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Tire wear particles in the marine environment: sources, migration, ecological risk and control strategy

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With the global proliferation of vehicular transportation, tire wear particles (TWPs) have emerged as a pervasive class of emerging contaminants in the environment. Primarily originating from terrestrial road networks, these anthropogenic particulates undergo complex environmental transport through atmospheric deposition and hydrological processes, ultimately accumulating in marine compartments through seawater column retention, benthic sedimentation, and bioaccumulation within marine trophic webs. The environmental impacts of TWPs manifest through multiple mechanisms including physically effects on marine organisms, chemically leaching of toxic tire components, and ecologically bioaccumulation and biomagnification. Current research priorities emphasize the development of standardized monitoring protocols for TWPs quantification and the implementation of source control strategies through green material engineering. This review systematically examines the environmental fate, ecological impacts, and risk mitigation approaches associated with marine TWPs pollution, providing critical insights for developing evidence-based management frameworks.

KEYWORDS

tire wear particles(TWPs), migration and transformation, marine ecosystem, ecotoxicological risk, emission reduction

1 Introduction

Vehicular transport, particularly passenger cars, has significantly enhanced human mobility and modern living standards. Meanwhile, the global dependence on rubber-based tires—composed of both natural elastomers and synthetic polymers—has engendered persistent environmental burdens. Tire Wear Particles (TWPs), microscopic particles generated through interfacial abrasion between vehicular tires and road pavements, have become an escalating contamination concern due to their toxic effects on ecosystems and increasing abundance worldwide (Rogge et al., 1993; Kreider et al., 2010; Kole et al., 2017; Wagner et al., 2018; Tian et al., 2021; Mian et al.,2022). With annual release of over 6 million tons into the environment, TWPs as major contributors of microplastics (MPs), are being one of the hot topics in environmental researches (Evangeliou et al., 2020; Xu et al., 2020; Gehrke et al., 2023).

The occurrence of TWPs in the environment is governed by multiple factors, including vehicle driving behaviors, road surface types, tire specifications, and ambient conditions, resulting in the pollution characteristics like broad size distributions (0.1 µm to 5 mm), heterogeneous morphologies, and complex chemical compositions (Kole et al., 2017; Chen et al., 2022). Notably, TWPs possess high specific surface areas and marked hydrophobicity, enabling strong adsorption affinities for co-occurring pollutants such as vehicular exhaust particulates, heavy metals (e.g., Zn, Pb), and antibiotics (e.g., tetracycline). The synergistic interactions between TWPs and adsorbed contaminants may amplify their combined ecotoxicological impacts (Hüffer et al., 2019; Ding et al., 2021; Glaubitz et al., 2023).

Environmental monitoring data have confirmed the pervasive distribution of TWPs in global atmospheric, terrestrial, and marine compartments like air, road dusts, soil, snow, stormwater runoff, wastewater treatment systems, rivers, lakes, seas, and sediments (Baensch-Baltruschat et al., 2021; Goßmann et al., 2021; Järlskog et al., 2022; Müller et al., 2022a; Zhao et al., 2024). It is estimated that TWPs may contribute to 26–74% of total MPs loads in the environment (Xu et al., 2020; Gehrke et al., 2023). Bioavailable additives (e.g., zinc, benzothiazoles) leached from the TWPs may induce sublethal impairments in growth allometry, developmental homeostasis, and reproductive fitness in organisms (Goßmann et al., 2021; Ertel et al., 2023). TWPs can also be ingested by organisms and then transfer through food webs (Auta et al., 2017; Parker et al., 2020). Thus TWPs pose significant potential threats to global ecosystems and human health especially in a background that the vast global TWPs emissions every year (Baensch-Baltruschat et al., 2020). TWPs has been a priority emerging pollutant requiring monitoring and control in the environment (Wik and Dave, 2009; Rødland et al., 2023; Kole et al., 2017; Wagner et al., 2018; Baensch-Baltruschat et al., 2020; Gehrke et al., 2023; Mayer et al., 2024).

Although TWPs mainly originate from land, their small size and light weight make them easily transport for long distance through air circulation, precipitation, and surface/subsurface runoff, ultimately enter the marine environments (Figure 1) (Evangeliou et al., 2020; Baensch-Baltruschat et al., 2021). TWPs contribute up to 15% of marine MPs pollution, meaning up to 350,000 tons of TWPs entering the ocean each year and the annual input of TWPs to the ocean is suggested to increase year by year (Meng et al., 2020; Kushwaha et al., 2024). The ocean covers about 71% of the earth's surface, thus the ocean is a potentially significant accumulation site for TWPs and serves as the ultimate sink for TWPs.

TWPs could induce profound and unpredictable impacts on marine ecological systems because of their particular physical and chemical properties, resulting in higher environmental and health risks compared with other types of MPs (Galafassi et al., 2019; Halle et al., 2020; John et al., 2022). An increasing number of studies have demonstrated that TWPs might interfere with marine biogeochemical cycles and undermine the equilibrium of marine ecosystems (Roch et al., 2019; Cunningham et al., 2024; Wang et al.,

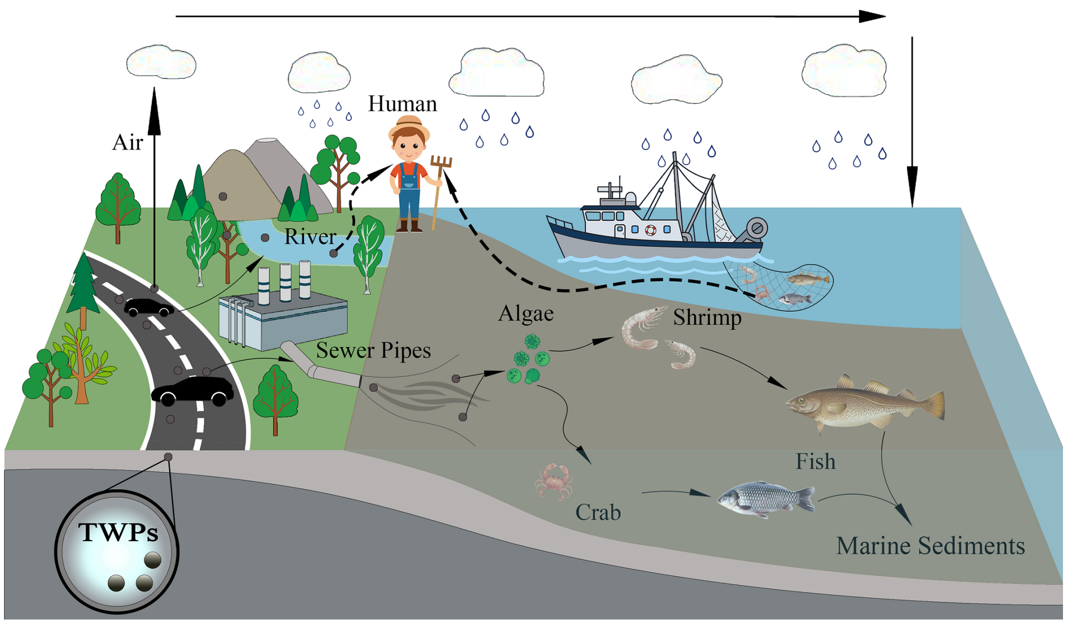

FIGURE 1
Transport and distribution of tire wear particles in the environment.

2024c; Zhang et al., 2025b). Tires are usually compounded with a large number of chemical additives, including plasticizers and vulcanizing agents. These chemical additives contain heavy metals such as chromium and nickel, as well as a variety of organic contaminants such as PAHs and N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine (6PPD), many of which are leachable in water (Kushwaha et al., 2024; Chen et al., 2025). Moreover, TWPs could also adsorb contaminants such as antibiotics, thus inducing joint toxicity (Wen et al., 2024; Ganie et al., 2025). Studies have shown varying degrees of toxic effects of TWPs and their leachates on different kinds of marine life, including bacteria, algae, phytoplankton, zooplankton, crustaceans, and fishes (Halle et al., 2021; Yu et al., 2023; Boisseaux et al., 2024; Li et al., 2024d). Emerging evidence confirmed the pervasive presence of TWPs in marine biota, possibly cause risks to ecosystem through trophic transfer (Wang et al., 2023a). TWPs also tend to accumulate progressively with increased intake and become amplified through bioaccumulation in the food chain and ultimately detected in the human body, posing potential threats to human health (Roch et al., 2019; Chai et al., 2024). Consequently, TWPs pollution in marine ecosystems cannot be ignored.

A total of 249 relevant publications since 2018 to 2025 maily in the Web of Science, PubMed, and the China National Knowledge Infrastructure (CNKI) were selected. 'Tire wear particles' was used as the subject headings coupled with key words marine, sea, microplastic particle, behavior, toxicity, monitoring, control, etc. Based on literature review, global research on TWPs has predominantly focused on regions spanning from the east coast of North America to the west coast of Europe, including Arctic areas. Additionally, investigations are gradually expanding to coastal zones near South Korea, Japan, and parts of China's Pacific coastline. This article comprehensively reviews the sources, migration of TWPs and their ecological risks in the marine environments, and explores the potential control strategies, aiming to provide support for the comprehensive understanding of the environmental behaviour of TWPs and effective control measures of TWPs pollution in marine ecosystems.

2 Sources

TWPs are primarily generated through the tribological interaction between tires and road surfaces, thus particles detached from tires during transportation activities are the greatest contributors to TWPs emission (Zhang et al., 2024a). At the global scale, around 3 billion new tires are produced, and 800 million are reaching end-of-life status annually (Kole et al., 2017; Mayer et al., 2024). During operational use, 10–30% of the tire tread mass undergoes progressive attrition, forming TWPs that would disperse into environmental matrices through mechanical shear and aerosolization processes (Wagner et al., 2018). The annual release of TWPs exceed 6 million tons globally, with per capita emissions ranging from 0.20 to 5.5 kg/year (mean: 0.81 kg/year) across different economic regions (Baensch-Baltruschat et al., 2020; Evangeliou et al., 2020; Kole et al., 2017).

Research revealed TWPs emissions occur across all vehicle types at varying levels, with light-duty passenger cars averaging 100 mg/vehicle·km, while heavy-duty trucks reaching up to 1,200 mg/vehicle·km (Baensch-Baltruschat et al., 2020; Lee et al., 2020). Notably, hybrid electric vehicles demonstrate 18-22% higher tire wear rates compared to conventional internal combustion engine vehicles, attributable to increased mass from battery systems and regenerative braking-induced torque variations (Liu et al., 2022a; Arole et al., 2023). While accelerating electric vehicle deployment is pivotal for decarbonizing transportation in the world, the concomitant increase in TWPs emissions presents an emerging environmental challenge. The extreme operational conditions of aircraft tires are likely to exacerbate the generation rates of TWPs by 3–5 times compared to those of highway vehicles. A case study at Frankfurt Airport (2014) quantified annual aircraft TWPs emissions at 83 metric tons (Spanheimer and Katrakova-Krüger, 2022). Coastal regions, which concentrate the world's highest population densities, significantly contribute to transport-related TWPs emissions due to intensified vehicular traffic and increased aviation activity.

Factors encompassing tire composition (natural/synthetic rubber ratios, tread design), pavement characteristics (surface roughness, hardness), vehicular parameters (axle load, velocity profiles), and driver behavior patterns (braking intensity, acceleration frequency) would strongly affect the emissions and characteristics of TWPs (Kole et al., 2017; Zhang et al., 2023b). TWPs are typically composed of synthetic rubber, fillers (e.g., carbon black), plasticizers, and road-derived particulates (Table 1), ranging in size from micrometers to millimeters (averaging 10–100 micrometers) (Kreider et al., 2010; Wagner et al., 2018; Amelia et al., 2021). Nascent TWPs typically demonstrate sub-aqueous densities (0.95-1.05 g/cm³), whereas in the environment, TWPs show elevated density ranges (1.20-1.70 g/cm³) due to their agglomerating with high-density road-derived materials (e.g., asphalt particles: 2.3-2.5 g/cm³) through thermomechanical adhesion processes (Kole et al., 2017; Baensch-Baltruschat et al., 2020; Kovochich et al., 2021). Once emitted, TWPs release into the ambient air or settle on the road surface, where their diameter and density play a critical role in determining their environmental behavior and fate (Wagner et al., 2018; Baensch-Baltruschat et al., 2021).

TABLE 1 Composition of TWPs (Kreider et al., 2010; Wagner et al., 2018).

Chemical family	Main substances	Quantity contained (%)
Additives and process oils	Mineral oil, antioxidants, plasticizers, softeners, etc.	10
Rubber	Natural rubber, styrene butadiene rubber	16
Filler	Carbon black, silicon dioxide	13
Pavement materials	Minerals	61

Synchronized monitoring data indicate that TWPs within various forms are primarily transported to marine environment through atmospheric dry/wet deposition and surface runoff. Fine TWPs with diameters below 10 μm (accounting for 0.1–10% of the total emissions) could remain suspended in the atmosphere for long periods of time due to their aerodynamic properties, increasing their potential for transboundary environmental impacts (Järlskog et al., 2020; Baensch-Baltruschat et al., 2021; Goßmann et al., 2023; Li et al., 2024e). The airborne concentrations of TWPs were reported in the range of 0.4-11μg/m^3 (Wik and Dave, 2009). Wind-driven suspension facilitates long-range atmospheric transport of TWPs (Kole et al., 2017), with concentrations reaching 35 ng/m^3 in coastal air over Norway (Goßmann et al., 2023). Evangeliou et al. (2020) suggested that direct deposition of airborne road TWPs was likely the most important source for the ocean, and about 30% of the emitted TWPs (140 kt yr^{-1}) were deposited in the world ocean through atmospheric transport. Atmospheric dispersion enables TWPs to deposit in remote marine environments far from emission sources, contributing to their global distribution.

The high mobility of water enables fluvial long-distance transport of TWPs, thus surface runoff is another key pathway for TWPs to enter marine ecosystems (Leads and Weinstein, 2019). Large TWPs with particle sizes between 10 μm and 500μm undergo transient deposition on road surfaces or adjacent soils, subsequent rainfall can transport them into urban drainage systems, thus TWPs would enter into waters via stormwater runoff (Huber et al., 2016; Baensch-Baltruschat et al., 2020). Urban street cleaning activities could accelerate TWPs entry into aquatic systems (Wik and Dave, 2009; Huber et al., 2016; Smyth et al., 2025). TWPs retained on road surfaces typically form heterogeneous aggregates with dust and road particles during runoff events. These aggregates undergo coagulation, aging, and co-transport with pollutants before entering roadside streams or wastewater treatment systems (Unice et al., 2019; Dupasquier et al., 2023; Li et al., 2023a; Li et al., 2024b; Li et al., 2024c). TWPs have reportedly reached concentrations of up to 179 mg/L in stormwater drainage (Parker-Jurd et al., 2021, 2025). It was estimated that 2.8–18.6% of micron-sized TWPs were discharged from land into freshwater bodies and rivers, and the high mobility of water enabled long-distance transport of TWPs to the ocean (Jambeck et al., 2015; Essel et al., 2015; Leads and Weinstein, 2019; Lebreton et al., 2017; Wang et al., 2024c). Siegfried et al. (2017) estimated that European rivers discharge approximately 1.2 kt of TWPs annually into the Atlantic Ocean. Continental modeling confirmed that terrestrial TWPs, particularly sub-100 μm particles, are efficiently transported via fluvial systems to marine ecosystems, with annual global fluxes estimated at 1.3-4.7 teragrams (Essel et al., 2015). Parker-Jurd et al. (2021) pioneered a flux quantification framework using benzothiazole biomarkers, identifying treated wastewater effluent, urban surface runoff, and atmospheric fallout as three dominant TWPs entry routes into marine systems. Later they quantified TWPs entering estuaries in stormwater drainage, surface waters and sediments in the marine environment, at concentrations of 0.4 mg/L, 0.00063 mg/L, and 0.96 g/kg, respectively (Parker-Jurd et al., 2025). At present, TWPs have been commonly detected in aquatic environments around the world (Wang et al., 2024b).

The global marine input flux of TWPs exhibits significant spatial heterogeneity. Current research on TWPs predominantly focuses on waters of developed countries, such as the United States, Sweden, Germany, Japan, and Norway (Siegfried et al., 2017; Goßmann et al., 2023), where substantial TWPs in marine environments have been consistently documented (Table 2). Among continents, the North America and the Europe are among the largest contributors, and China's rapid motorization contributed significantly to TWPs emissions in Asia (Evangeliou et al., 2020; Wu et al., 2024). In contrast, data about the developing countries were not so comprehensive becasuse of the relative poor

TABLE 2 Global distribution and abundance of TWPs in the marine environment.

Region/Country	Research medium	Abundance	References
Charleston Harbor Estuary, South Carolina, USA	Sea surface microlayer	0.513–6.1 particle/L	(Leads and Weinstein, 2019)
Charleston Harbor Estuary, South Carolina, USA	Intertidal sediment	0 to 111.5 particle/m^2	(Leads and Weinstein, 2019)
Charleston Harbor Estuary, South Carolina, USA	Subtidal sediment	0.51–748.1particle/kg wet weight	(Leads and Weinstein, 2019)
Seine River Estuary, France	Surface water	330 mg/kg	(Barber et al., 2024)
Seine River Estuary, France	Sediment	90 mg/kg	(Barber et al., 2024)
Osaka Bay, Japan	Surface water	231 μg/g	(Barber et al., 2025)
Osaka Bay, Japan	Sediment	312 μg/g	(Barber et al., 2025)
Mediterranean Sea Atlantic Ocean	Marine salt	1-1815μg/kg	(Goßmann et al., 2021)
North Atlantic	Marine atmosphere	1–35 ng/m^3	(Goßmann et al., 2023)
North Pole	Ice and snow	1–80 ng/kg	(Goßmann et al., 2023)
Alpine region	Ice and snow	0.3–0.84 ng/kg	(Evangeliou et al., 2020)
Greenland	Ice and snow	3.4 ng/kg	(Evangeliou et al., 2020)

robust monitoring systems in those countries (Wang et al., 2024b). With the growing evidence base for TWPs distribution across various environmental compartments, the pathways of TWPs entering the ocean have been largely elucidated. However, current calculations of TWPs fluxes and their inputs into the ocean predominantly rely on modeling estimations and lack empirical data (Pan et al., 2023; Xu et al., 2024b; Zheng et al., 2025; Parker-Jurd et al., 2025). Future research should prioritize establishing a global monitoring network to quantify the generation, transport, and environmental distribution of TWPs. This initiative is critical for obtaining accurate mass balance data, which will enable a more scientifically grounded allocation of national responsibilities.

3 Migration

The migration behaviors of TWPs in marine systems, were primarily governed by both their physicochemical properties and the environmental factors of the ocean (Kole et al., 2017). The size, shape, density, surface charge, and other characteristics of TWPs affected the rate of suspension, settling, and dispersion of TWPs in the sea. Lighter and little TWPs can remain suspended in water for extended periods, dispersing via ocean currents, while denser and large particles are more prone to sedimentation on the seabed (Unice et al., 2019; Wang et al., 2024c). Most of TWPs ultimately assumbled in coastal sediments, with concentrations 2–5 orders of magnitude higher than those in pelagic regions (Unice et al., 2019; Lee et al., 2020; Roychand and Pramanik, 2020; Rauert et al., 2022a, 2022b). A hydrodynamic modeling revealed 67–89% TWPs entering the ocean from land would deposit in bays retained in estuarine transition zones, with resuspension rates inversely correlated to sediment organic carbon content (Parker-Jurd et al., 2021). Studies found that synthetic rubber-containing carbon particles predominantly derived from TWPs constituted 15-38% microplastics in coastal sediments, with the 1.6-20 μm size fraction representing >60% of total TWPs mass (Kole et al., 2017; Ziajahromi et al., 2020; Gaggini et al., 2024). Semi-enclosed coastal systems (e.g., urban estuaries in the U.S.) exhibit significant TWPs accumulation, with sediment concentrations reaching 7,515 particles/kg, exceeding those in open oceans (Klöckner et al., 2020; Zhu et al., 2021). Of particular concern is the detection of microplastics (MPs), including TWPs, in Pacific abyssal sediments (4,900-7,016 m depth), where maximum concentrations reach 111.3 ± 75.1 items/kg dw as well as the spatial distribution patterns correlated strongly with the Great Pacific Garbage Patch (GPGP) location and current systems (Deng et al., 2025). Tidal dynamics facilitated the transport of TWPs from terrestrial sources to coastal zones through periodic water level fluctuations. Accelerated sea-level rise has increased the frequency of tidal flooding in coastal cities, potentially elevating TWPs fluxes to marine ecosystems by 23-41% (Ertel et al., 2023). Global ocean circulation patterns further contribute to the wide distribution of TWPs in the ocean, with polar regions acting as potential sinks. The deposition of TWPs on Arctic ice exacerbated ice melt through radiative forcing—a mechanism analogous to black carbon impacts (Materić et al., 2022).

Environmental variations induce the transformation of TWPs in marine systems, thereby significantly altering their fate. Temperature changes, UV irradiation and microbial activity, could affect the stability and transport of TWPs (Weyrauch et al., 2023; Zhao et al., 2024). TWPs might undergo physical fragmentation processes such as weathering and water shear, chemical oxidation processes such as photo-oxidation, ozone decomposition, thermal oxidation and biodegradation in the ocean. These processes lead to decomposition, further fragmentation and physicochemical property changes of TWPs, exchanging their morphology, density, and elemental composition, thereby enhancing their mobility in aquatic systems (Chen et al., 2022; Shin et al., 2023; Wagner et al., 2022; Weyrauch et al., 2023; Li et al., 2024a). Notably, aged TWPs demonstrate enhanced adsorption and transport capacities compared to pristine particles (Wagner et al., 2022; Weyrauch et al., 2023; Li et al., 2024f). As effective adsorbents, TWPs interact with environmental contaminants through their polymer-rubber and carbon-black components (Hüffer et al., 2019). Their adsorption affinity for antibiotics resembles that of carbonaceous materials, with aging further amplifying antibiotic adsorption efficiency (Fan et al., 2021; Wen et al., 2024). Xu et al. (2024) revealed that TWPs and their leachates substantially increase the abundance and diversity of antibiotic resistance genes (ARGs) and virulence factor genes (VFGs) in coastal sediments (Xu et al., 2024a).

TWPs that enter marine environment can be re-emitted into the atmosphere or transported back to terrestrial ecosystems via multiple pathways. Sea spray aerosols generated by wave-breaking processes can reintroduce suspended TWPs into the atmospheric boundary layer (Sha et al., 2024). Through ingestion and bioaccumulation in marine organisms, TWPs may transfer back to land-based ecosystems via animal and human consumption (Weinstein et al., 2022; Laubach et al., 2025; Lian et al., 2025). According to McIntyre's TWPs exposure experiment, the cumulative rate of TWPs in Coho tissues can reach more than 35% (McIntyre et al., 2021). This finding highlights the urgency of conducting environmental risk assessments throughout the life cycle of TWPs, especially the need to quantify their global fluxes through "ocean-atmosphere-land" multi-media migration and "aquatic food chain-human" exposure pathways.

To our knowledge, data on the migration of TWPs in the marine environment were still scarce because of some technological and environmental issues, posing significant challenges in tracking their transport pathways, spatial distribution, and potential accumulation in the ocean. Limited large-scale application of tracing technologies (e.g., stable isotope labeling) hindered comprehensive quantitative analysis of TWPs migration pathways. Although Py-GC/MS coupling technology has reduced the detection limit of TWPs to 0.02 μg/g, complex marine matrices still result in 30-45% false-negative rates (Rauert et al., 2022a). Dynamic ocean currents, water movements, biological activities, and sediment deposition patterns, further complicates the detection and quantification of TWPs. These factors collectively obscure our understanding of how TWPs transport and transformation in the marine environment.

4 Toxicology & ecological risks

TWPs exhibit certain physical and chemical properties like conventional MPs, whereas their primary compositions differ from those of MPs as well as they contain much more toxic chemical additives (Halle et al., 2020; Wang et al., 2023b; Rizwan et al., 2024), resulting in greater potential environmental and health risks. Distinct from conventional MPs, to enhance vehicular safety parameters such as traction efficiency and mechanical durability, tires are reinforced by synthetic rubber matrices and specialized additive formulation, governing post-consumption environmental interactions, including contaminant leaching kinetics and ecotoxicological impacts (Halle et al., 2020; Guo et al., 2024). The cross-linked polymer networks and stabilized additive packages in TWPs confer superior environmental persistence (Barbara et al., 2023), leading to progressively release of complex leachates as well as posing heightened ecological risks through bioaccumulation and interference with biogeochemical cycles (Wik and Dave, 2009; Jambeck et al., 2015; Laubach et al., 2025). TWPs can also readily adsorb environmental contaminants such as heavy metals and PAHs (Cassandra et al., 2022), and such adsorption processes have been suggested to exert greater chemical impacts on water quality than the particles themselves (Vogel et al., 2024; Ganie et al., 2025).

Commonly, TWPs leachates contain measurable concentrations of toxic chemical additives such as heavy metals (Zn, Pb, Cd) and organic pollutants including polycyclic aromatic hydrocarbons and benzothiazole derivatives (Table 3). Wu et al. (2024) systematically investigated the chemical components of TWPs through controlled abrasion experiments using a standardized tire profile simulator, quantifying 18 elements and 20 PAHs from 17 commercially dominant tire models in China. In an environmental leaching study, about 60% of 203 organic compounds identified in TWPs were observed within aqueous-phase mobilization potential (Müller et al., 2022). These chemical additives dissolved in the leachates accounted for 72-89% of observed toxicity, surpassing physical toxicity from particle presence (Rødland et al., 2023). For example, TWPs leachates exhibited higher toxicity (EC_{50}=0.04-8.60 mg/L) than intact particles on *Chlorella vulgaris* and biphenylamine derivatives were observed much more toxic (Jiang et al., 2024). An acute exposure study demonstrated 3.2-fold higher toxicity of TWPs leachates compared to particulate matter itself, attributable to enhanced bioavailability of dissolved contaminants (Caballero-Carretero et al., 2024). 6PPD (N-(1,3-dimethylbutyl)-N′-phenyl-pphenylenediamine), an antioxidant ubiquitously employed in tire formulations was observed undergoing rapid quinoid transformation to 6PPD-quinone (6PPD-Q), a compound demonstrating acute aquatic toxicity at ng/L concentrations (LC_{50}=0.62 μg/L for *Oncorhynchus mykiss*) (Ihenetu et al., 2024; Jiang et al., 2024; Calle et al., 2025). Tian et al. (2020) found that 6PPD-Q induced 100% mortality in coho salmon (*Oncorhynchus kisutch*) at environmental concentrations. Multi-continental surveys detected 6PPD-Q in nearly 90% of urban stormwater samples, with concentrations exceeding ecotoxicological thresholds by 2–3 orders of magnitude (Tian et al., 2021; Yao et al., 2024).

A growing body of research confirms that TWPs exert toxic effects on various marine organisms through multiple pathways, posing a potential threat to marine ecosystems (Siddiqui et al., 2022; Bournaka et al., 2023; Wang et al., 2024c; Yang et al., 2025b). TWPs could change the microbial community composition and function like MPs (Peng et al., 2024; Zhang et al., 2025b). Ding et al. (2022) revealed that environmentally relevant concentrations of TWPs (1% weight/dry weight) could significantly change the microbial community structure, decrease community diversity, and inhibit nutrient cycling processes, including carbon fixation and degradation, nitrification, denitrification, and sulfur cycling in coastal sediments. Liu et al. (2022b) found that exposure to TWPs (150g/kg) could lead to a shift in bacteria community and affect nitrogen metabolism in marine sediments. The effects of TWPs on aquatic organisms showed a significant dose-dependent effects. At low concentrations (0.6 and 3 mg/L), TWPs stimulated the growth of microalgae *Phaeodactylum tricornutum*, whereas higher concentrations (15 and 75 mg/L) significantly inhibited growth, reduced chlorophyll-a content, and induced oxidative damage in algal cells (Lv et al., 2024). Page et al. (2022) revealed a significant negative impact of TWPs leachates on the growth rates of three marine phytoplankton species. The 72-h median effect concentration (EC50) values were determined to be 0.23 g/L for the cryptophyte R*hodomonas salina*, 0.64 g/L for the diatom *Thalassiosira weissflogii*, and 0.73 g/L for the dinoflagellate *Heterocapsa steinii*. Notably, leachate concentrations equivalent to or exceeding 90% of 1 g/L TWP resulted in 100% mortality for all three species within 72 h. Since primary production plays a key role in the marine food web, the growth inhibition and ethality of primary producers such as phytoplankton and macroalgae by TWPs might disrupt the primary production network and the stability of aquatic food webs in the ocean (Wang et al., 2024c). Studies showed that TWPs and their leachates also exhibited toxic effects on higher marine trophic levels such as zooplankton, mollusca and fish (Table 4), posing a profound impact on the structure and function of the entire aquatic ecosystem. For example, exposure of *Tigriopus japonicus* to 0.17 g/L TWPs leachate for 48 hours caused severe oxidative stress, and activities of superoxide dismutase (SOD), glutathione (GSH) and glutathione-S-transferase (GST) decreased to 44.5%, 7.08% and 15.6%, respectively (Yang et al., 2025b); the water filtration rate and respiration rate of juvenile oyster (*Crassostrea gigas*) decreased by 52% and 16% within 40.5 hours at 1 μg/mL TWPs leachates (Tallec et al., 2022); 500 mg/L of TWPs significantly prolonged the burial time of *Eriocheir sinensis*, affecting the antioxidant defense system and energy metabolism (Ni et al., 2023, 2024); 6PPD led to developmental abnormalities in

TABLE 3 The main leachables of TWPs and their toxicity.

Substances		Concentration range	Sources	Toxicity	References
Heavy metal	Zinc (Zn)	8,000-1,5000 mg/kg	Zinc oxide is used as a core component of vulcanization accelerators and participates in the chemical reaction of rubber vulcanization (cross-linking) to enhance the strength, elasticity and abrasion resistance of rubber	Tire wear particle leachate is toxic to earthworms, with zinc being the dominant component	(Ding et al., 2023)
				Zinc (21900 mg/kg) in tire particles is the major trace element and affects soil animal health	(Selonen et al., 2021)
				TWPs leachate is more toxic to bacteria in marine sediments than the TWPs themselves, and Zinc is the main toxicant in the leachate	(Liu et al., 2022b)
				The 96-hour LC50 of TWPs leachate on Jellyfish Hydra was 0.4583 g/L. Zinc was the main toxicant in the leachate	(Yi et al., 2023)
				Constituents such as Zinc in leachate may be important contributors to microalgal toxicity	(Lv et al., 2024)
	Lead (Pb)	10–50 mg/kg	Some older tires or special rubber products may have used lead-containing compounds (e.g., lead salts) as stabilizers or vulcanization accelerators, but this has been largely eliminated in modern tires	Accumulation and transport in soil plants can affect soil ecology	(Li et al., 2020)
				Affects the growth and development of aquatic organisms e.g. zebrafish	(Kiper and Freeman, 2022)
				Inhalation of lead-containing dust samples may be hazardous to human health	(Haque et al., 2024)
	Chromium(Cr)	20–100 mg/kg	Carbon black (filler) contains traces of chromium impurities. Antioxidants/anti-ozonates (e.g., some chromium-containing compounds) may be used to improve rubber weathering (but are less commonly used in modern tires)	Affecting photosynthesis in marine algae	(Zhang et al., 2023c)
				Significant negative effects on the growth and physiological and biochemical processes of *Phyllostachys obliqua*	(Li et al., 2025)
	Cadmium (Cd)	1–5 mg/kg	Exists as an impurity in some additives, such as ZnO (zinc oxide) as an additive in tires	Cadmium is a Group I carcinogen. Cadmium accumulates in marine organisms and seriously affects marine ecological health	(Laubach et al., 2025)
				Cadmium causes oxidative stress and immunotoxic effects in fish.	(Lee et al., 2023)
				Cadmium may cause kidney damage, bone disease and cancer in humans	(Fulke et al., 2024)
	Nickel (Ni)	5–30 mg/kg	Nickels are sometimes used to improve the heat resistance of tires, especially those used at high speeds or high temperatures	Leachate from TWPs contains a variety of metals, including titanium, which may have toxic effects on freshwater organisms	(Roubeau Dumont et al., 2023)
				Metals detected in the leach solution included zinc, titanium and strontium	(Boisseaux et al., 2024)

(Continued)

TABLE 3 Continued

Substances		Concentration range	Sources	Toxicity	References
				(Daphnia magna 21-day LC50 was 60 mg/L)	
	Titanium (Ti)	50–500 mg/kg	Titanium in tire wear particles primarily originates from titanium dioxide additives (pigments/anti-aging agents) and potential titanate coupling agents.	Leachate from TWPs contains a variety of metals, including titanium, which may have toxic effects on freshwater organisms	(Roubeau Dumont et al., 2023)
				Metals detected in the leach solution included zinc, titanium and strontium (Daphnia magna 21-day LC50 was 60 mg/L)	(Boisseaux et al., 2024)
	Copper(Cu)	10–100 mg/kg	Some compounds of copper may act as antioxidants to help protect tire materials from oxidation and UV damage	Copper leaching from TWPs has potential toxic effects on aquatic and soil organisms, which may lead to oxidative stress, inflammatory responses, and ecosystem disruption	(Cui et al., 2024a; Wang et al., 2024b)
				Toxic effects of copper in leachate from tire wear particles show cumulative effects over multiple generations of exposure and may lead to long-term declines in Brachionus plicatilis populations	(Lian et al., 2025)
Polycyclic aromatic hydrocarbon (PAH)	Phenylnaphthalene	50-200 μg/kg	Tires contain rubber, carbon black and certain additives. Tires undergo wear and aging during use, and these processes may release PAHs. Factors such as ultraviolet light, heat and mechanical stress may promote the production and release of these substances	Phenylnaphthalene is the most abundant PAHs in tire wear particles. Phenylnaphthalene and its derivatives cause inflammatory reactions in human cells and are cytotoxic	(Wu et al., 2024)
	Chrysene	100-500 μg/kg		It is toxic to plants, fish and other organisms	(Xiu et al., 2014, 2016)
				Induction of liver injury in mice	(Tao et al., 2021)
	Benzo[a]pyrene	10-100 μg/kg		Benzo[a]pyrene affects the stability of marine ecosystems	(Zapelini de Melo et al., 2022)
				Benzo[a]pyrene affects the growth and development of zebrafish	(Zhang et al., 2023a)
				Benzo[a]pyrene has been linked to lung, skin, and bladder cancers, among other diseases	(Porwisiak et al., 2023)
				Induces lung fibrosis and EMT in mice and acts through the Ahr-mediated Nrf2-p62 signaling pathway	(Li et al., 2024h)
	Fluoranthene	200-800 μg/kg		Fluoranthene has an impact on marine fish and other organisms	(Othman et al., 2023)
				Significant effects on the growth and physiological characteristics of Chlamydomonas reinhardtii	(Narayanan et al., 2024)

(Continued)

TABLE 3 Continued

Substances		Concentration range	Sources	Toxicity	References
				Significantly reduced the viability of human alveolar epithelial cells (A549) in a dose-dependent manner	(Takam et al., 2024)
	Phenanthrene	300-1000 μg/kg		Phenanthrene affects plant metabolism and marine fish growth and reproduction	(Othman et al., 2023)
				Phenanthrene affects the growth and reproduction of aquatic fish	(Karami et al., 2016)
				Phenanthrene affects the growth and reproduction of soil fauna, such as earthworms	(He et al., 2021)
				Phenanthrene causes oxidative stress and changes in immune function in zebrafish	(Xu et al., 2021)
Other organic pollutants	6PPD and 6PPD-Q	100-2–500 mg/kg	6PPD is an antioxidant widely used in tires to prevent oxidative degradation and prolong the service life of tires.6PPD reacts with ozone during tire use to form 6PPD-Q, a highly toxic compound	6PPD-Q is acutely toxic to coho salmon. Coho salmon has a 24-hour LC50 of 0.041 μg/L. The LC50 of coho salmon is 0.041 μg/L. The LC50 of coho salmon is 0.041 μg/L	(Hiki et al., 2021; Brinkmann et al., 2022; Prosser et al., 2023)
				A 400 μg/L concentration of 6PPD-Q inhibited the growth of Chlorella vulgaris	(Liu et al., 2024a)
				After 28 days of incubation, the LC50 of 6PPD-Q against Folsomia candida was 16.31 μg/kg	(Xu et al., 2023)
				Impacts on the growth of marine plankton	(Calle et al., 2025)
				The brook trout died within 3 hours of exposure, and the 24-hour LC50 was 0.59 μg/L The brook trout was also found to have a high LC50 at 24 hours	(Brinkmann et al., 2022)
	Benzothiazole	30-1200mg/kg	Used as a vulcanizing agent in tire manufacturing	Affecting the development of zebrafish embryos	(Zhang et al., 2023d)
				Zinc and benzothiazole are the most common compounds in leachate. They are toxic to aquatic organisms, including acute toxicity to fish and Daphnia	(Kim et al., 2023)
				The contaminant with the highest concentration in the tire leachate was benzothiazole, 4,875 μg/L in the winter tire sample	(Li et al., 2024i)
				The content of BTHs extracted from TWPs ranged from 35.4 to 106 mg/kg, with BTH and OHBT being the major components. BTHs had significant adverse effects on	(Peng et al., 2024)

(Continued)

TABLE 3 Continued

Substances		Concentration range	Sources	Toxicity	References
				soil fungal biomass and community structure	
				High concentration of organic compounds of benzothiazole in tire particles (89.2 mg/kg) and affects soil animal health	(Selonen et al., 2021)
				Benzothiazole consistently leached from TWPs into water and affected the survival of H. azteca	(Halle et al., 2021)
	Benzotriazole	5–150 mg/kg	The role of benzotriazole in tires is mainly as a corrosion inhibitor and rust inhibitor to protect the metal parts in tires from corrosion, thus improving the overall performance and durability of tires	Benzothiazole is not only present in the oceans, but also in the coastal atmosphere, posing a threat to the marine environment	(Franklin et al., 2021)
				Causes apoptosis in rainbow trout cell lines RTgill-W1 and RTL-W1 cells	(Zeng et al., 2016)
				The presence of Benzothiazole increases the accumulation of copper in earthworms and affects their growth	(Xing et al., 2018)
	1,3-Diphenylguanidine (DPG)	10–400 mg/kg	As a rubber accelerator, it is used to increase the speed of rubber vulcanization and improve the physical and mechanical properties of rubber	1,3-DPG affects bacterial growth at high concentrations	(Saifur and Gardner, 2023)
				Tire leachate affects the growth and development of Pimephales promelas, and DPG may play a key role	(Chibwe et al., 2022)
				DPG is released into water and may pose a threat to the aquatic environment	(Sieira et al., 2020)
	Hexamethoxymethylmelamine (HMMM)	20–600 mg/kg	HMMM is mainly used as resin cross-linking agent in rubber vulcanization system, especially in tire tread rubber and rubber products to improve the hardness, abrasion resistance and heat resistance of rubber	Affects bacterial growth at high concentrations	(Saifur and Gardner, 2023)
				HMMM is present in the aquatic environment and may affect aquatic ecology	(Alhelou et al., 2019)
				HMMM and its derivatives derived from tire wear are found in a wide range of environmental media	(Johannessen and Parnis, 2021)

zebrafish (*Danio rerio*) embryos (Cunningham et al., 2022) and reproductive impairment in *Daphnia magna* (Boisseaux et al., 2024; Cunningham et al., 2024).

Emerging evidence confirmed the pervasive presence of TWPs in marine biota, possibly cause ecological risks through trophic transfer (Wang et al., 2023a). Numerous aquatic species have been documented to ingest TWPs, and the accumulation of those absorbed pollutants in marine organisms may exacerbate their adverse effects on marine ecosystems, potentially compromising food web stability and ecosystem health (Halle et al., 2021; Boisseaux et al., 2024; Philibert et al., 2024). 6PPD and 6PPD-Q, have been detected in various fish species including bighead carp (*Hypophthalmichthys nobilis*), sea bream (*Sparidae*), and mackerel (*Scomberomorus* spp.) (Ji et al., 2022). Foscari et al. (2025) revealed significant bioaccumulation of tire additives in blue mussels (*Mytilus edulis*), and all quantifiable 21 tire-related chemicals were found at significantly higher concentrations in mussel's tissue than in tested water, with N,N′-diphenyl-1,4-phenylenediamine(DPPD), N,N′-di-(p-tolyl)-p- phenylenediamine(DTPD) and 4-Hydroxydiphenyl amine(4-HDPA) concentrations more than 50 times higher than water levels. Suspect and non-target screening found 37 additional transformation products of tire additives, many of which did not decrease in concentration during depuration. Chai et al. (2024) demonstrated that the ecotoxicity of TWPs leachate can be transferred and amplified across multi-generations and different trophic levels through food chain (microalgae-zooplankton-fish).

TABLE 4 Toxic effects of TWPs on different marine species.

Genus	Species	Toxic effects	References
Zooplankton	*Daphnia magna*	High exposure (62.5 mg/L) inhibited the growth and development of *Daphnia magna*, reduced its survival and delayed its reproduction.	(Cunningham et al., 2022; Kim et al., 2022; Wang et al., 2022; Boisseaux et al., 2024; Jiang et al., 2023; Liu et al., 2023; Roubeau Dumont et al., 2023; Cunningham et al., 2024)
	Daphnia pulex	TWPs and their leachate had significant negative effects on the survival and reproduction of *Daphnia pulex*, and the toxicity was enhanced with increasing extraction time	(Li et al., 2023b; Liu et al., 2024b)
	Marine copepods	TWPs leachate was acutely toxic to marine copepods and the toxicity increased with increasing leachate concentration and exposure time	(Bournaka et al., 2023)
	Acartia tonsa	TWPs leachate was acutely toxic to all life stages of *Acartia tonsa*, with 48-hour LC_{50} ranging from 0.4 to 0.6 g/L	(Moreira et al., 2024)
	Brachionus calyciflorus	TWPs leachate had a significant negative effect on the reproduction and longevity of *B. calyciflorus*, which was exacerbated with increasing concentration	(Adeolu et al., 2024; Chai et al., 2024, 2025; Lian et al., 2025)
	Mysid shrimp	The mysid shrimp growth was reduced in a concentration-dependent manner upon exposure to micrometer-scale TWPS	(Siddiqui et al., 2022)
	Tigriopus japonicus	Aggregation of TWPs of specific sizes (90-110 μm) may lead to intestinal damage and lipid peroxidation. TWPs leachate produced significant chronic toxic effects on the survival, development, and reproduction of *T. japonicus*	(Yang et al., 2022; Song et al., 2024; Yang et al., 2025b)
	Brachionus plicatilis	TWPs leachate had an acute toxicity test LC_{50} of 0.601 g/L. Significant reductions in reproduction and population growth were observed at concentrations of 0.3 and 0.4 g/L	(Shin et al., 2022)
	H. azteca	48 h LC_{50}: 0.91 ± 0.06 g/L	(Halle et al., 2021)
Aquatic fish species	*Danio rerio*	Exposure to TWPs and their leachate resulted in abnormal development of zebrafish embryos with an EC_{50} value of 0.8865g/L. TWPs can remain in zebrafish gills and intestines	(Cunningham et al., 2022; Weinstein et al., 2022; Jiang et al., 2023; Kim et al., 2023; Wang et al., 2023c; Magni et al., 2024; Moreira et al., 2024; Song et al., 2025; Wen et al., 2025; Zhang et al., 2025a)
	Pimephales promelas	The embryos exposed to TWPs leachate exhibited decreased heart rate, reduced hatching success, shorter body length, increased number of malformations, and decreased eye and body pigmentation	(Kolomijeca et al., 2020; Chibwe et al., 2022)
	Cyprinus carpio	The survival, body weight, body length and feeding rate were significantly suppressed	(Chai et al., 2024)
	Menidia beryllina	Exhibited growth inhibition upon exposure to nanoTWPs, and larvae showed significant alterations in swimming behavior	(Siddiqui et al., 2022)
	Carassius gibelio and Carassius carassius	Chronic exposure to sublethal doses of TWPs may lead to behavioral changes in fish that affect their interactions with predators	(Siddiqui et al., 2022; Gorule et al., 2024)
	Oreochromis niloticus	TWPs had significant negative effects on growth, metabolism and antioxidant capacity of *Oreochromis niloticus*	(Banaee et al., 2023)
	Clarias gariepinus	Decrease in *Clarias gariepinus* erythrocyte and hemoglobin levels as well as an increase in leukocyte counts, while varying degrees of pathological changes were observed in gill and liver tissues	(Adeolu et al., 2024)
	Oncorhynchus mykiss	Tire-related compounds are bioaccessible in fish digestive fluids, which may have potential toxic effects	(Masset et al., 2021, 2022; Dudefoi et al., 2024)
	Fundulus heteroclitus	Chronic exposure may lead to increased DNA damage and oxidative stress in *Fundulus heteroclitus*	(LaPlaca et al., 2022)

(Continued)

TABLE 4 Continued

Genus	Species	Toxic effects	References
	Oncorhynchus kisutch	The lowest concentration of 100 mg/L of TWPs leachate resulted in the death of 25-50% of *Coho Salmon* within 24 hours, while concentrations of 320 mg/L and 1000 mg/L resulted in the death of all Coho Salmon	(McIntyre et al., 2021)
	Cyclopterus lumpus	TWPs were ingested by *Cyclopterus lumpus* and retained in the gut for a long period of time, possibly up to several weeks	(Hägg et al., 2023)
	Rhopilema esculentum	96 h LC_{50}: 0.4586 g/L	(Yi et al., 2023)
Aquatic Bottom Sacrificial Species	*Eriocheir sinensis*	High concentrations of TWPs (500 mg/L) and leachate (30%) significantly prolonged the burrowing time of *Eriocheir sinensis*, and TWPs and its leachate affected the antioxidant defense system and energy metabolism of *Eriocheir sinensis*	(Ni et al., 2023, 2024)
	Diadema africanum	LC_{50}: 0.46 g/L	(Rist et al., 2023)
	Paracentrotus lividus	EC_{50}: 0.16 g/L	(Rist et al., 2023)
	Arbacia lixula,	EC_{50}: 0.35 g/L	(Rist et al., 2023)
	Hediste diversicolor	Under high (5%) exposure to TWPs, the health of *Hediste diversicolor* was affected	(Garrard et al., 2022)
	Silurana tropicalis	Chronic exposure to low concentrations of TWPs leachates negatively affected the survival and behavior of *Silurana tropicalis*	(Cheong et al., 2023)
	Pelophylax nigromaculatus	TWPs leachate can cause toxic effects on the liver of the *Pelophylax nigromaculatus* via the gut-hepatic axis	(Liu et al., 2024c)
	Lumbriculus variegatus	No significant effects on growth, survival or reproduction	(Carrasco-Navarro et al., 2021b)
	Chironomus riparius	High concentrations of TWPs had a significant effect on gene expression in Chironomidae, increasing the germline mutation rate of *Chironomus riparius*	(Carrasco-Navarro et al., 2021a, 2021; Caballero-Carretero et al., 2024; Rigano et al., 2025)
Mollusca	*Mytilus galloprovincialis*	TWPs leachate significantly reduced lysosomal membrane stability (LMS) of *Mytilus galloprovincialis* in a concentration range of 10-100%	(Capolupo et al., 2020, 2021)
	Crassostrea gigas	Tire leachate significantly reduced clearance (52%) and respiration (16%) of *Crassostrea gigas*	(Tallec et al., 2022)
	Crassostrea virginica	*Crassostrea virginica* is capable of accumulating TWPs, but the load of TWPs can be reduced by prolonging excretion	(Weinstein et al., 2022)
	Magallana gigas	100 TWPs mL-1 had affected the energy metabolism and stress response in *Magallana gigas*	(Bernardini et al., 2024)
	Scrobicularia plana	The feeding rate of *Scrobicularia plana* was significantly reduced by exposure to 0.2% and 1% TWPs	(Garrard et al., 2022; Woodhouse et al., 2025)
	Mytilus edulis	TWPs and their leachate significantly reduced the water filtration rate of *Mytilus edulis*	(Thomsen et al., 2024)
Algae	*Chlorella pyrenoidesa*	The growth of *Chlorella pyrenoidesa* was significantly inhibited by TWPs leachate at concentrations of 1500 mg/L and above	(Roubeau Dumont et al., 2023; Chai et al., 2024)
	Scenedesmus obliquus	*Scenedesmus obliquus* is sensitive to TWPs leachate (96 h EC_{50}: 24.1 g/L)	(Jiang et al., 2023)
	Skeletonema costatum	Inhibition of *Skeletonema costatum* growth by TWPs leachate (EC_{50}:15.2 g/L)	(Capolupo et al., 2020)
	Raphidocelis subcapitata	EC_{50}: 0.4 g/L	(Capolupo et al., 2020)
	Alexandrium pacificum	72 h EC_{50}: 465.27 mg/L	(Wang et al., 2024a)
	Chlorella vulgaris	Exposure to TWPs leads to oxidative stress in *Chlorella vulgaris*, which may compromise cell membrane integrity	(Yang et al., 2024; Ganie et al., 2025)

(Continued)

TABLE 4 Continued

Genus	Species	Toxic effects	References
	Phaeodactylum tricornutum	TWPs inhibit *Phaeodactylum tricornutum* growth by decreasing chlorophyll content, increasing photosynthetic efficiency, causing oxidative damage, and disrupting the metabolome	(Lv et al., 2024)
	Rhodomonas salina	72 h EC_{50}: 0.64 g/L	(Page et al., 2022)
	Thalassiosira weissflogii	72 h EC_{50}: 0.73 g/L	(Page et al., 2022)
	Heterocapsa steinii	72 h EC50: 0.23 g/L	(Page et al., 2022)
	Lemna minor	TWPs not only cause direct physical damage to *Lemna minor*, but can also indirectly affect the ecological niche of *Lemna minor* by carrying algae, posing a potential threat to aquatic ecosystems	(Putar et al., 2025)
	Isochrysis galbana	Significant effect of leachate on physiological and nutritional metabolism of *Isochrysis galbana*	(Li et al., 2024g)
	Microcystis aeruginosa	The 100 mg/L TWPs treatment group achieved 89.4% inhibition at 96 hours	(Cui et al., 2024b)
Marine Sediment Bacteria	*Bacillus subtilis and Haliotidis lutimaris*	*G+B. subtilis* and *G-H. lutimaris* were sensitive to exposure to TWPs and their growth rates were significantly inhibited	(Liu et al., 2022b)

The study showed that the growth of microalgae (*Chlorella pyrenoidosa*) was significantly inhibited at TWPs leachate concentration ≥1500 mg/L. For rotifers (*Brachionus calyciflorus*) fed with TWPs-contaminated microalgae, the 500 mg/L group showed reduced reproductive capacity starting from the 3rd generation and the 1000 mg/L group went extinct after the 5th generation.When carp larvae consumed contaminated rotifers from the group higher than 250 mg/L, their mortality increased, and body length/weight decreased by over 30%. Yu et al. (2023) have specifically addressed the ecological risks and potential human health implications via dietary exposure. Primary producers in marine environment, phytoplankton may initiate trophic transfer through uptake of TWPs from aquatic matrices. Filter-feeding organisms subsequently ingest these particles through consumption of suspended particulate matter. The bioaccumulation process continues through higher trophic levels, ultimately affecting marine mammals. As apex consumers, humans may be exposed to TWPs through consumption of contaminated seafood (Roch et al., 2019). Due to their bioaccumulation potential and persistent release of toxic additives, TWPs represent a significant ecological threat in marine environments that demands urgent scientific attention (Youn et al., 2021).

Current information indicated TWPs could cause the abnormality and death of marine life at certain concentrations in the lab, but there are still great challenges in comprehensively assessing TWPs ecological risks. Most of exposure experiments conducted were predominantly limited to short-term studies, which failed to reflect the long-term ecological effects of TWPs. Moreover, the tested concentrations often significantly exceed those found in real-world environments. Whether death occur in the real marine environment has not been investigated, thus long-term in-field studies are called for further investigation.

5 Control strategy

5.1 Detection methods

The establishment of standardized protocols encompassing rational sampling systems, robust analytical methodologies, and validated testing procedures constitutes a critical prerequisite for characterizing TWPs emissions, developing reduction strategies, and formulating regulatory frameworks (Zhang et al., 2023b). Currently, three fundamental challenges impede TWPs analysis (Wagner et al., 2018; Thomas et al., 2022a): (1) light-absorbing properties arising from carbon-black constituents, (2) polydisperse size distributions spanning three orders of magnitude (10 nm - 500 µm), and (3) complex chemical matrices containing >400 additive compounds. Furthermore, environmental interactions with mineral particulates, bituminous materials, or co-pollutants frequently result in surface encapsulation phenomena, thereby substantially complicating analytical characterization (Halle et al., 2021; Mattonai et al., 2022).

Appropriate sampling methods are the prerequisite for accurate quantification of marine TWPs concentrations, and the selection of sampling methods depends on the geographical location, environmental matrix and research objectives (Goßmann et al., 2023; Tariq et al., 2025). Researchers have collected samples from air, water, sediments, sea salt and marine organisms across various global marine regions to conduct environmental concentration analyses (Table 5). Goßmann et al. (2023) utilized active sampling equipment to collect aerosol samples over the North Atlantic. Leads and Weinstein (2019) collected subtidal sediment samples using an Ekman dredge and sea surface microlayer samples with a 0.5 m mesh screen. Particles were separated via NaCl density separation

TABLE 5 Sampling methods of TWPs in different marine environmental media.

Environmental Medium	Sampling method	References
Marine Atmosphere	Active air sampler (e.g., HVAS) combined with quartz fiber membrane (0.45-1.0 µm) UAV with aerosol sampling system (for offshore atmosphere) Passive atmospheric deposition collector (for polar remote transmission studies)	(Goßmann et al., 2023)
Seawater	Stainless steel filter (20-100 µm) Trawl sampling (e.g. Manta trawl) Pump filtration system (0.2-5 µm glass fiber membrane) Niskin water collection (combined with on-site filtration) Surface microlayer sampler (studying sea-air interface exchange)	(Barber et al., 2025; Leads and Weinstein, 2019)
Marine sediments	According to depth, it is divided into surface sampling, deep sampling and layered sampling. Surface sampling (0–5 cm) uses tools like Box/Gravity Samplers or stainless steel shovels. Deep sampling employs Gravity Corers or Vibrocorers to reach several meters down. Layered sampling with Piston Corers or Multi-Corers collects samples from multiple depths. Sediment traps are also used for long-term flux monitoring.	(Barber et al., 2025; Leads and Weinstein, 2019; Tariq et al., 2025)
Ice and snow	Clean ice core drilling (to avoid surface contamination) Stainless steel tool sampling under low temperature conditions	(Chand et al., 2024; Goßmann et al., 2023; Seiwert et al., 2022)
Marine Biology	In marine biology, sampling methods vary based on the target organisms and environment. For plankton, net tows (e.g., zooplankton nets) are commonly used to collect samples at different water depths. Benthic organisms are often sampled using grabs (e.g., Van Veen grab) or corers for soft-bottom habitats, and SCUBA diving or remotely operated vehicles (ROVs) for hard-bottom or deeper areas. Fish and larger marine animals are studied using techniques like trawling, longlining, or tagging and tracking methods.	(Kovochich et al.; Leads and Weinstein, 2019)

and identified under a stereomicroscope. It should be noted that TWPs concentrations in water bodies are usually low, so large amounts of water samples need to be collected and rapidly filtered and enriched in the field. Additionally, minimizing sediment disturbance and re-suspension during sampling is very important (Tariq et al., 2025). At present, the absence of unified process for the sampling of TWPs restricts the reliability and comparison of data between different studies (Zhang et al., 2023b; Tariq et al., 2025). Therefore, formulating standardized guidelines for sampling will help to improve the accuracy and comparability evaluation of TWPs research results under different conditions.

Current detection methodologies are broadly categorized into two paradigms: single-particle methods and mass-based methods (Wagner et al., 2018; Klöckner et al., 2021; Kovochich et al., 2021). Single-particle methods are methods that can be used to identify the presence of TWPs based on, for example, the number of particles, size, morphology, surface texture, and color, focusing on identifying and analyzing individual TWPs mainly using microscopic observations and spectroscopic techniques (e.g., infrared, Raman spectroscopy) (Kovochich et al., 2021). By using the single-particle method, the mass of TWPs in a sample can be calculated based on the number, size, and density of particles. However, this methods can only measure two-dimensional characteristics of particles meaning that the actual volume and mass may be underestimated, and it is difficult to confirm TWPs without additional chemical markers for identification (Khan et al., 2024). Mass-based methods identify the presence of TWPs using chemical markers and quantify their mass based on the amount of standard chemical markers in the sample (Table 6). These methods significantly improve the accuracy of TWPs assessment and quantification (Wagner et al., 2018; Klöckner et al., 2021). The chemical markers can be rubber polymers (e.g., natural rubber, styrene butadiene rubber, etc.) or components added to the tire. Specific mass-based methods used for the analysis of TWPs include inductively coupled plasma mass spectrometry (ICP-MS), liquid chromatography-mass spectrometry (LC-MS), gas chromatography-mass spectrometry (GC-MS), pyrolysis and thermal desorption-coupled gas chromatography-mass spectrometry (PYR-GC/MS, TED-GC/MS)) (Eisentraut et al., 2018). The improved microfurnace pyrolysis-GC-MS method is suitable for the analysis of complex environmental samples and can improve the reliability of TWPs concentration measurements (More et al., 2023).

The development of robust chemical markers remains a central challenge in mass-based quantification approaches, particularly regarding their environmental stability and analytical specificity (Klöckner et al., 2021). Ideal TWPs markers should exhibit three critical characteristics: (1) minimal leaching potential from tire matrices, (2) source specificity distinguishing tire-derived particles from co-occurring brake wear particles, and (3) detectability using conventional analytical platforms (Thomas et al., 2022b; Wagner et al., 2018). Müller et al. (2022b) identified 6-PPD transformation products as promising candidate markers, demonstrating their utility in environmental impact assessments through systematic degradation studies. Goßmann et al. (2021) utilized synthetic rubber vinylcyclohexene and SBB (phenyl [4.4.0] bicyclodecene) of synthetic rubber, and 2,4-dimethyl-4-vinylcyclohexene (DMVCH) and pinene (dipentene, DP) of natural rubber as molecular markers to determine the amount of TWPs in marine salts. By employing ^{13}C-labeled styrene-butadiene rubber (SBR) as an internal standard in PYR-GC/MS analysis, researchers attained 89% recovery efficiency for 15 tire samples, yielding accurate TWPs concentration estimates through polymer-specific mass ratio calibration (Jeong et al., 2024). Using PYR-GC/MS and particulate zinc (Zn) as markers, a Japanese research consortium

TABLE 6 Advantages and disadvantages of the main quantification and assessment methods for TWPs.

Methods	Advantages	Disadvantages	Solution	References
Microscopic method	The morphology, structure and size of tire wear particles can be visually observed, providing important clues for analyzing the causes of tire wear. The operation is simple, and the cost is relatively low	Requires specialized personnel to operate and analyze and is highly subjective. Relatively low precision and may not accurately measure the specific size and composition of particles	High-resolution microscopes are used to improve measurement accuracy in combination with image analysis software, while operator training is strengthened to reduce subjective errors	(Eisentraut et al., 2018; Barbara et al., 2023)
Laser particle sizing method	High precision can quickly and accurately measure the particle size distribution of tire wear particles. High degree of automation reduces human error	The equipment is costly and requires specialized personnel for operation and maintenance. High demands on samples, requiring pre-treatment to remove impurities and interferences	Optimize equipment maintenance processes and reduce equipment costs, while developing automated pretreatment systems to reduce sample preparation	(Kovochich et al., 2021)
Spectroscopy, chromatography, mass spectrometry and energy spectrometry	High sensitivity and fast analysis	Expensive equipment with high operating and maintenance costs. The analysis process is complex and requires specialized personnel to operate	Choose equipment with automation functions to reduce manual operation steps and reduce operation difficulty. Establish a regional-level analysis platform (to share the cost of equipment use through membership.	(More et al., 2023; Rdland et al., 2022; Rødland et al., 2023)
Thermogravimetric analysis method (TG-DSC)	TG-DSC can provide both mass change (TG) and heat change (DSC) information of the sample, which helps to understand the thermal properties of materials more comprehensively. Wide range of applications	The shape and size of the sample needs to be suitable for the instrument and is not suitable for analyses that require many samples	Priority is given to the use of standardized tablet micro crucibles to improve the reliability of single data. Introduced an automated injection system to achieve 30 sample throughput per day. Deploy an AI-triple system for full characterization of complex matrix TWPs.	(Spanheimer and Katrakova-Krüger, 2022)
X-ray diffraction method (XRD)	XRD analysis is usually not destructive to the sample and can be performed multiple times without changing the state of the sample	Proper operation of XRD instruments and interpretation of diffraction patterns requires specialized knowledge and experience. For very small amounts of samples, it may be difficult to obtain X-ray diffraction signals of sufficient intensity	Establish online XRD operations training courses (such as Malvern Panalytical's XRD Academy) covering instrument operation, sample preparation, and data analysis. Promote the regional shared laboratory model, centralize high-precision equipment (such as synchrotron radiation sources), and test it uniformly by professional teams. Intelligent software (e.g., Jade, HighScore Plus) is used to automatically match diffraction peaks and generate phase analysis reports, reducing manual intervention.	(Mohammad, 2023)
Marker analysis method	TWPs can be accurately detected by detecting specific compounds or elements in TWPs as markers. Highly sensitive, the marker analysis method usually uses high-precision instruments for detection, such as gas chromatography-mass spectrometry (GC-MS), inductively coupled plasma mass spectrometry (ICP-MS), etc., and can detect markers at very low concentrations	It requires the use of high-precision instrumentation and specialized operating skills and is therefore relatively costly. A complex pre-processing process for TWPs is usually required before marker analysis. Selection of appropriate markers is one of the key steps in the marker analysis method. Due to the complex and variable chemical composition of TWPs, the selection of suitable markers may be challenging	Develop markers with good optimization and stability effects. Standardize operating processes and train professionals to improve instrument efficiency and reduce the cost per analysis. An open database of TWPs markers (such as TireChem DB) was established to include the chemical composition profiles of different tire brands to assist in marker screening. The algorithm was used to analyze a large amount of TWPs composition data to predict the most stable marker combination.	(Wagner et al., 2018; Chae et al., 2021; Klöckner et al., 2021; Müller et al., 2022b; Thomas et al., 2022a)

documented TWPs accumulation in Osaka Bay (Barber et al., 2025). Notably, the study revealed that Py-GC/MS overestimateed TWPs mass by 12–18% relative to Zn-based methods in high-salinity waters, addressing key methodological inconsistencies. Complementary approaches utilizing zinc isotopes and other heavy metal signatures show potential for discriminating tire-derived particles from geogenic sources, though matrix interference remains a limitation (Klöckner et al., 2020; Pan et al., 2023).

While analytical methods for detecting TWPs in marine systems remain underdeveloped compared to general microplastics research (Yadav et al., 2025), emerging technologies show promise for TWPs measurement. Scholars in China have developed a chemometric model combining attenuated total reflectance-FTIR (ATR-FTIR) with partial least squares discriminant analysis (PLS-DA), achieving 92% classification accuracy for 23 tire brands across four polymer categories (Qiu and Meng, 2019), suggesting spectroscopic techniques cost-effective alternatives for TWPs identification. Chae et al. (2021) advanced TWPs quantification through ole-amide derivatization-GC/MS, achieving superior sensitivity over traditional markers. Validation across 12 riverine and marine sediment samples showed strong concordance with μFTIR particle counts, demonstrating cross-matrix applicability. (Zhang et al., 2024b) demonstrated the potential of machine learning-enhanced satellite remote sensing for retrieving marine particulate organic carbon (POC), particularly through Data Interpolating Empirical Orthogonal Functions (DINEOF) for gap-filling in satellite datasets. These methodologies could be adapted for TWPs tracking, given their analogous transport pathways to other marine microparticles.

Monitoring TWPs in marine environments is crucial for assessing pollution levels and evaluating risks to marine biota. Current monitoring methodologies require integration of comparable and validated techniques, due to the challenges in simultaneous identification of TWPs with diverse sizes, shapes, and chemical compositions using a single analytical approach (Wang et al., 2024c). A critical challenge lies in the absence of internationally harmonized protocols for both quantitative and qualitative characterization of TWPs in oceanic systems (Foscari et al., 2024; Jones, 2024). This methodological inconsistency compromises global assessment efforts of TWPs contamination in marine ecosystems. Several international initiatives are underway to address these challenges and establish standardized methodologies for assessing TWPs emissions and their environmental impacts. The Euro 7 regulations, agreed upon in December 2023, will for the first time include limits on TWPs emissions, alongside brake and tailpipe emissions, extending regulatory oversight to electric vehicles as well (European Commission, 2023). The United Nations Economic Commission for Europe (UNECE) through its Noise and Tyres Working Group (GRBP) is conducting field tests to refine wear measurement techniques, including real-world driving simulations across urban, rural, and highway conditions (UNECE, 2023). While current TWPs monitoring in marine environments remains fragmented, these international efforts—particularly under Euro 7 and UNECE frameworks—are paving the way for standardized, globally applicable methodologies. The integration of these regulatory and scientific advancements will enhance the accuracy of TWPs pollution assessments and support mitigation strategies in marine ecosystems. Future research priorities should focus on establishing standardized analytical frameworks with interlaboratory validation and adaptive remote sensing algorithms for coastal TWP tracking, particularly for multimodal particle characterization as proposed in recent studies (Thodhal Yoganandham et al., 2024; Wang et al., 2024c).

5.2 Control measures

Effective marine TWPs management requires integrated strategies combining source reduction, process control, and terminal treatment. Preventing and controlling the release of TWPs at the source is the most effective approach (Pottinger et al., 2024; Wang et al., 2024c). Optimized chemical formulation and material substitution represents a crucial approach to mitigating TWPs generation. For instance, by incorporating advanced tread compounds, such as silica-reinforced elastomers and graphene-enhanced rubber, could reduce TWPs production by up to 40% under laboratory conditions (Amelia et al., 2021). Ternary rubber systems incorporating transformed 1,4-poly (isoprene-co-butadiene) rubber (TBIR) with natural rubber (NR) and cis-1,4-polybutadiene rubber (BR) have demonstrated improved NR/BR compatibility, optimized filler dispersion, and 35-40% reduction in TWPs generation (Yang et al., 2025a). Incorporation of carbon nanotube (CNT)-reinforced rubber composites into tire has been shown significantly improvement in abrasion resistance, thermal conductivity, and tear strength, enhancing rubber hardness and reducing fine particulate emission rates by 32-45% (Pei et al., 2022). Furthermore, the adoption of sustainable or eco-friendly tire materials represents an effective strategy for mitigating environmental hazards associated with tire use. Tire formulations utilizing biobased polymers and non-toxic plasticizers are emerging as promising alternatives (Pottinger et al., 2024). The tire industry is also actively pursuing alternative sustainable materials, including dandelion root-derived rubber (Taraxacum kok-saghyz) by Continental and guayule-based elastomers by Bridgestone and Nokian (Whba et al., 2024), with major manufacturers committing to ambitious sustainability targets - Michelin plans to incorporate 40% sustainable materials by 2030 and achieve 100% circular tire production by 2050 (Wang and Yong, 2025). However, the widespread adoption of these advanced or green materials may be constrained by higher production costs and limited commercial availability. To address this, cost and scalability assessments are crucial. While these innovations show promise, further research is needed to evaluate their economic feasibility and potential for large-scale production (Amelia et al., 2021; Chen et al., 2022).

Multifaceted source mitigation approaches should integrate not only advanced material engineering solutions, but also traffic flow optimization (e.g., reduced speed limits, congestion management) and road pavement improvement (Wang and Yong, 2025; Chen et al., 2022; He et al., 2024). Moreover, policy tools such as extended producer responsibility (EPR) can play a significant role in TWPs management. EPR policies can incentivize manufacturers to prioritize environmentally benign designs and take responsibility for the entire lifecycle of their products (Wang and Yong, 2025; Chen et al., 2022; Rødland et al., 2024). This includes mandating tire composition disclosures, encouraging low-emission product labeling and recycling of scrap tires, which can further drive the adoption of sustainable materials and technologies. While these policy and design strategies offer promising avenues for reducing TWPs emissions, it is important to recognize that a comprehensive solution requires a multifaceted approach.

Control of the migration and diffusion process of TWPs entering the marine environment is another effective way. The mitigation of TWPs pollution in marine ecosystems requires effective control of particle migration and diffusion processes. Strategic implementation of particle collection infrastructure, including retention ponds and constructed wetlands, in high-risk zones such as roadways and parking facilities can significantly reduce TWPs transport to aquatic systems through optimized hydraulic design and sedimentation processes (Foscari et al., 2024; Rasmussen et al., 2024). Stormwater management infrastructure, including infiltration basins and detention ponds, serves a critical function in urban hydrology by both regulating runoff volume and reducing particulate contaminant fluxes to receiving water (De Oliveira et al., 2024). Systematic sampling using wet dust samplers (WDS) combined with density separation and stereomicroscopy has revealed that optimized street sweeping protocols can reduce TWPs loads in urban runoff by 40-60% (Järlskog et al., 2020). Permeable pavement systems, particularly those modified with cured carbon fiber reinforcement, demonstrate dual benefits of mechanical strength enhancement and TWPs capture capacity (Mitchell and Jayakaran, 2024). Emerging nature-based solutions including bioengineered wetlands and modular bioretention systems show promise for TWPs removal (60-85%) (Wei et al., 2023). However, drainage systems may serve as temporary TWPs reservoirs, with delayed maintenance potentially leading to downstream contamination events (Mengistu et al., 2021). Municipal wastewater treatment plants currently intercept approximately 65-80% of TWPs through multiple processes, though the resulting sludge-bound particles raise concerns regarding agricultural applications (Sun et al., 2024). Roadside vegetation systems, particularly those incorporating high-deposition tree species, can achieve atmospheric TWPs removal efficiencies of 30-50% (Foscari et al., 2024). These integrated control strategies collectively contribute to reducing marine TWPs inputs.

Once in the marine environment, TWPs needs end-of-pipe treatment. Adding exogenous adsorbents to capture TWPs is a potential method for reducing their concentration in seawater. Recent studies have demonstrated that activated carbon derived from poplar pruning waste showed excellent adsorption capacity for TWPs (Lladó et al., 2025). Advanced abiotic degradation methods, including accelerated UV exposure and cyclic freeze-thaw/wet-dry treatments, have been developed to enhance TWPs breakdown rates (Thomas et al., 2022b). Marine microorganisms, including specialized bacterial strains (e.g., *Rhodococcus ruber* and *Gordonia polyisoprenivorans*) and fungal species, have shown capability to colonize TWPs surfaces and initiate biodegradation through extracellular enzyme secretion, effectively depolymerizing high molecular weight rubber components into lower molecular weight oligomers (Calarnou et al., 2023; Saifur and Gardner, 2023). Genomic analysis of rubber-degrading bacteria such as *Rhizobacter gummiphilus* NBRC 109400 has identified key functional genes (e.g., latex-clearing protein *lcp*, Mw ~50 kDa) responsible for rubber degradation, providing molecular targets for strain optimization (Chang et al., 2019). Microbial desulfurization technologies have demonstrated sulfur removal efficiencies of 65-80% from waste tires, enabled rubber regeneration while maintained 85-90% of original mechanical properties (Xie et al., 2024). While approximately 60-75% of organic compounds in tire leachate can be biodegraded under optimal conditions, persistent transformation products remain resistant to microbial degradation, suggesting the need for combined physical-chemical-biological treatment systems (Foscari et al., 2024; Rasmussen et al., 2024). Pretreatment methods significantly enhance microbial degradation efficiency, with ozone oxidation increasing biosurfactant production by 40-55% and improving subsequent biodegradation rates in *Candida methanosorbosa* BP-6 cultures (Marchut-Mikołajczyk et al., 2019). Photo- and thermo-oxidative pretreatment generates carbonyl (CI=0.15-0.25) and hydroxyl (HI=0.08-0.12) functional groups that facilitate subsequent microbial assimilation, with selected bacterial strains achieving 30-45% mineralization of oxidized styrene-butadiene rubber within 60 days (Calarnou et al., 2024). Although the current understanding of marine TWPs biodegradation remains incomplete, these findings establish fundamental principles for developing engineered remediation routes. Future research will likely focus on screening and optimizing microorganisms capable of efficiently degrading TWPs, as well as exploring how these microbial degradation techniques can be applied in the marine environment (Calarnou et al., 2023; Saifur and Gardner, 2023).

Addressing the issue of marine TWPs pollution necessitates a collaborative approach involving governments, enterprises, the public, and other stakeholders (Jones, 2024). A comprehensive strategy integrating advanced source control technologies (including silica-reinforced tire formulations demonstrating 40% lower abrasion rates), AI-enhanced process management systems for real-time TWP monitoring, SDG-aligned policy frameworks incorporating extended producer responsibility schemes, and behaviorally-informed public education campaigns has been shown to reduce marine TWP fluxes by 55-72% in coastal urban environments while maintaining cost-effectiveness (<$0.15 per capita annual implementation cost), thereby significantly enhancing the resilience of marine ecosystems against particulate pollution (Zhou et al., 2023). This integrated approach, requiring transnational knowledge-sharing platforms and circular economy innovations, offers a replicable model for addressing particulate pollution crises while advancing the UN Decade of Ocean Science (2021-2030) objectives, though challenges persist in standardizing global monitoring protocols (Jones, 2024; Tariq et al., 2025). By integrating source control, process management, policy and regulatory frameworks, as well as public education and awareness initiatives, the generation and discharge of TWPs can be significantly curtailed, contributing to safeguarding the health and sustainability of marine ecosystems (Jones, 2024; Zhang et al., 2024b).

6 Conclusions

Given the substantial global production and disposal volumes of tires, TWPs represent an emerging contaminant that has significant impacts on the marine environment. Studies have shown that the introduction of TWPs into the ocean may change the composition of organic matter in the marine system, which in turn will interfere with the stability and function of the marine ecosystem. This alteration in the composition of marine organic matter may lead to the accumulation of harmful substances in marine organisms, ultimately affecting human health through the food chain, and there is also a potential risk of direct or indirect harm to human health from the chemicals that TWPs themselves may carry. A comprehensive understanding of TWPs sources, transport mechanisms, and ecological impacts is essential for knowing their marine environmental behavior assessing associated ecological risks, and developing targeted control measures. The complex characteristics of TWPs like heterogeneous chemical composition, morphological diversity, polydisperse size distribution, and variable density characteristics, created substantial challenges for environmental monitoring, risk quantification, and pollution management of TWPs. Current environmental behavior and exposure studies predominantly utilize laboratory-synthesized or commercially procured TWPs, which may not accurately represent natural environmental conditions. Consequently, investigating TWPs environmental impacts requires integrated approaches that account for multiple interacting factors to better predict their environmental fate and effects. Presently, insufficient data exists regarding intrinsic TWPs properties and environmental parameters to comprehensively evaluate alterations in marine ecosystem induced by TWPs. This knowledge gap underscores the need for more rigorous assessment of TWPs impacts on environmental health.

Based on current research progress of TWPs in the marine environment, the following future research priorities are recommended to address critical knowledge gaps and emerging challenges:

(1) At the technological scopes, accurate monitoring technologies are suggested to be improved to fulfill the quantification need overcoming the present shortages such as...... For example, a novel TGA-GC/MS shows promise but requires further validation and standardization for broader application. Using remote sensing to accomplish source apportionment and large-scale mapping. And of course, the coming economic cost should be decreased to make more researchers at the global scale using and charging those applications or skills.

(2) At the research area scopes, studies should be conducted in those barely investigated marine zones especially those around developing countries focusing on the TWPs in the coasts, bays, estuaries, oceans and so on.

(3) At the research object scopes, field studies are suggested caring about the occurrence, transport, transformation and fate of TWPs in both environmental and biotic matrixes. The behaviors, such as bioaccumulation, biomaginfication, and biodegradation of TWPs in marine species should be well investigated. The potential being bio-indicator of TWPs in marine ecosystems of typical aquatic species such as fish, shellfish, crustacean or algae should be discussed.

(4) At the toxicology scopes, much more biomarkers are suggested to be used in more marine species focusing on more TWPs and their leaches and absorbed chemicals, such as the nanoplastics of TWPs, toxic additives like 6-PPD and its transformation product 6PPD-quinone, which pose severe risks to aquatic organisms even at low concentrations.

(5) At the management scopes, integrating TWPs into existing marine pollution monitoring frameworks to inform regulatory policies, establishing universal protocols for TWP collection, isolation, and analysis, and strengthening the allocation of responsibilities for TWPs emissions and international cooperation are urgently needed to facilitate comparative studies and regulatory assessments.

Author contributions

YW: Conceptualization, Investigation, Writing – original draft. JX: Investigation, Writing – review & editing. YZ: Writing – review & editing. YP: Writing – review & editing. ZZ: Writing – review & editing, Visualization. SL: Writing – review & editing. XC: Writing – review & editing. JZ: Writing – review & editing, Investigation, Supervision. TW: Supervision, Writing – review & editing, Visualization.

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Conflict of interest

Authors YZ and XC were employed by the company Shandong Wudi Gold Turn Land Development and Construction Co., LTD.

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