Atmospheric Environment 351 (2025) 121184

Contents lists available at ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

Mass flow analysis of tire-wear particles, including carbon black, and implications for road dust management

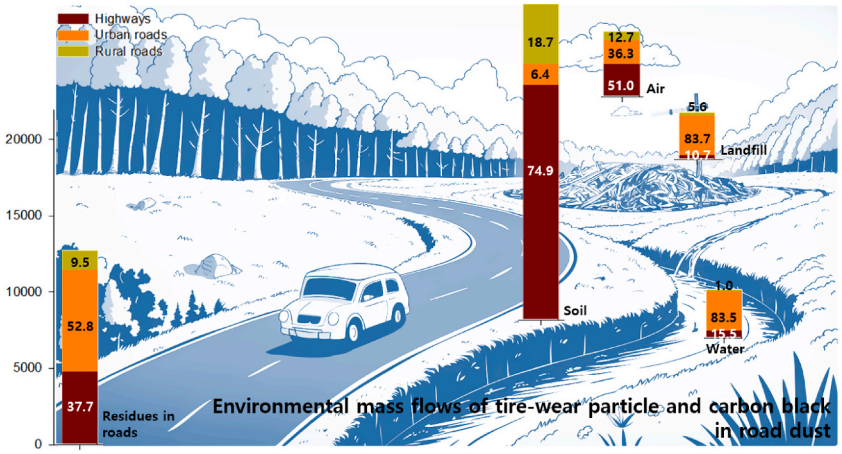
Min Gyu Lee [a,1], Hyeonjung Ryu [a,1], Minseung Hyun [a], Woosuk Chung [a], Jaehwan Hong [a,b], Hyunook Kim [b], Jung-Taek Kwon [c], Jaewoong Lee [c], Younghun Kim [a,*]

[a] Department of Chemical Engineering, Kwangwoon University, Seoul, 01897, South Korea
[b] Department of Environmental Engineering, University of Seoul, Seoul, 02504, South Korea
[c] Department of Environmental Health, Korea Environment Institute, 370 Sicheong-daero, Sejong, 30147, South Korea

HIGHLIGHTS

- Mass flow analysis highlighted significant exposure of TWP from highways to both the soil and air.
- TWP concentrations in soil decrease exponentially with increasing distance from roads.
- TWPs and CB in water bodies are identified as long-distance sources of contamination.
- Predicted environmental concentration (PEC) values align with literature ranges.

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
Tire-wear particle
Carbon black
Mass flow analysis
Road type
Vehicle type
Predicted environment concentration

ABSTRACT

Representative non-exhaust emissions from automobiles and tire-wear particles (TWPs) accumulate in road dust, causing their dispersion into the surrounding environment. TWPs undergo fragmentation due to continuous abrasion and weathering, resulting in the release of carbon black (CB), a major component of tire rubber. Although previous studies have conducted mass flow analyses (MFAs) for TWPs based on vehicle and road types, **a detailed MFA specifically addressing road dust, including CB-bound TWPs, has not been performed.** In this study, annual emissions of TWPs, including CB, from road dust were estimated based on different vehicle and road types. Mass flow diagrams were constructed to illustrate the distribution of these particles across technical and environmental compartments for each road type. The environmental mass of TWPs, including CB, in each compartment was calculated based on sewage system type, runoff ratio, road cleaning efficacy, and UV exposure. The MFA results indicate that highways contribute significantly to TWP exposure in soil and air, despite a

* Corresponding author.
E-mail addresses: molar1023@kw.ac.kr (M.G. Lee), korea1@kw.ac.kr (Y. Kim).
[1] Contributed equally.

https://doi.org/10.1016/j.atmosenv.2025.121184
Received 16 January 2025; Received in revised form 15 March 2025; Accepted 20 March 2025
Available online 21 March 2025

substantial portion of TWPs also accumulating as road residue. Additionally, the endpoint mass flow of the environmental media (**4241 t/a to air, 3140 t/a to water bodies, 20,602 t/a to soil, 3044 t/a to landfill, and 12,712 t/a to road residue)** and the predicted environmental concentration values **(1.16 μg/m^3 in air, 0.37 mg/L in water, and 1824 mg/kg in soil)** were consistent with literature ranges, supporting the reliability of the findings. These results underscore the importance of targeted road dust management strategies to mitigate the environmental impact of non-exhaust emissions.

1. Introduction

With the surge in traffic volume, particularly in urban areas, research has focused on pollutants originating from vehicular activities (Sharmilaa and Ilango, 2022). Traditionally, air quality management strategies have primarily targeted exhaust emissions to mitigate their impact on ambient air quality (Liu et al., 2022). However, particulate matter (PM) generated by non-exhaust sources, particularly from tire wear, influences not only air quality but also the broader environmental compartments, including roadside rivers, oceans, and soil (Baensch-Baltruschat et al., 2021; Timmers and Achten, 2016). As a major component of non-exhaust emissions, tire-wear particles in road dust have attracted increasing attention due to their contribution to microplastic pollution, their role in transporting heavy metals and organic pollutants, and their persistence in the environment (Baensch-Baltruschat et al., 2021; Kovochich et al., 2021; Wagner et al., 2022). Awareness of the environmental implications of TWPs has prompted a shift in focus toward understanding their sources, distribution, and fate in the environment (Wagner et al., 2022). This includes investigating the factors influencing TWP generation, transport pathways, and the potential risks they pose to both the environment and human health (Baensch-Baltruschat et al., 2020).

TWPs represent a complex mixture of micro-sized particles that originate from the wear and tear between tires and road surfaces (Wagner et al., 2018). The abrasion process is influenced by a multitude of factors, including sudden braking, acceleration, road roughness, tire type, vehicle load, and degree of wear (Järlskog et al., 2022; Kim and Lee, 2018). As tires interact with the road surface during vehicle operation, frictional forces cause a gradual breakdown of the tire treads, resulting in the release of TWPs into the surrounding environment. Notably, TWPs are not static entities; their physical and chemical properties change dynamically under environmental conditions. For instance, TWPs undergo photooxidation and further wear while on roads, leading to alterations in their size distribution and surface characteristics (Wagner et al., 2022; Luo et al., 2024a). Furthermore, carbon black (CB), a key reinforcing filler in tire rubber, significantly influences the environmental behavior of TWPs. During weathering, CB detaches from TWPs and contributes to the particulate load in road dust, exhibiting distinct physical and chemical properties compared to bulk TWPs (Kim et al., 2021). CB particles are finer and more chemically stable, allowing for prolonged atmospheric suspension and increased potential for inhalation exposure. Additionally, CB has a high surface area, enabling it to adsorb organic pollutants and heavy metals, which may enhance contaminant transport in aquatic and soil environments (Wi et al., 2023). Given these unique properties, understanding CB transport and accumulation is essential for assessing the broader environmental risks associated with TWP emissions. While tires also contain trace amounts of heavy metals, such as zinc and lead, their environmental behavior differs from that of CB and TWPs. Since this study focuses on the mass flow analysis of particulate emissions, heavy metal estimation was not included in the scope of this work (Lopez et al., 2023).

Despite the growing concern over TWPs and CB as sources of environmental contamination, comprehensive quantitative assessments of their transport and fate remain limited (Sieber et al., 2020). A robust evaluation of their environmental impact requires an in-depth mass flow analysis (MFA) to trace their movement across technical and environmental compartments. Moreover, the predicted environmental concentration (PEC) values are essential for estimating actual exposure levels and assessing potential risks to air, water, and soil. However, previous studies have largely focused on individual aspects of TWP emissions without integrating MFA and PEC to provide a holistic understanding of their distribution in different environmental media.

TWPs are initially generated on roads, and CB derived from weathering processes primarily accumulates in road dust. Subsequently, they can become airborne due to wind and vehicular movements, acting as resuspended dust particles (Wi et al., 2023). This resuspended dust can transport vehicle-derived particles not only in the vicinity of roads but also to more distant locations, including aquatic environments. As rubber is the primary component of TWPs, this material inherently possesses hydrophobic characteristics (Kovochich et al., 2021). However, weathering processes, such as photo-oxidation, can alter their surface properties, enhancing their dispersibility in water and allowing them to persist in aquatic environments for extended periods (Wagner et al., 2022). Additionally, as TWPs undergo further abrasion, their particle size may decrease, facilitating easier penetration into soil (Rødland et al., 2023). While TWPs and CB can adsorb pollutants, their adverse environmental effects outweigh any potential benefits. Adsorbed contaminants are eventually released due to desorption and degradation, and TWPs themselves leach toxic substances, including PAHs and benzothiazoles, which persist in ecosystems and contribute to aquatic toxicity (Foscari et al., 2023; Glaubitz et al., 2023). The dominant factor influencing aquatic exposure to TWPs and CB is likely transportation via runoff from road dust to roadside drains, sewers, and waterways, particularly during rainfall events or flooding (Zafra et al., 2017). In addition, periodic road-cleaning activities can also contribute to transportation (Amato et al., 2010). Therefore, understanding the quantitative fate of TWPs and CB in air, water, and soil is critical for developing effective mitigation strategies. A well-structured MFA can provide insights into their major transport routes, while PEC values enable the validation of estimated environmental concentrations against real-world data, improving the reliability of pollution management approaches.

The transport of these diverse environmental materials through road dust varies depending on the type of roadway infrastructure (Baensch-Baltruschat et al., 2021). For instance, highways, urban roads, and rural roads exhibit differences in the presence and design of roadside drainage systems, depending on whether they have combined or separate systems for handling sewage and stormwater. Moreover, the installation of separate stormwater drains along roads can affect the flow of road dust particles into water bodies and other environmental compartments (Rasmussen et al., 2024). Furthermore, the characteristics of road dust dispersion are influenced by the types of vehicles frequenting the roads. Variations in vehicle types can lead to differences in tire wear rates owing to factors such as vehicle weight and traffic volume (Chen et al., 2023). For example, heavy-duty vehicles exert more pressure on tires, increasing tire wear and subsequently raising TWP levels in road dust. Conversely, lighter vehicles or less frequently used vehicles may contribute to relatively lower TWP levels in road dust. These factors collectively contribute to the complex dynamics of road dust dispersion and its environmental impact on different types of roadways.

To address these gaps, this study aims to perform an MFA of TWPs and CB across different road types and vehicle categories, considering key transport mechanisms such as runoff, atmospheric resuspension, and road-cleaning processes. Additionally, the PEC values for air, water,

and soil are derived to assess the environmental relevance of the estimated TWP and CB distributions. To investigate the contribution of TWP generation by road and vehicle types and to quantify the migration of TWPs and CB from their sources to environmental compartments nationwide, this study analyzes their distribution patterns. Additionally, the effects of various technical components, including runoff treatment facilities, wastewater treatment plants (WWTP), waste incineration plants (WIP), street sweeping, and rainfall, on TWP migration into water bodies are examined. To design these scenarios, some of the required variables were directly determined through experiments. The dispersion stabilities of TWPs and CB in water were tested under UV irradiation, and their sedimentation rates were measured (van Os et al., 2025). Through MFA based on the scenario, the contributions of TWPs and CB generated at the national level by road and vehicle types were assessed, and fundamental data were gathered for designing management strategies.

2. Materials and methods

2.1. Configuration of technical and environmental compartments

To interpret the mass flow of the TWPs generated from roads and the CB detached from the TWP, a flow diagram consisting of technical and environmental compartments was constructed, as shown in Fig. 1. The endpoints were primarily defined as air, water, and soil systems, whereas the basic technical compartments included the WWTP and WIP. The TWPs generated from roads either flows into the surrounding soil or remains on the road surface (Baensch-Baltruschat et al., 2021). For the runoff generated from road dust due to rainfall or road cleaning, a flow diagram was constructed to incorporate runoff treatment plants and separate or combined sewage systems (SSS and CSS) (Sewage Information System, 2022). Runoff treatment methods for urban roads are differentiated based on the design of sewage systems as separate or combined systems within urban areas. In general, WWTPs efficiently remove fine particles, primarily by using activated sludge processes that concentrate most particulates in the sludge for removal. Based on the waste disposal status, waste sludge from WWTP is partially transported to landfills and WIP, whereas some of the remaining untreated TWPs are discharged into untreated water (Gehrke et al., 2023). In the case of WIP, most TWPs are incinerated and removed. However, some microplastics (MP) may remain and are subsequently transported to landfills along with the incineration residues (Yang et al., 2021). The compartments representing sludge and sludge removal in the flow diagram are considered items removed from the external system of interest.

2.2. Resuspension ratio of CB from road dust

A diesel particulate matter cyclone with a quartz filter (Zefon International) was used to evaluate the atmospheric resuspension ratio of CB contained in road dust along with TWPs. Airborne dust particles were sampled at a flow rate of 8 L/min over 4 h, and the mass change in the filter was used to calculate the total suspended particles (TSP). Measurements were conducted at 15 roadside air monitoring stations in Seoul, where the CB concentrations averaged 4291 mg/kg (Shin et al., 2024). The atmospheric CB concentration was calculated using the triangular method proposed in our previous report (Kim et al., 2022a), and the resuspension ratio of CB was evaluated by comparing it to a scenario in which all the CB in the road dust was assumed to be resuspended in the atmosphere.

2.3. Dispersion stability of TWPs and CB in water

To assess the environmental fate of TWPs and CB runoff into water bodies due to rainfall or street washing, their dispersion in water was examined. Laboratory-synthesized TWPs (Kim et al., 2021) and commercially available CB (30–40 nm, Graphene Supermarket, USA) were tested under different light conditions to evaluate their water dispersibility. The selected CB size (30–40 nm) aligns with typical CB particles found in road dust alongside TWPs, as confirmed by our previous study (Kim et al., 2022b). This similarity ensures that the experimental CB accurately represents real-world CB characteristics in urban environments. Changes in the water dispersion tendencies based on light exposure were observed using ultraviolet A radiation (UVA-340, Sankyo, wavelength: 295–365 nm). The sedimentation rate of the particles was measured using multiple light scattering (MLS, Turbiscan Lab, Lean Tecah), and particle size analysis was conducted using a particle size analyzer (PSA, Mastersizer 3000, Malvern). The zeta potentials of the samples were measured via electrophoretic light-scattering spectroscopy (ELS, ELSZ-2000ZS, Otsuka).

2.4. Predicted environmental concentration (PEC)

To calculate the PEC of the TWPs in air, the relevant air volume in a 1-km-long terrestrial ecosystem was multiplied by the total area of South Korea (National Geography Information Institute, 2022), resulting in a total air volume of 10,038 km^3 (0.1 km of relevant height). For the aquatic compartment, the total area of surface water (2860 km^2) was multiplied by a relevant mixing depth of 3 m, yielding a resulting water volume of 8.58 km^3, and the concentration of TWPs in this water volume was determined (Mueller and Nowack, 2008). The variation in soil mixing depth based on soil type led to the consideration of mixing depths of 20, 5, and 5 cm for agricultural land (18,833 km^2), natural land (66,854 km^2), and urban land (8524 km^2). Consequently, the corresponding soil volumes, as per the data provided by the National Geography Information Institute (2022), were calculated to be 7.54 km^3.

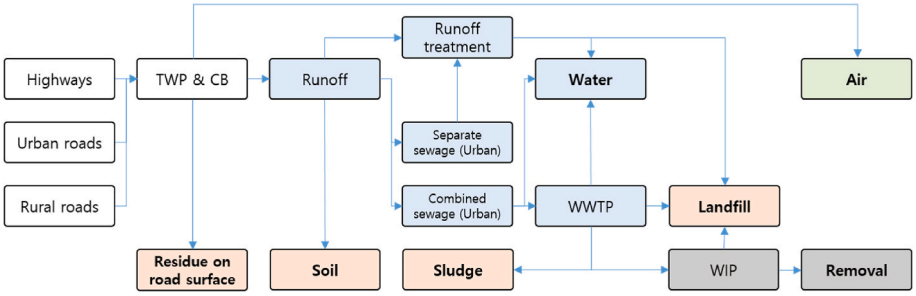


Fig. 1. Flow chart illustrating the origins of TWPs and CB and their pathways through environmental and technical compartments. Main sources and intermediate pathways are represented by white boxes, while waste treatment and removal processes (WIP and Removal) are shown in gray. Compartments related to water, air, and soil are represented by blue, green, and brown boxes, respectively, with end-points of flows indicated in bold letters for clarity.

3. Results and discussion

3.1. Percentage of atmospheric emission of TWPs and CB

The flow chart presented in Fig. 1, which depicts the mass flow of the TWPs and CB across the environmental compartments, is influenced by various factors within the technical compartments, leading to variations in the mass change of the individual components. Initially, the atmospheric resuspension ratios or emission factors (EF) of the TWPs were determined based on the existing literature. For instance, the EF contributing to PM_{10} from tires, brakes, roads, and resuspension particles were 6.1, 9.3, 7.5, and 40 mg/vkm, respectively (Timmers and Achten, 2016). Based on this information, TWPs were assumed to account for approximately 9.7 % of the total particles suspended in the atmosphere.

Although some parameters can be obtained from the existing literature, others require direct experimentation. Owing to the lack of data in the literature on the resuspension ratio of CB on road surfaces, our study estimated the amount of CB by capturing the TSP resuspended from the vicinity of roads using a cyclone impactor. Airborne CB concentrations at the three measurement points were calculated using the EF method; the methodology has been proposed in our previous study (Kim et al., 2022a). The airborne CB concentrations at Stations #2, #7, and #8 were calculated to be 22.9, 19.7, and 13.1 ng/m^3, respectively. To quantify the nano-CB derived from TWPs in road dust, this study integrated four distinct analytical techniques (Kim et al., 2022b). Assuming that all the CB in the road dust were resuspended, the airborne CB concentrations at the three locations were 1237, 1306, and 7762 ng/m^3. Thus, the resuspension ratios of CB at the three sites were 1.8 %, 1.5 %, and 0.2 %, respectively, and the resuspension ratio of CB was set at 1.8 %.

3.2. Parameters setting for MFA

To conduct a comprehensive MFA of the emissions of TWPs and CB originating from roads, we must define and adjust the parameters (Table 1) that influence the transportation of these materials across various environmental compartments (Fig. 1). Initially, TWPs emerged as a result of mechanical abrasion between the tires and the road surface. Subsequently, these particles undergo continuous fragmentation owing to ongoing wear and tear processes, along with exposure to environmental factors such as weathering (shear stress, thermo-oxidation, and photooxidation) (Wagner et al., 2022). During this fragmentation process, approximately 30 wt% of the tires (Palos et al., 2021), primarily composed of CB particles, are released into the surrounding environment. Experimental studies employing milling techniques have demonstrated that within a span of 4 weeks, spherical nano-particles resembling grape-like structures, constituting 4.8–5.7 wt% of the total TWP, detach from the tire surface (Kim et al., 2021). Further analysis using electron microscopy and electron diffraction confirmed the presence of CB within the detached particles.

The amount of TWPs generated varies, depending on the type of vehicle and road characteristics (Baensch-Baltruschat et al., 2020; Gao et al., 2022). Therefore, the EF values of the TWPs must be adjusted differently for each road type and vehicle category. Highways typically exhibit significant TWP levels as they are associated with high vehicular speeds and long driving distances. Because highway surfaces are relatively smooth and clean compared with other roads, TWP generation tends to be relatively small. Conversely, urban roads may experience lower driving speeds but frequent stops and starts, along with driving at intersections and crosswalks, leading to TWP generation (Rødland et al., 2022). In addition, urban road surfaces often have more cracks and dirty patches than highways, which results in a higher occurrence of TWPs. In rural roads, vehicles typically travel at lower speeds, contributing to TWP generation (Rødland et al., 2023). Furthermore, rural roads tend to have more cracks and irregular surfaces than highways or urban roads, which increases the likelihood of TWP generation. Therefore, TWP generation on highways, urban areas, and rural areas varies according to the driving and road conditions.

Table 1
Parameters and assumptions for calculating the mass of TWPs and CB in environmental compartments.

Parameters	Data applied	Unit	Remarks	References
Detachment ratio of CB from TWPs	4.8–5.7	wt %	CB detached from TWPs via additional weathering on roads	Kim et al. (2021)
Percentage of atmospheric emissions	9.7	%	Contribution of TWPs to total EF value for PM_{10}	Timmers and Achten (2016)
	1.8	%	CB content in the dust recovered by the impactor	This work
Percentage of runoff from road surfaces	70, 35, 70	%	Highways, urban, and rural roads	Baensch-Baltruschat et al. (2021)
Recovery of road dust via street sweeper	10–40	%	Removal of road dust by street sweeping on urban roads	Amato et al. (2010)
Ratio of combined to separate sewage system	44.3 vs. 55.7	%	Separation type is compared by the length of the rainwater pipe	Sewage information system (2022)
Effluent ratio from runoff treatment plants	60, 70, 15	%	Highways, urban, and rural roads	Baensch-Baltruschat et al. (2021)
Removal efficiency of WWTP	80	%	70–90 % of removal rates for 20–300 μm microplastics	Gehrke et al. (2023)
Sewage sludge treatment	0.2, 33.9, 60.8	%	Landfill, incineration, and recycle	Sewage information system (2022)
Removal efficiency of WIP	99	%	Termination of microplastics in WIP	Yang et al. (2021)

The EF of TWPs for various vehicle types on different roads can be found in the literature (Chen et al., 2023). By categorizing vehicles into three types, we adjusted the literature values of EF with vehicle types in this study. For example, the EF for trucks in Table 2 was determined using the average EF of trucks and lorries, as described in the literature (Baensch-Baltruschat et al., 2021). To estimate the total amount of TWPs generated annually, we determined the annual driving distance for each road type and vehicle category. Data were obtained from the Korean Statistical Information Service and Statistics Korea (2022). The annual amount of TWPs generated was calculated by multiplying the annual driving distance with the corresponding EF of the TWPs. Additionally, the EF of the CB was calculated from the detachment ratio listed in Table 1 relative to the TWP levels. These two values are summarized in Table 2, and the annual emissions of TWPs and CB were 45,505 and 2184 t/a, respectively, exhibiting a similar range to the TWP levels in Germany reported in a previous study (Baensch-Baltruschat et al., 2021). According to domestic data, calculated based on the warranty period of tires and annual mileage by vehicle type, the TWP level was estimated to be 53,188 t/a (Lee et al., 2020). This result demonstrates that the TWP levels calculated in this study are reasonably consistent.

As summarized in Table 2, highways exhibited the highest TWP emissions (23,201 t/a, 51.0 %), followed by urban roads (16,510 t/a, 36.3 %) and rural roads (5794 t/a, 12.7 %). Despite having the longest total length, rural roads showed the lowest emissions due to lower vehicle traffic and reduced tire abrasion. Highways accounted for the largest emissions due to approximately three times greater vehicle

Table 2
Annual travel distance with road and vehicle types and amount of TWPs and CB calculated with emission factor.

Road types	Vehicle types	Travel distance (x10^6 vkm/a)[a]	EF of TWPs (mg/vkm)[b]	Amount of TWPs (t/a)	Amount of CB (t/a)
Highways 4924 km	Passenger cars	65,089	104	6769	325
	Buses	1920	326	626	30
	Trucks	26,654	593	15,806	759
	Sum	93,662	–	23,201	1114
Urban roads 12,702 km	Passenger cars	47,511	132	6271	301
	Buses	876	415	364	17
	Trucks	13,097	754	9875	474
	Sum	61,484	–	16,510	792
Rural roads 14,353 km	Passenger cars	23,213	85	1973	95
	Buses	694	267	185	9
	Trucks	7496	485	3636	175
	Sum	31,403	–	5794	278
	Total sum	186,549	–	45,505	2184

[a] Data obtained from the Korean Statistical Information Service (2022).
[b] EF values of TWPs by vehicle type are based on the EF values for passenger cars and Buses from Baensch-Baltruschat et al. (2021), whereas the EF for Trucks is the average value of the EF for lorries and trucks.

traffic volume than rural roads, coupled with higher driving speeds that intensify tire wear. Vehicle type significantly influenced TWP emissions, with trucks contributing 29,317 t/a (64.4 %) of total emissions despite making up only 38.4 % of total travel distance. On highways alone, trucks emitted 15,806 t/a of TWPs—2.3 times more than passenger cars (6769 t/a), despite traveling only 2.4 times farther. This pattern reflects the higher emission factors (EFs) of trucks due to their greater tire load and continuous operation at high speeds. Urban roads accounted for over one-third of total TWP emissions, driven by frequent acceleration, braking, and stop-and-go traffic, which increase tire wear despite lower speeds. Meanwhile, rural roads exhibited the lowest emissions, but long-term TWP accumulation in roadside soil remains a concern due to limited removal processes.

Similarly, CB emissions were estimated at 2184 t/a, constituting 4.8–5.7 wt% of total TWP emissions (Kim et al., 2021). Among road types, highways contributed the largest share (1114 t/a, 51.0 %), followed by urban roads (792 t/a, 36.3 %) and rural roads (278 t/a, 12.7 %). Trucks were responsible for 1408 t/a (64.5 %) of total CB emissions, with highway truck-related CB emissions (759 t/a) being 2.3 times higher than those from passenger cars (325 t/a), despite trucks traveling only 41 % of the highway distance covered by passenger cars. The high fraction of CB emissions from trucks is due to their larger tire sizes, higher axle loads, and more intense road-surface interactions, which accelerate tire tread wear. In urban areas, CB emissions were lower than on highways, despite greater traffic volume, due to stop-and-go traffic reducing sustained abrasion. However, CB accumulation per unit area may be higher in urban settings due to limited dispersion compared to open highways. Rural roads exhibited the lowest CB emissions (278 t/a), but retention in soil remains a concern due to low resuspension rates. While absolute emissions are lowest in rural environments, CB deposition in soil may persist for extended periods due to minimal disturbance and limited removal processes.

Particles originating from traffic activities on roads are transported to the surrounding soils or drainage systems by rainfall, heavy rainfall, or road cleaning, ultimately resulting in the migration of a substantial amount of TWPs into water bodies (Luo et al., 2021; Wagner et al., 2022). Urban roads are equipped with sewage systems that allow particle runoff to be treated along with wastewater. However, highways and rural roads lack these facilities. Therefore, the particle fraction in runoff from road surfaces for highways and rural roads was set to 70 % (Baensch-Baltruschat et al., 2021). In urban areas, two types of treatment facilities are available, SSS and CSS, wherein sewage collected in the CSS is transported to the WWTP. According to national data, the installation rates of SSS and CSS are 55.7 % and 44.3 %, respectively (Sewage Information System, 2022). Assuming that approximately 20 % of the runoff remained in the surrounding soil, considering the proportion of the CSS, it was assumed that 35 % of the runoff would be directed to the CSS.

Another method for cleaning roads involves the use of street sweepers that utilize vacuum suction, in addition to water flushing. Vacuum suction cleaning is particularly effective at removing dust suspended in the air (Amato et al., 2010), however, this form of cleaning is primarily performed on urban roads. Moreover, the effectiveness of this method may vary depending on factors, such as traffic volume, road surface conditions, and weather patterns (Järlskog et al., 2022). Despite being performed regularly, the cleaning frequency can be irregular owing to the dynamic nature of roads and weather conditions. The efficiency of road dust removal through suction cleaning can be adjusted, offering flexibility for controlling particle mitigation. This investigation focused on determining whether adjusting the removal efficiency from 0 % to 40 % effectively reduced TWP emissions from road surfaces.

For highways and rural roads, the runoff treatment plant and urban road SSC movement quantities were set to 60 %, 15 %, and 70 %, respectively, based on the values assumed in the literature (Baensch-Baltruschat et al., 2021). Wastewater entering WWTP is primarily treated using an activated sludge process, which generally exhibits high treatment efficiency. However, recent reports on the treatment efficiency of MP indicate a removal efficiency of 70–90 % for MP ranging within 20–300 μm (Gehrke et al., 2023). Therefore, in this study, the removal efficiency of TWPs from WWTP was set at 80 %. Wastewater sludge is mainly disposed of through landfilling, incineration, and reuse, with proportions of 0.2 %, 33.9 %, and 60.8 %, respectively (Sewage Information System, 2022).

Finally, the wastewater sludge transported to the WIP was largely removed by the high-temperature treatment. However, according to previous report (Yang et al., 2021), even after incineration, some MPs persist in the bottom ash. Specifically, The MP concentration ranges from 1.9 to 565 ng/kg in the bottom ash. Therefore, the incineration efficiency of WIP was set to 99 %.

3.3. Flow diagram analysis based on MFA

Based on the variable settings (Tables 1 and 2) and flow diagram (Fig. 1) outlined above, the mass flows of the TWPs and CB across the different technical and environmental compartments were determined. The mass flows of the TWPs and CB detaching from the TWP on roads were calculated as depicted in Fig. 2. CB, which does not exist independently of TWPs in road dust (Kim et al., 2022a), coexists as a composite substance during its migration from roads to compartments. The residual TWPs left on the roads amounts to 119.38 t/a, representing 26.2 % of the initial generation, while 36.4 % of CB remains on the roads. The remaining mass was transferred to the subsequent technical compartment through runoff and street-cleaning processes.

The amount exposed to the atmosphere, totaling 4241 t/a, accounted for a relatively small fraction (9.3 %) of the generated TWPs. This lower exposure is attributed to the nature of the micrometer-sized particles composed of TWPs and grape-like aggregated CB (Kim et al., 2021), which settle easily even when resuspended from road dust, resulting in minimal atmospheric exposure. Smaller microparticles (CB aggregates and small-sized TWPs) tend to be less resuspended than larger particles because they are readily trapped within the microstructure of the road surface (Vogelsang et al., 2019). The quantity removed through road cleaning operations was notably low at 1651 t/a, representing only 3.6 % of the road dust generation. This highlights the preference for vacuum suction cleaning over water-based road cleaning for effective road dust removal, particularly in urban areas (Gehrke et al., 2023).

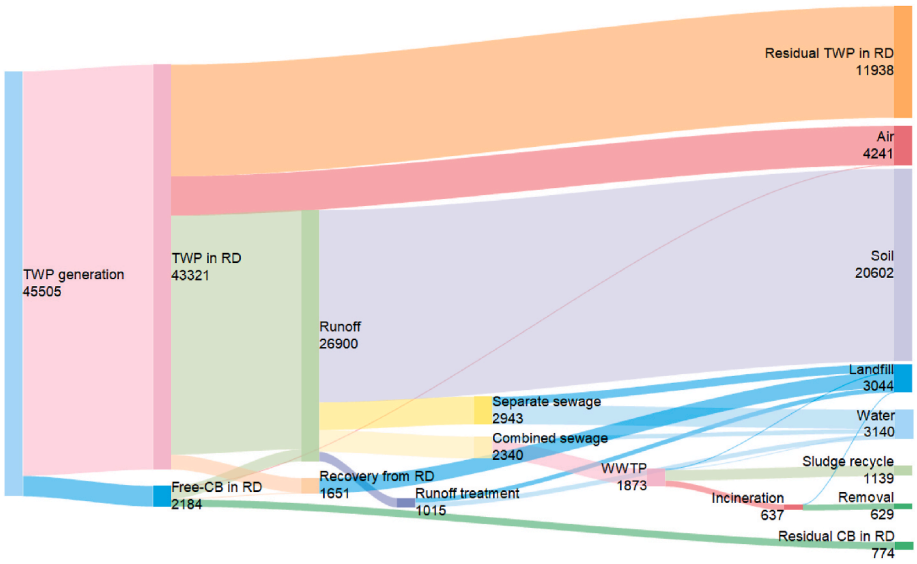


Fig. 2. Emissions of TWPs and CB (t/a) generated on roads, along with their dispersion into technical and environmental compartments.

Rainfall or road-cleaning runoff of TWPs into rivers or surrounding soil demonstrated a significantly higher dominance compared to other compartments. Approximately 26,900 t/a, accounting for 59.1 % of the generated amount, was runoff. Among these, 76.6 % of the generated amount migrated to the surrounding soil, whereas the remainder moved to runoff treatment systems or the SSS and CSS in urban areas. Substances moving from urban areas to water bodies pass through the CSS to the WWTP, thereby increasing the load on the WWTP during rainfall events. Specifically, 80 % of the 2340 t/a entering the CSS is treated at WWTP, enhancing the total treatment capacity of WWTP by mixing with runoff from roads and surfaces. In contrast, SSS separates sewage from surface runoff, relieves the WWTP burden, and provides pathways to landfills and water bodies (Nishimagi et al., 2023). In WWTP, TWPs and CB are generally removed by activated sludge processes, and particulate substances are incinerated or buried in the waste sludge and eventually eliminated from the system.

Overall, the endpoint mass flow of the environmental media corresponded to 4,241, 3,140, 20,602, 3,044, and 12,712 t/a for air, water bodies, soil, landfill, and road residue, respectively. If the mass of the landfill is considered as the soil compartment, the mass ratio of air to water bodies to soil is 1:0.74:3.72. This result highlights the dominant movement of particles into the soil, which is consistent with the findings of studies conducted in Germany (Baensch-Baltruschat et al., 2021). TWPs deposited in soil accumulates persistently unless its environmental fate is considered. Without soil erosion from heavy rainfall, the TWPs that accumulated in the soil remained relatively stationary and persisted for a considerable duration. According to a study in Switzerland, approximately 170,000 t of TWPs in roadside soil accumulation was analyzed within a 30-year span at a distance of 5 m from the road (Sieber et al., 2020). This accumulation may undergo partial degradation owing to soil environmental conditions and microbial activity (Klun et al., 2023; Wagner et al., 2022), potentially leading to the leaching of organic compounds composed of TWPs (Yang et al., 2022). Conversely, TWPs have been reported to exhibit potential for the adsorption of heavy metals such as Pb^{2+} and Cd^{2+}, thereby rendering positive effects on the removal of heavy metals from the soil (Glaubitz et al., 2023).

Regular urban road cleaning can significantly remove a substantial portion of the TWPs and CB remaining on roads. However, the collected road dust is disposed of in landfills, leading to its accumulation in soil. Increasing the recovery ratio of road dust from 10 % to 40 % resulted in 39.5 % and 30.6 % reductions in the remaining TWPs and CB on the roads, respectively (Fig. 3). The road dust removed from the roads was

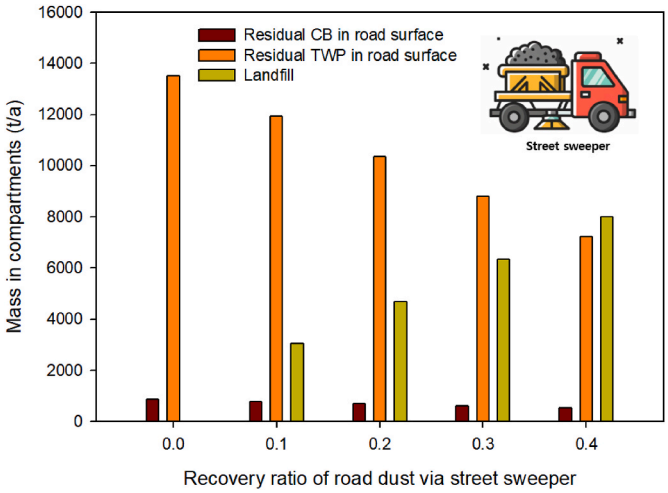


Fig. 3. Mass change of TWPs and CB with the change of the recovery ratio of road dust using street sweepers.

transported to landfills, resulting in a 1.6-fold increase in landfill volume (from 3042 to 7995 t/a). To prevent an excessive burden on landfills, proper management of road dust recovery ratios is necessary. However, periodic urban road cleaning is essential to minimize human inhalation exposure and particle dispersion in the surrounding road environment through the resuspension of particles remaining on the roads (Kreider et al., 2020). Although modeling studies suggest that enhanced road cleaning can significantly reduce road dust accumulation, direct empirical verification remains limited. Amato et al. (2010) found that vacuum-assisted and regenerative-air sweepers are more effective than traditional mechanical sweepers at removing fine particulate matter, with sediment removal efficiencies reaching up to 75 %. While some studies indicate that combining vacuum sweeping with water flushing can reduce PM concentrations by up to 24 %, the persistence of these reductions is short-lived (Chang et al., 2005; Chou et al., 2007). To substantiate the claim that urban road cleaning can mitigate TWP pollution, future research should incorporate before-and-after measurements of TWP residues in cities implementing different cleaning strategies, ideally comparing traditional sweeping with vacuum-based systems under varying traffic and weather conditions.

The exposure of water bodies to TWPs and CB is significantly

influenced by sewage systems. The SSS in urban areas plays a crucial role, similar to the runoff treatment from highways and rural roads. Through this pathway, the sediment is transported to waste landfills for disposal, whereas the remainder is discharged into water bodies. Conversely, the CSS combines rainwater and wastewater in a single pipe. Consequently, TWP runoff from urban roads, along with rainwater, is transported to WWTP, where some of the runoff remains untreated and is directly discharged into water bodies. Therefore, the ratio of SSS to CSS also affects the environmental distribution of materials.

The government is striving to convert urban sewage pipelines from CSS to SSS (Sewage Information System, 2022), aiming to increase the proportion of SSSs from 42.3 % in 2013. This transition alleviates the burden on WWTP by separating sewage and rainwater. However, an increase in SSC leads to elevated TWP and CB runoff into water bodies through stormwater drains, consequently augmenting their migration to water bodies. If the SSS ratio is increased from the current 55.7 % to the projected 70 % by 2030, the volumes of TWPs and CB transported to landfills and water bodies will increase by 6.83 % and 9.97 %, respectively. Conversely, this decrease would result in a reduction of 604 t/a of contaminated water entering the WWTPs. A high percentage of SSS in sewage pipelines, constrained by the handling capacity of WWTP for TWP-containing wastewater, inevitably escalates its migration to water bodies. Therefore, instead of converting all urban sewage pipelines entirely into SSS, maintaining an appropriate CSS level while ensuring the removal and treatment of TWP-containing wastewater in WWTP is a more pragmatic strategy. The findings in Fig. 3 demonstrate how road cleaning efficiency directly influences the environmental fate of TWPs and CB. By improving road dust recovery, the accumulation of TWPs and CB in road environments can be significantly reduced, ultimately shifting their distribution across environmental compartments, as shown in Fig. 4.

The contributions of TWPs and CB to the total amount in the environmental compartments varied depending on the type of road. Fig. 4 illustrates the mass flows for the three road types, each exhibiting a distinct composition. Urban roads have a complex composition (Fig. 4b), whereas highways and rural roads have relatively simple structures (Fig. 4a and c). The complexity arises from the inclusion of additional technical compartments, such as the recovery of road dust and the distinction between the two sewage pipelines, WWTP and WIP. The contribution of road type to each environmental medium is plotted in Fig. 4d. On highways, the predominant contributors were air (51.0 %) and soil (74.9 %). Most of the TWPs generated on highways seems to

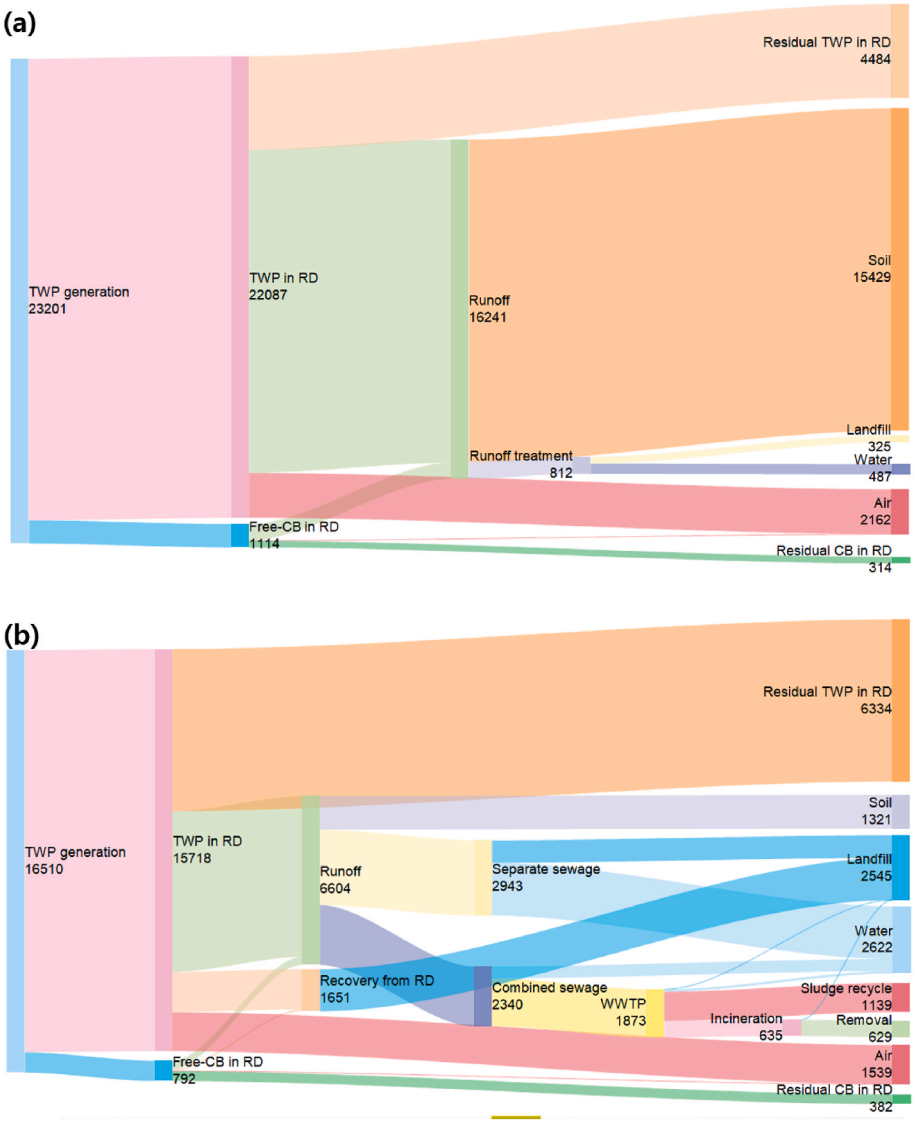


Fig. 4. Annual emissions of TWPs and CB in (a) highways, (b) urban, and (c) rural roads. (d) Illustrate for total annual mass by environmental compartments with road type.

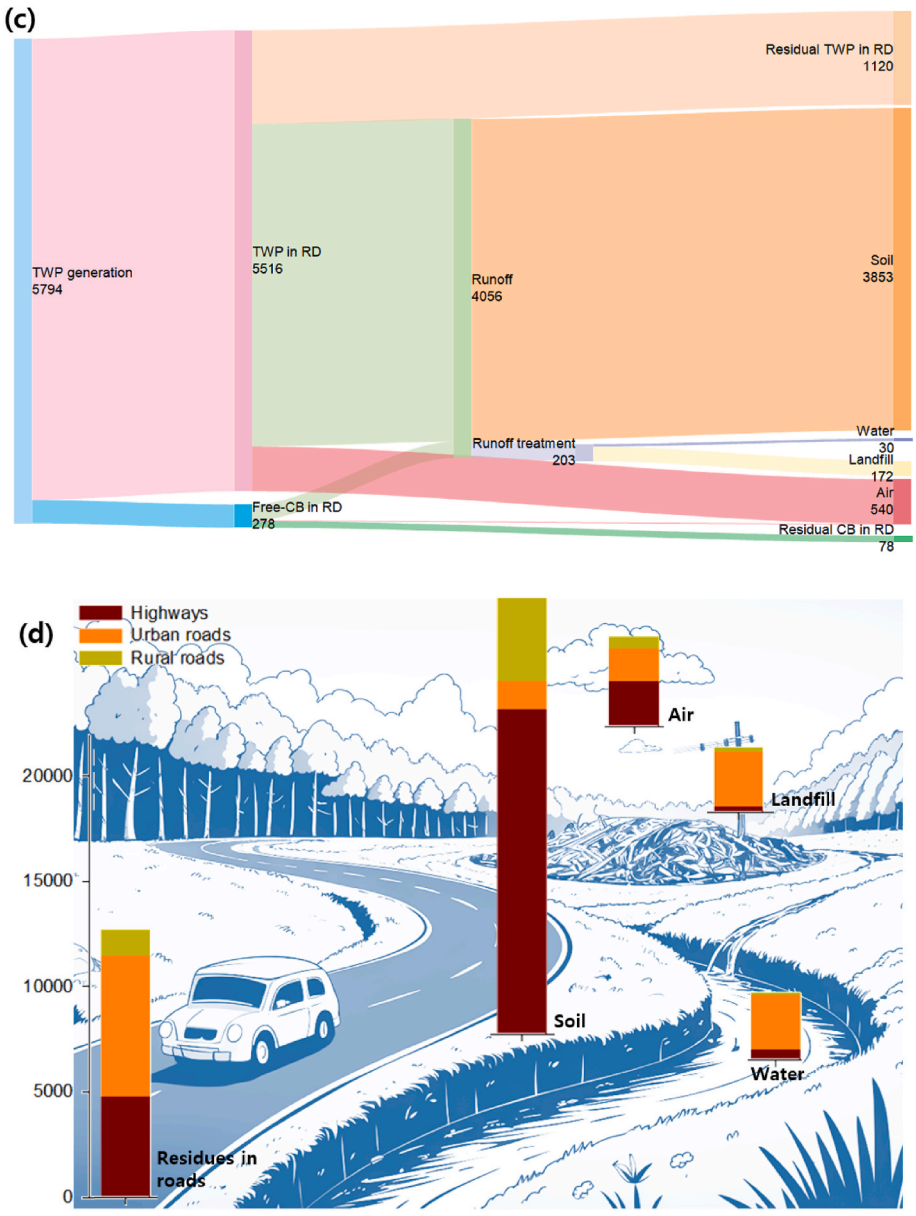


Fig. 4. (*continued*).

accumulate in the surrounding soil (Rødland et al., 2023). Urban roads made high contributions from water bodies (83.5 %), landfills (83.7 %), and road residue (52.8 %). Despite the lower annual travel distance compared with highways (Table 2), the relatively lower runoff resulted in a higher retention of TWPs on roads (Fig. 4b). Rural roads exhibited a relatively high contribution to soil (18.7 %) but lower contributions to other environmental media. This trend is attributed to the lower traffic volume and, consequently, lower TWP generation (Table 2).

3.4. PEC of TWPs and CB in environmental media

The PEC values were calculated based on the spatial volume of the air, water, and soil systems derived from the calculated levels of the TWP and CB mixtures per environmental compartment and geographic information. This methodology uses a simplified MFA (Liu and Nowack, 2022; Mueller and Nowack, 2008). Geographic information relied on data from our previous studies (Hong et al., 2023). PEC values of TWP, including CB, in the air, water, and soil were calculated as 1.16 μg/m^3, 0.37 mg/L, and 2.73 mg/kg, respectively. The PEC values for the air and water systems lie within the range of literature values, which typically vary from 0.5 to 11 μg/m^3 (air) and 0.03–17.9 mg/L (water) (Wik and Dave, 2009).

However, the PEC value in the soil is considerably lower than the literature values; road-adjacent soil exhibits a wide range of concentrations from 600 to 117,000 mg/kg, sharply decreasing beyond 30 m from the road to below 100 mg/kg of TWPs (Wik and Dave, 2009). To calculate the PEC of the soil, only the area of the soil adjacent to the road was considered. Thus, the PEC value of TWPs in the soil was recalculated based on the total length of national roads (112,977 km) (National Geography Information Institute, 2022). Furthermore, the proportion of TWPs varies depending on the soil depth, with the majority of TWPs being found within a depth of 10 cm (Müller et al., 2022). This information was used to define the volume of soil adjacent to the road. The PEC of TWPs in the soil within 5 m of the road was calculated as 364.7 mg/kg. As plotted in Fig. 5, the PEC of the soil decreases exponentially with increasing distance from the road, ranging from 364 to 1824 mg/kg. This exponential decrease in the TWP concentration with distance from the road aligns with the tendency of roads to act as sources of

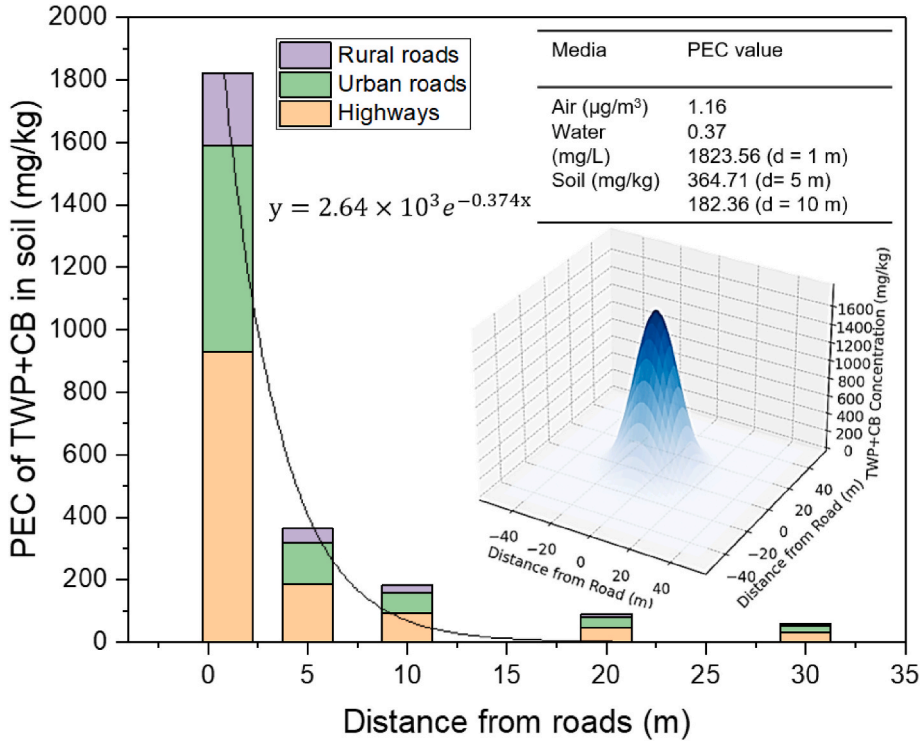


Fig. 5. PEC of TWP + CB in soil with distance from roads. (Inset: summary table for PEC values for environmental media, and 3D spatial distribution of PEC of TWP + CB in soil).

heavy metals in the soil (Werkenthin et al., 2014). Vehicular traffic near roads generates TWP, and as the distance from the road increases, the primary source diminishes. Additionally, there is minimal long-distance transport of TWPs to soil near roads, resulting in higher concentrations of TWPs in the soil adjacent to roads.

3.5. Water dispersibility of TWPs and CB

TWPs typically exhibit a non-polar surface as carbonaceous polymers predominantly containing C-C and C-H bonds; this renders them less prone to interactions with polar water molecules. However, partial oxidation of the rubber surface through photooxidation and thermo-oxidation activates -OH and -C=O groups, thereby reducing hydrophobicity and enhancing dispersibility in aqueous environments (Rosso et al., 2023). Despite unintentional exposure, as depicted in the mass flow diagram, the amount of TWPs and CB entering the water bodies accounted for 6.9 % of the initial generation but was still considerable at 3140 t/a (Fig. 2). Enhanced hydrophilicity enables these particles to disperse effectively in water bodies, facilitating long-distance transport along water flows, which converts them into mobile pollutants rather than localized sources of contamination. Considering these aspects, the water dispersibilities of TWPs and CB were evaluated under UV irradiation.

Under dark conditions without UV irradiation, TWPs showed no dispersion in water, even after 4 h (Fig. 6a). Slight dispersion began after 6 h, and after 16 h of stirring, only 12 % (by mass input) was dispersed in the water, with the remainder floating on the water surface. In contrast, when subjected to UV irradiation, 32 % of TWPs was dispersed after 6 h of stirring, which increased to 47 % after 24 h. The size of dispersed particles under dark conditions averaged 243 μm, whereas under irradiation conditions, the particle size reduced to 145 μm, attributed to some cracking on the particle surface during irradiation, resulting in further fragmentation. The zeta potentials of TWPs under each condition were −7.58 and −11.16 mV, respectively, indicating lower dispersion stability of TWPs under dark conditions. Additionally, the sedimentation velocities measured by MLS averaged 33.1 and 43.6 mm/h after 6 h, respectively, indicating improved dispersion stability of TWPs in water bodies under irradiation.

Comprising more than 95 % elemental carbon, CB possesses inherent hydrophobicity (Hong et al., 2021). However, because of its small size (30–40 nm), CB aggregates tend to disperse easily in water bodies, even via simple methods such as ultrasound treatment or stirring. Under conditions without irradiation, as shown in Fig. 6b, the water dispersion of CB occurred within 2 h. Under environmental exposure conditions, CB did not undergo ultrasonic dispersion but could disperse within 1 h of ultrasonic treatment. Although the primary particles of CB are on the nanometer scale, the secondary particles exhibit agglomerated structures, resulting in micrometer-sized particles in water bodies. The particle size during water dispersion was approximately 12.5 μm, with a high zeta potential of −36 mV, indicating excellent dispersion stability. The sedimentation velocity was 9.74 mm/h, considerably slower than that for TWPs.

The dispersion behavior of TWPs and CB in water bodies (Fig. 6) aligns with the mass flow trends observed in Fig. 4, highlighting their persistence in aquatic environments and potential for long-range transport. Initially, these particles exhibit hydrophobic properties, limiting their interaction with water. However, surface oxidation, mechanical agitation, and environmental weathering progressively alter their surface chemistry, enhancing their dispersibility and mobility. This transformation facilitates their migration from road surfaces into water bodies, increasing their environmental persistence and potential ecological impact. By integrating the findings from mass flow analysis and dispersion behavior, it becomes evident that TWPs and CB do not remain confined to road surfaces but actively enter aquatic systems, where they can be transported over considerable distances. This highlights the need for more accurate exposure assessments in both soil and water bodies to better understand their long-term fate. Developing improved methodologies for quantifying their transport and transformation will provide critical data for road pollution management and policy development, ensuring more effective mitigation strategies.

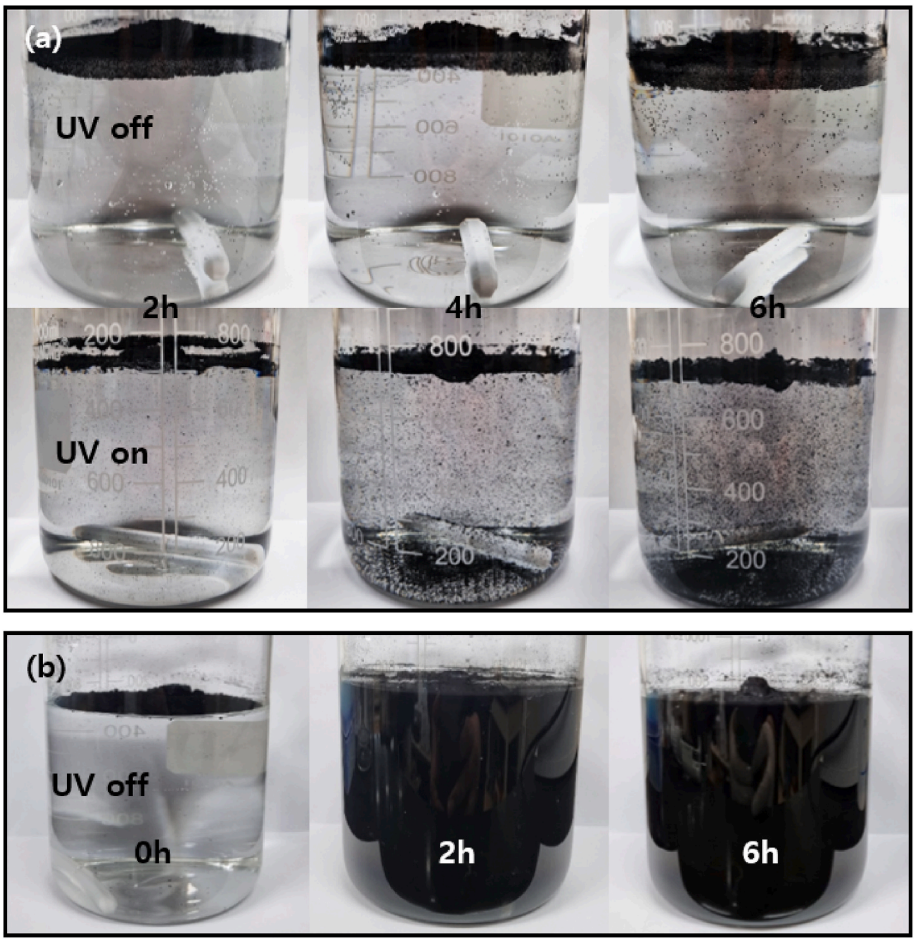


Fig. 6. Dispersion of (a) TWPs and (b) CB with or without UV irradiation under different string periods.

4. Conclusions

In summary, the annual generation of TWPs and CB was calculated using annual vehicle travel data based on road type and vehicle category. Based on the MFA results, the PEC values of TWP in air, water, and soil were determined as 1.16 μg/m^3, 0.37 mg/L, and 1824 mg/kg, respectively, aligning with literature ranges. The TWP concentration in the soil decreased exponentially with increasing distance from roads. Although TWPs and CB exposed to soil cannot undergo long-distance transport, the carbonaceous particles (TWPs and CB) experience surface property changes through photooxidation, and the string exhibits improved dispersibility in water. Consequently, the TWPs and CB present in water bodies can act as long-distance sources of contamination. These results highlight the substantial impact of non-exhaust emissions from roads on environmental pollution, particularly in urban areas where runoff can carry pollutants into water bodies. This underscores the urgency of implementing measures to mitigate road-related pollution, such as improving road-cleaning practices, enhancing wastewater treatment efficiency, and promoting the use of low-emission vehicles. Furthermore, the findings emphasize the need for ongoing research to refine the models and gather more accurate data, enabling better-informed policymaking aimed at minimizing the environmental impact of road transportation. Despite the comprehensive approach used in this study, limitations exist due to the reliance on available data and the estimation accuracy of certain parameters. In addition, the assumptions and simplifications of the model may introduce uncertainties in the results.

CRediT authorship contribution statement

Min Gyu Lee: Methodology, Conceptualization. **Hyeonjung Ryu:** Data curation. **Minseung Hyun:** Software. **Woosuk Chung:** Formal analysis. **Jaehwan Hong:** Investigation. **Hyunook Kim:** Data curation. **Jung-Taek Kwon:** Formal analysis. **Jaewoong Lee:** Resources. **Younghun Kim:** Writing – review & editing, Writing – original draft, Conceptualization.

Declaration of generative AI in scientific writing

None.

Funding

This study was supported by the Korea Environmental Industry and Technology Institute (KEITI; MOE-2022003120002) and the National Institute of Environmental Research (grant numbers NIER-2023-04-02-163 and NIER-2023-01-01-081, respectively).

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Younghun Kim reports financial support was provided by National Institute of Environmental Research. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

References

Amato, F., Querol, X., Johansson, C., Nagl, C., Alastuey, A., 2010. A review on the effectiveness of street sweeping, washing and dust suppressants as urban PM control methods. Sci. Total Environ. 408, 3070–3084. https://doi.org/10.1016/j.scitotenv.2010.04.025.

Baensch-Baltruschat, B., Kocher, B., Stock, F., Reifferscheid, G., 2020. Tyre and road wear particles (TRWP) – a review of generation, properties, emissions, human health risk, ecotoxicity, and fate in the environment. Sci. Total Environ. 733, 137823. https://doi.org/10.1016/j.scitotenv.2020.137823.

Baensch-Baltruschat, B., Kocher, B., Kochleus, C., Stock, F., Reifferscheid, G., 2021. Tyre and road wear particles – a calculation of generation, transport and release to water and soil with special regard to German roads. Sci. Total Environ. 752, 141939. https://doi.org/10.1016/j.scitotenv.2020.141939.

Chang, Y.M., Chou, C.M., Su, K.T., Tseng, C.H., 2005. Effectiveness of street sweeping and washing for controlling ambient TSP. Atmos. Environ. 39, 1891–1902. https://doi.org/10.1016/j.atmosenv.2004.12.010.

Chen, C.Y., Lu, T.H., Wang, W.M., Liao, C.M., 2023. Assessing regional emissions of vehicle-based tire wear particle from macro-to micro/nano-scales with pandemic lockdowns and electromobility scenarios implications. Chemosphere 311, 137209. https://doi.org/10.1016/j.chemosphere.2022.137209.

Chou, C., Chang, Y., Lin, W., Tseng, C., 2007. Evaluation of street sweeping and washing to reduce ambient PM_{10}. Int. J. Environ. Pollut. 31, 431–448. https://doi.org/10.1504/IJEP.2007.016507.

Foscari, A., Schmidt, N., Seiwert, B., Herzke, D., Sempéré, R., Reemtsma, T., 2023. Leaching of chemicals and DOC from tire particles under simulated marine conditions. Front. Environ. Sci. 11, 1206449. https://doi.org/10.3389/fenvs.2023.1206449.

Gao, Z., Cizdziel, J.V., Wontor, K., Clisham, C., Focia, K., Rausch, J., Jaramillo-Vogel, D., 2022. On airborne tire wear particles along roads with different traffic characteristics using passive sampling and optical microscopy, single particle SEM/EDS, and μ-ART-FTIR analysis. Front. Environ. Sci. 10, 1022697. https://doi.org/10.3389/fenvs.2022.1022697.

Gehrke, I., Schläfle, S., Bertling, R., Öz, M., Gregory, K., 2023. Review: mitigation measures to reduce tire and road wear particles. Sci. Total Environ. 904, 166537. https://doi.org/10.1016/j.scitotenv.2023.166537.

Glaubitz, F., Rocha Vogel, A.R., Kolberg, Y., von Tümpling, W., Kahlert, H., 2023. Detailed insights in adsorption process of heavy metals on tire wear particles. Environ. Pollut. 335, 122293. https://doi.org/10.1016/j.envpol.2023.122293.

Hong, J., Moon, H., Kim, J., Lee, B.C., Kim, G.B., Lee, H., Kim, Y., 2021. Differentiation of carbon black from black carbon using a ternary plot based on elemental analysis. Chemosphere 264, 128511. https://doi.org/10.1016/j.chemosphere.2020.128511.

Hong, J., Park, H.N., Lee, S., Song, M.K., Kim, Y., 2023. Material flow analysis-based assessment of polypropylene-fiber-containing microplastics released from disposable masks: characterizing distribution in the environmental media. Sci. Total Environ. 892, 164803. https://doi.org/10.1016/j.scitotenv.2023.164803.

Järlskog, I., Jaramillo-Vogel, D., Rausch, J., Gustafsson, M., Strömvall, A.M., Andersson-Sköld, Y., 2022. Concentrations of tire wear microplastics and other traffic-derived non-exhaust particles in the road environment. Environ. Int. 170, 107618. https://doi.org/10.1016/j.envint.2022.107618.

Kim, G., Lee, S., 2018. Characteristics of tire wear particles generated by a tire simulator under various driving conditions. Environ. Sci. Technol. 52, 12153–12161. https://doi.org/10.1021/acs.est.8b03459.

Kim, J., Yang, S.I., Moon, H., Hong, J., Hong, J., Choi, W., Son, H., Lee, B.C., Kim, G.B., Kim, Y., 2021. Potential release of nano-carbon black from tire-wear particles through the weathering effect. J. Ind. Eng. Chem. 96, 322–329. https://doi.org/10.1016/j.jiec.2021.01.036.

Kim, J., Park, E., Moon, H., Son, H., Hong, J., Wi, E., Kwon, J.T., Seo, D.Y., Lee, H., Kim, Y., 2022a. Estimation of the concentration of nano-carbon black in tire-wear particles using emission factors of PM_{10}, $PM_{2.5}$, and black carbon. Chemosphere 303, 134976. https://doi.org/10.1016/j.chemosphere.2022.134976.

Kim, J., Wi, E., Moon, H., Son, H., Hong, J., Park, E., Kwon, J.T., Seo, D.Y., Lee, H., Kim, Y., 2022b. Quantitative analysis of the concentration of nanocarbon black originating from tire-wear particles in the road dust. Sci. Total Environ. 842, 156830. https://doi.org/10.1016/j.scitotenv.2022.156830.

Klun, B., Rozman, U., Kalčíková, G., 2023. Environmental aging and biodegradation of tire wear microplastics in the aquatic environment. J. Environ. Chem. Eng. 11, 110604. https://doi.org/10.1016/j.jece.2023.110604.

Korean Statistical Information Service, Statistics Korea, 2022. Vehicle kilometer by vehicle type and fuel. https://kosis.kr/eng.

Kovochich, M., Parker, J.A., Oh, S.C., Lee, J.P., Wagner, S., Reemtsma, T., Unice, K.M., 2021. Characterization of individual tire and road wear particles in environmental road dust, tunnel dust, and sediment. Environ. Sci. Technol. Lett. 8, 1057–1064. https://doi.org/10.1021/acs.estlett.1c00811.

Kreider, M.L., Unice, K.M., Panko, J.M., 2020. Human health risk assessment of tire and road wear particles (TRWP) in air. Hum. Ecol. Risk Assess. 26, 2567–2585. https://doi.org/10.1080/10807039.2019.1674633.

Lee, H., Ju, M., Kim, Y., 2020. Estimation of emission of tire wear particles (TWPs) in Korea. Waste Manag. 108, 154–159. https://doi.org/10.1016/j.wasman.2020.04.037.

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. Resour. Conserv. Recycl. 179, 106071. https://doi.org/10.1016/j.resconrec.2021.106071.

Liu, Y., Chen, H., Li, Y., Gao, J., Dave, K., Chen, J., Li, T., Tu, R., 2022. Exhaust and non-exhaust emissions from conventional and electric vehicles: a comparison of monetary impact values. J. Clean. Prod. 331, 129965. https://doi.org/10.1016/j.jclepro.2021.129965.

Lopez, B., Wang, X., Chen, L.A., Ma, T., Mendez-Jimenez, D., Cobb, L.C., Frederickson, C., Fang, T., Hwang, B., Shiraiwa, M., Park, M., Park, K., Yao, Q., Yoon, S., Jung, H., 2023. Metal contents and size distributions of brake and tire wear particles dispersed in the near-road environment. Sci. Total Environ. 883, 163561. https://doi.org/10.1016/j.scitotenv.2023.163561.

Luo, Z., Zhou, X., Su, Y., Wang, H., Yu, R., Zhou, S., Xu, E.G., Xing, B., 2021. Environmental occurrence, fate, impact, and potential solution of tire microplastics: similarities and differences with tire wear particles. Sci. Total Environ. 795, 148902. https://doi.org/10.1016/j.scitotenv.2021.148902.

Luo, C., Xu, R., Wu, D., Zhang, X., Cheng, X., Wang, H., Yin, X., Xu, J., Ma, Q., Chen, F., 2024. Sulfate radical-based advanced oxidation process effects on tire wear particles aging and ecotoxicity. Sci. Total Environ. 906, 167497. https://doi.org/10.1016/j.scitotenv.2023.167497.

Mueller, N.C., Nowack, B., 2008. Exposure modeling of engineered nanoparticles in the environment. Environ. Sci. Technol. 42, 4447–4453. https://doi.org/10.1021/es7029637.

Müller, A., Kocher, B., Altmann, K., Braun, U., 2022. Determination of tire wear markers in soil samples and their distribution in a roadside soil. Chemosphere 294, 133653. https://doi.org/10.1016/j.chemosphere.2022.133653.

National Geography Information Institute, 2022. The national atlas of Korea. http://nationalatlas.ngii.go.kr.

Nishimagi, C., Yanagihara, M., Fang, Y., Sakai, H., 2023. Occurrence of tire-derived microplastics (TMPs) focusing on driving behavior. H2Open J. 6, 52–62. https://doi.org/10.2166/h2oj.2023.058.

Palos, R., Gutiérrez, A., Vela, F.J., Olazar, M., Arandes, J.M., Bilbao, J., 2021. Waste refinery: the valorization of waste plastics and end-of-life tires in refinery units. A review. Energy Fuel. 35, 3529–3557. https://doi.org/10.1021/acs.energyfuels.0c03918.

Rasmussen, L.A., Liu, F., Klemmensen, N.D.R., Lykkemark, J., Vollertsen, J., 2024. Retention of microplastics and tyre wear particles in stormwater ponds. Water Res. 248, 120835. https://doi.org/10.1016/j.watres.2023.120835.

Rødland, E.S., Lind, O.C., Reid, M.J., Heier, L.S., Okoffo, E.D., Rauert, C., Thomas, K.V., Meland, S., 2022. Occurrence of tire and road wear particles in urban and peri-urban snowbanks, and their potential environmental implications. Sci. Total Environ. 824, 153785. https://doi.org/10.1016/j.scitotenv.2022.153785.

Rødland, E.S., Heier, L.S., Lind, O.C., Meland, S., 2023. High levels of tire wear particles in soils along low traffic roads. Sci. Total Environ. 903, 166470. https://doi.org/10.1016/j.scitotenv.2023.166470.

Rosso, B., Gregoris, E., Litti, L., Zorzi, F., Fiorini, M., Bravo, B., Barbante, C., Gambaro, A., Corami, F., 2023. Identification and quantification of tire wear particles by employing different cross-validation techniques: FTIR-ATR Micro-FTIR, Pyr-GC/MS, and SEM. Environ. Pollut. 326, 121511. https://doi.org/10.1016/j.envpol.2023.121511.

Sewage information system, 2022. Operation status of sewage pipeline installation. https://www.hasudoinfo.or.kr/stat/statNormal.do.MinistryoftheEnvironment, Korea.

Sharmilaa, G., Ilango, T., 2022. Vehicular air pollution based on traffic density - a case study. Mater. Today Proc. 52, 532–536. https://doi.org/10.1016/j.matpr.2021.09.278.

Shin, H., Hyun, M., Jeong, S., Ryu, H., Lee, M.G., Chung, W., Hong, J., Kwon, J.T., Lee, J., Kim, Y., 2024. A correlation study of road dust pollutants, tire wear particles, air quality, and traffic conditions in the Seoul (South Korea). Atmos. Pollut. Res. 15, 102309. https://doi.org/10.1016/j.apr.2024.102309.

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.

Timmers, V.R.J.H., Achten, P.A.J., 2016. Non-exhaust PM emissions from electric vehicles. Atmos. Environ. 134, 10–17. https://doi.org/10.1016/j.atmosenv.2016.03.017.

van Os, M.F., Nooijens, M.G.A., van Reness van Duivenbode, A., Tromp, P.C., Höppener, E.M., Grigoriadi, K., Boersma, A., Parker, L.A., 2025. Degradation rates and ageing effects of UV on tyre and road wear particles. Chemosphere 372, 144121. https://doi.org/10.1016/j.chemosphere.2025.144121.

Vogelsang, C., Lusher, A., Dadkhah, M.E., Sundvor, I., Umar, M., Ranneklev, S.B., Eidsvoll, D., Meland, S., 2019. Microplastics in road dust – characteristics. Pathways and Measures. Norwegian Institute for Water Research, Oslo, Norway.

Wagner, S., Hüffer, T., Klöckner, P., Wehrhahn, M., Hofmann, T., Reemtsma, T., 2018. Tire wear particles in the aquatic environment – a review on generation, analysis, occurrence, fate and effects. Water Res. 139, 83–100. https://doi.org/10.1016/j.watres.2018.03.051.

Wagner, S., Klöckner, P., Reemtsma, T., 2022. Aging of tire and road wear particles in terrestrial and freshwater environments – a review on processes, testing, analysis and impact. Chemosphere 288, 132467. https://doi.org/10.1016/j.chemosphere.2021.132467.

Werkenthin, M., Kluge, B., Wessolek, G., 2014. Metals in European roadside soils and soil solution – a review. Environ. Pollut. 189, 98–110. https://doi.org/10.1016/j.envpol.2014.02.025.

Wi, E., Park, E., Shin, H., Hong, J., Jeong, S., Kwon, J.T., Lee, H., Lee, J., Kim, Y., 2023. Overall distribution of tire-wear particles, nanocarbon black, and heavy metals in size-fractionated road dust collected from the steel industrial complexes. Sci. Total Environ. 884, 163878. https://doi.org/10.1016/j.scitotenv.2023.163878.

Wik, A., Dave, G., 2009. Occurrence and effects of tire wear particles in the environment – a critical review and an initial risk assessment. Environ. Pollut. 157, 1–11. https://doi.org/10.1016/j.envpol.2008.09.028.

Yang, Z., Lü, F., Zhang, H., Wang, W., Shao, L., Ye, J., He, P., 2021. Is incineration the terminator of plastics and microplastics? J. Hazard Mater. 401, 123429. https://doi.org/10.1016/j.jhazmat.2020.123429.

Yang, K., Jing, S., Liu, Y., Zhou, H., Liu, Y., Yan, M., Yi, X., Liu, R., 2022. Acute toxicity of tire wear particles, leachates and toxicity identification evaluation of leachates to the marine copepod, Tigriopus japonicas. Chemosphere 297, 134099. https://doi.org/10.1016/j.chemosphere.2022.134099.

Zafra, C., Temprano, J., Suárez, J., 2017. A simplified method for determining potential heavy metal loads washed-off by stormwater runoff from road-deposited sediments. Sci. Total Environ. 601–602, 260–270. https://doi.org/10.1016/j.scitotenv.2017.05.178.