpleBox4Plastics.

Particle radius
Distributions for particle radii for TRWPs was derived from the particle size distribution as measured in a combination of 11 environmental measurements, for details see section 6.2.3.
The particle radii for most other polymer microplastics (HDPE, LDPE, PP, PS, PVC, PA, PET, ABS, PC and PMMA) were based on the wear and impaction model from (Boersma et al., 2023), see Table S13. The impaction size is taken as lower end and

wear size as higher end of a uniform distribution per polymer type. For the polymers acryl, EPS, PUR, RUBBER and other (non-identified polymer), this data was not available and we assumed the broadest possible size range. Particle radius is an important driver for deposition processes from air to soil or water and in water to sediment.

Particle density
Particle density for all polymer types is specified based on readily available sources (Bremmer and van Engelen, 2007; Omnexus, 2024; RSC, 2024). All densities follow a uniform distribution, see table S9. Density, similar to particle size, also drives the deposition processes. However, low density particles do not sediment out of the water phase, but remain suspended and accumulate. This only applies to HDPE, LDPE, PP, EPS and PUR of the known polymers. Processes such as biofouling are currently not included in the sedimentation process in SimpleBox4plastics.

Attachment efficiency
The attachment efficiencies of microplastic particles to natural particles is highly variable. This is a result of differing environmental conditions, i.e. natural organic matter content, salt types and ionic strength (Shams et al., 2020). Therefore, attachment efficiencies were assumed to follow a uniform distribution between $1e^{-4}$ and 1 (Quik et al., 2023) (Table S8).

2.4.3 Model availability and application

The SimpleBox model is available from https://github.com/rivm-syso/SBooScripts and https://github.com/rivm-syso/SBoo. SBooScripts contains the data and scripts used to perform the calculations, see version 2024.11.1. The scripts apply the model functions and structure as provided in SBoo (Version 2024.11.0).

Calculations for this study were executed on a high performance cluster for parallel calculations, as the stochastic analysis of the 100 year timespan results in long calculation times (~3 minutes per run). Data analysis requires at least 64gb of RAM for the 17000 unique runs required for this study (n=1000 and 17 types of polymer particles).

2.5 Measurements

2.5.1 Field data

TWP data from the field measurements are used for comparison to concentrations estimated using the modelling approach (Tromp and Esveld, 2023). Field measurements were performed in 2022 at six roadside locations; in the Netherlands: highway A2, highway A27/A28, city intersection Rotterdam and regional background/provincial road N210 Cabauw, in Sweden: highway E18 and in Germany highway A61. Measurements consisted of sampling of air (TSP, PM10, PM2.5, PM1 and PM0.1), surface water, sediment, road runoff, atmospheric deposition and soil. To evaluate the dispersion of TWP deposition and soil samples were taken at increasing

distance from the road. In addition surface water and sediment samples were taken from the river Meuse (Eijsden and Rotterdam) and Rhine (Lobith). To determine the particle size distribution of TWP in the different environmental compartments, selective samples were fractionated. All samples were analysed with TED-GCMS (thermal extraction and desorption gas chromatography mass spectrometry) to determine the concentration of natural rubber, styrene butadiene and total TWP. In *Figure* 9 an overview is given from the main results of the field measurements.

2.5.1 Degradation half lives

Data from the accelerated UV aging experiments and subsequent biodegradation experiments (Parker et al., 2024; van Os et al., 2024) are used as basis for the fragmentation and degradation rate constants. Sieve fractions (50-200µm) of cryomilled car and truck TWP and car TRWP obtained from the road simulator experiments (Mathissen et al., 2023) were subjected to accelerated photothermal ageing for 0, 160, 505 and 1000 h; the latter corresponds to a simulated environmental ageing time of 8.2 (5 – 10) years. The UV aged car and truck TWP (0, 160, 505 and 1000h) were further tested in accordance with ISO 14851:2019 in sealed reaction bottles with activated sludge microbial inoculum (25.0 mg TSS/L). Only 0h and 160h UV aged TWP showed noticeable degradation after 28 days (0h: 6 – 8%, 160h: 2 – 6.5%) due to the presence of NR. The older samples have negligible NR as this is already degraded by the UV exposure. For this reason it is assumed that degradation of SBR is about 100 times slower than NR at best or has almost negligible degradation. The observed UV degradation rates for TRWP including SBR are assumed to be representative for the fragmentation of particles from the attached (P) and aggregated (A) species to solid species.

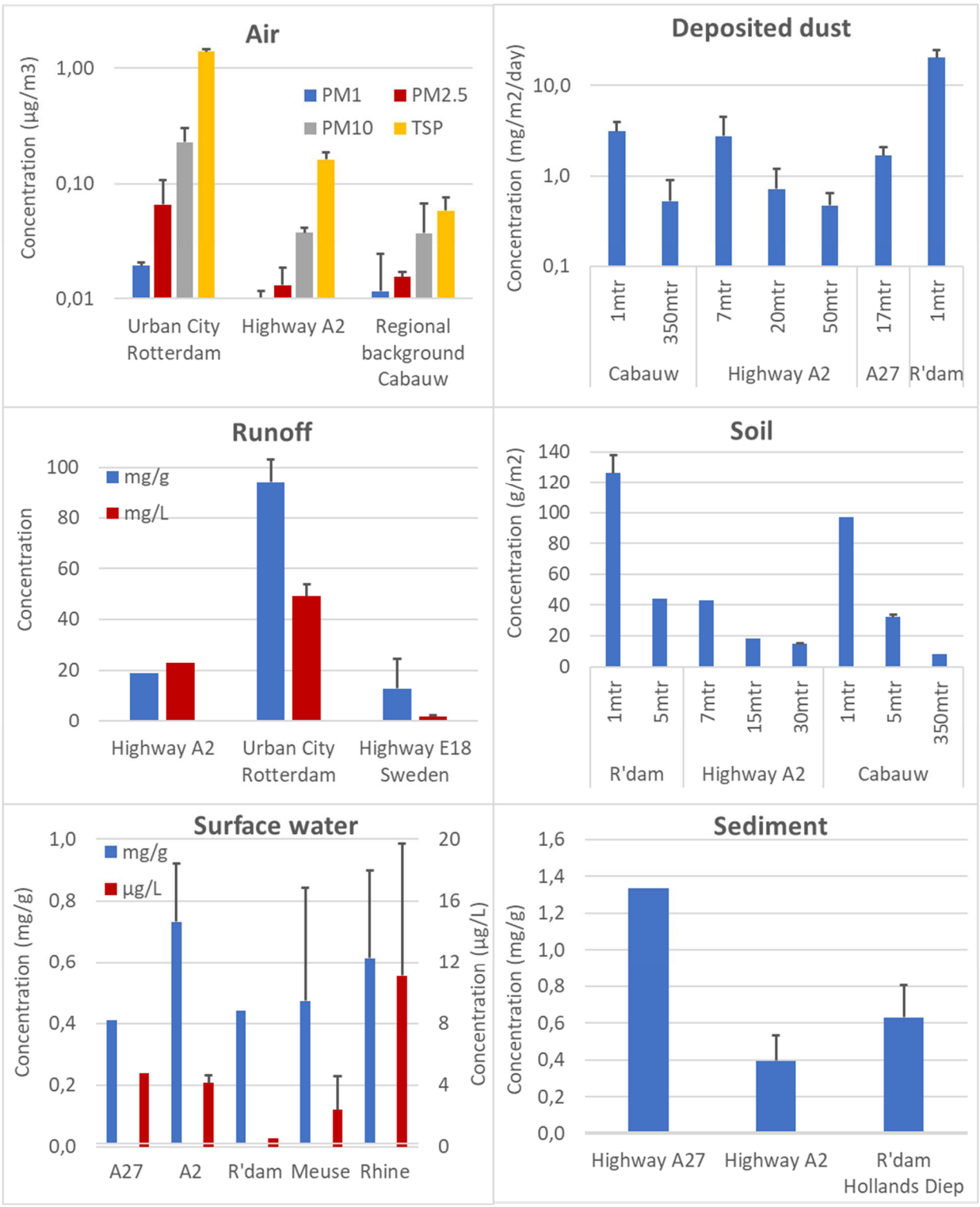

Figure 9 TWP concentrations in different environmental compartments obtained from the field measurement campaign in 2022 at several roadside locations and the rivers Rhine and Meuse, source: (Tromp and Esveld, 2023).

3 Results and Discussion

3.1 Tyre wear release

The tyre wear released in the EU (Figure 10) and Netherlands (Figure 11) is in line with other estimates. For instance one of the latest estimates for the EU reports an average release of 450 kton (360-540 kton) for 2019 for EU-27 (EC, 2023). This lies within the the low (300 kton) and high (552 kton) estimates used in this study, even though population scaling was applied instead of vehicle mileage. The range in the high and low estimates for the Netherlands is solely due to variability of the tyre wear emission factors of tyres. The actual variability is a lot higher, as many other characteristics affect tyre wear, such as vehicle and road characteristics or driver behaviour (Giechaskiel et al., 2024; Meesters and Quik, 2024).

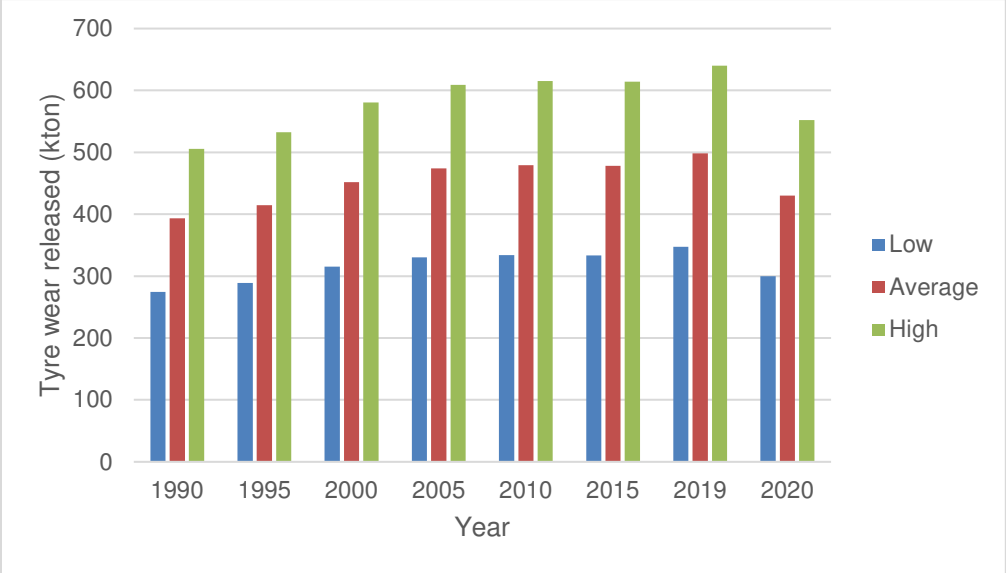

Figure 10 Tyre wear released in the EU-27 from which the low and high values are used in the DPMFA model, scaled using population from NL estimates.

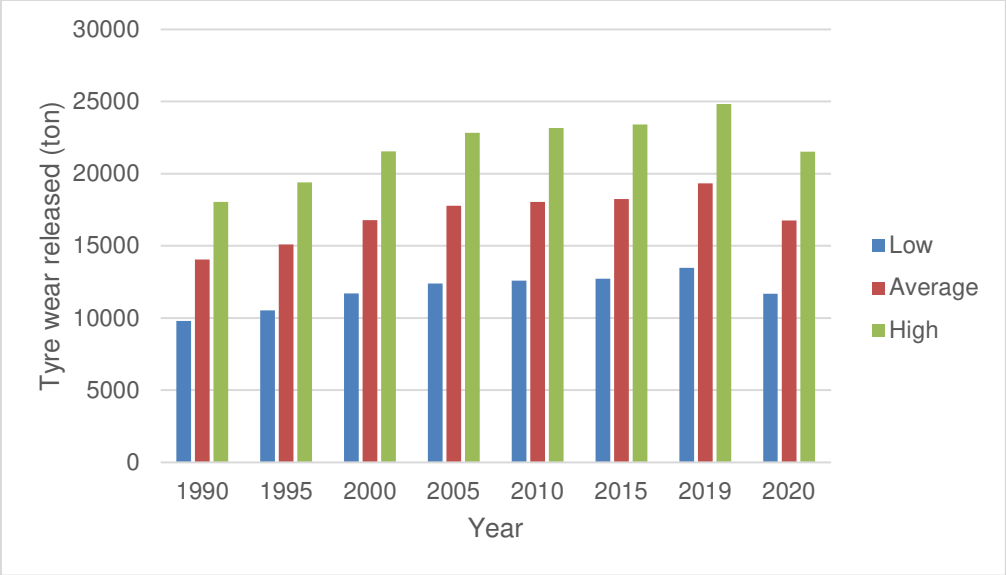

Figure 11 Tyre wear released in the Netherlands from which the low and high values are used in the DPMFA model.

3.2 Emission estimates EU and NL

Tyre wear emissions are on average the largest source of microplastics to the environment in the EU (Figure 10). Although there is some uncertainty, at best, the lowest emission estimate of tyre wear would still contribute a large part of microplastics emissions to the environment. The emissions of tyre wear are uncertain due to uncertainty in release of tyre wear (Figure 10 and Figure 11), but also in the uncertainty and variability of road-runoff treatment and the effectivity of capturing TWP in porouse asphalt (Figure 5 and Figure 6). The effect of almost 95% porous asphalt in the Netherlands compared to only 5% at EU scale is the cause for the relative lower contribution of Tyre Wear to overall microplastics emissions to the environment compared to EU as a whole. However, the assumptions leading to the commonly applied 80-90% cleaning efficiency of porous roads should be further investigated as data supporting this is scarce and given this is one of the most important variables in assessing the emissions of tyre wear important to reduce it's uncertainty.

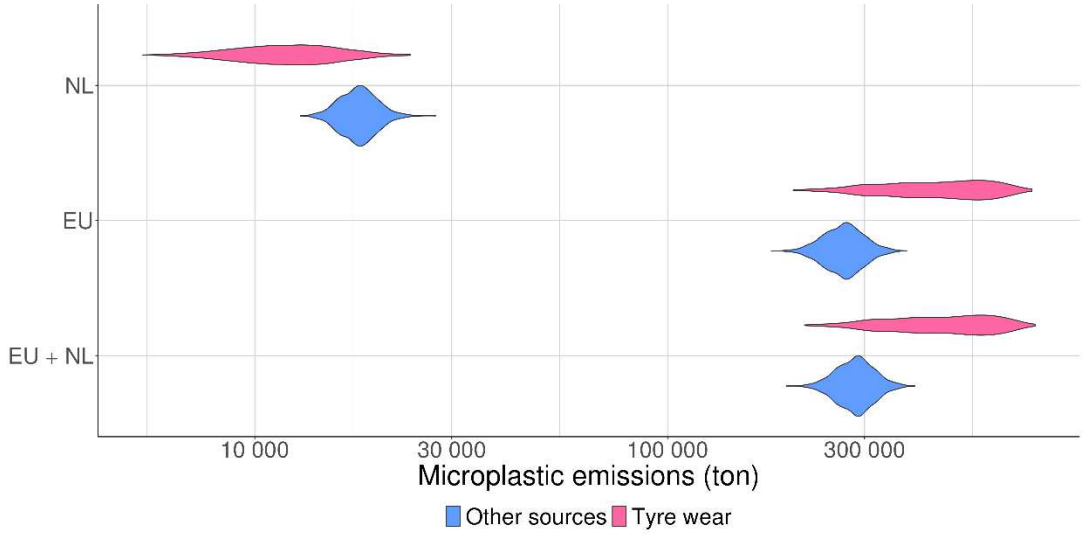

Figure 12 Violin plot of microplastic emission to the environment from tyre wear and other microplastics sources. DPMFA at different scales in 2019 (EU+NL=EU-27, EU = EU-26 (without NL)). Logarithmic scale on x-axis.

About 90% of tyre wear particles are emitted to environmental compartments (Table S5). Compared to other sources of microplastics, a little more Tyre Wear is emitted to the water compartment (Figure 13). Although soil is the largest receiving compartment for all microplastics, Tyre Wear is largely emitted to road side soil (Figure 14), while other microplastics are much more emitted to other soil types such as agricultural, residential or natural soils. This is important to consider as the protection goals are usually differentiated per land/soil use type.

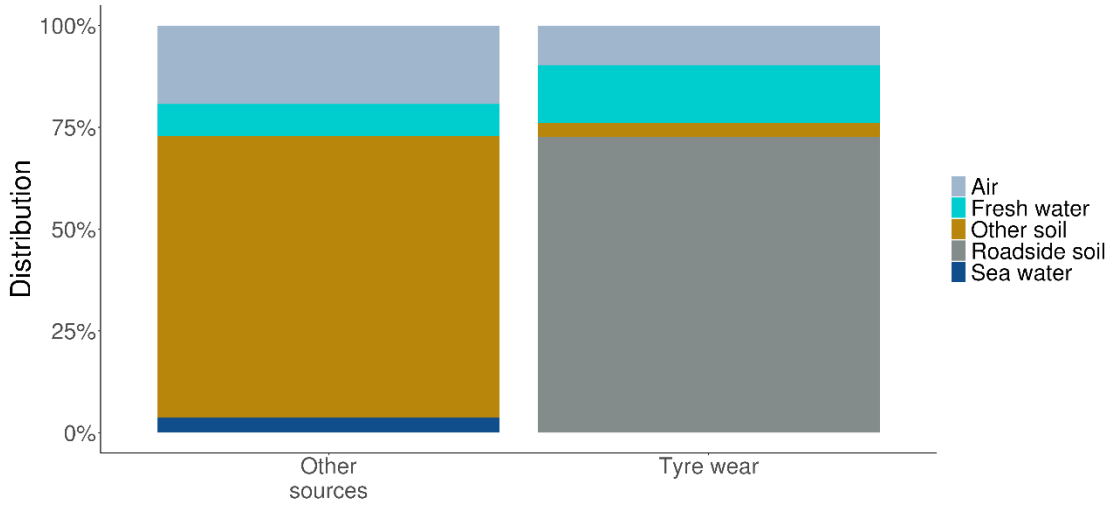

Figure 13. Distribution of microplastic emissions to soil, air and water compartments for microplastics from tyre wear versus other sources based on the average emission. DPMFA EU-27 in 2019.

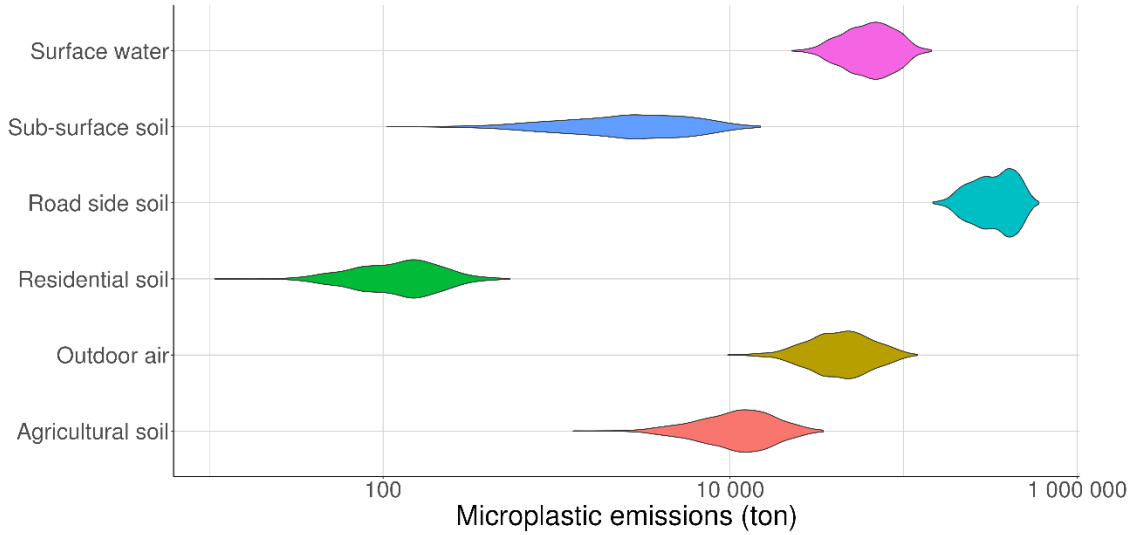

Figure 14. Microplastic emissions from tyre wear to different environmental sinks. DPMFA EU-27 in 2019. Logarithmic scale.

A historic and future scenario is used to address the past and future emissions (Figure 15). This is for estimating the environmental load. The emission dynamics in time are relevant for assessing the buildup of microplastics in different environmental compartments in time due to the persistence of many polymer particles considered microplastics. This needs to be considered when comparing model estimates of concentrations to measured concentrations, which due to the persistence and potential continued buildup might not represent steady state.

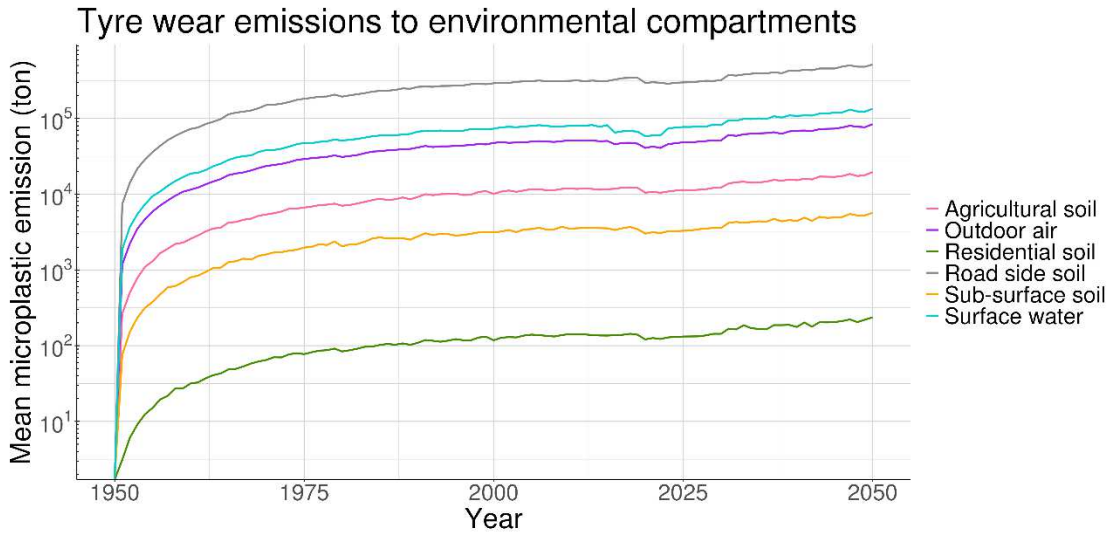

Figure 15. Mean tyre wear emissions to environmental compartments over time. DPMFA EU-27. Logarithmic scale on y-axis.

3.3 Distribution among environmental compartments

3.3.1 Environmental distribution Tyre Wear

The distribution of TRWP through the environment is dependent on the route of emission (Figure 16). TRWP emitted to air accumulates in a lot more different environmental compartments due to wet and dry deposition to water and soil compartments. This leads to a large fraction eventually ending up in marine sediments (74%). A direct emission to Fresh water results in Tyre and Road Wear rubber almost all ending up in sediments (97%), with a small fraction remaining in the water column (<3%) and no accumulation in soil. An emission to road side soil would result in the majority of Tyre Wear remaining in this compartment (68%) with some accumulation in sediments (30%) due to erosion and runoff to surface waters and subsequent sedimentation.

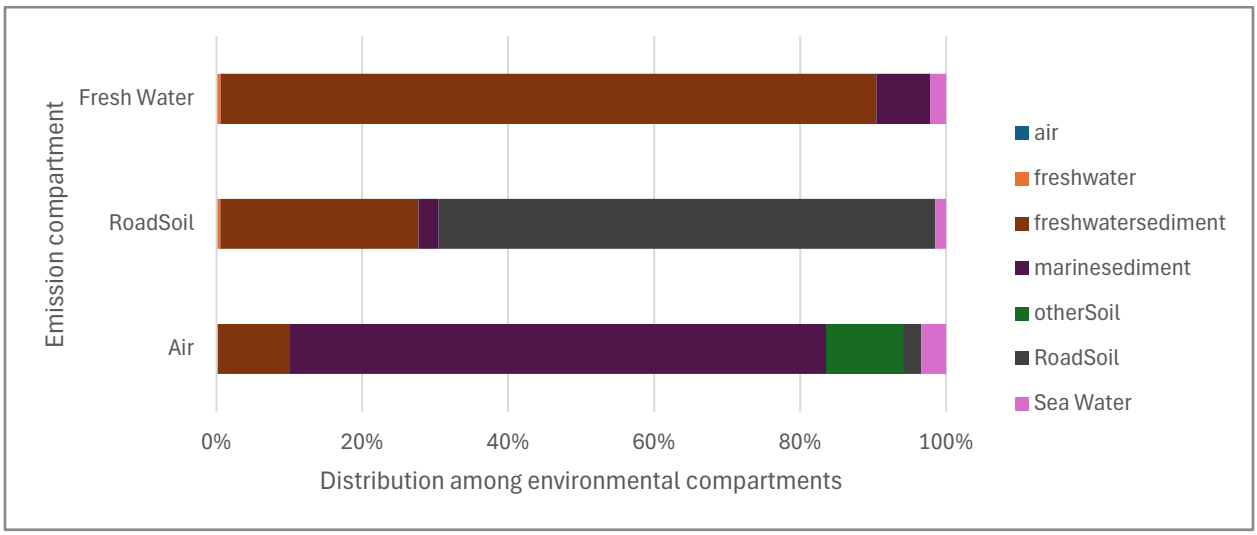

Figure 16. Distribution of Tyre Wear based on SBR and NR emissions to Air, Soil (road side) and Water (Fresh) using SimpleBox4Plastics at EU scale. FF is in days based on a kg emitted per kg accumulated in the compartment. A higher value means more accumulation of Tyre Wear based on the route of emission.

It should be kept in mind that the provided distribution of Tyre Wear, based on the mass of SBR and NR in these compartments is an average situation for the whole of Europe based on an infinite time horizon (steady state). These types of results thus indicate the fate that TRWP have in our environment, but location and time specific effects are excluded.
Furthermore, the model is based on several assumptions that influence the outcome and in future updates can still be better implemented when data becomes available. One such assumption, which might have an influence on the results, is that all TRWP and TWP particles will travel with run-off to the water compartments. This might not be realistic as particles can be held back by fauna as a sort of filter. This would result in an overestimation of the TWP ending up in water and sediment.

Unice et al. (2019) also studied road and tyre wear particle fate in the environment. They estimated that between 1.4 and 4.9% of TRWP are exported to the estuary (sea water/marine sediment) system. This is lower than the average 9% fraction of TRWP ending up in sea water (2%) and marine sediment (7%) using SimpleBox4Plastics and the expected division of emissions between air, water and soil. This could at least partially be explained by an overestimation of the runoff flow from soil to water. This should be investigated further in future work.

3.3.1.1 Fate factors

When the mass of microplastics in each environmental compartment is considered per kg of emitted microplastic to that compartment, the result is a fate factor [kg compartment/kg emitted day]. A fate factor can be seen as representing the residence time of microplastics in each compartment. However, as the emission route to the environment affects the emission factor, e.g. an emission to air build-up in sediment only after deposition to soil or water, run-off from soil to water and subsequent sedimentation. For an emission to soil this is a different pathway to sediment, excluding for instance deposition from air. Here we present a table of fate factors for emissions to air, fresh water (incl. lake) and road-side soil of Tyre wear particles. These emission factors can be applied as proxy for application of SimpleBox4plastics when the buildup of tyre wear in different environmental compartments is of interest.

Table 1. Fate factors in $kg_{emitted}/kg_{compartment}$ days for tyre wear considering SBR and NR for three emission routes.

	Air	**Fresh-water**	**Freshwater-sediment**	**Marine-sediment**	**Other Soil**	**Road Soil**	**Sea water**
Emission to Air	3.03E-01	7.86E+00	4.08E+02	3.07E+03	4.42E+02	1.01E+02	1.41E+02
Emission to Road Soil	0	1.64E+01	8.14E+02	8.23E+01	0	2.04E+03	4.40E+01
Emission to Water	0	1.17E+01	1.80E+03	1.48E+02	0	0	4.38E+01

3.3.2 Concentration estimates Tyre Wear and other microplastics

In order to understand the contribution of Tyre Wear to overall microplastics load in the environment, the concentration buildup over time is estimated based on an emission scenario of Tyre Wear and other microplastics emission going back to 1950. This was possible due to a novel coupling of the microplastics DPMFA model to SimpleBox4Plastics while accounting for physio-chemical properties of the microplastic particles. This provides a screening level estimate of environmental concentrations of microplastics for the whole of Europe and the Netherlands. The difference in degradation

half-life, size and density of SBR, NR and 13 other polymers on their environmental fate are considered here for the first time in such an integrated modelling study.

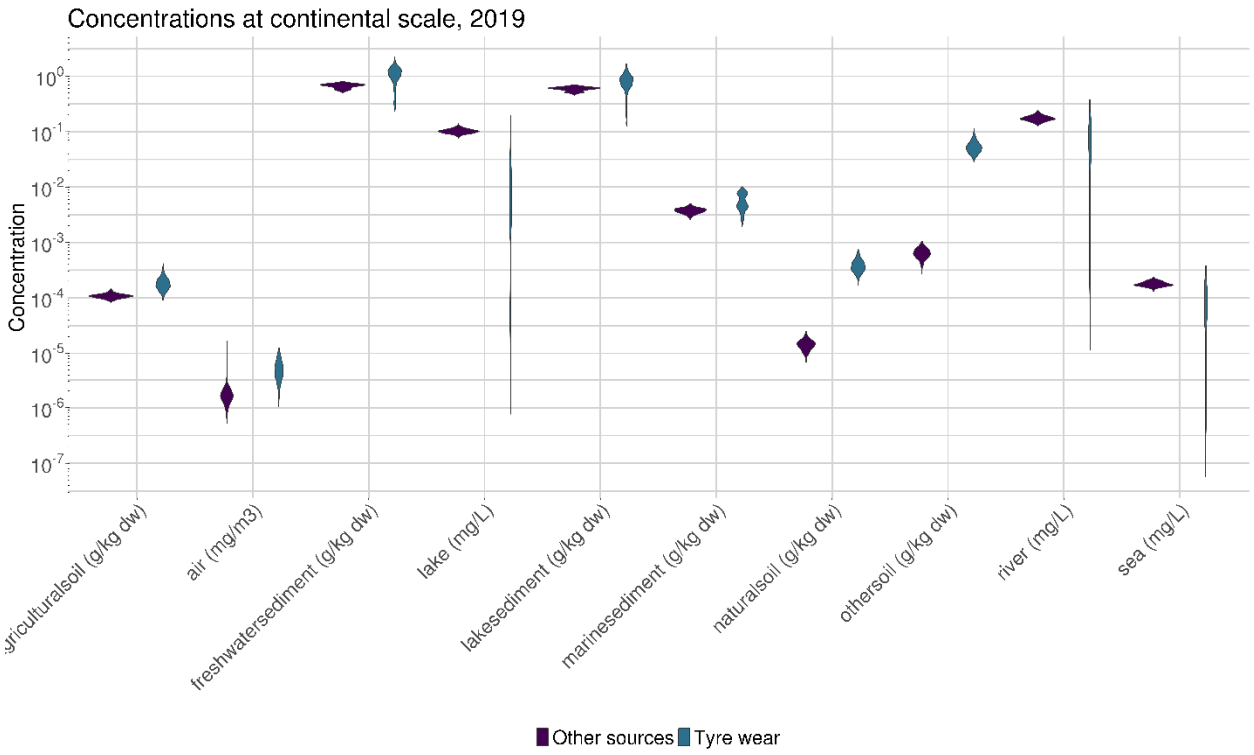

Figure 17. Comparison of the concentration microplastics from tyre wear (blue) versus other microplastic sources (purple) in environmental compartments at continental scale (EU) for 2019. Logarithmic scale on y-axis.

Estimates of the concentration of Tyre Wear particles in the environmental compartments agricultural soil, air and freshwater or marine sediments are slightly higher than the concentrations of other microplastics (Figure 17). However, overall they cover a similar order of magnitude. The largest differences are observed for the water compartments, natural soil and the other soil compartment. The other soil compartment includes Road Side soil and for this reason it is not surprising that here Tyre Wear has almost a two orders of magnitude higher concentration compared to other microplastics. Interestingly also in natural soil, Tyre Wear has a higher concentration compared to other microplastics. This is due to the relatively high emission of tyre wear to air compared to many other microplastics and this results in a relatively higher deposition of tyre wear in natural soils. This is in line with observations that Tyre Wear has also been found in more remote area's (Herzke et al., 2024).

A remarkable result is the very high range of uncertainty in the concentration of Tyre wear in the water compartment compared to the other microplastics. The estimated concentrations of tyre wear span almost 5 orders of magnitude. The high uncertainty in the concentrations of Tyre Wear in the water compartments is due to the specific particle size distribution of Tyre Wear affecting deposition and transport through the water compartments. It is clear that the fate processes affecting the Tyre Wear concentrations in other compartments is less relevant (Figure 18). This also illustrates the importance of considering the uncertainty in model parameters. And because the

wear size estimates from Boersma et al. (2023) were applied to most of the polymers (See section 6.2.2) this resulted in a much narrower size distribution in comparison.

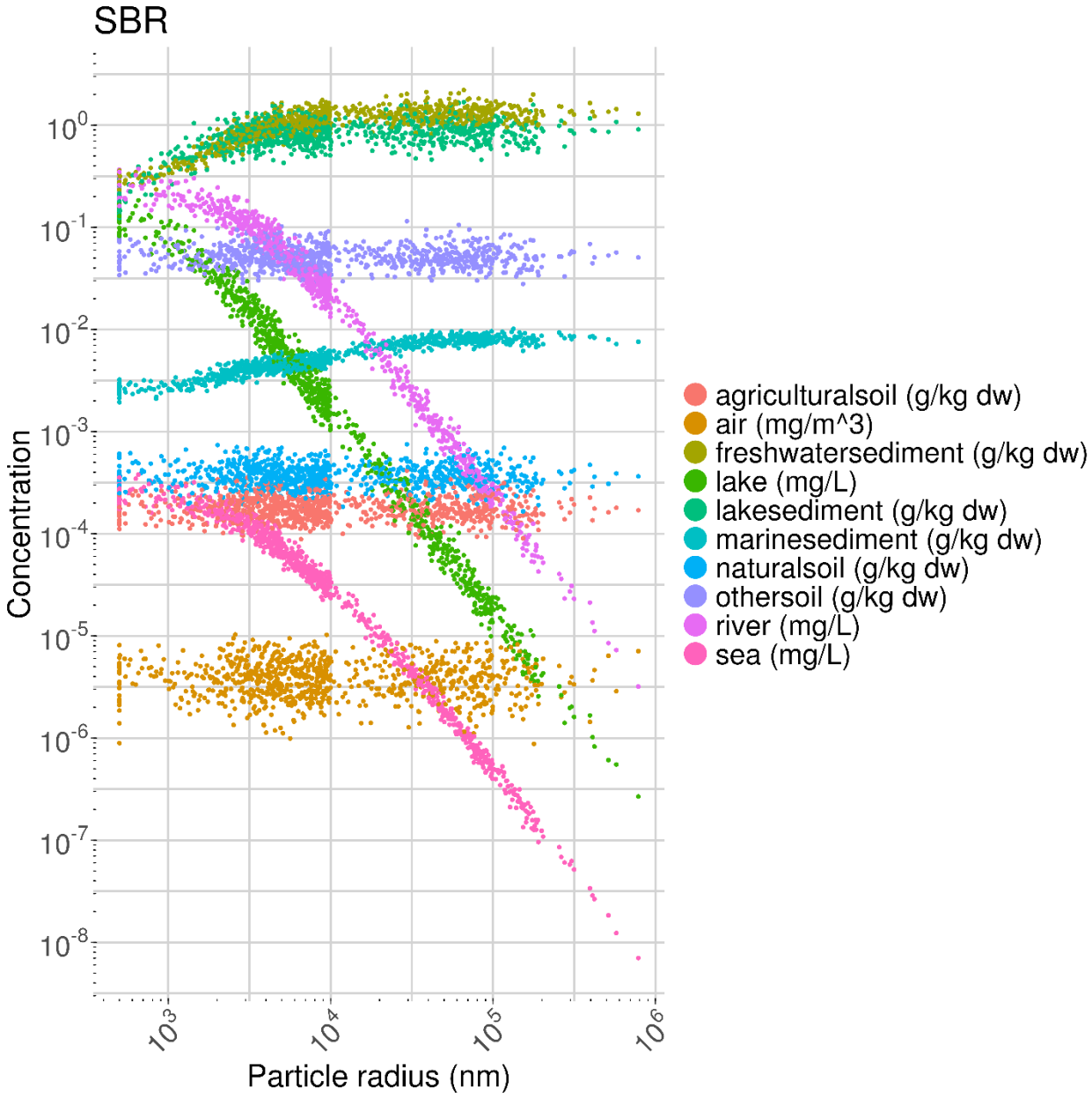

Figure 18 SBR concentrations versus Radius of Tyre Wear particles for different environmental compartments at EU scale. Logarithmic scale on x-axis and y-axis.

One of the main concerns of microplastics is their persistence, meaning degradation to basic organic or mineral molecules is very low, taking tens to thousands of years. From the degradation tests being conducted on Tyre wear it became apparent that Natural Rubber had a significantly higher degradation rate compared to SBR (Parker et al., 2024; van Os et al., 2024). For this reason a much higher degradation half-life is

assumed for NR, between ~25 days and 1 year compared to SBR, between ~8 years and more than a 1000 years.
Based on the model estimate the concentration NR is expected to be about 1% of the SBR concentration in road-side soils (Figure 19). This is about 10 fold lower than what is being found in soil and sediment measurements during a limited field measurement campaign (Tromp and Esveld, 2023). This could mean that the degradation half-lives are too high for NR, the SBR half lives are taken too low or the estimate of NR emission is estimated too low. The difference in NR composition of different tyre types mostly linked to vehicle types is not considered in the DPMFA model. It is for instance known that heavy-duty truck tyres have on average a much larger NR content compared to passenger car tyres (Giechaskiel et al., 2024) Follow-up research should focus on increasing the uncertainty in these type of studies in order to better understand the NR and SBR content in our environment and the difference of impact on the ecosystem based on this. This can contribute to a better understanding of future development of safe and sustainable materials for tyres.

The concentration dynamics of Tyre Wear in time (Figure 20) are closely related to the emission dynamics (Figure 15). They clearly illustrate the increase of concentrations in time due to the increased emissions, but also the dip after 2020 can still be distinguished.

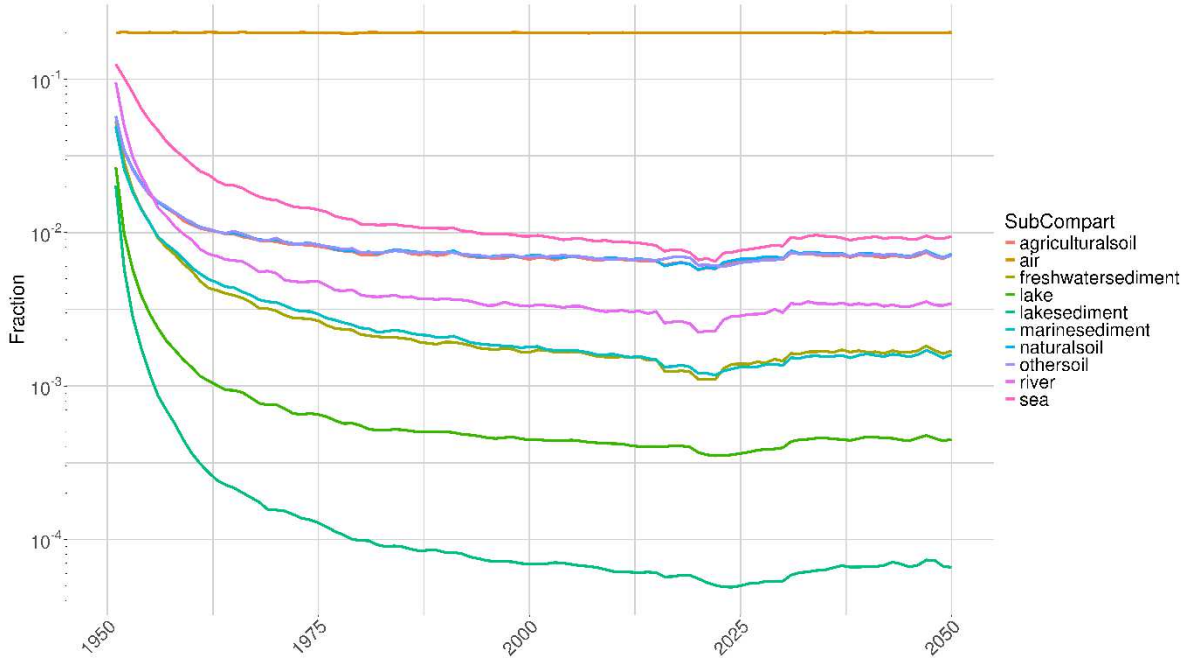

Figure 19. The average Natural rubber fraction over time per environmental compartment for EU scale. Logarithmic scale on y-axis.

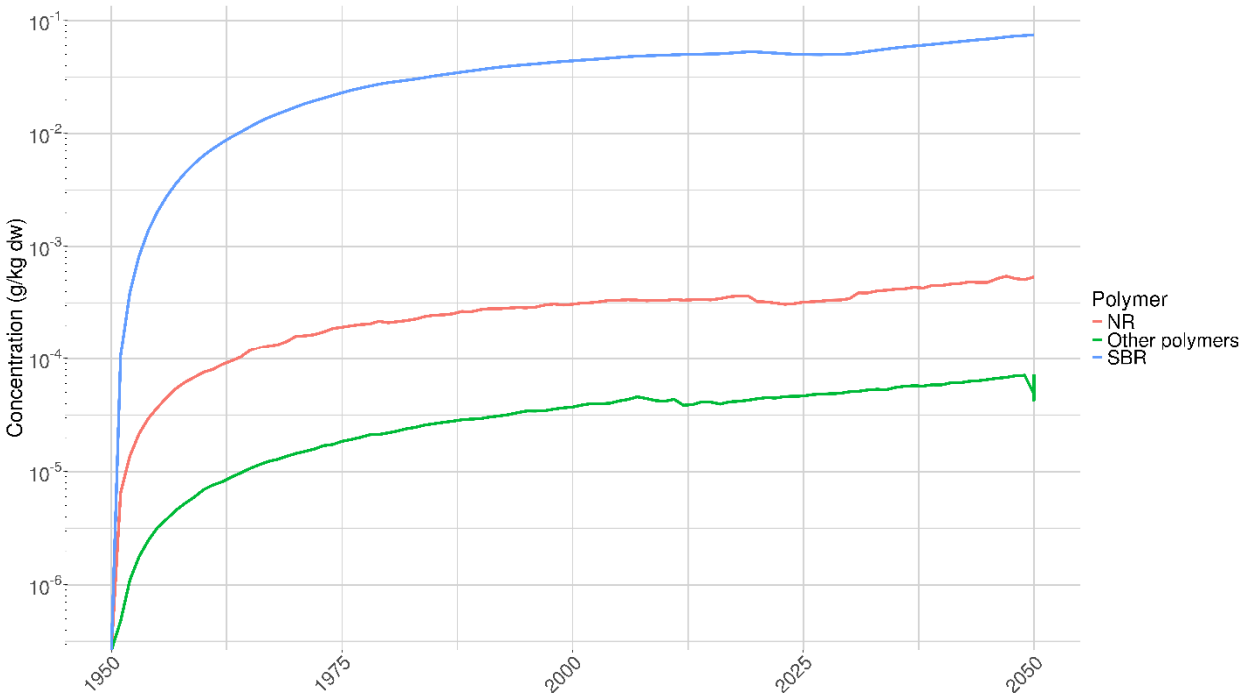

Figure 20 NR and SBR concentration over time in other soil compartment at continental scale. Logarithmic scale on y-axis.

3.3.3 Comparison to field measurements

The SimpleBox4plastics estimates for Tyre Wear rubber in air, freshwater and sediments correspond well to the measured concentrations (Figure 21). Although for air a lot higher concentrations are also measured, one should keep in mind that the model estimates are to be considered background concentrations. For other soil, which includes road side soil, the model estimates are a lot lower compared to measurements. As discussed above this might be due to an overestimation of the runoff rate from soil to freshwater. Additionally, it is common for local measurements close to the emission source to have a lot higher concentrations compared to remote area's which would be more in line with the background concentrations. This is for instance the one point for agricultural soil, which was measured in rural area 350 meters from a provincial road. Even this concentration is higher compared to the othersoil concentration. However, the depth of the considered soil compartment could also play a role. SimpleBox by default considers a soil depth of 20 cm for agricultural soil (ploughing depth) and 5 cm for other soil and natural soil. Nevertheless, this is something which requires some further research.

Both measurements and model estimates include uncertainty and variability. The comparison between model estimates and measurement illustrate that adjusting for the emission dynamics in time produces concentration estimates for air, water and sediment that correspond well. Some attention is needed to better estimating the soil concentration. Nevertheless, one should make sure the model is fit to the purpose and in this case one can argue that prioritization of mitigation measures can already be supported by taking into account the dispersion of microplastics through the environment using SimpleBox4Plastics.

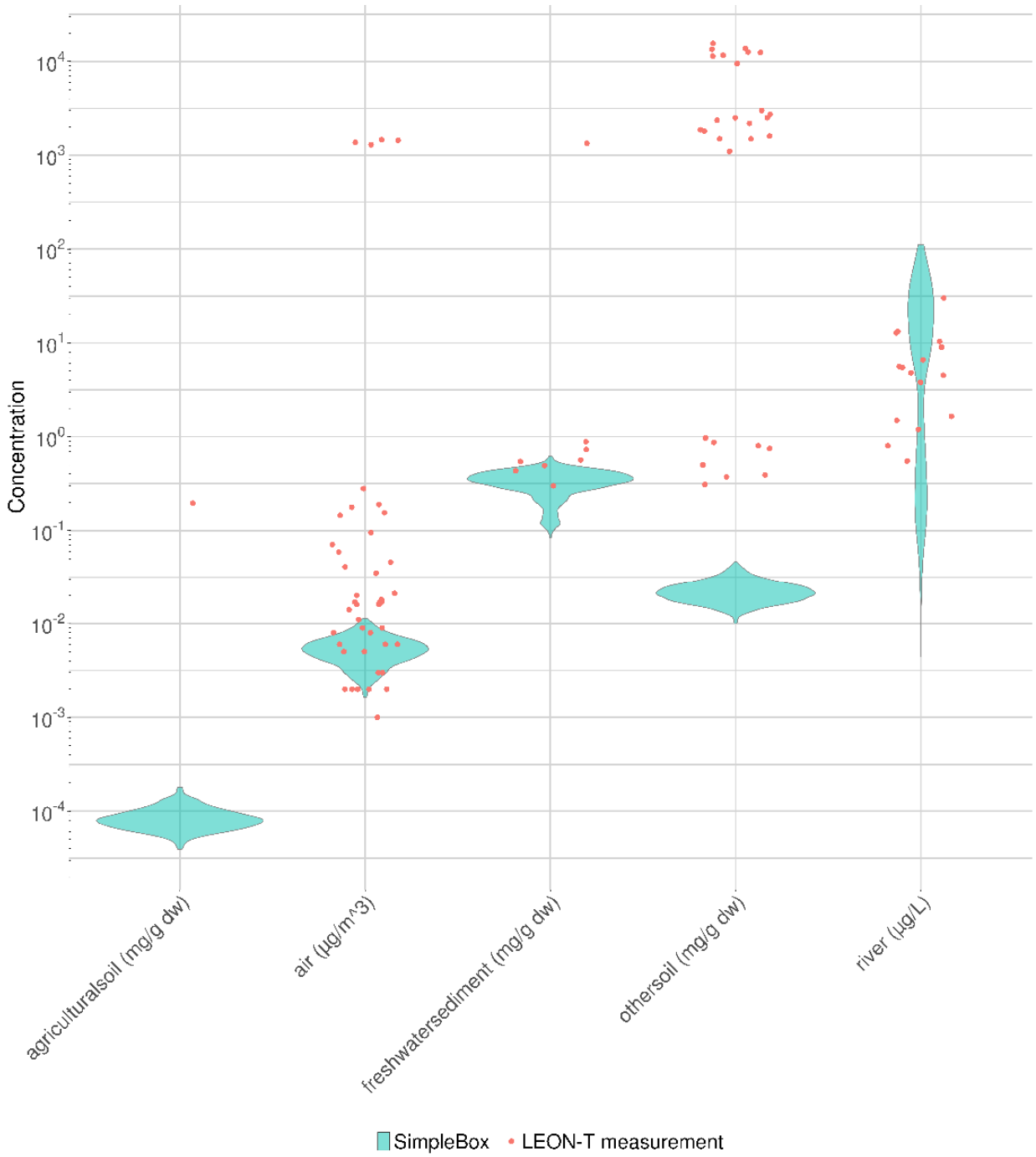

Figure 21. Comparison between modelled and measured concentrations for the Netherlands (Regional scale). Using emissions for 2019, Logarithmic scale on y-axis.

4 Conclusions

This study showed the utilization of tyre wear release, emission and fate modelling in order to better understand the dispersion of tyre wear particles and other microplastics in the natural environment. The modelling approach developed takes most of the known uncertainty and variability into account in order to quantify the potential under or overestimation of microplastics concentrations in our environment. It is clear however,

that the significant amount of microplastics entering our environment will lead to large build up in soils and sediments as clear sinks of tyre wear particles and most microplastics. Some work on estimating local tyre wear and other microplastics buildup could be useful for more site-specific data, but overall this modelling study illustrates important aspects of microplastic fate and transport.

The model outcome was compared to measured Tyre wear concentrations which clearly overlap, but for soils some additional work is needed to get better model estimates.

These results further highlight the need to apply such modelling for the appropriate goals. For instance, one can apply this for prioritization of mitigation measures, but less so for applications where an absolute estimate is relevant until the models uncertainty and performance for soil can be reduced. Overall, the model approach linking emissions modelling to fate modelling can provide a powerful tool in linking observed microplastics to sources. Assisting in further mitigating plastic pollution.

5 References

ADAC, 2022. Tyre wear particles in the environment Tyre abrasion: wear and burden on the environment. Allgemeiner Deutscher Automobil-Club, München.

Bergmann, M., Mutzel, S., Primpke, S., Tekman, M.B., Trachsel, J., Gerdts, G., 2019. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. Sci Adv 5, eaax1157. https://doi.org/10.1126/sciadv.aax1157

Boersma, A., Grigoriadi, K., Nooijens, M.G.A., Henke, S., Kooter, I.M., Parker, L.A., Dortmans, A., Urbanus, J.H., 2023. Microplastic Index—How to Predict Microplastics Formation? Polymers 15, 2185. https://doi.org/10.3390/polym15092185

Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., McGivern, A., Murphy, E., Jambeck, J., Leonard, G.H., Hilleary, M.A., Eriksen, M., Possingham, H.P., De Frond, H., Gerber, L.R., Polidoro, B., Tahir, A., Bernard, M., Mallos, N., Barnes, M., Rochman, C.M., 2020. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. Science 369, 1515–1518. https://doi.org/10.1126/science.aba3656

Bremmer, H., van Engelen, J., 2007. Paint Products Fact Sheet. To assess the risks for the consumer. (No. 320104008). RIVM.

Centraal Bureau voor de Statistiek, 2017. Bodemgebruik; uitgebreide gebruiksvorm, per gemeente [WWW Document]. Centraal Bureau voor de Statistiek. URL https://www.cbs.nl/nl-nl/cijfers/detail/70262ned (accessed 10.30.24).

Chamas, A., Moon, H., Zheng, J., Qiu, Y., Tabassum, T., Jang, J.H., Abu-Omar, M., Scott, S.L., Suh, S., 2020. Degradation Rates of Plastics in the Environment. ACS Sustainable Chemistry & Engineering 8, 3494–3511. https://doi.org/10.1021/acssuschemeng.9b06635

EC, 2023. Preliminary analysis of the main sources of microplastic emissions, ANNEX 15 of the IMPACT ASSESSMENT REPORT on Combatting microplastics pollution in the Eurpean Union.

Eurostat, 2024. Sewage sludge production and disposal from urban wastewater (in dry substance (d.s)). https://doi.org/10.2908/TEN00030

Faizah, R., Priyosulistyo, H., Aminullah, A., 2019. The Properties of Waste Rubber Tires in Increasing the Damping of Masonry Wall Structure. IOP Conf. Ser.: Mater. Sci. Eng 650, 012041. https://doi.org/10.1088/1757-899X/650/1/012041

Geilenkirchen, G., Bolech, M., Hulskotte, J., Dellaert, S., Ligterink, N., Sijstermans, M., Geertjes, K., Felter, K., 't Hoen, M., 2023. Methods for calculating the emissions of transport in the Netherlands. PBL Netherlands Environmental Assessment Agency, The Hague.

Giechaskiel, B., Grigoratos, T., Mathissen, M., Quik, J., Tromp, P., Gustafsson, M., Franco, V., Dilara, P., 2024. Contribution of Road Vehicle Tyre Wear to Microplastics and Ambient Air Pollution. Sustainability 16, 522. https://doi.org/10.3390/su16020522

Herzke, D., Schmidt, N., Evangeliou, N., Eckhardt, S., Schulze, D., 2024. Comparison of Atmospheric Microplastic in remote and urban locations in Norway; occurrence, composition and sources. Presented at the MICRO, Lanzarote.

Hoeke, S., Van Wijnen, J., Krikke, H., Löhr, A., Ragas, A.M.J., 2024. Mapping the tire supply chain and its microplastics emissions using a multi-stakeholder approach. Resources, Conservation and Recycling 203, 107389. https://doi.org/10.1016/j.resconrec.2023.107389

Hollander, A., Schoorl, M., van de Meent, D., 2016. SimpleBox 4.0: Improving the model while keeping it simple…. Chemosphere 148, 99–107. https://doi.org/10.1016/j.chemosphere.2016.01.006

Kaandorp, M.L.A., Dijkstra, H.A., van Sebille, E., 2021. Modelling size distributions of marine plastics under the influence of continuous cascading fragmentation. Environmental Research Letters. https://doi.org/10.1088/1748-9326/abe9ea

Kawecki, D., Scheeder, P.R.W., Nowack, B., 2018. Probabilistic Material Flow Analysis of Seven Commodity Plastics in Europe. Environ Sci Technol 52, 9874–9888. https://doi.org/10.1021/acs.est.8b01513

Kawecki, D., Wu, Q., Gonçalves, J.S.V., Nowack, B., 2021. Polymer-specific dynamic probabilistic material flow analysis of seven polymers in Europe from 1950 to 2016. Resources, Conservation and Recycling 173, 105733. https://doi.org/10.1016/j.resconrec.2021.105733

Kim, T., Eo, S., Shim, W.J., Kim, M., 2024. Qualitative and quantitative assessment of microplastics derived from antifouling paint in effluent from ship hull hydroblasting and their emission into the marine environment. Journal of Hazardous Materials 477, 135258. https://doi.org/10.1016/j.jhazmat.2024.135258

Koelmans, A.A., Kooi, M., Law, K.L., van Sebille, E., 2017. All is not lost: deriving a top-down mass budget of plastic at sea. Environmental Research Letters 12, 114028. https://doi.org/10.1088/1748-9326/aa9500

Lau, W.W.Y., Shiran, Y., Bailey, R.M., Cook, E., Stuchtey, M.R., Koskella, J., Velis, C.A., Godfrey, L., Boucher, J., Murphy, M.B., Thompson, R.C., Jankowska, E., Castillo Castillo, A., Pilditch, T.D., Dixon, B., Koerselman, L., Kosior, E., Favoino, E., Gutberlet, J., Baulch, S., Atreya, M.E., Fischer, D., He, K.K., Petit, M.M., Sumaila, U.R., Neil, E., Bernhofen, M.V., Lawrence, K., Palardy, J.E., 2020. Evaluating scenarios toward zero plastic pollution. Science 369, 1455–1461. https://doi.org/10.1126/science.aba9475

Liu, Z., Nowack, B., 2022. Probabilistic material flow analysis and emissions modeling for five commodity plastics (PUR, ABS, PA, PC, and PMMA) as macroplastics and microplastics☆. Resources, Conservation and Recycling 179, 106071. https://doi.org/10.1016/j.resconrec.2021.106071

Materic, D., Ludewig, E., Brunner, D., Rockmann, T., Holzinger, R., 2021. Nanoplastics transport to the remote, high-altitude Alps. Environ Pollut 288, 117697. https://doi.org/10.1016/j.envpol.2021.117697

Mathissen, M., Gramstadt, S., Gustafsson, M., Svensson, N., Tromp, P., 2023. Final results from the assessment and characterization of tyre particle emissions (No. LEON-T Deliverable 2.2).

Meesters, J.A.J., Koelmans, A.A., Quik, J.T.K., Hendriks, A.J., van de Meent, D., 2014. Multimedia Modeling of Engineered Nanoparticles with SimpleBox4nano: Model Definition and Evaluation. Environ. Sci. Technol. 48, 5726–5736. https://doi.org/10.1021/es500548h

Meesters, J.A.J., Quik, J.T.K., 2024. Updated emission model for local and EU scale (No. LEON-T Deliverable 3.4). RIVM.

OECD, 2022. Global Plastics Outlook: Economic Drivers, Environmental Impacts and Policy Options. OECD. https://doi.org/10.1787/de747aef-en

Omnexus, 2024. Density of Plastics Material: Technical Properties Table [WWW Document]. URL https://web.archive.org/web/20240712211553/https:/omnexus.specialchem.com/polymer-property/density (accessed 10.31.24).

Parker, L., van Os, M., Hoppener, E.M., Tromp, P., Boersma, A., van Renesse van Duivenbode, A., Hartmann, N.B., Afshar, S.V., 2024. Degradation and analysis of tyre and road wear particles (TRWP) (No. LEON-T Deliverable 3.3 incl. Addendum). TNO.

Quik, J.T.K., Hids, A.R., Steenmeijer, M.A., Mellink, Y., van Bruggen, A., 2024. Emission of Microplastics to Water, Soil, and Air. What can we do about it? (RIVM report No. 2024– 0106). Rijksinstituut voor Volksgezondheid en Milieu RIVM, Bilthoven, The Netherlands.

Quik, J.T.K., Meesters, J.A.J., Koelmans, A.A., 2023. A multimedia model to estimate environmental fate of microplastic particles. Science of The Total Environment 163437. https://doi.org/10.1016/j.scitotenv.2023.163437

RSC, 2024. Identifying polymers by density [WWW Document]. Identifying polymers by density. URL https://web.archive.org/web/20240716024904/https://edu.rsc.org/experiments/identifying-polymers-by-density/385.article (accessed 10.31.24).

Rutgers, M., Faber, M., Waaijers-van der Loop, S., Quik, J., 2022. Microplastics in soil systems, from source to path to protection goals. State of knowledge on microplastics in soil. Rijksinstituut voor Volksgezondheid en Milieu RIVM. https://doi.org/10.21945/RIVM-2021-0224

Rutsch, M., Rieckermann, J., Krebs, P., 2006. Quantification of sewer leakage: a review. Water Sci Technol 54, 135–144. https://doi.org/10.2166/wst.2006.616

RWS, 2022. Bandenslijtage wegverkeer (No. Versie juli 2022), Emissieschattingen Diffuse bronnen Emissieregistratie.

Schwarz, A.E., Lensen, S.M.C., Langeveld, E., Parker, L.A., Urbanus, J.H., 2023. Plastics in the global environment assessed through material flow analysis, degradation and environmental transportation. Science of The Total Environment 875, 162644. https://doi.org/10.1016/j.scitotenv.2023.162644

Shams, M., Alam, I., Chowdhury, I., 2020. Aggregation and stability of nanoscale plastics in aquatic environment. Water Res 171, 115401. https://doi.org/10.1016/j.watres.2019.115401

Sieber, R., Kawecki, D., Nowack, B., 2020. Dynamic probabilistic material flow analysis of rubber release from tires into the environment. Environ Pollut 258, 113573. https://doi.org/10.1016/j.envpol.2019.113573

Thompson, R.C., Courtene-Jones, W., Boucher, J., Pahl, S., Raubenheimer, K., Koelmans, A.A., 2024. Twenty years of microplastic pollution research—what have we learned? Science 386, eadl2746. https://doi.org/10.1126/science.adl2746

Tromp, P., Esveld, J., 2023. Results from Monitoring TWP (results from selective field measurements) in different environmental compartments) (No. LEON-T Deliverable 3.2). TNO.

Unice, K.M., Weeber, M.P., Abramson, M.M., Reid, R.C.D., van Gils, J.A.G., Markus, A.A., Vethaak, A.D., Panko, J.M., 2019. Characterizing export of land-based microplastics to the estuary - Part II: Sensitivity analysis of an integrated geospatial microplastic transport modeling assessment of tire and road wear particles. Sci Total Environ 646, 1650–1659. https://doi.org/10.1016/j.scitotenv.2018.08.301

van Os, M., Nooijens, M.G.A., van Renesse van Duivenbode, A., Tromp, P.C., Hoppener, E.M., Grigoriadi, K., Boersma, A., Parker, L.A., 2024. Degradation rates and ageing effects of UV on tyre and road wear particles. Chemosphere Submitted.

Verschoor, A.J., de Poorter, L.R.M., Droge, R., Kuenen, J., de Valk, E., 2016. Emission of microplastics and potential mitigation measures - Abrasive cleaning agents, paints and tyre wear (No. 2016– 0026).

Verschoor, A.J., de Valk, E., 2018. Potential measures against microplastic emissions to water (No. RIVM Report 2017-0193). RIVM.

Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., Sillanpää, M., 2020. Atmospheric microplastics: A review on current status and perspectives. Earth-Science Reviews 203. https://doi.org/10.1016/j.earscirev.2020.103118

6 Supplementary information

6.1 Material Flow Analysis modelling

6.1.1 Sewage sludge disposal in the EU

In the last version of the DPMFA_NL_EU model all sewage sludge was assumed to be incinerated for the Netherlands and the EU. While this holds true for the Netherlands, sewage sludge in other EU countries is also used for other applications. To better reflect this in the emission modelling, additional data on sewage sludge disposal in other EU27 countries was collected (Eurostat, 2024). This data was consequently used to calculate transfer coefficients to other compartments.
The transfer coefficients were calculated by aggregating the data on sewage sludge disposal for the EU27 countries excluding the Netherlands and calculating the fractions from the total for each application (Table S1**Error! Reference source not found.Error! Reference source not found.**).

Table S1 Sludge applications and corresponding transfer coefficients. TCs are assumed to be the same for sludge containing microplastics and sludge containing macroplastics.

Application	Mass (kt)	TC	To DPMFA compartment
Sludge disposal - agricultural use	1064.5	0.26	Agricultural soil (micro) / agricultural soil (macro)
Sludge disposal - compost and other applications	661.94	0.16	Compost (micro) / compost (macro)
Sludge disposal - landfill	231.43	0.06	Landfill
Sludge disposal - incineration	1707.41	0.42	Incineration
Sludge disposal - other	500.36	0.12	Secondary material reuse
Total	4076.83	1	-

6.1.2 Adjustments DPMFA model for SimpleBox

The input data entered into the MainInputFile for the DPMFA model is ‘pretreated’ in the DPMFA model. In the script Input2csv.py, the input data from the MainInputFile is first interpolated from the chosen start year to the chosen end year using the OECD plastics outlook (OECD, 2022). This interpolation results in a dataframe with inputdata for all years between the start year and end year of the model run. After this step in the script, the input data for NL is subtracted from the input data for EU for each category, material and year.

One combination of category, year and material can have one or two masses: a high and a low estimate. If there were two input masses for one combination, the rules in table S3 were followed.

Table S2 Rules for subtracting NL values from EU values.

# Entries NL	# Entries EU	Rule
1	1	Value EU - value NL
1	2	High value EU – value NL, low value EU – value NL
2	1	EU value – high value NL, EU value – low value NL
2	2	High value EU – high value NL, low value EU – low value NL

6.1.3 Tyre wear release estimates for NL and EU

Table S3 Emission factors for different vehicle types and road types (ADAC, 2022; Geilenkirchen et al., 2023). Emission factors from Geilenkirchen et al. (2023) were multiplied with high (1.66) and low (0.57) factors from ADAC (2022).

Vehicle type	Urban (mg/km)	Rural (mg/km)	Highway (mg/km)
Passenger cars	92-169	59-109	73-134
Motorcycles	42-77	27-50	33-60
Mopeds	9-17	6-50	7-60
Delivery vans	111-204	71-131	87-161
Lorries	593-1091	381-701	466-858
Busses	289-533	186-343	227-419

Table S4. Low and high estimates of tyre wear release (kt) for the Netherlands and EU-27.

Year	Scale	Release low (kt)	Release high (kt)	Source
1990	NL	9.8	18.0	Calculation based on (RWS, 2022)
1995	NL	10.5	19.4	Calculation based on (RWS, 2022)
2000	NL	11.7	21.5	Calculation based on (RWS, 2022)
2005	NL	12.4	22.8	Calculation based on (RWS, 2022)
2010	NL	12.6	23.2	Calculation based on (RWS, 2022)
2015	NL	12.7	23.4	Calculation based on (RWS, 2022)
2019	NL	13.5	24.8	Calculation based on (RWS, 2022)
2020	NL	11.5	21.5	Calculation based on (RWS, 2022)
1990	EU	274.4	505.2	Scaling from NL to EU using population
1995	EU	289	532.2	Scaling from NL to EU using population
2000	EU	299.7	580.3	Scaling from NL to EU using population
2005	EU	315.1	608.6	Scaling from NL to EU using population
2010	EU	330.5	615.0	Scaling from NL to EU using population
2015	EU	333.4	613.9	Scaling from NL to EU using population
2019	EU	347.5	640.0	Scaling from NL to EU using population
2020	EU	299.7	551.8	Scaling from NL to EU using population

6.1.1 Tyre wear emissions results

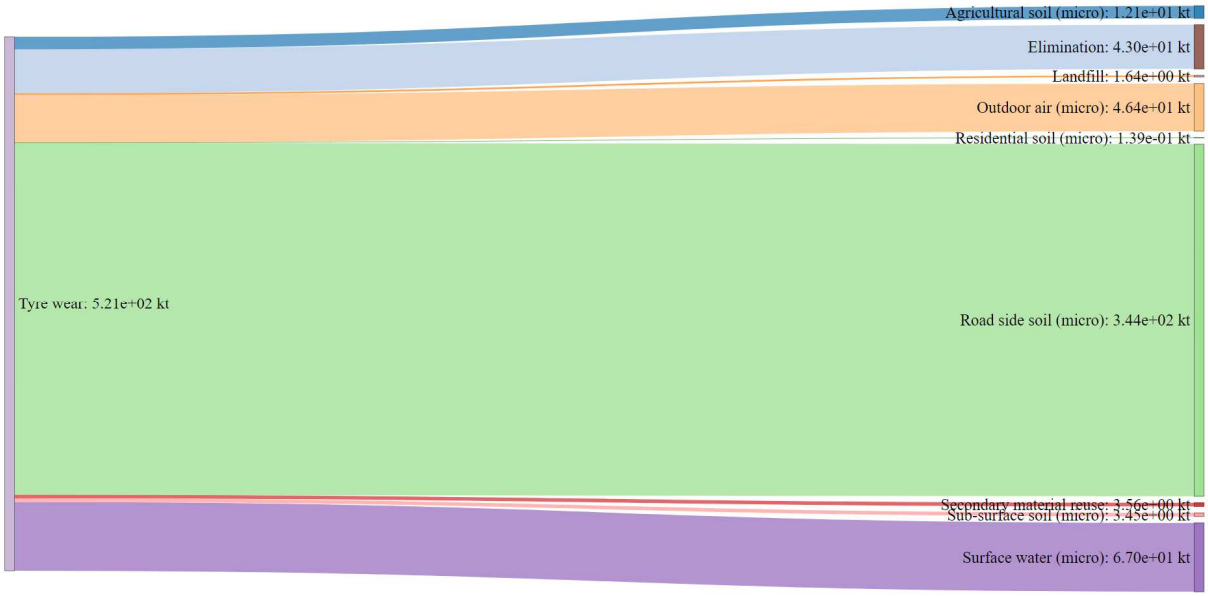

Figure S1 Sankey diagram depicting the mean tyre wear rubber emissions to different sink compartments.

Table S5 Mean tyre wear masses (kt) to different sinks, including percentages of total emissions.

Sink	**Mean mass (kt)**	**Percentage of total emissions**
Agricultural soil (micro)	12.1	2.32%
Elimination	43	8.25%
Landfill	1.64	0.31%
Outdoor air (micro)	46.4	8.90%
Residential soil (micro)	0.139	0.03%
Road side soil (micro)	344	65.99%
Secondary material reuse	3.56	0.68%
Sub-surface soil (micro)	3.45	0.66%
Surface water (micro)	67	12.85%
Total	**521.289**	**100.00%**

6.2 Environmental Fate modelling

6.2.1 Rescale regional scale SimpleBox to the Netherlands

By default, the regional scale of SimpleBox is set to the river Rhine catchment area. The DPMFA model however produces emission data for the Netherlands and the EU. Thus, the regional scale of SimpleBox should be rescaled to the Netherlands. The land, sea, freshwater, and their subcompartment areas were obtained from Centraal Bureau voor de Statistiek (2017) (Table S4, Table S5). The areas of the regional scale were adjusted in SimpleBox by using the MutateVar function. Based on a yearly average inflow of 2430 m^3/s for freshwater in the Netherlands from the Meuse and Rhine rivers a regional discharge fraction is applied of 0.1.

Table S6 Sea, freshwater and land area in the Netherlands (Centraal Bureau voor de Statistiek, 2017).

	Area (km^2)	**Fraction of total area**
Total sea	4152	0.100
Total freshwater + Total land	37391	0.900
Total area	**41543**	**1**

Table S7 Soil and freshwater compartment areas and their fractions (Centraal Bureau voor de Statistiek, 2017).

	Area (km^2)	**Fraction of land + freshwater area**
Agricultural soil	22304	0.597
Lake	1984	0.053
Natural soil	5015	0.134
Other soil	6328	0.169
River	1760	0.047
Total	**37391**	**1**

6.2.2 Specification of SimpleBox variables

Table S8 Attachment efficiency based on (Quik et al., 2023) and applied to all polymers.

Subcompartment	**Species**	**Distribution**	**Min**	**Max**
Water	Small/large	Uniform	1e-4	1
Soil/sediment	Small/large	Uniform	1e-4	1

\

Table S9 Degradation rate constant for other polymers. Distributions based on (Chamas et al., 2020) as applied in (Quik et al., 2023).

Subcompartment	**Species**	**Distribution**	**Min**	**Peak**	**Max**	**Unit**
Water	Any	Triangular	1e-20	2e-10	2e-9	s^{-1}

Soil/sediment	Any	Triangular	1e-20	3e-11	1e-9	s^{-1}

Table S10 Degradation rate constant for TRWP and NR/SBR from TRWP, applied to all Species. Distributions based on LEON-T Deliverable 3.3.

Subcompartment	Polymer	Distribution	Min	Peak	Max	Unit
Water	NR	Triangular	2.3E-08	4.6E-08	3.1E-07	s^{-1}
Soil/sediment	NR	Triangular	2.3E-08	3.1E-07	4.6E-07	s^{-1}
Water	SBR	Triangular	1.0E-20	2.89E-10	6.42E-10	s^{-1}
Soil	SBR	Triangular	1.0E-20	2.89E-09	6.42E-09	s^{-1}
Sediment	SBR	Triangular	1.0E-20	2.1E-11	3.0E-11	s^{-1}

Table S11 Fragmentation rate constant for other microplastics. Distributions based on (Kaandorp et al., 2021; Koelmans et al., 2017) as applied in (Quik et al., 2023).

Subcompartment	Species	Distribution	Min	Peak	Max	Unit
Water	Small/large	Triangular	1.6e-9	2.7e-8	1.3e-7	s^{-1}
Water	Solid	Uniform	1.0e-20		1.0e-19	s^{-1}
Soil/sediment	Small/large	Triangular	1.0e-20	2.7e-8	1.3e-7	s^{-1}
Soil/sediment	Solid	Uniform	1.0e-20		1.0e-19	s^{-1}

Table S12 Fragmentation rate constant for TRWP to TWP. Distributions based on LEON-T Deliverable 3.3.

Subcompartment	Species	Distribution	Min	Peak	Max	Unit
Water	Small/large	Triangular	2.3e-9	2.8e-9	4.5e-9	s^{-1}
Soil/sediment	Small/large	Triangular	2.3e-9	2.8e-9	4.5e-9	s^{-1}
Water/Soil/Sediment	Solid	Uniform	1.0e-20		1.0e-19	s^{-1}

Table S13 Size distributions for radius of other microplastic particles, mostly based on the wear and impaction particle size generation as calculated by (Boersma et al., 2023).

Polymer	Distribution	Min (µm)	Max (µm)	Source
HDPE	Uniform	3.15	120	(Boersma et al., 2023)
LDPE	Uniform	2.50	136	(Boersma et al., 2023)
PP	Uniform	2.20	36	(Boersma et al., 2023)
PS	Uniform	0.50	1.90	(Boersma et al., 2023)
PVC	Uniform	1.30	12	(Boersma et al., 2023)
Acryl	Uniform	20.00	100	(Kim et al., 2024)
PA	Uniform	0.45	38	(Boersma et al., 2023)
PET	Uniform	1.65	26	(Boersma et al., 2023)
ABS	Uniform	1.05	20.50	(Boersma et al., 2023)
EPS	LogUniform	1e-3	2.50e3	Assumption
PC	Uniform	1.70	23	(Boersma et al., 2023)
PMMA	Uniform	0.15	1.60	(Boersma et al., 2023)
PUR	LogUniform	1e-3	2.50e3	Assumption
Rubber	Uniform	0.50	2500	Assumption
Other	LogUniform	1e-3	2.50e3	Assumption
TRWP	Emperical	-	-	See SI 6.1.7

Table S14 Distributions of Polymer density, Rubber density also applied to NR and SBR.

Polymer	Distribution	Min	Max	Unit	Source
HDPE	Uniform	940	960	kg/m^3	(RSC, 2024)
LDPE	Uniform	910	930	kg/m^3	(RSC, 2024)
PP	Uniform	890	910	kg/m^3	(RSC, 2024)
PS	Uniform	1040	1110	kg/m^3	(RSC, 2024)
PVC	Uniform	1200	1551	kg/m^3	(RSC, 2024)
Acryl	Uniform	1200	1500	kg/m^3	(Bremmer and van Engelen, 2007)
PA	Uniform	1030	1600	kg/m^3	(Omnexus, 2024)
PET	Uniform	1380	1400	kg/m^3	(RSC, 2024)
ABS	Uniform	1020	1250	kg/m^3	(Omnexus, 2024)
EPS	Uniform	20	30	kg/m^3	(RSC, 2024)
PC	Uniform	1150	1590	kg/m^3	(Omnexus, 2024)
PMMA	Uniform	1100	1250	kg/m^3	(Omnexus, 2024)
PUR	Uniform	15	17	kg/m^3	Assuming foam
Rubber	Uniform	1090	1136	kg/m^3	(Faizah et al., 2019)
Other	Uniform	800	1500	kg/m^3	Largest possible range, excluding foams

6.2.3 TRWP size distribution

Mass fractions of size ranges were reported by TNO in Deliverable 3.2. This data consisted of 11 samples (2 soil, 3 deposition, 3 runoff and 3 sediment). The fractionation was performed with a wet sieving method for fractions: 25-50, 50-100, 100-200, 200-400 and 400-1000µm. The fraction < 25µm was further fractionated with

cascade filtration using track-etched membrane polycarbonate filters in four fractions: <2, 2-5, 5-10 and 10-25µm. All fractions were analysed with TED-GCMS. A probability density functions was fitted to this data by calculating the normalized cumulative sum of the mass fraction per size, and using the approxfun function to fit a distribution to this data. The distribution was used to determine the particle size of TRWP particles in each SimpleBox run (Figure S2).

Table S15 Mass based size distribution used as basis for the size distribution in simplebox.

Particle size range TNO (µm)	Particle size SimpleBox (µm)	Mass fraction
0.5-2	0.5	0.026111
2-5	2	0.074014
5-10	5	0.246252
10-25	10	0.23123
25-50	25	0.064539
50-100	50	0.103705
100-200	100	0.127457
200-400	200	0.074421
400-1000	1000	0.05385

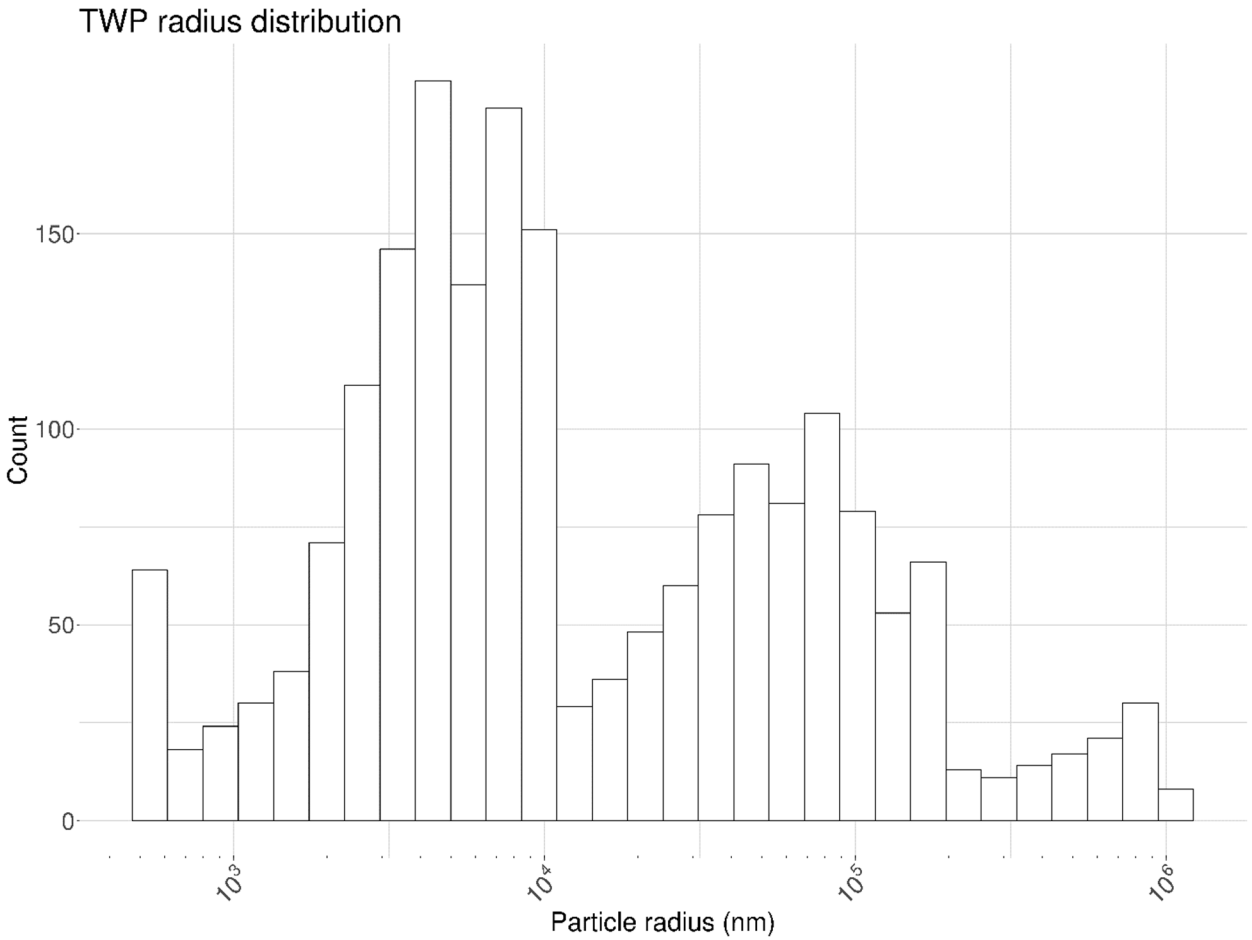

Figure S2 Distribution of sampled radii for TWP (n=1000). Logarithmic scale on x-axis.

6.2.4 Distribution between NR and SBR

Average NR fractions in TRWPs were reported by TNO in Deliverable 3.2. From these average fractions, the minimum (9.8%), mean (17.8%) and maximum (32.6%) values were derived. A triangular distribution was made with these values, and this distribution was used to determine the mass of NR and SBR per run of the DPMFA model for tyre wear (Figure S3). This results in an emission estimate of the NR and SBR fraction of TRWP particles. Which is modelled separately to distinguish the degradation rate of these two components.

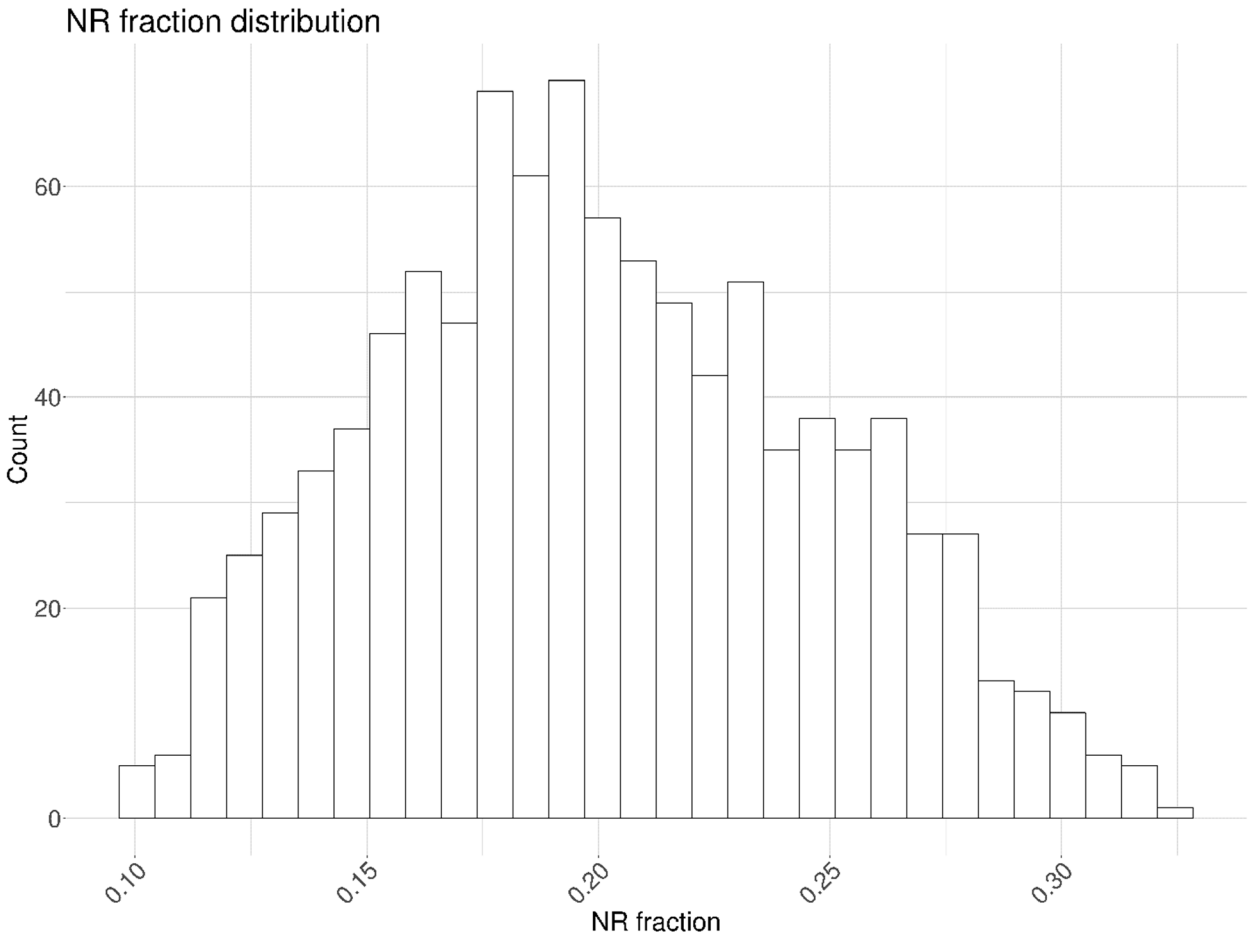

Figure S3 Histogram of the used NR fractions. 1000 runs.

Table S16 Fraction of NR compared to SBR in TWP samples.

Size range Fraction (µm)	Average fraction NR
2-5	9.79E-02
5-10	9.91E-02
10-25	1.27E-01
25-50	1.30E-01
50-100	2.05E-01
100-200	2.31E-01
200-400	3.26E-01
400-1000	2.09E-01

6.2.5 SimpleBox output completeness

Due to the uncertainty of several variables, not all runs were successful for all polymers. For Other sources, there were 510 unique runs successful for all polymers combined. For Tyre wear, 893 runs were successful for NR and SBR combined with data for 2050. As shown in Table

Table S17 Number of successful runs per polymer.

Source	Polymer	Number of complete runs
Other sources	ABS	1000
Other sources	Acryl	1000
Other sources	EPS	917
Other sources	HDPE	1000
Other sources	LDPE	973
Other sources	Other	972
Other sources	PA	1000
Other sources	PC	1000
Other sources	PET	1000
Other sources	PMMA	1000
Other sources	PP	1000
Other sources	PS	1000
Other sources	PUR	857
Other sources	PVC	1000
Other sources	Rubber	683
Tyre wear	NR	946
Tyre wear	SBR	945