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SOS-ZEROPOL2030

D4.1 Knowledge and outcomes from T4.1-T4.4 summarised in the 'integrated analysis' section of the individual case study pollutant template.

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Part B: Tyre Wear Particles (TWPs)

Public

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Table of Contents

1. Executive Summary

The SOS-ZEROPOL2030 project aims to deliver a stakeholder-led European Seas zero-pollution framework to help achieve the European Union's long-term ambition of 'Zero Pollution' in European seas. The SOS-ZEROPOL2030 project focuses on marine pollution , where (i) per- and polyfluoroalkyl substances (PFAS) and (ii) tyre wear particles (TWPs) were selected as example pollutant case studies for 'chemical' and 'microplastic' pollution, respectively. It is important to note that these two very complex case study pollutants were intentionally chosen to allow the zero-pollution framework to be stress-tested under the most challenging scenarios. As a part of Work Package 4 (Integrated Case Study Pollutant Assessments) within the SOS-ZEROPOL2030 project, this deliverable report (D4.1. Part B) provides an integrated assessment for TWPs, while a separate report (D4.1 Part A) is available for PFAS. The integrated assessment comprises four primary components: (i) Mapping of primary emission sources along the value chain, (ii) Determination of environmental risk, (iii) Mapping of existing value chain and technological actions and strategies for TWP emission reduction, and (iv) Mapping of current governance strategies/efforts/arrangements for TWP pollution.

Emission sources along the value chain: TWPs are primarily produced through the friction and wear of tyres upon driving vehicles, meaning that urban, highway and rural roads are the main point sources of TWP emissions, with a direct link between increasing traffic volumes and higher TWP emissions. Heavier vehicles emit more particles by distance travelled, while braking, accelerating and turning cause increased emissions. Emissions per km travelled are largest in urban areas (due to increased braking, accelerating and turning), followed by highways and rural areas. In addition to releasing particles, tyres are also the primary sources of many tyre-related chemicals (TWCs), where almost 800 additives and non-intentionally added substances (NIAS) are known to be used and/or present in tyres and TWPs. Roadside soil receives 45-80% of TWPs and surface waters receive 6-26%, while the smaller TWPs can remain airborne for long periods. TWPs and low mobility TWCs will accumulate in marine sediments, while smaller TWPs and mobile TWCs may be transported away from deposition sites. Estuaries and coastal zones are the primary accumulation zones, especially those closest to urban areas that act as a point source of TWPs. Recommendations include:

(i) Policy-based mitigation efforts that develop actions to reduce TWP emissions, with the following focus areas: tyre reformulation to reduce emissions, increased driver awareness campaigns regarding the impact of driving style on TWP emissions, improved emission capture technologies and road runoff treatment methods, and consideration of TWP emissions in future road design.

(ii) Reduction in the emission of hazardous chemicals via leaching into water through reformulation of vehicle tyres, with an emphasis of removing and/or replacing the most hazardous ingredients and those which leach most readily.

(iii) Emission mitigation actions for the marine environment should target those areas which are considered as hotspots for TWP emissions.

Environmental risk: The TWP risk assessment process also involved a detailed exposure assessment based on reported exposure concentrations and modelled data across different European regional seas and hazard assessment based on database and literature toxicity data and modelling. Owing to the complex nature of the pollutant, the risk assessment was conducted separately for both TWPs and TWCs/tyre wear leachates (TWLs). A method to integrate the exposure and toxicological data into the most comprehensive scenario-based environmental risk assessment for TWPs and TWCs was developed. The outcomes of the work highlighted that there is insufficient empirical TWP. TWL and TWC concentration/exposure data for the marine environment for it to be used for risk assessment purposes and modelling approaches are currently needed. The TWP risk assessment indicated that current particle emission concentrations in natural waters are unlikely to be above critical threshold values, but this does not account for TWP accumulation in sediments, nor does it account for TWCs/TWLs, which are much more toxic. TWLs comprise complex mixtures of TWCs vary in composition and cannot be directly measured in environmental samples, only individual TWCs. Hazard assessment could be conducted for a limited (~10) number of TWCs. However, many TWCs have other sources, attributing individual TWC concentrations in environmental samples is not really possible and they cannot be reliably used in risk assessment. Recommendations include:

(i) Improved analytical methods for TWP quantification and identification of tyre-specific TWCs.
(ii) Generation of significantly more TWP and TWL/TWC toxicity data for use in hazard assessment, including sublethal and chronic toxicity, as well as elucidation of the relative contribution of each to observed toxicity.
(iii) Inclusion of TWCs into environmental monitoring programs.

Existing value chain and technological actions and strategies for TWP emission reduction: When looking at existing value chain and technological actions and strategies for TWP emission reduction, it is only the fraction of TWPs ending up in wastewater treatment plant sludge that is incinerated that will be permanently removed from the environment. Several technical mitigation strategies are currently in place or in development that reduce the emission of TWPs to waterways and the natural environment, including the use of porous asphalt concrete, capture measures for road runoff, capture of airborne TWPs and reformulation of tyres to reduce emissions and/or the impact of emissions. The generation of TWPs could also be reduce by lowering speed limits, improving road pavement, lowering car weight, and limiting torque at low speeds or limiting braking power where it is safe to do so. However, there are no potential or future technological measures known to reduce the emission of TWPs that have not already been explored and that appear to be technically feasible in the coming years. Recommendations include:

(i) A focus on measures that intervene the least with human behaviour while driving as these are in general more effective and acceptable than those that require drivers to adapt their driving behaviour.

(ii) Technological measures to reduce generation and emission of TWPs and leaching of TWCs into the environment should first be taken at hotspots (e.g. coastal cities and metropolitan areas).

Governance strategies/efforts/arrangements for TWP pollution: Microplastics in general, and TWPs in particular, are a pollutant of emerging concern, and governance and regulation for TWPs is only beginning to emerge. Governance efforts were mapped for TWPs and pollution in the Mediterranean Sea region. Based on existing governance efforts at EU, regional and national levels, as well as industry responses to increased regulatory attention to TWP emissions and pollution, we draw two main conclusions. First, the governance of pollutants of concern currently seems much stronger at the EU level than at the regional sea and national level. Second, the regulatory focus on TWPs at the regional and EU level highlights different pace of agenda-setting and policy making between the Barcelona Convention and EU-level governance of TWPs, where EU-level regulatory developments currently happen rather independently from the regional sea convention. Recommendations include:

(i) Governance measures to effectively tackle TWP emissions need to consider sources in addition to solely addressing TWP emissions and release.

(ii) Enhanced cooperation and exchange between the EU and Barcelona Convention is needed to ensure governance in relation to TWPs in the Mediterranean Sea becomes aligned with EU policy developments.

2. Introduction

2.1. General introduction to tyre wear particles (TWPs)

Tyre wear particles (TWPs) have been identified as an important source of microplastics (MPs; <5 mm in 2 or more dimensions) but reported particulate emission volumes and environmental concentrations are highly variable, both spatially and temporally. Car tyres and the emitted TWPs are known to contain a high level and diversity of potentially hazardous chemicals (tyre wear chemicals; TWCs), many of which have been shown to partition or leach into environmental matrices. Here, we have collated the available literature data with the aim of providing state of the art insights for TWPs/TWCs into: (i) the chemical composition of TWPs, and (ii) the major emission sources across Europe, (iii) transport and accumulation, (iv) hazards and risks, (v) mitigation actions and (vi) governance strategies.

2.2. Background to WP4 of the SOS-ZEROPOL2030

As a part of Work Package 4 (WP4) within the SOS-ZEROPOL2030 project, this deliverable report document aims to provide both factual information and best-case risk assessment approaches to project partners and other work packages. This deliverable report (D4.1. Part B) focuses on one of the two case study pollutants (TWPs) within the SOS-ZEROPOL2030 project, with a separate report (D4.1 Part A) also available for per- and polyfluoroalkyl substances (PFAS).

This deliverable report document provides an overview of the findings in relation to emissions of tyre wear particles (TWPs) from:

i. Mapping of primary emission sources along the value chain (Section 3),
ii. Determination of environmental risk (Section 4),
iii. Mapping of existing value chain and technological actions and strategies (Section 5),
iv. Mapping of current governance strategies/efforts/arrangements (Section 6).

The deliverable report is specifically designed to be accessible to the intended end users within the project, as well as a diverse group of interested stakeholders. As such, the main report is a condensed summary of the findings and outcomes, while a detailed overview is presented in the accompanying Annexes for each of the 4 thematic areas described above. The main report contains brief summaries of the background and outcomes related to TWPs/TWCs, as well as the main conclusions and recommendations for end users (research community, policy makers, value chain stakeholders). In-depth analysis of available and relevant information, descriptions of the used methodology, detailed data summaries and references to data sources can be found in the Annexes.

3. Assessment of primary TWP emissions

3.1. Primary outcomes – TWP emissions

Please refer to Annex A.1 for further information and in-depth analysis regarding the emission of TWPs and their associated chemicals.

3.1.1. Main point source emissions for TWPs in Europe

Urban, highway and rural roads are considered the main point sources of TWP emissions. There is a direct link between traffic volumes and TWP emissions, with higher emissions with increasing traffic loads. 'Heavier' traffic (e.g., vans, buses, and trucks) emit more particles by distance travelled. In particular, braking and accelerating, as well as turning, causes increased emissions for all vehicle types. Summer tyres release more TWPs than winter tyres (while winter tyres, particularly studded tyres, cause road wear to a greater extent). Emissions per km travelled are largest in urban areas (due to increased braking, accelerating and turning), followed by highways and rural areas. Total TWP emissions from a few western European countries have been ranked in the following order: Germany > UK > Italy > Sweden > Netherlands > Norway > Denmark, with total emissions ranging from 6.7 (Denmark) to 92.6 (Germany) kTonnes/year[14]. As a proxy for TWP emissions, Figure 3.1-1 shows a map of the road network and per-country traffic intensity as stacked graphs (for countries where data is available). The traffic intensity data stems from the Conference of European Directors of Roads (CEDR) annual pan-European road network report (2021).

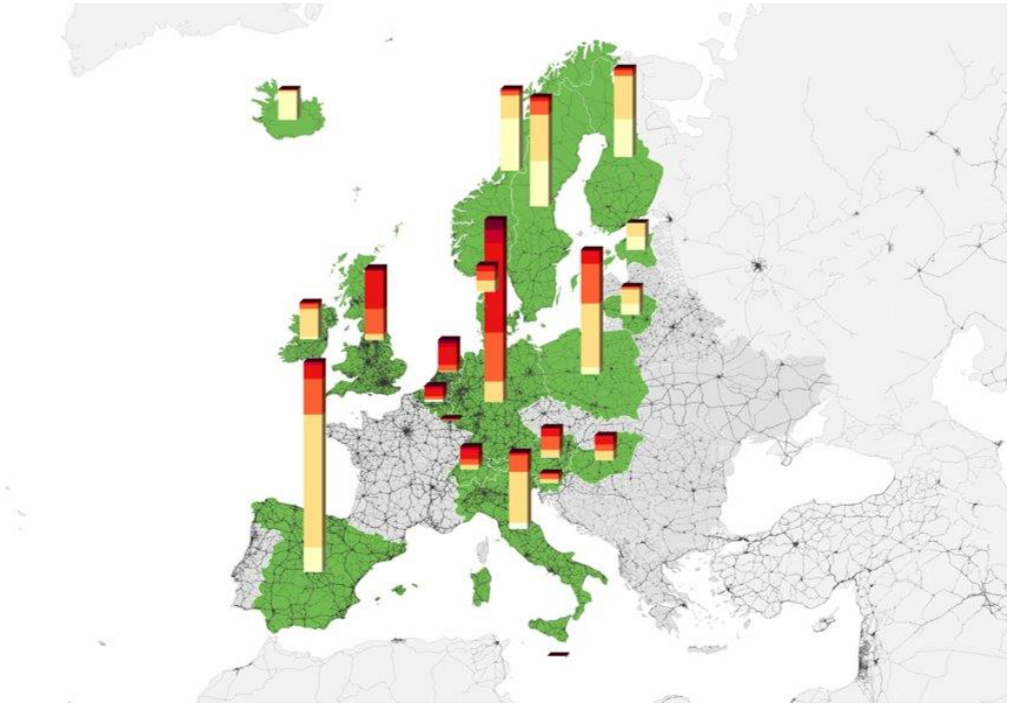

Figure 3.1-1. As a proxy for TWP, this map shows the road network in black, and per-country traffic intensity as stacked graphs where data is available. The total length of each bar represents the total road network length of the country (km).

Each individual shade indicates a different traffic intensity class, expressed as average annual number of vehicles passing (light yellow = fraction of the total network that has less than 5,000 cars passing, dark red = above 100,000). The traffic intensity data stems from the CEDR annual pan-European road network report (2021).

3.1.2. Most relevant diffuse sources of TWP emissions in Europe

Road run-off, atmospheric deposition of smaller TWPs, and re-distribution through wastewater treatment plants (WWTPs) may be considered relevant diffuse sources of TWPs. Re-purposing of roadside soil may also be a currently overlooked diffuse source of TWP emissions.

3.1.3. Largest contributors to overall TWP emissions

The act of driving vehicles with tyres is by far the largest source of TWP emissions. It is estimated that through its usable lifetime, a tyre loses between 10-30% of its initial mass due to the abrasion of the tread in contact with the road (Figure 3.1-2). Considering an average initial mass of 8.2 kg for a passenger car tyre and of 70 kg for a truck tyre, this means that around 1.2 kg (passenger car) and 12.6 kg (truck) of rubber are lost in the environment during the working lifespan of the tyre[12]. Due to their widespread use and higher vehicle numbers, passenger cars, followed by vans, lorries and trucks, are the largest contributors to TWP releases. In urban areas, mopeds, motorcycles and buses also contribute to overall TWP releases. Bikes and airplanes contribute minor, non-road vehicle-related, emissions.

Figure 3.1-2. Left: Conceptual image of TWP generation. Image reproduced from Oxford Indices: ***https://www.oxfordindices.com/news/2020/12/4/whats-in-a-tyre.*** *Right:* Comparison of tread loss between a new and old car tyre. Image reproduced from Tyroola: **https://www.tyroola.com.au/guides/what-are-bald-tyres/.**

In addition to releasing particles, tyres are also the primary sources of many tyre-related chemicals. However, many additive chemicals and non-intentionally added substances (NIAS) associated with tyres have other uses and sources. It is therefore difficult to distinguish the specific sources of these based on environmental concentration measurements, although the presence of multiple tyre-associated chemicals in a single sample could potentially increase the likelihood of tyres being a major source.

3.1.4. Most relevant TWP chemicals (additives and NIAS) in terms of production/usage volumes

Both unreactive/inert chemicals (e.g., polymers, oils, fillers, wax, resins and antioxidants) and reactive chemicals (e.g., sulphur compounds, retardants, accelerators, activators and glue) are used in the composition of the tyre tread (the part of the tyre where TWPs originate). Literature reviews have shown that nearly 800 chemicals are known to be used and/or present in tyres and tyre particles[1]. Chemical groups in descending order by number of known chemicals are: vulcanisation accelerators, antioxidants and antiozonants, polymerisation agents, plasticiser, other protectants, adhesion agents, nitrosamine compounds, reaction products, mastication agents, blowing agents, vulcanisation agents, vulcanisation retarders, vulcanisation activators. Notably, decomposition products and 'other' chemicals represent some of the largest groups of chemicals. As the exact tyre formulation for each tyre on the market in Europe is subject to intellectual property rights of the producers, there is limited transparency in terms of the use volumes of each of these chemicals in different tyres.

3.1.5. Most likely emission points for TWP chemicals along the value chain

It is the use phase of vehicle tyres (i.e., when driving) where TWPs are generated and emitted to the environment. As TWPs are formed through abrasion, other tyre-derived particles that might be generated in the production and end-of-life phases are not classified as TWPs, although they may contribute to chemical emissions. This is particularly true in the case of rubber granulates used as infill on artificial sports pitches and playgrounds. Similarly, chemical emissions from tyres can occur when the vehicle is stationary and still in contact with the environment (especially under wet conditions).

3.1.6. Primary sources of TWP and TWP chemical emissions

Vehicle tyres are the key contributors to TWP emissions. Similar particle and chemical emissions may be expected from other sources where very similar materials are used in consumer products, such as airplane tyres, bikes, conveyor belts etc. Crumb rubber artificial turf infill material produced from end-of-life tyres is different to TWPs (larger particle sizes and possibly different chemical compositions) and is therefore not considered in this study. However, crumb rubber does act as a source of TWL and TWC emissions.

3.1.7. Environmental compartment(s) most likely to receive TWP and TWP chemical emissions

Roadside soil is the main recipient of TWPs. European case studies have estimated 45-80% of TWP deposition occurs within the first few meters from the curb or roadside. Surface waters are the next largest recipient, receiving an estimated 6-26%. Smaller TWPs

(low-micron and nanosized) remain airborne for longer and may be subject to long-range transport. Similarly, small TWPs and mobile TWCs (i.e. those not adsorbing strongly to soil components) may be subject to wash-out from soils and transfer to groundwaters. TWPs are not buoyant in natural waters (freshwater or seawater) and will sink, thus accumulating in freshwater and marine sediments. However, it should be noted that their small size (0.05-100 µm) means that sinking rates can be quite slow as friction processes become more dominant than gravitational forces, and therefore smaller TWPs have the potential to be transported over longer distances.

3.1.8. Most likely transportation routes for TWPs and TWP chemicals (TWCs) towards their primary accumulation zone(s)

Most TWPs deposit directly onto the road or within a few metres of the roadside. Smaller TWPs may be airborne for longer and thus transported over longer distances before deposition occurs. Although limited data is available, some evidence of long-range transport of TWPs does exist. Water sources such as lakes, rivers and marine waters located close to main non-urban roads (e.g., highways) can act as accumulation zones and transport mechanisms for TWPs and TWCs to the marine environment (Figure 3.1-3). Similarly, coastal towns and cities may represent major point sources of emissions and transport of TWPs and TWCs to the marine environment via urban run-off (bypassing or through WWTP systems) and short-range atmospheric transport.

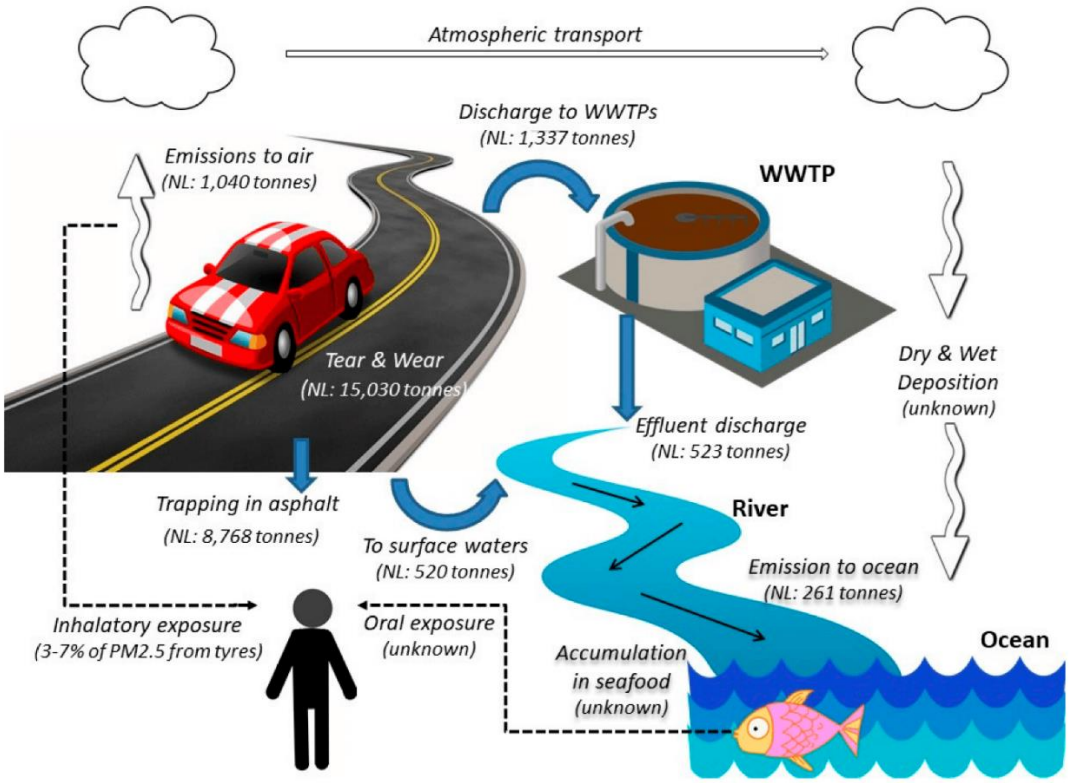

Figure 3.1-3. Primary transport routes for TWPs into the marine environment. Schematic reproduced from Kole et al.[14].

3.1.9. Marine environmental compartments are considered primary accumulation zones for TWPs and TWCs

For TWPs, estuaries and coastal zones are the primary accumulation zones, especially those closest to urban areas that act as a point source of TWPs. Sediments are expected to be the main marine environmental matrix in which the particles accumulate. For mobile TWCs, ocean currents will carry the chemicals away from marine deposition areas, while less mobile chemicals are likely to accumulate close to their entry point into the marine environment. Some chemicals are subject to biodegradation and photodegradation in the marine environment, although the degree of degradation depends strongly on their individual persistency.

4. Assessment of TWP risks

4.1. TWP risk assessment approaches

Please refer to Annex A.2 for further information and in-depth analysis regarding risk assessment of TWPs. This section describes the development of a risk assessment framework(s) that incorporated the following components:

- An exposure assessment of TWPs based on reported exposure concentrations and modelled data across different European regional seas and key EU regions, which is subsequently used to identify current data gaps.
- A toxicological assessment of TWPs based on database and literature data, which is subsequently used to identify current data gaps.
- A method to integrate this exposure and toxicological data into the most comprehensive environmental risk assessment for the specific case study pollutants, reflecting the current capability to evaluate the risks associated with this complex and emerging pollutant.

4.1.1. TWPs and their associated chemicals

Car tyres and the TWPs they emit are known to contain a high level and diversity of potentially hazardous chemicals, many of which have been shown to partition or leach into environmental matrices. The key ingredient groups of tyres and their weight contribution to a new tyre are summarised in Table 7.2-1 (Section A.1, Annexes). As such TWPs actually represent two quite separate and very different forms of pollution, the elastomer-based particle and the chemicals present in the bulk rubber material and emitted TWPs; herein further referred to as tyre wear chemicals/leachates (TWCs/TWLs). In this work, we have collated the available exposure (marine environment) and hazard (aquatic toxicity) data for TWPs and for TWCs. We have also decided, based upon the differing amounts of TWP and TWC/TWL exposure and hazard data to apply the risk assessment framework

separately to each. The outcomes of this work aim to provide a best possible risk assessment of TWPs and TWCs/TWLs across the European marine environment. Importantly application of the risk assessment framework to this complex form of pollution will serve to demonstrate how it can be used to identify current knowledge and data gaps that should be addressed for a more robust risk assessment to be conducted.

4.1.2. TWP, TWC and TWL hazard assessment using species sensitivity distributions

Toxicity data for TWPs and TWLs do not exist in the ECOTOXicology Knowledgebase, only data for individual chemicals (such as selected TWCs). **Owing to a lack of any TWP and TWL toxicity data in the ECOTOXicology Knowledgebase, only toxicity data for some TWCs could be retrieved and utilised for Species Sensitivity Distribution (SSD) assessment.** A detailed description of the SSD risk assessment approach and why this was selected for use with TWCs is provided in Section A.2.6 (Annex A.2). Entries in the ECOTOXicology Knowledgebase were found for 149 of the 295 TWCs included in this work (Table 7.2-8, Section A.2, Annexes). Most entries were found for formaldehyde (4996 entries, CAS 50-00-0), bisphenol A (4586 entries, CAS 80-05-7), bis(2-ethylhexyl) benzene-1,2-dicarboxylate (4186 entries, CAS 117-81-7), dibutyl phthalate (3481 entries, CAS 84-74-2), 4-nonylphenol (2713 entries, CAS 104-40-5), 3-(3,4-dichlorophenyl)-1,1-dimethylurea (2578 entries, CAS 330-54-1) and methylsulfinylmethane (2290 entries, CAS 67-68-5). Substantially more data were available from freshwater tests (total 20543 entries) than from seawater tests (total 4866 entries). A complete overview of entries in ECOTOXicology database categorized in freshwater and seawater data is provided in Table 7.2-8 (Section A.2, Annexes).

After selecting data for the toxicological endpoints LC_{50}, EC_{50} and NOEC where concentrations were reported as mg/L or µg/L (need for comparable data), a total of 8455 entries remained. Further filtering to include only seawater data, led to 1451 entries remaining spread across 43 chemicals (Figure 7.2-3, Section A.2, Annexes). These were dominated by bisphenol A (CAS 80-05-7, 288 entries), bis(2-ethylhexyl) benzene-1,2-dicarboxylate (CAS 117-81-7, 230 entries), formaldehyde (CAS 50-00-0, 159 entries), 3-(3,4-Dichlorophenyl)-1,1-dimethylurea (CAS 330-54-1, 152 entries) and 4-nonylphenol (CAS 104-40-5, 136 entries). After selecting only for LC_{50} data, 37 chemicals remained, ranging from 1 to 39 entries.

Using LC_{50} (endpoint) data reported in mg/L or µg/L (concentration unit), and with a species cut-off at a minimum of 4 species, we constructed log-normal SSDs (Figure 4.1-1) for 10 chemicals (1 was omitted, methylsulfinylmethane, CAS 67-68-5), of which all were in the TWP-associated chemical group. The log-normal SSD function was fitted to all the toxicity data after filtration, based on ranking of species by their mean LC_{50} values. In the SSD plots in Figure 4.1-1 we also included SSDs previously published[2] for the individual chemicals, for comparison. HC_5 was chosen as the level of protection and estimated based on the SSD, and these are reported for the 10 chemicals in Table 4.1-1. We also included

HC_{50} as these were reported in Posthuma et al (2019)[2]. Please see Section A.2.7 for an overview of that research.

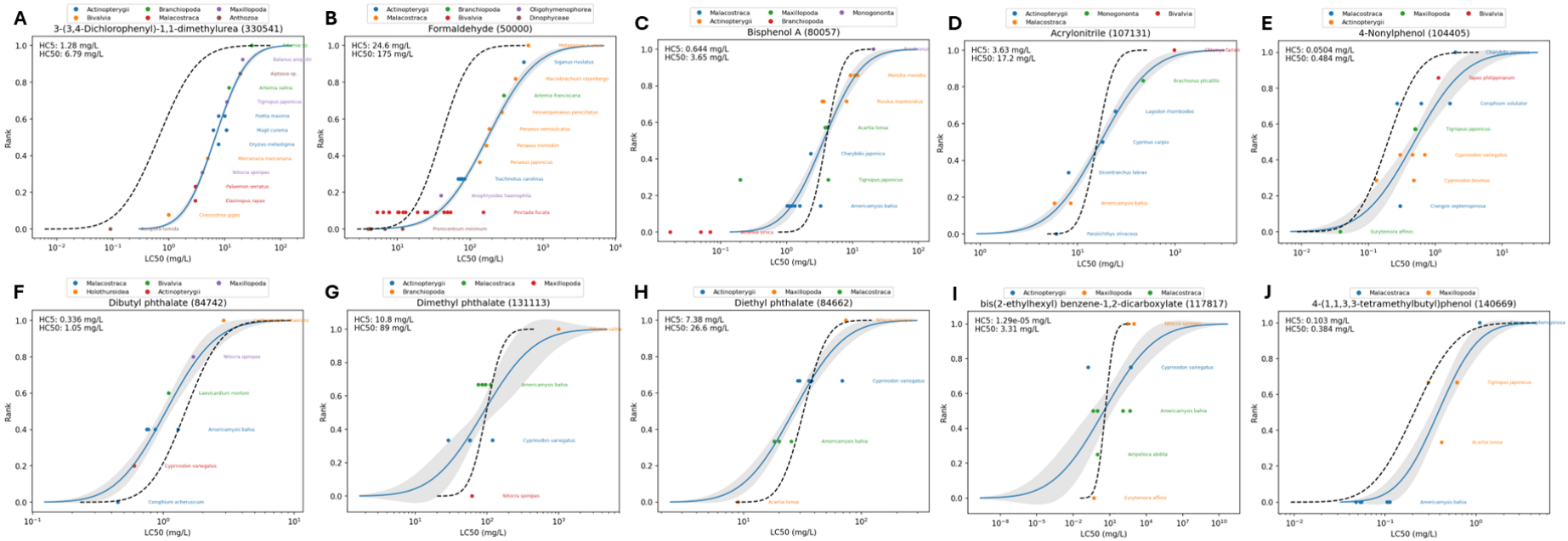

Figure 4.1-1. Species sensitivity distributions constructed from ecotoxicity data (seawater LC50) available in the ECOTOXicology Knowledgebase for A) 3-(3,4-Dichlorophenyl)-1,1-dimethylurea, B) Formaldehyde, C) Bisphenol A, D) Acrylonitrile, E) 4-Nonylphenol, F) Dibutyl phthalate, G) Dimethyl phthalate, H) Diethyl phthalate, I) bis(2-ethylhexyl) benzene-1,2-dicarboxylate and J) 4-(1,1,3,3-tetramethylbutyl)phenol. Latin names of test species are given in different colours based on taxonomic groups (class). Estimated HC_5 and HC_{50} benchmarks are also given in individual plots. Literature SSDs (Posthuma et al., 2019)[2] were also plotted for comparison (black dashed curves).

Table 4.1-1. Hazard concentration for 5th (HC_5) and 50th (HC_{50}) percentile for TWP-associated chemicals where SSDs were constructed. HC benchmarks were estimated based on SSDs using LC_{50} data for seawater tests available in the ECOTOX database. Number of species and taxonomic classes (based on taxonomical order) are included. HC_5 and HC_{50} from literature values (Posthuma et al., 2019)[2] are also included for comparison (indicated by the -P19 postfix).

CAS	Chem. group	Chemical name	HC5 (mg/L)	HC5-P19 (mg/L)	HC50 (mg/L)	HC50-P19 (mg/L)	Number of Species	No. of taxonomic classes
330541	TWP	3-(3,4-Dichlorophenyl)-1,1-dimethylurea	1.28	0.0832	6.79	0.661	14	6
50000	TWP	Formaldehyde	24.6	12.3	175	39.8	12	6
80057	TWP	Bisphenol A	0.644	1.9	3.65	3.98	8	5
107131	TWP	Acrylonitrile	3.63	9.41	17.2	15.8	7	4
104405	TWP	4-Nonylphenol	0.0504	0.049	0.484	0.195	8	4
84742	TWP	Dibutyl phthalate	0.336	0.651	1.05	1.48	6	5
131113	TWP	Dimethyl phthalate	10.8	48.7	89	97.7	4	4
84662	TWP	Diethyl phthalate	7.38	18	26.6	32.4	4	3
117817	TWP	bis(2-ethylhexyl) benzene-1,2-dicarboxylate	1.29E-05	0.52	3.31	4.27	5	3
140669	TWP	4-(1,1,3,3-tetramethylbutyl)phenol	0.103	0.0525	0.384	0.209	4	2

The HC_5 benchmark concentrations estimated for the 10 TWCs, for which there were sufficient data for SSD construction, were spread over three orders of magnitude[1]. Based on the current assessment, bis(2-ethylhexyl) benzene-1,2-dicarboxylate, 4-nonylphenol, 4-(1,1,3,3-tetramethylbutyl)phenol were the three most acutely toxic chemicals of the 10 TWCs subject to hazard assessment (i.e., they had the lowest HC_5 values). In contrast, formaldehyde, dimethyl phthalate and diethyl phthalate were the three least acutely toxic TWCs (i.e., they had the highest HC_5 values). It is important to mention, however, that the fit for bis(2-ethylhexyl) benzene-1,2-dicarboxylate is poor, providing a very low and probably unreasonable HC_5. By omitting bis(2-ethylhexyl) benzene-1,2-dicarboxylate, good correlations between our obtained HC5s (seawater only) with literature data[2] were observed (Figure 4.1-2).

[1] Excluding bis(2-ethylhexyl) which exhibited a poor SSD curve fit to the data.

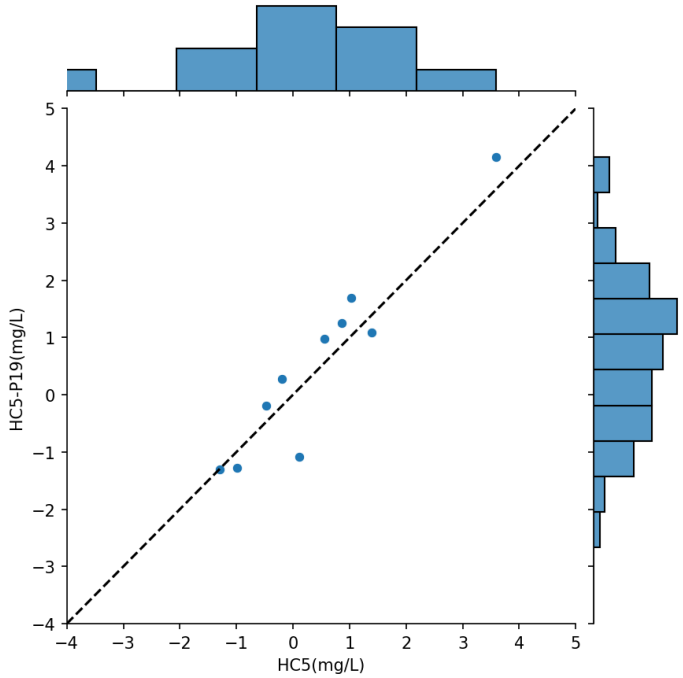

Figure 4.1-2. HC_5 values from literature[2] plotted as a function of our HC_5 values. Axes are log-scaled. A 1:1 line is given.

SSDs can currently only be constructed for a small number of TWCs, i.e., those for which sufficient aquatic toxicity data is currently available. It is generally considered possible to construct SSDs when >10 toxicity tests and their associated LC_{50} values (or another selected toxicity endpoint) are available. Ideally, these should represent a minimum of three different trophic levels, as well as both sensitive and non-sensitive species, although modelling allows fewer trophic levels and species to be used. The lack of dedicated TWP toxicity data means that SSDs cannot be generated. For TWLs, there is a growing amount of toxicity data available, but this is severely compromised and limited by a lack of standardisation in the production methods used to generate car tyre and TWP leachates. Several environmental parameters, including temperature, salinity, pH, UV-exposure and turbulence are known to impact the leaching of chemicals from micronized rubber particles[3,4] (See Section A.2.11, Annexes). The wide range of conditions employed to generate TWLs for toxicity testing (Table 7.2-10, Section A.2, Annexes) means that the TWL composition can vary significantly, rendering the resulting data from different studies incomparable. This issue is further compounded by the likely variations to TWC composition associated with the different source materials used to generate the TWLs. As a result, we have only attempted to generate SSDs and to derive HC_5 values for individual TWCs where sufficient data was available in toxicity databases.

4.1.3. Methodology for risk assessment

The scenario-based risk assessment concept that has been developed in WP4 of SOS-ZEROPOL2030 reflects the hypothesis that insufficient exposure and toxicity (hazard) data would be available for the chosen case study pollutants. In an ideal situation (**Scenario 1**),

there is already sufficient empirical exposure and hazard data available on which to base or conduct a robust risk assessment (Figure 4.1-3). Such situations are mostly applicable to single legacy chemical pollutants that have been studied extensively over many years. However, for some pollutants, especially those which are (i) complex (e.g. mixtures or a combination of particles and chemicals), (ii) difficult to measure in the environment, and/or (iii) classified as emerging contaminants, it is rare that there is either sufficient exposure and or hazard data to enable a robust risk assessment. In such cases, there is potential to utilise modelling approaches to 'fill in the gaps'. Within SOS-ZEROPOL2030, **Scenario 2a** is defined as a situation where there is sufficient exposure data available for a risk assessment to be conducted, but insufficient hazard data. In this scenario, any risk assessment would need to rely upon the use of models to generate the necessary hazard input data (Figure 4.1-3). Similarly, **Scenario 2b** is defined as a situation where there is sufficient hazard data available for a risk assessment to be conducted, but insufficient exposure data (Figure 4.1-3). In this scenario, any risk assessment would need to rely upon the use of models to generate the necessary exposure input data. For Scenarios 2a and 2b the robustness of the risk assessment for the selected pollutant is considerably reduced, although the overall robustness is strongly influenced by the quality of the modelled data. In the final and 'worst case' situation (**Scenario 3**), there is insufficient empirical exposure data and insufficient empirical hazard data (Figure 4.1-3). In this scenario, any risk assessment of a specific pollutant would need to rely upon the use of models to generate the necessary exposure and hazard input data. For Scenario 3 the robustness of the risk assessment for the selected pollutant is most likely to be the lowest in terms of robustness, although the ability to generate high quality modelled data could potentially mitigate some of the uncertainty.

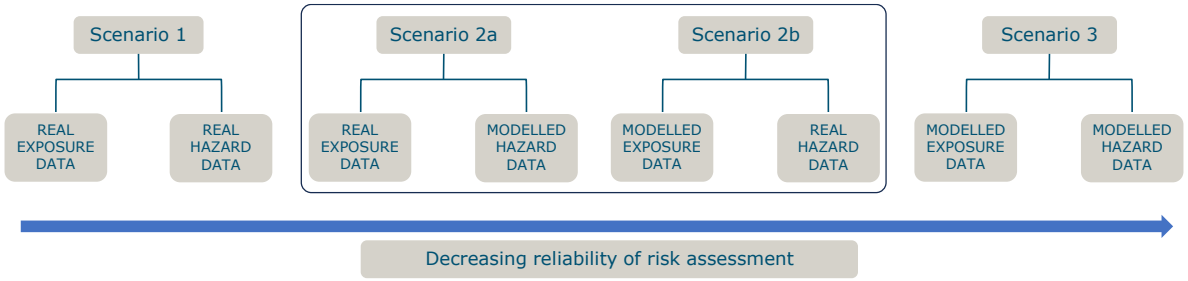

Figure 4.1-3. Overview of the 3 risk assessment scenarios developed and utilised within SOS-ZEROPOL2030. Each scenario reflects the availability of empirical exposure and hazard data for a specific pollutant and indicates when modelled data needs to be utilised. It also demonstrates how the reliability of the risk assessment decreases with an increasing reliance upon modelled data. Importantly, this can be used as a basis for identifying knowledge and data gaps that need to be addressed before a robust risk assessment can be achieved.

It is important to highlight that the goal of this work in SOS-ZEROPOL2030 is not to generate a final or complete risk assessment for either of the case study pollutants. Rather, PFAS and TWPs were selected as two of the most complex emerging pollutants from a European (and global) perspective and the goal is to develop a risk assessment

framework that forms an important and integrated component of the overall SOS-ZEROPOL2030 pollution mitigation framework and roadmap. In a sense, we use PFAS and TWPs as 'worst case' pollutants from a risk assessment perspective, to determine if it would be possible to conduct a meaningful and robust risk assessment provided that sufficient real/empirical exposure and hazard data are available. By applying the scenario-based risk assessment framework to the two case studies we also demonstrate how it can be used to (i) identify current knowledge and data gaps for these pollutants, and (ii) how modelling approaches can be utilised to (at least partially) fill in these knowledge gaps until the necessary empirical data can be generated.

4.2. Primary outcomes – TWP risk assessment

4.2.1. TWP exposure assessment in the (marine) environment

A total of 27 studies from the literature review claimed to report concentrations of TWPs and/or TWCs measured in environmental matrices sampled across Europe. However, only 2 of those 27 studies (~7%) reported data for the marine environment. As a result, there is insufficient empirical TWP concentration/exposure data for the marine environment for it to be used for risk assessment purposes. Furthermore, this extends to the different regional seas in the EU, preventing region specific analysis and comparison. Due to this lack of data, modelling approaches based on emission and dispersion modelling are required to 'guestimate' exposure concentrations for use in risk assessment. Based solely on the availability of exposure data, risk assessment would be either Scenario 2b or Scenario 3 (Figure 4.2-1). A detailed overview of the current knowledge regarding TWP concentrations in the marine environment and other environmental matrices is presented in Section A.2.3.

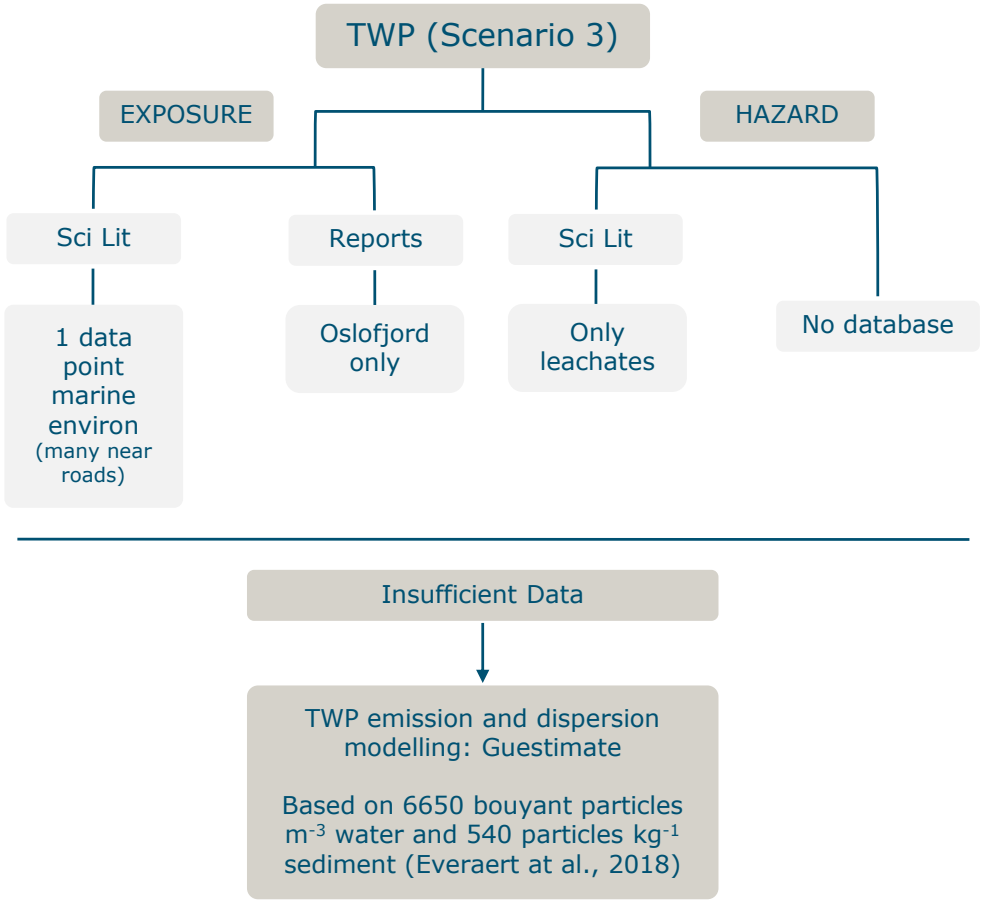

Figure 4.2-1. Assessment of the available TWP data indicated that there was currently insufficient empirical exposure and empirical hazard data for a robust risk assessment to be conducted on TWPs. As a result, TWPs can be considered as 'Scenario 3' pollutants. Assessment of the available exposure data indicated an insufficient amount of exposure data is currently available and that this would need estimated using a combination of emission and dispersion modelling.

In the absence of a European wide exposure dataset, we have created an exposure model based on TWP emission per capita (See Section A.2.4, Annexes). The exposure model estimates the number of TWP particles emitted via natural freshwaters (rivers, river deltas) into the European regional seas. Due to the lack of a pre-existing dataset for TWP exposure, a population-based model was used to estimate low, medium and high TWP production/emission scenarios across European and Non-European countries sharing the coastline of each European regional sea. For each TWP emission/production level, three additional emission scenarios (low, medium and high) were modelled for the release of TWPs into surface waters. A detailed description of the model and the emission/exposure model development is described in Section A.2.4 (Annex), while the outputs of the annual TWP emissions (number of TWP/year) and concentrations (TWP/m^3 per year) across the 4 European regional seas and 9 scenarios are presented in Table 7.2-5.

It is important to note that the emission load of TWPs refers to the total number of particles released into the environment over a specified time period, while the concentration of

TWPs represents the amount of these particles per unit volume (e.g., mass per cubic meter) emitted by a catchment area, calculated by dividing the emission load by the volume of water in which the particles are dispersed. Under the low, medium and high emission scenarios, the Mediterranean Sea exhibited the highest TWP annual TWP emission values (Table 7.2-5). The 20 catchment areas estimated to release the highest numbers of TWP to European seas annually are presented in Figure 4.2-2 and Table 4.2-1. These are distributed across all 4 European Seas, with a predominance of catchment areas in the Northeast Atlantic. Similarly, the Mediterranean Sea exhibited the highest TWP release concentrations to marine waters (Table 7.2-5). The 20 catchment areas estimated to release the highest numbers of TWP to European seas annually are presented in Figure 4.2-3 and Table 4.2-2. A high percentage of these catchment areas are located on the North African coast of the Mediterranean Sea. However, it is important to note that the higher concentrations are strongly influenced by lower levels of rainfall in the Mediterranean Sea region than in the other regional seas. Furthermore, the model only estimates emissions to the marine environment and not actual TWP concentrations in marine water or marine sediments, where accumulation of TWPs is likely to occur for the latter.

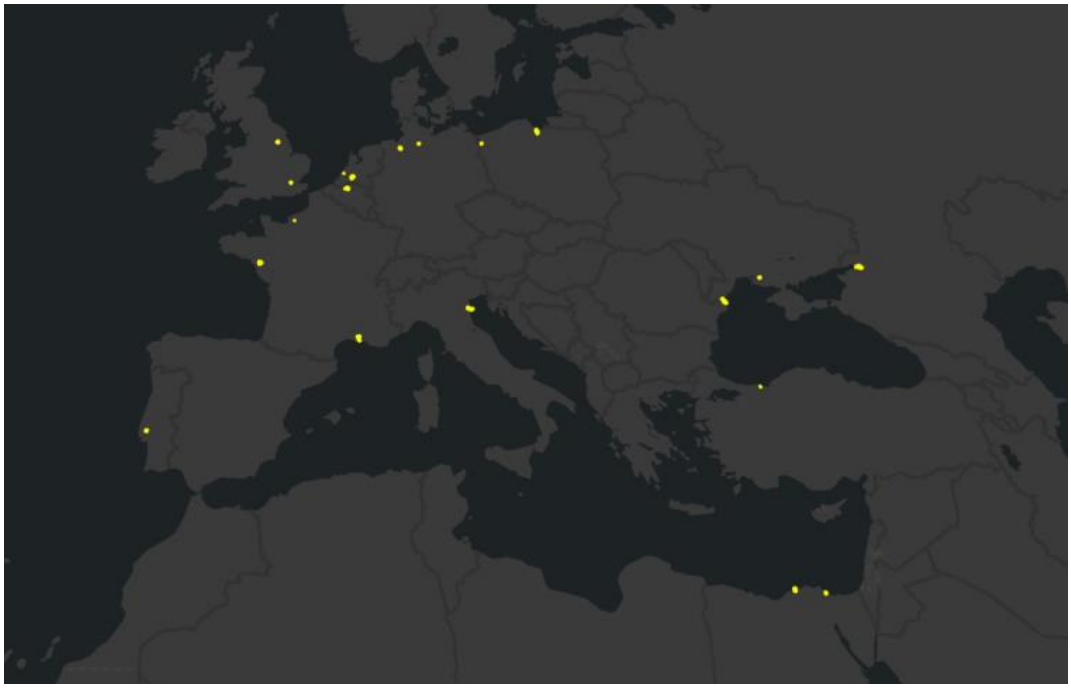

Figure 4.2-2. Top 20 catchment areas discharging TWPs within European regional seas.

Table 4.2-1. Top 20 catchment areas discharging TWPs loads within European regional seas.

Rank	Catchment Area	Water Catchment Type	Country
1	Nile	River	Egypt
2	Danube	River Delta	Ukraine, Romania
3	Rhine	River Delta	The Netherlands
4	Dnipro	River Delta	Ukraine
5	Vistula	River Delta	Poland
6	Elbe	River	Germany
7	Suez Canal	Canal	Egypt
8	Po	River	Italy
9	Don	River	Russia
10	Seine	River	France
11	Oder	River	Poland
12	Thames	River	England
13	Rhone	River	France
14	Scheldt	River	Belgium
15	Tagus	River	Lisbon
16	Karasu	River	Turkey
17	Weser	River	Germany
18	Thure	River	The Netherlands
19	Loire	River	France
20	Trent	River	England

Figure 4.2-3. Top 20 catchment areas discharging TWP concentrations within European Seas.

Table 4.2-2. Top 20 catchment areas discharging TWP concentrations within European Seas.

Rank	Catchment Area	Catchment Type	Country
1	Baltim	River Delta	Egypt
2	Tripoli	City	Libya
3	Alexandria	City	Egypt
4	Sidi Salem	River Delta	Egypt
5	Sousah	City	Tunisia
6	Barcelona	City	Spain
7	New Damietta	River Delta	Egypt
8	Gulf of Sidra		Libya
9	Zawiyah	City	Libya
10	New Damietta	River Delta	Egypt
11	Alexandria	City	Egypt
12	Athens	City	Greece
13	Marsa Matrouh	City	Egypt
14	Tripoli	City	Libya
15	Gran Canaria	Island	Spain
16	Mostaganem	City	Algeria
17	Oran	City	Algeria
18	Marsa Matrouh	City	Egypt
19	Odesa	City	Ukraine
20	Bani Sat	City	Algeria

4.2.2. TWP hazard assessment in the (marine) environment

The available evidence strongly suggests that it is the TWCs and their associated TWLs that are the drivers of TWP toxicity to marine organisms. There is currently no direct evidence indicating the physical TWP particles cause toxicity, but this has hardly been studied as it is difficult to test the particle and chemical toxicity of TWPs separately. Most of the data obtained from the literature study investigated the toxicity of TWLs rather than that of the TWP particles themselves. As such, there is insufficient empirical TWP hazard data (or bioaccumulation data) for the marine environment for it to be used for risk assessment of TWP specifically. The lack of sufficient TWP toxicity data at the pan-European level, means it is not possible to conduct a TWP hazard assessment at the regional sea level. As a result, modelling approaches are required to 'guestimate' hazard and toxicity data for use in risk assessment. Based on the combined availability of exposure and hazard data, risk assessment of TWPs in the marine environment are classified as Scenario 3 in our SOS-ZEROPOL20230 risk assessment framework (Figure 4.2-1). A detailed summary of the current knowledge regarding TWP hazards in the marine environment and other environmental matrices is presented in Section A.2.9 and Section A.2.10. From a meta-analysis with effect data for MP particles available in literature, Everaert et al., 2018[5] derived a safe water concentration of 6650 particles m^{-3} below which adverse effects are not likely to occur.

4.2.3. TWP risk assessment in the (marine) environment

A TWP risk assessment tool (RAT) can serve as an essential resource for identifying areas at risk of environmental contamination by mapping the distribution and concentration of TWP arriving in European Seas. Through this approach, it enables risk assessment based on predefined threshold concentrations that may pose risks, particularly in sensitive habitats such as coastal zones. The tool can highlight high-risk areas, where TWP pollution may impact the environment, supporting targeted mitigation efforts and informing policy decisions on TWP management.

A link to the interactive TWP RAT is provided here:
https://experience.arcgis.com/experience/ee9d22b305604ea59d976fd71c0e463f/

The exposure model based on TWP emissions per capita indicates that mean concentrations of TWPs arriving in the European regional seas, depending on the scenario, range between 0.0001 particle/m^3 and 0.016 particle/m^3 (see Table 7.2-5). The European regional sea with the highest yearly load of TWPs is the Mediterranean, followed by the Northeast Atlantic, the Black Sea and the Baltic Sea.

In the absence of particle and chemical toxicity data for TWPs and TWCs/TWLs, we have applied the Everaert et al., 2018[5] safe water concentration of 6650 particles m^{-3} as a threshold (Figure 4.2-1). While it is important to consider the simplification of our theoretical model, our risk assessment (excluding the potential role of TWP as chemical and/or biological vectors) suggests that this safe concentration will not be exceeded at the

river basin level, upon arrival in the marine environment. As such, when excluding chemical and biological effects, no direct effects of TWP emissions in water are occurring. However, this does not account for total microplastic concentrations, to which TWPs a contributor, nor does it account TWP accumulation in the marine environment.

Despite the scarcity of effect data, especially in relation to TWCs/TWLs, we expect adverse ecological effects to occur in the marine environment. The total load of TWPs in certain European seas, especially in those which are closed basins, is extremely high. Depending on the scenario and European sea, these loads range between 82 million and 1.2 trillion particles per year (see Table 7.2-5, Annex). It is essential to exercise caution when interpreting these results, as they are highly dependent on basin resolution.

4.2.4. TWC/TWL exposure assessment in the (marine) environment

TWPs contain a complex mixture of chemicals, with approximately 800 chemicals known to have been used at some point in the production of vehicle tyres (N.B. not all in the same formulation and not all are still used today). Given the complex chemical composition of TWLs, and the fact they can vary due to the original tyre composition and the environmental conditions influencing the leaching process, it is not possible to measure TWLs directly in the marine environment. It is, however, possible to measure the concentrations of individual TWCs that are known to be present in tyres and their resulting TWLs, but to date, very few TWCs have been reported in environmental samples. For those that have been reported, multiple sources in addition to car tyres and TWPs may be contributing to the measured environmental concentrations. As such, attributing individual TWC concentrations entirely to TWPs in marine environmental samples is not possible in most cases. There is currently insufficient empirical concentration/exposure data for TWCs or TWLs in the marine environment for it to be used for risk assessment purposes. This also means that there is insufficient TWL or TWC exposure data to be region specific across the different regional seas in the EU. As a result, modelling approaches based on emission and dispersion modelling are required to 'guestimate' TWL and TWC exposure concentrations for use in risk assessment. Based solely on the availability of TWL and TWC exposure data, any risk assessment would need to be model based, in line with Scenario 3 (Figure 4.2-4).

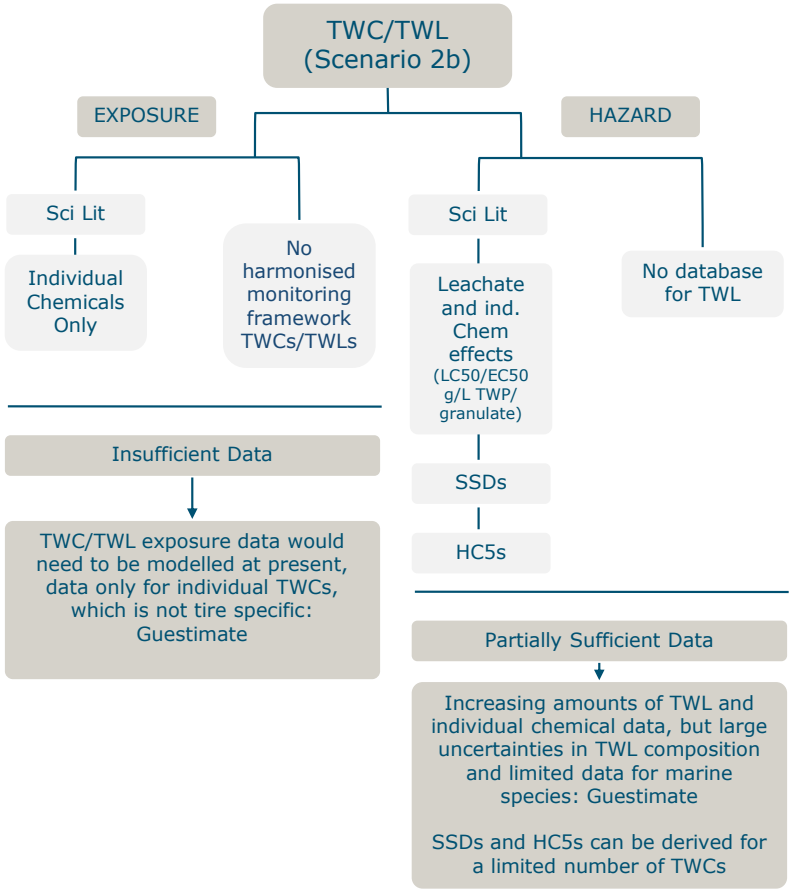

Figure 4.2-4. Assessment of the available TWC/TWL data indicated that there was currently insufficient empirical exposure data for a robust Scenario 1 level risk assessment to be conducted for TWLs especially, and so modelling is required to generate predicted exposure data for use in risk assessment. The non-specific nature of many TWCs, means that it is almost impossible to attribute individual chemical concentrations in the marine environment to TWPs only. However, there is (partially) sufficient empirical hazard data available for TWLs and some individual TWCs, although this remains quite limited for marine organisms. As a result, TWCs/TWLs can be considered as 'Scenario 2b' pollutants. Assessment of the available data indicated an insufficient amount of marine aquatic species hazard data, as well as an insufficient amount of spatiotemporal exposure data, is currently available to allow a European-wide assessment for the marine environment.

4.2.5. TWC/TWL hazard assessment in the (marine) environment

The available evidence strongly suggests that it is the TWCs and their associated TWLs that are the drivers of TWP toxicity to marine organisms (Table 7.2-10, Section A.2, Annexes). Many studies have attempted a toxicity identification evaluation (TIE) of TWLs, with fractionation into inorganic and organic fractions. Both fractions have proven toxic, with zinc (Zn) proposed as a key toxicant in the inorganic fraction, while a wide range of chemicals have been identified as potentially hazard-driving substances in the organic fraction. In comparative studies, variation in the toxicity of TWLs between different tyres has been demonstrated, but there is insufficient evidence to identify the chemicals responsible for the observed variation in toxicity. There is a sufficient amount of empirical acute (LC_{50}) marine species toxicity data available for only a small number of TWCs for it to be used in hazard assessment and for subsequent risk assessment purposes. If freshwater toxicity data is also utilised for the TWC hazard assessment, there is additional

data covering 96 TWCs in total (Table 7.2-8, Section A.2, Annexes). For these 96 TWCs, there 2235 total LC_{50} entries. There is considered to be a robust data set for a large number of the 96 TWCs in terms of size. There is also a reasonable amount of sublethal toxicity (EC_{50}) data available for a small number of TWCs (Table 7.2-8, Section A.2, Annexes), with 123 EC_{50} data entries for marine species representing 16 chemicals. However, it is important that these values are based on relevant and comparable toxicological endpoints for them to be considered useable for hazard and risk assessment. For freshwater, there are 880 EC_{50} data entries for freshwater species representing 89 chemicals. EC_{50} values have not been utilised in the SOS-ZEROPOL2030 hazard assessment but should be considered in the future.

In contrast, there is insufficient hazard data available for a hazard assessment of TWLs, as they represent a complex mixture of chemicals that vary due to tyre formulation and the environmental conditions under which they are produced. As a result, there is currently insufficient hazard data for key marine species regarding TWLs and most TWCs to be region specific across the different seas in the EU. Using SSD approaches to derive HC_5 values for both TWLs and individual TWCs could potentially identify the drivers of toxicity observed for TWLs (Figure 4.2-5). As SSDs cannot be constructed for the majority of identified TWCs due to insufficient marine toxicity data, modelling approaches are required to 'guestimate' marine toxicity values for TWLs and for the majority of known TWCs for hazard assessment and further use in risk assessment. Based on the availability of marine exposure data and hazard data, risk assessment of a small number of TWCs/TWLs in the marine environment can be conducted according to Scenario 2b in our SOS-ZEROPOL20230 risk assessment framework (Figure 4.2-4). A detailed summary of the current knowledge regarding TWL hazard and toxicity data in the marine environment and other environmental matrices is presented in Table 7.2-10 (Section A.2.12, Annexes). Importantly, these data are typically derived from TWLs that have been generated by different methods, meaning they are not directly comparable and cannot be used as input data for SSDs.

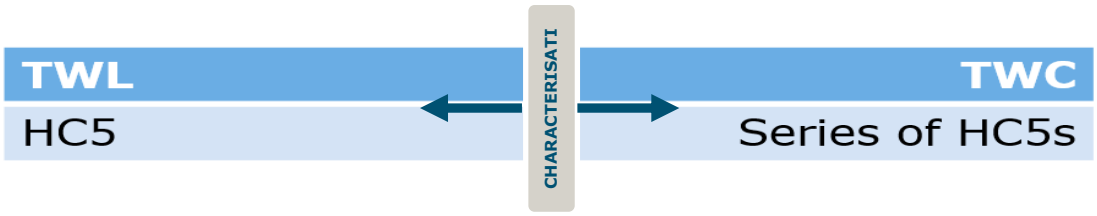

Figure 4.2-5. As TWLs are a complex mixture of chemicals it makes risk assessment more challenging. Although it has not been directly attempted in the current work due to a lack of data, it is suggested that available HC_5 data for TWL mixtures and HC_5 data for individual TWCs generated using an SSD approach could be used to identify the key drivers of TWLs .

4.2.6. TWC/TWL risk assessment in the (marine) environment

Owing to a lack of (i) exposure data for TWLs, (ii) toxicity data for TWLs, (iii) exposure data for TWCs that can be linked exclusively to tyres, and (iv) only sufficient marine toxicity for

a very small number of TWCs, it was not possible to conduct a risk assessment for TWLs and TWCs.

5. Assessment of TWP emission reduction measures

5.1. Primary outcomes – Existing product chain and technological actions for emission reduction of TWPs

Please refer to Section A.3 (Annexes) for further information and in-depth analysis regarding the reduction of TWP emissions.

5.1.1. The product chain of tyres and TWPs

Figure 5.1-1 depicts the lifecycle of tyres, proceeding from manufacturing on the left to emissions into the natural environment on the right. The focus of this lifecycle is on the formation of TWPs and on emissions of chemicals from recycled rubber products made from end-of-life tyres. The intermediate products are shown as green hexagons.

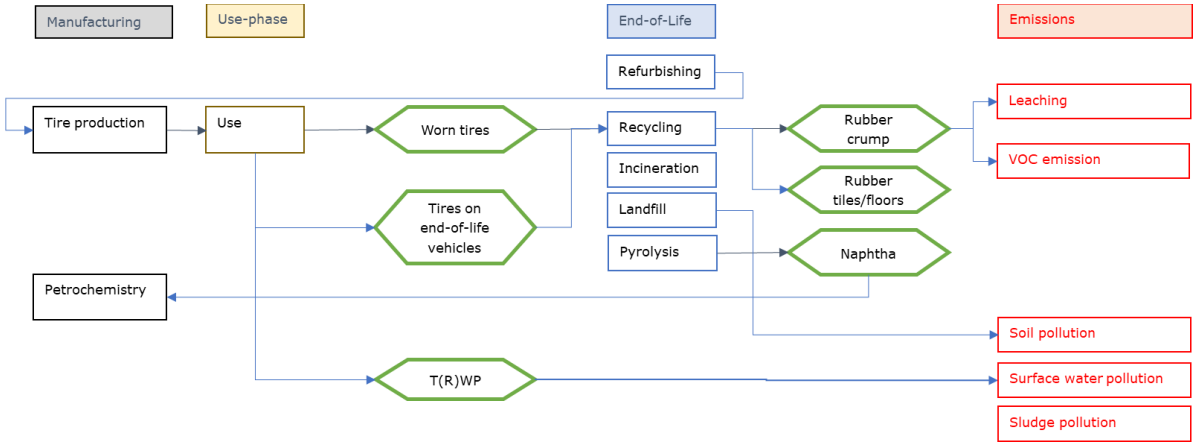

Figure 5.1-1. A schematical depiction of the life cycle of tyres, proceeding from the manufacture on the left to the emissions into natural environment on the right.

The manufacturing of tyres involves the mixing of many rubbers, fillers and chemical additives in the correct ratios and sequence into compounds. These compounds, together with metal wires and industrial textiles, are used to make well-performing tyres. Therefore, manufacturing already requires a complex network of multiple companies.

During the use phase tyres lose about 30% of their weight as TWPs[6]. These TWPs combine with road particles to form tyre and road wear particles (TRWPs) and which are deposited on and close to the roads. Due to the action of wind and rain, these particles will transport further to either road run-off catchment systems, the soils next to the road and the vegetation in proximity of the roads. Crude estimations indicate that at least half of the TRWPs settle in roadside soils[7].

Most urban roads have separate water-run-off catchment systems that connect via sewer systems and drains, while other roads have neighbouring gutters, creeks and other waterways. Some of the waterways are treated to remove pollutants and sludge, and if treated, a large proportion of the TRWPs will settle in the sludge. The sludge from waterways and sewage systems has traditionally been used as fertilizer in agriculture. However, due to rising levels of various contaminants (heavy metals, organic persistent substances, macroplastics, etc.), this use was restricted to sludge with sufficiently low levels of contaminants [EEC 1986/278] in the 1980s. In 2021, EurEau reported that 48% of European sewage sludge was used in agriculture. In some countries (Portugal, Bulgaria, Ireland), almost all sludge is used in agriculture, whereas in other countries (almost) no sludge is used for agriculture (Malta, Netherlands, Slovenia Slovakia)[8–10]. Where sludge is not used for agriculture, it needs to be dewatered, incinerated and the ashes need to be landfilled in a controlled manner[11–13]. When the directive was written in 1986, TRWPs were not originally a subject of regulation, as their presence in sludge was at that time not known or registered. Nonetheless, TRWPs are very likely one of the culprits causing many European sludges to exceed contamination threshold levels and therefore indirectly contribute to the loss of organic material and additional greenhouse gas emissions. In summary, TRWPs will spread out over soils, surface waters and sludges. Only the fraction of TRWPs that ends up in sludge that is incinerated will be permanently removed from these environmental compartments; in all other cases TRWPs will accumulate in the environment.

Worn tyres and end-of-life tyres can be processed in many different ways, with the method of choice depending on the legal and industrial context. In countries that mandate recycling, tyres with reasonable quality are refurbished and get a new tread. Strongly worn tyres are shredded and used in crumb and flooring applications. Rubber crumb has been used in playground and in artificial grass turfs for sport fields. The latter applications have been scrutinized as these rubber crumbs emit polyaromatic hydrocarbons and heavy metals[14,15] and this could negatively affect the health of the children and sports people.

5.1.2. Current technological actions to limit TWPs and their environmental effects

Several technical mitigation strategies are currently in place that reduce the emission of TWPs to waterways and the natural environment. These interventions have previously been taken to control other emissions (such as heavy metals, polyaromatic hydrocarbons and mineral oils)[16] and also reduce the emission of TWPs.

5.1.2.1. Porous asphalt concrete

To improve the drainage of rainwater from motorways and reduce roadside noise, in most European countries conventional tarmac is being replaced with open (porous) asphalt concrete. This porous road cover also acts as a filter, removing up to 80% of road particles and dirt, trapping it in its porous structure[16]. The TWPs that are trapped in the open asphalt

concrete will be recycled together with the asphalt into largely new asphalt. In this recycling process, most of the TWPs will be melted into the new asphalt.

5.1.2.2. Capture measures for road runoff

Many urban roads and motorways either have their own separate sewer system or are connected to the municipal sewage system. The sludge in these systems is separated from the water in gully pots or in settling tanks. Gully pots must be emptied regularly to avoid overflowing. Sludge from road runoff catchment systems is usually highly contaminated with heavy metals and must be dewatered and incinerated or dewatered and landfilled[17]. The removal effectivity of gully pots for particles is strongly dependent on many factors, including density, particle size, rainfall, and operational aspects such as frequency of emptying. Nonetheless, where these pots are well managed a substantial fraction of TWPs could be trapped[17].

In rural areas, road-water runoff is seldomly captured and treated. Usually, the runoff flows off the road to the sides. Often swales (gently sloping shallow channels filled with vegetation) are present along rural roads, but in mountainous regions gutters can be present to channel off the water. In most cases, the runoff ends up in the neighbouring soil and vegetation[17]. In some cases, the water is channelled to ponds and traps, although the sludge and sediment in these runoff ponds and waterways can be so contaminated that they must be treated as contaminated soil (for example by incineration)[18]. Where the wayside vegetation is composted, the trapped TWPs will contaminate the compost and be dispersed over agricultural fields and gardens. The same holds true when the sludge of the gully pots and sewage sludge is composted.

5.1.2.3. Capture of airborne TWPs

A fair share of TWPs are emitted into the air as particular matter. In the Netherlands, it is estimated that roughly equals 7%[19]. Particulate matter in the air is measured in two size classes (below 2.5 µm and below 10 µm; abbreviated as PM2.5 and PM10, respectively). There are multiple sources of this particulate matter, such as diesel engine exhausts, industries, wood fires, agriculture, TRWPs. Vegetation is known to absorb a part of these emissions and therefore trees and shrubs along the wayside can probably help to reduce the aerial emission of TWPs to some extent, but the level of the effect is unknown (https://edepot.wur.nl/22878). In general, the larger particles are trapped more efficiently than the smaller particles[17].

5.1.3. Reformulation of tyre compound aimed to reduce the emission of TRWPs

The technological challenge to reformulate tyres to reduce the generation of TWPs often results in dilemmas with respect to increasing rolling resistance (fuel usage) and reduced wet grip (safety) [19]. The only exception that is relatively widely acknowledged is substituting carbon black particles with silica particles as the abrasion resistance is improved without compromising on wet grip and rolling resistance[17]. The usage of highly dispersible silica

particles instead of carbon black is effective and is used already by many tyre manufacturers[17]. Many other options to improve tyre compounding have been explored. They vary from new types of self-healing rubbers and anti-oxidants, but their actual contribution to either lowering TWP generation or the dispersion of toxic chemicals into the environment remains unclear.

Context

Besides technological measures, many non-technical policy interventions could reduce TWP emissions. For instance, speed limits intended to reduce road casualties and nitrous oxide emissions also effectively reduce TWP emissions[17,20]. Interventions intending to encourage drivers to keep the tyres of their vehicles inflated at the correct pressure (and avoid under-pressured tyres) will not only reduce fuel usage but also reduces TWP formation. Measures to reduce the usage of winter tyres during summer could also reduce TRWP formation[17]. A recent article reviews a comprehensive set of technological and policy measures[17].

5.1.4. Potential and future technological measures to limit TWPs and their environmental effects

There are no potential or near-future technological measures known to reduce the emission of TWPs that have not already been explored and that appear to be technically feasible in the coming years. Many different mitigation strategies have been proposed, but the potential efficiency of these measures is either unknown or context dependant. Two types of strategies can be discerned: emission reduction measures and capture measures. Both strategies cannot deliver complete emission control and have their own sets of downsides, so an appropriate combination of measures needs to be sought.

Emission reduction measures include for instance:

- Speed limits
- Improved road pavement
- Improved tyre compositions
- Lower car weight
- Electronically limited engine torque at low speeds
- Smart software in vehicles to control the above and other relevant parameters
- Reformulation to remove the most toxic components of tyres and therefore reduce the impact of emissions

Capture measures:

- Separate sewer systems for road runoff, combined with sludge separation and incineration
    - Road runoff treatment with biofilters (swales) and/or roadside gullies in which the sludge settles

5.1.5. TWP chemicals emission reduction by reformulation

Rubber tread compounds for tyres are complex mixtures that have to fulfil many criteria. The technical challenge is to reformulate a tyre tread compound to raise the abrasion resistance (with the hope to produce less MPs) without compromising the rolling resistance, the wet grip, durability, etc. This proves to be very difficult in reality. One of the few improvements that we have observed over the last decades is the transition from carbon black to silica particles as fillers. These appear to be successful in improving the abrasion resistance without compromising other essential properties. It is, however, not confirmed that using these silica fillers actually reduces the formation of TWPs, as the limited experimental data is inconclusive[17]. Potentially this replacement will result in less pollution with carbon black and hence polyaromatic hydrocarbons.

The situation is even less clear for other tyre ingredients, as most industrial research is unpublished, and it is not publicly disclosed which replacement attempts have been made or what results were obtained. The fact that little is known about these industrial attempts, suggests that there are dilemmas that are not easily resolved. As discussed by Neupert *et al*, the Emission Analytics report suggests that the release of toxic chemicals from tyre wear particles differs strongly between different tyres[21].This suggests that the release of toxic chemicals from TWPs can be significantly decreased by altering the formulation of tyres.

5.1.6. Perspectives on TWP reduction potential

The emission of TWPs to natural environments can only be estimated from scientific abrasion studies combined with data about actual usage of tyres. In the air, particle contamination is measured with PM values, of which airborne TWPs form an unknown share. In waterways and sludges heavy metals and polyaromatic hydrocarbons are monitored, and again TWPs form an unknown share. Hence, in the final environmental compartments it is challenging to pinpoint the origin of these contaminants, and thus to link them to tyre usage. Nonetheless, tyre wear is one of the known sources for particles in the environment. This elusive situation makes it more challenging for politicians and industry to act on TWPs.

It is also difficult to assess the replacement potential for ingredients in rubber tyre tread compounds with the intention to reduce TWP generation, as most of this research is executed in industries and little progress is published. Most likely it is challenging to navigate the many dilemmas between the various technical demands that are placed on tyres. The absence of clear substitutes suggests that there are no easy alternatives available. Nevertheless, some data suggest large differences in the amounts of toxic materials in tyres from different manufacturers.

These observations and conclusions lead to the following summary:

- The formation of TWPs per driven kilometre can be decreased by both technological and policy measures. The increasing population, increasing mobility and increasing

car weight will however lead to higher volumes of TWPs. Without taking drastic measures with large social impact, TWP formation levels will likely not decrease significantly.

- Measures to prevent emitted TWPs from entering into the environment (e.g. road design, open asphalt concrete usage, separate sewer for roadside runoff capture) are likely more effective and acceptable than measures to reduce the formation potential.
- Regulations that force tyre producers to provide insight in formulations could help to reduce chemical pollution from TWPs.
- Future research is expected to clarify the efficiency of identified mitigation strategies.
- Smart software in vehicles to detect under-inflated tyres is likely to be adopted by a larger share of the vehicle fleet.
- Hopefully, more public research will be conducted on tyre compound reformulation, so the challenges and potential options can be better understood also by non-industrial researchers.

6. Overview of TWP governance strategies

Microplastics in general, and TWPs in particular, are a pollutant of emerging concern, and governance and regulation for TWPs is only beginning to emerge. This section focuses on how EU governance arrangements that are currently developing EU regulation to combat TWP emissions and pollution, which were identified as part of WP2 of SOS-ZEROPOL2030 (Van Leeuwen et al., 2023), are transcribed, implemented and complemented with governance efforts at regional and national level. This section also includes some examples of industry responses addressing the issue of environmental pollution by TWPs.

In consideration of the objectives of the SOS-ZEROPOL2030 project and the level of public awareness of the pollutant in the region (ref. Section A.4 Annexes), TWPs were selected as the case study pollutant in the Mediterranean Sea region. In particular, two life cycle stages were identified as the primary focus for the Mediterranean Sea region: tyre production and end-of-pipe collection through improved wastewater treatment facilities. These were both chosen to be able to cover important sources of TWPs across the pollutant life cycle. To understand governance developments at national level, we chose to focus on two countries, France and Italy, as both are EU Member States and represent significant car manufacturing and tyre producing industry. More information on the underlying rationale and methodologies can be found in Section A.4 (Annexes). A full deliverable report on regional and national governance strategies for TWPs will be published in November 2024.

6.1. Primary outcomes – Emerging TWPs governance in the Mediterranean Sea

6.1.1. Governance strategies by the EU and Barcelona Convention

The EU has developed or is developing a number of different regulations and directives directly or indirectly addressing TWPs (see Van Leeuwen et al., 2023). For example, recent policy developments targeted at TWPs include wastewater treatment and the production of tyres. The Euro 7 regulation sets thresholds for abrasion limits from tyres, to ban the most environmental harmful tyres from the market. Based on the methodology currently under development by the United Nations Economic Commission for Europe (UNECE) World Forum for Harmonization of Vehicle Regulations (WP.29) a tyre abrasion measurement method will be established and applied under Euro 7.

Furthermore, the Ecodesign for Sustainable Products Regulation will set minimum eco-design requirements for tyres. The Tyre Labelling Regulation sets information requirements to be reflected on the tyre label based on tyre performance standards, including safety and energy-efficiency that have to be fulfilled by tyre manufacturers when placing a tyre on the market.

The end of pipe stage includes measures such as monitoring at the inlets and outlets of urban wastewater treatment plants and in sludge as per the Urban Wastewater Treatment Directive (UWWTD).

At the Regional Sea level there is currently no regulation on TWPs developed under the UNEP/MAP - Barcelona Convention. However, relevant policy developments include the definition of TWPs in the Updated Regional Plan for Marine Litter Management (under the Barcelona Convention). As such, TWPs are defined as secondary MPs. Furthermore, guidelines for monitoring MPs coming from wastewater treatment plants are underway.

6.1.2. National level examples of governance strategies

This section explores national regulations, measures and initiatives in place and under development to reduce and mitigate TWPs at production and end-of-pipe stages with relevance to the Mediterranean Sea region.

Both France and Italy do not have explicit regulations addressing TWPs in place yet. Under the EU Plastics Strategy and its associated legislation, Member States are urged to create incentives for the use of energy-efficient and safe tyres. Italy implements the UWWTD and Water Framework Directive through legislative decree 152/2006. In consequence, the monitoring of MPs and the assurance of optimal water quality at the national level are in accordance with EU policy developments pertaining to the incorporation of MPs into the UWWTD and the Water Framework Directive (WFD). The same applies for France, which in addition has introduced the Anti Waste and Circular Economy Law (loi contre le gaspillage et pour l'économie circulaire) and the National Roadmap against Marine Litter

"Zero Plastic Reaching the Sea 2020-2023"; both policy/legislative tools currently focus more on single-use plastics than MPs.

Regarding the Euro7 regulation, both Italy and France together with five other EU countries initially pushed back the Euro7 proposal due to negative effects on investments in the sector. In 2023, France joined a joint statement urging the European Commission to acknowledge that MPs pollution is a cross-border challenge that requires ambitious EU legislation and measures included in an upcoming United Nations (UN) treaty to end plastic pollution, as well as to be incorporated into other EU policies, including eco-design rules, the waste framework directive, the single-use plastics directive and the packaging waste law.

6.1.3. Industry Examples of governance strategies

This section provides examples of industry approaches and initiatives for influencing TWP thresholds and reducing emissions from the production and end-of-pipe of TWPs. Industry involvement is defined by lobbying of tyre and car manufacturers represented through different lobby groups at EU level, including the European Tyre and Rubber Manufacturers Association (ETRMA), European Tyre and Rim Technical Organisation (ETRTO), European Automobile Manufacturers Association (ACEA), European Council for Automotive R&D (EUCAR) actively present in technical groups and public consultations[22]. The tyre industry, represented by ETRMA at EU level, is actively engaged in developing a tyre abrasion measurement method.

Other examples of industry initiatives at the EU level include the European Tyre and Road Wear Particles Platform, as well as Horizon Europe Projects, e.g. LEON-T project, Life Blue Lakes project in Italy, which contribute to the reduction and measurement of TWP emissions. Individual tyre manufacturers explore alternative materials and chemical formulations to improve tyre design. The French tyre manufacturer Michelin for example, developed a system for analysing TWPs that is complementary to Euro 7 standard.

7. Conclusions and recommendations

7.1. Conclusions

7.1.1. Conclusions – Emissions

- TWPs are considered an important source of MPs, representing a high proportion of total MP emissions globally.
- TWPs are primarily produced through the friction and wear of tyres upon driving vehicles (cars, buses, lorries etc). Although the production phase and the processing of end-of-life tyres can also generate particles (e.g., crumb rubber used on artificial sports fields, playground surfaces and as an additive material to asphalt), these waste products are not considered to be TWPs as no wear takes place.

- A wide variety of chemicals are used in tyres, with ~800 different chemicals reported. Some of these tyre wear chemicals (TWCs) can leach into the environment from both the tyres and the emitted TWPs. Pollution by TWPs therefore occurs both at the particle and chemical levels as complex mixtures of chemical leachates (TWLs).
- Soils and freshwater bodies located a few meters from roads are the major sinks for TWPs. However, most TWPs are relatively small (<100 µm), which makes them prone to longer-range transport and accumulation in areas further away from the emission location. This transport of TWPs can occur through aqueous pathways as well as through the atmosphere.
- The exposure model based on TWP emissions per capita indicates that mean concentrations of TWPs arriving in the European regional seas, depending on the scenario, range between 0.0001 particle/m^3 and 0.016 particle/m^3 (see Table 7.2-5). The European regional sea with the highest yearly load of TWPs is the Mediterranean, followed by the Northeast Atlantic, the Black Sea and the Baltic Sea.
- In marine environments, TWPs are expected to eventually settle due to having a higher density than seawater. Nothing is known about nano-sized TWP particles, and this represents a critical knowledge gap. Marine ecosystems adjacent to coastal towns and cities with significant vehicle volumes are expected to receive the highest TWP emissions.

7.1.2. Conclusions – Risk Assessment

- There is insufficient empirical TWP concentration/exposure data for the marine environment for it to be used for risk assessment purposes, and so modelling approaches are required to estimate environmental concentrations. Results from the exposure modelling indicate that TWP concentrations in the marine environment are likely to be highly localised around high traffic density areas close to the coast and at river mouths.
- The lack of TWP exposure data means that a robust pan-European or a European regional seas risk assessment cannot be conducted at present.
- While it is important to consider the simplification of our theoretical model, our risk assessment (excluding the potential role of TWP as chemical and/or biological vectors) suggests that this safe concentration will not be exceeded at the river basin level, upon arrival in the marine environment. As such, when excluding chemical and biological effects, no direct effects of TWP emissions in water are occurring. However, this does not account for total microplastic concentrations, to which TWPs a contributor.
- Despite the scarcity of effect data, especially in relation to TWCs/TWLs, we expect adverse ecological effects to occur in the marine environment. The total load of TWPs in certain European seas, especially in those which are closed basins, is extremely high. Depending on the scenario and European sea, these loads range between 82 million and 1.2 trillion particles per year (see Table 7.2-5, Annex). It is

essential to exercise caution when interpreting these results, as they are highly dependent on basin resolution.

- TWLs vary in composition due to the original tyre formulation and the environmental conditions under which leaching occurs. TWLs are therefore not a parameter that can be directly measured in environmental samples and the risk assessed. Only individual TWCs can potentially be risk assessed
- Many TWCs are typically not specific to tyres (or TWPs) and have other sources, which can complicate allocation of environmental occurrence only to tyres and TWPs; attributing individual TWC concentrations in marine environmental samples is not really possible.
- There is insufficient empirical concentration/exposure data for most TWCs in the marine environment for it to be used for robust risk assessment purposes at the pan-European or regional sea levels.
- Of the reported TWCs, 149 had some level of associated toxicity data registered in the largest ecotoxicity database (e.g. ECOTOXicology Knowledgebase). Most entries (>100 entries per individual chemical) were found for a small group of approximately 40 TWCs.
- Substantially more TWC ecotoxicity data were available from freshwater tests (total 20543 entries) than for seawater tests (total 4866 entries). Using only ecotoxicity data for marine species limits the amount of available data and reduces the robustness of the Species Sensitivity Distribution (SSD) hazard assessment.
- There was sufficient LC_{50} ecotoxicity data for marine species to be able to construct SSDs for 10 individual TWCs (around 3% of the included TWCs), meaning there is little understanding of the overall TWL/TWC toxicity, but that there is good knowledge on these specific chemicals.
- To conduct a risk assessment for TWPs as a form of microplastic pollution, modelled exposure data and modelled ecotoxicity data had to be utilised. This was limited only to assessing the risk associated with the aquatic TWP levels present in freshwaters (rivers) entering the marine environment, and did not include risk assessment of TWP accumulation in sediments. The TWP risk assessment **excluded the role of TWLs/TWCs as contributors to ecotoxicity**. The results indicated suggests that no acute effects of free-floating TWPs arriving in the marine environment are to be expected. **However, this does not reflect the true situation of TWP accumulation in sediments over time and the continued release of toxic TWLs and individual TWCs into the marine environment.**
- Hazard and risk assessment of TWCs and resulting TWLs is not possible to address at this stage.
- Over time, due to the continuous accumulation of TWPs in regional seas, it is possible that chronic environmental and human health effects might occur, but further study is needed on this issue.
- It is currently not possible to conduct a meaningful hazard assessment or risk assessment for TWLs, nor for the vast majority of TWCs. For TWCs where there is

exposure and/or toxicity data, an accurate risk assessment is not possible due to the uncertainty in attributing the presence of an individual chemical exclusively to vehicle tyres.

7.1.3. Conclusions – Technological measures

Technological measures can be split in measures to reduce generation of TWPs, measures to decrease the emission into the environment, and measures to limit leaching of TWCs from the particles.

- The generation of TWPs could be reduce by lowering speed limits, improving road pavement, lowering car weight, and limiting torque at low speeds or limiting braking power where it is safe to do so.
- Measures to decrease the emission of TWPs into the environment include tyre design optimized towards lower friction, using systems on cars that can catch a large share of the formed particles while driving, and building roads with open surfaces that capture particles.
- Capturing emitted particles could help to reduce spreading of TWPs and leaching of chemicals from TWPs into the environment. Potential measures include using a separate sewer system for road runoff combined with effective treatment of the collected runoff, using roadside gullies or swales designed to cause particles to settle, and reformulating tyres such that they contain fewer harmful substances and at lower concentrations.

7.1.4. Conclusions – Governance

Based on the presented overview of governance efforts at EU, regional and national levels as well as industry responses to increased regulatory attention to TWP emissions and pollution, we draw two conclusions.

- The governance of TWPs currently seems much stronger at the EU level than at the regional sea and national level. Expert stakeholders often suggest that the EU is the natural focus for many of these challenges, since the pollutants are not national in character, and the economic players are European or global in nature. Pro-active national and regional initiatives rather take the shape of adopting EU regulatory developments quicker or by adopting stricter standards. For TWPs there is currently no example of legislation or policy development at the national level targeting this specific pollutant.
- The regulatory focus on TWPs at the regional and EU level highlights differences between the UNEP/MAP - Barcelona Convention and EU governance approaches of TWPs. Currently, EU-level regulatory developments related to TWPs happen rather independently from the regional sea convention. This is because the Barcelona Convention and EU have different legal mandates and timelines for agenda-setting and development of measures around microplastics and TWPs. For example, the regional sea convention in relation to microplastics focuses on protecting the coastal and marine environment, through prevention measures aiming to promote

regulation on the use of primary microplastics or standards for product labelling or voluntary agreements, while the EU recently included the role of eco-design and developing thresholds for tyre abrasion as part of developing a more far reaching governance approach.

7.2. Recommendations

7.2.1. Recommendations – Emission

- Policy-based mitigation efforts should focus on developing actions that reduce TWP emissions. This should be conducted in multiple ways to achieve maximum impact in emission reductions. Specific focus areas could include tyre reformulation to reduce emissions, increased driver awareness campaigns regarding the impact of driving style on TWP emissions, improved emission capture technologies and road runoff treatment methods, and consideration of TWP emissions in future road design.
- The emission of hazardous chemicals via leaching into water can be reduced through reformulation of vehicle tyres, with an emphasis of removing and/or replacing the most hazardous ingredients and those which leach most readily.
- Emission mitigation actions for the marine environment should target those areas which are considered as hotspots for TWP emissions (e.g. cities, roadsides).

7.2.2. Recommendations – Risk Assessment

- There is an urgent need to develop analytical methods that allow for TWP concentrations to be robustly determined in marine and aquatic environmental samples and an urgent need for such data to be generated through routine monitoring (with inclusion in appropriate data bases).
- Identification of tyre-specific TWCs (or least TWCs with limited use in other applications) should be prioritised to identify marker chemicals for TWP pollution in environmental samples. While methods for isolation and quantification of TWPs from environmental samples are improving all the time, there is a need for harmonisation and wider application of such methods in marine pollution monitoring so that data on TWP levels can be generated.
- Future studies should prioritise the selection of specific locations and hotspots at a more localised scale to enhance the accuracy and relevance of the findings regarding TWP exposure and risks.
- To enable more accurate hazard assessment, it is critical to improve on existing knowledge about particle-driven (TWP) and leachate-driven (TWL) effects of tyre wear specifically, and MP in general.
- There is an urgent need to include multiple TWCs in environmental monitoring programs to generate the exposure data currently lack for TWP/TWC risk assessment in the marine environment.
- One of the main recommendations is the generation of robust toxicity data for more TWCs that is added to relevant ecotoxicity databases (e.g. ECOTOXicology

Knowledgebase) and not 'lost' in the grey literature. Adding existing TWC toxicity data from additional sources (e.g. grey literature) and/or through modelling-based methods will also help to enrich data and SSD coverage.

- It is recommended to combine all available ecotoxicity data into 'global' SSDs, which will have higher robustness and statistical strength. By filtering only for TWC LC_{50} data for use in the SSDs (as in this study), other toxicological endpoints are omitted that may provide additional insights into TWC hazards. As such, there is potential for including EC_{50} data available for individual TWCs in the data sets used for SSD construction in the futureEC_{50} data (sublethal toxicity and chronic toxicity) for TWCs should be included in the future, as this will increase the size and number of available data points. In turn, these larger toxicity datasets will lead to an improvement in the robustness and accuracy of the resulting SSDs, particularly for algae studies where EC_{50}s are exclusively reported. *It is important to note that EC_{50} data can be based on any biological variable/parameter that can be measured (from molecular to ecologically relevant variables) and would need substantial curation before being used, representing a highly time consuming task.*
- It would be of value to re-assess and curate the available data for chronic effects, which probably will warrant a more robust HC_5 (more data) and more protective benchmarks (ECs and NOECs are lower than LCs). Conversion of acute SSDs to chronic SSDs through AFs is an option that can also be considered.
- Additional ecotoxicological research in which marine species are chronically exposed to realistic environmental TWP concentration series are also urgently needed.

7.2.3. Recommendations – Product chain and technological actions for emission reduction of TWPs

- Measures that intervene the least with human behaviour while driving are hypothesized to be in general more effective and acceptable than those that, for example, require drivers to adapt their driving behaviour. Measures with low interference to behaviour include improved road design, runoff collection, and tyre formulation based on lower amounts and fewer types of harmful chemicals. These measures can be initiated and enforced by implementing regulations.
- Technological measures to reduce generation and emission of TWPs and leaching of TWCs into the environment should first be taken at hotspots (with coastal cities and metropolitan areas as focal point). When implementing these measures, monitoring campaigns can help to understand the effect of individual measures as well as sets of measures.

7.2.4. Recommendations – Governance

- Governance measures to effectively tackle the TWPs problem need to consider TWP sources in addition to solely addressing TWP emissions and release. As such, different governance approaches and best practices should be considered, ranging from EU, national, and regional sea convention level in the design and

implementation of TWP-relevant policies addressing not just TWPs, but also tyre design and abrasion. Bridging policy-making addressing various sources and sites of pollution requires enhanced communication channels by either creating joint TWPs working groups, regular forums or establish stakeholder platforms for dialogue between EU representatives and regional sea convention bodies to share TWPs updates, conflicting policies, challenges and best practices, as well as address specific issues affecting different levels of governance.

- In addition, enhanced cooperation and exchange between the EU and UNEP/MAP -Barcelona Convention is needed to promote governance developments in relation to TWPs in the Mediterranean Sea .

A. Annexes

A.1. Assessment of primary TWP emissions

A.1.1. Methodology

To address the thematic areas highlighted above, peer-reviewed literature was accessed using Web of Science in February 2023[9]. Formulated search strings (key terms combined by Boolean operators) were used to retrieve bibliographic data (with abstracts) from the databases. Collated peer-reviewed literature was screened using Rayyan[10, 11], applying a combination of soft and hard include/exclude criteria.

To address ***chemical composition*** of tyres and TWPs, the following search strings were used: care ti(y)re OR ti(y)re wear particle* OR TWP AND chemical composition OR reaction* OR composition OR chemical* OR element* OR byproduct OR vulcanization OR stabilizer* OR filler*. 415 unique articles were from the last 10 years were exported and subject to screening by two double-blind reviewers. Include criteria were: Rubber*, Zn, Metals, Cu, Carbon*, Metal*, Natural*, Zinc*, Reaction*, Hydrocarbons, Oil*, Additives, Copper*, Phenylenediamine, Synthetic*, Additive*, 6PPD*, filler*, PAH*, Carbon black*, Reactions, Steel*, Silica*, Elastomer*, Polyester*, Catalyst*, Sulphur*, Rubbers, Binder*, Agents, Amines, Hydrocarbon*, Cyclic amines, By-product*, Zinc oxide*, Phenylenediamines, Benzotriazoles, By-products, Phthalate*, Catalysts, Amine*, Nylon*, Resin*, Chlorinated paraffins, Natural and synthetic, Anti-oxidants, Carbon blacks, Aryl-amines, Binders, Resins, Steels.

Exclusion criteria were: titles and abstracts not mentioning either ti(y)res or TWP, articles only reporting concentration in the environment and not directly in tyres or tyre particles, articles only addressing particle emissions, articles only addressing toxicity without chemical characterisation or addressing toxicity of individual chemicals without a direct link to tyres. A total of 31 articles were left after inclusion/exclusion criteria were implemented, 25 of which were accessed and reviewed for information. Further information was also obtained from cascading literature, as well as overviews provided by tyre manufacturers.

To address ***sources of TWP emissions***, the following search strings were used: ti(y)re wear particles OR AND emission sources OR emissions OR sources AND environment. Refinement of the search was performed to only include papers published from 2013 onwards, all in English language and from European countries. A total of 992 unique articles were exported and subject to screening by one reviewer. Include criteria were: Tyre wear, tyre wear, TWP.

Exclusion criteria were: non-European study area, articles only reporting method development or laboratory assessments without field sample demonstration. A total of 45 resulting articles were accessed and reviewed for information, two of which were deemed not relevant and subsequently omitted.

A.1.2. Tyre wear particles

Every year, almost 2 billion new vehicle tyres are produced world-wide. At the same time >1 billion tyres reach their end-of-life. During use, millions of TWPs form from each tyre through abrasion of the rubber material in contact with the road surface. Friction between vehicle tyres and the road surface causes tiny particles of the tyre to wear of, forming so-called TWPs. TWPs have many definitions, depending on the perspective they are studied in, but the following may be the most generic and easy to understand: ***TWPs are small particles abraded from tyres during driving due to friction between the tyres and the (road) surface***[75, 76]. In terms of vehicle emissions, TWPs are defined as *non-exhaust particles that are transported to and through the environment primarily through run-off and via air*. It is important to distinguish TWPs from 'tyre particles' generated through shredding, milling and grinding of tyre tread in the laboratory or as part of repurposing of end-of-life tyres[77, 78]. Other types of MPs or microparticles may be released from vehicle use on roads, including road wear particles and MPs from road markings[19]. TWPs and road wear particles are often grouped together and defined as 'tyre and road wear particles (TRWPs).

TWPs have been identified as being 'roundish', 'kidney'-shaped' or elongated[79]. They are chemically and physically affected by the interaction with the road surface and friction energy, changing shapes and inclusion of road wear and mineral particles on the TWP surface[79, 80]. The most commonly detected size ranges of TWPs are 0.05-100 µm, with most of the particles being below 2.5 µm[14, 81-83].

A.1.3. Tyre and TWP chemical composition

Car tyres contain a complex mixture of components ('ingredients'). The production of car tyres is a multi-step process, with a multitude of materials, chemicals and processes involved. As a basis for the tyre tread, synthetic and natural rubber, as well as a multitude of other chemicals are used. Steel for cords and bead cores, and textiles for reinforcement are also used. The key ingredient groups of tyres and they weight contribution to a new tyre are summarised in Table 7.2-1.

Table 7.2-1. Main components in a general tyre formulation[76, 77]

Group	Content	Function	Wt.%
Natural and synthetic rubbers	Poly-butadiene (BR), styrene-butadiene (SBR), neoprene-isoprene, polysulphide	Main components that give elastic properties providing force, friction and comport.	40-60
Fillers	Carbon black, silicon dioxide, precipitated calcium carbonate (PCC), calcium and aluminium silicates, clay, resins	Strengthens the rubber by improving resistance to strain and wear and tear.	20-35

Oils	Mineral oils	Lubrication.	12-15
Reinforcement materials	Textile (rayon, nylon, polyester and aramid fibres) or metal nets.	Reinforcement to provide strength and resistance.	5-10
Additives	Preservatives, antioxidants, drying agents, plasticisers, processing aids.	Various.	5-10
Vulcanisation agents	ZnO, ZnS, S, Se, Te, thiazoles, organic peroxides, nitro compounds	Transforms rubber to a solid product through vulcanisation (hardening of the rubber) by forming cross-links between polymer chains.	1-2

A.1.3.1. Tyre tread composition

Tyre tread composition varies depending on the purpose or intended use of the specific tyre and the area of use, for example summer and winter tyres. For winter tyres, the tread pattern is aligned across the tyre to provide a good grip on snow and ice, while summer tyres have a tread pattern that runs around the tyre to reduce the risk and effect of hydroplaning/aquaplaning. Non-studded winter tyres have a type of tread called sipes, which contributes to providing the best possible grip. To optimise the desired properties for each tyre category, tyre manufacturers use different combinations of compounds in different tyres. In winter tyres, the rubber mixture is softer than that used in summer tyres, to ensure that winter tyres remain soft at low temperatures. In contrast, a harder mix of rubber is used in summer tyres to prevent them from becoming too soft in very warm conditions[77]. An example of the difference in composition of three different types of summer tyres is shown in the Table 7.2-2 (reproduced from [8423]).

Table 7.2-2. Example of composition of summer tyre samples from different producers (wt.%)[84]

Component	Tyre A	Tyre B	Tyre C
SBR	39.0	-	20.0
NR	-	41.3	45.0
BR	-	22.3	25.0
Carbon black	36.9	23.2	-
Oil	19.5	4.1	-
Other	4.5	9.1	10.0

A.1.3.2. Rubber

Rubber is an elastic substance either gathered from natural latex through exudation or collected from certain tropical plants ('natural rubber') or derived from petroleum and natural gas ('synthetic rubber'). The elastic and tough properties of rubber make it extremely suited as a main component in vehicle tyres. The rubber used in vehicle tyres is a mixture of different compounds, where the most important are elastomers – large, long-chained molecules that can be stretched and still retain their original shape once tension is removed. These elastomers contribute largely to the strength of rubber and its ability to be stretched repeatedly without tearing or breaking. Synthetic rubber does not have all the properties of natural rubber, and in many cases synthetic rubber cannot substitute natural rubber. For instance, natural rubber provides better resistance to tears and fatigue cracks in tyres. A mixture of natural and synthetic rubber is therefore often used to make vehicle tyres.

A.1.3.3. Synthetic rubber production

Synthetic rubber is produced via different methods, including polymerisation, mixing and latex treatment. Further chemical steps to modify the rubber polymer may involve treatments such as churning (mastication), extrusion, calendaring, and vulcanisation. The two most important synthetically produced rubber polymers used in vehicle tyres are butadiene rubber (BR, made from 1,3-butadiene monomers) and styrene-butadiene rubber (SBR, made from styrene and butadiene monomers).

A.1.3.4. Natural rubber production

Natural rubber is flexible, a natural electrical isolator and resistant to many corrosive chemicals. It is obtained from latex, a white fluid consisting of proteins, starch, and alkaloids produced by certain plants. Approximately 20,000 different plants produce latex, but only about 2500 of these produce latex with natural rubber. The main monomer that makes the polymer in natural rubber is *cis*-1,4-polyisoprene. The processing of natural rubber starts with the harvesting of latex from the plant. Acid is added to the latex to cause the material to coagulate/from lumps, before water is removed by a mill, simultaneously as the remaining components (lumps) are turned into sheets. After this the sheets undergo pre-vulcanisation through specific chemical reactions at low temperatures. Subsequent heating after this step hardens the rubber and finalises the vulcanisation process.

A.1.3.5. Chemicals in tyres

Both unreactive chemicals (e.g., polymers, oils, fillers, wax, resins, and antioxidants) and reactive chemicals (e.g., sulphur compounds, retardants, accelerators, activators and glue) are used in the composition of the tyre tread. Most of these reactive chemicals are used in tyre production during the vulcanisation and hardening processes. The tyre tread usually consists of natural rubber and polybutadiene rubber in specific ratios, as these components provide resistance to low heat build-up, as well as resistance to cracks and wear and tear. Natural rubber composites are preferred over conventional metals as they have favourable mechanic and chemical properties when fillers are added[85-87]. An

overview of the main chemical groups present in vehicle tyres, together with examples of specific chemicals within each group and their function is presented in Table 7.2-3.

Table 7.2-3. Examples of tyre compound groups, typical chemicals and their function[77]

Compound	Examples of chemicals	Function
Vulcanising agents	Sulphur	Transforms rubber to a solid product through vulcanisation (hardening of the rubber) by forming cross-links between polymer chains.
Activators	ZnO and stearic acid, resins.	Activates hardening during vulcanisation.
Accelerators	Sulphenamides (*N*-cyclohexyl-2-benzothiazolsulphenamide, CBS)	Speeds up the vulcanisation process.
Inhibitors	Nitrosodiphenylamine, salicylic acid, benzoic acids.	Prevents too rapid/early vulcanisation.
Antioxidants	Trimethyl quinoline (TMQ), amines, phenols, quinoline, phosphites	Prevents deterioration of rubber due to temperature or oxygen.
Antiozonants	Monocrystalline wax and chemical antiozonants. Phenylenediamines (DPPD), Paraphenylenediamines (PPDA)	Prevents weathering caused by exposure to ozone.
Peptisation agents	Zinc salt of pentachlorophenol	Accelerates the churning process (when viscosity is reduced to aid the mixture).
Softeners	Aromatic petroleum process oils (containing polycyclic aromatic hydrocarbons; PAHs), MES-oils (mild extracted solvate oil), rapeseed oil and resins.	Softens and aids the process by reducing viscosity during processing
Other process aids	Rosin, synthetic resins, process oils, pine tar, fatty acids, esters of fatty acids, low molecular weight resins.	Aids the mixing and processing.

Müller *et al*[88] compiled a list of 792 substances known to be used in car tyre formulations or previously detected in car tyre particles. The list included 163 vulcanisation accelerators, 10 vulcanisation activators, 15 vulcanisation agents, 14 vulcanisation retarders, 111 antioxidants and antiozonants, 91 polymerisation agents, 80 plasticiser, 46 other protectants, 53 decomposition products, 40 adhesion agents, 23 nitrosamine compounds, 20 reaction products, 18 mastication agents, 17 blowing agents and 91 other agents.

A.1.4. Vulcanisation and the chemicals involved

During tyre production, vulcanisation occurs after 'building' the tyre and assembling the structure. Vulcanisation is a chemical reaction discovered in the first half of the 19th century. It involves a reaction between elementary sulphur and diene rubber; the rubber is mixed with activators (inorganic oxides), accelerators (such as sulphenamides), coactivators (fatty acids) and sulphur at high temperatures. Tensile strength and resistance to weathering is increased by cross-links between long rubber molecules. Cross-links between unsaturated rubber are initiated by vulcanisation using sulphur, the most widely used vulcanising agent. Using sulphur alone causes slow reaction rates and results in the rubber being susceptible to corrosion. To counter these negative effects, an accelerator is used that becomes a part of the rubber compound in the end[24]. Zinc oxide (ZnO) is often added in vulcanisation processes as an activator. It enhances the curing process sand increase the density of cross-links between rubber composites. Quinones, amines, benzothiazoles, sulphenamides, thiurames and dithiocarbamates are among the most commonly used accelerators. Mercaptobenzothiazole (MBT), *N*-cyclohexyl-2-benzothiazole sulphonamide (CBS) and tetramethyl thiuram disulfide (TMTD) are examples of such accelerators[89, 90].

A.1.5. Emissions of TWPs

There is limited available data on TWP emissions from direct measurements of environmental compartments[75]. Emissions are either estimated based on material flow analysis MFA), applying the expected mass loss of tyres over their lifespan, or from the limited available data on TWP emissions from controlled road simulations[75, 82]. Emission factors of 30-214 mg/km per passenger car are typically reported[14, 75]. Release per km per vehicle type is reported to decrease in the order lorry > truck > bus > van > passenger car > motorcycle > moped, i.e. increasing with increasing vehicle weight[14]. Total TWP emissions from a few western European countries have been ranked in the following order: Germany > UK > Italy > Sweden > Netherlands > Norway > Denmark, with total emissions ranging from 6.7 (Denmark) to 92.6 (Germany) kTonnes/year[14].

In addition to the vehicle tyre composition and environmental temperature in which it is being used, factors such as tyre dimensions, strain, tyre pressure, wheel adjustment, speed, driving behaviour and the properties of the road surface will all influence the wear of tyres (e.g., rate, particle type emitted). Tyres with air pressures that are lower than recommended will cause internal heat development and increase the damage, while

simulations/modelling have shown that higher pressures reduce the wear. In general, the wrong pressure will decrease the lifespan of the tyre, as the tyre is worn down irregularly and thus more rapidly[77].

A.1.6. Transport and accumulation of TWPs and TWCs in the environment

Environmental monitoring data for TWPs is limited compared to other types of MPs and chemical pollutants, mainly due to analytical challenges related to quantification. Only in recent years have methods that directly identify TWPs based on spectroscopic or mass spectrometry-based identification of NR/SBR been developed and reported[79, 81, 91][25–27]. Earlier works have typically used the elemental composition fingerprint, focusing on the large content of Zn and sometimes sulphur or organic carbon, to distinguish 'TWP' contribution to particle loads[92-97]. Even today, there are challenges with the identification of TWPs in environmental matrices (waters, soils, sediments, biota), as rubber marker compounds are non-specific to vehicle tyres and are also often found in analytically challenging low concentrations[80, 98-100]. Soils located within a few meters of the roadside are the main sink of TWPs, followed by surface waters and soil at greater distances[78, 101, 102]. TWPs smaller than 100 µm (i.e. the majority of TWP emissions) are generally more prone to longer range transport, while small TWPs and mobile TWCs may also leach by rain and into groundwater and other receiving waters[83, 103-105]. TWPs make up a significant fraction of airborne particulate matter in urban areas, mostly in the smaller (<2.5 µm) fractions[93, 95, 96]. TWPs also make up a significant fraction of road dust[92, 94], making their way into street run-off[81] and into WWTPs[106]. Similarly, coastal towns and cities can represent major point sources of direct emissions of TWPs and TWCs to the marine environment, as well as their further transport within the marine ecosystem. However, there are very limited studies looking at direct TWP and TWC emissions to the marine environment, which represents a significant knowledge gap regarding their current exposure levels and potential for impacts. Both tyre tread and TWPs are denser than fresh and seawater, meaning they will mostly sink and accumulate in sediments[77]. It should be noted that the small size of many TWPs (0.05-100 µm) means that sinking rates can be quite long as friction processes become more dominant for particles in water in this size range.

A.2. Environmental risk

A.2.1. Composition of tyres

Car tyres contain a complex mixture of components ('ingredients'). The production of car tyres is a multi-step process, with a multitude of materials, chemicals and processes involved. As a basis for the tyre tread, synthetic and natural rubber, as well as a multitude of other chemicals are used. Steel for cords and bead cores, and textiles for reinforcement are also used. In addition to vulcanization agents, chemicals used as activators, accelerators, inhibitors, antioxidants, antiozonants, peptization agents, softeners and other processing aids are expected in tyres[28]. The key ingredient groups of tyres and their weight contribution to a new tyre are summarised in Table 7.2-4. Müller *et al*[29] compiled a list of 792 substances known to be used in car tyre formulations or previously detected in

car tyre particles. The list included 163 vulcanisation accelerators, 10 vulcanisation activators, 15 vulcanisation agents, 14 vulcanisation retarders, 111 antioxidants and antiozonants, 91 polymerisation agents, 80 plasticiser, 46 other protectants, 53 decomposition products, 40 adhesion agents, 23 nitrosamine compounds, 20 reaction products, 18 mastication agents, 17 blowing agents and 91 'other' chemicals. Several of these chemicals have been reported in the marine environment, even accumulating in marine organisms[30].

Table 7.2-4. Main components in a general tyre formulation[76, 77]

Group	Content	Function	Wt.%
Natural and synthetic rubbers	Poly-butadiene (BR), styrene-butadiene (SBR), neoprene-isoprene, polysulphide	Main components that give elastic properties providing force, friction and comport	40-60
Fillers	Carbon black, silicon dioxide, precipitated calcium carbonate (PCC), calcium and aluminium silicates, clay, resins	Strengthens the rubber by improving resistance to strain and wear and tear	20-35
Oils	Mineral oils	Lubrication	12-15
Reinforcement materials	Textile (rayon, nylon, polyester and aramid fibres) or metal nets.	Reinforcement to provide strength and resistance	5-10
Additives	Preservatives, antioxidants, drying agents, plasticisers, processing aids.	Various.	5-10
Vulcanisation agents	ZnO, ZnS, S, Se, Te, thiazoles, organic peroxides, nitro compounds	Transforms rubber to a solid product through vulcanisation (hardening of the rubber) by forming cross-links between polymer chains.	1-2

A.2.2. TWP/TWC/TWL exposure and hazard data collection

Peer-reviewed literature was accessed using Web of Science[31], aiming to address two topics: i) ***concentrations*** of TWP and TWC in environmental matrices, ii) environmental ***hazards*** associated with TWPs and TWCs. Formulated search strings, for ***concentration*** literature: *TWP OR tyre wear particle* OR tyre wear particle* OR tyre OR tyre AND chemical* OR associated chemical* AND/OR concentration AND/OR environment* OR freshwater OR sediment* OR biota OR air OR matrices**, for ***hazard*** literature: *tyre wear particle* OR tyre wear particle* OR TWP OR tyre wear OR tyre wear AND toxicity*) to retrieve

bibliographic data (with abstracts) from the databases. Only scientific, peer-reviewed papers or reviews written in English were selected for initial screening. Due to the quantity of papers, the collection was also limited to papers published starting 2013. After removal of duplicates, 666 (***concentration***) and 1643 (***toxicity***) papers were screened by two blind reviewers using Rayyan[32,33]. Inclusion criteria for the ***concentration*** dataset was the presence of the following words in the title or abstract: *air, soil, sediment, marine, freshwater, river*, terrestrial, biological, precipitation, fish, environmental matrices, biota, lake*, rivers, organism, lakes, bird, biological**, while exclusion criteria were clear indications that concentrations reported were not measured in environmental matrices or were no plausible link to tyre or TW emissions was demonstrated. Inclusion criteria words for the ***toxicity*** dataset were *tyre wear, tyre wear, TWP, effect, toxicity*. From the ***toxicity*** dataset, papers were excluded if they did not demonstrate direct effects data or where only human health effects were presented. This left 38 (***concentration;*** Table 7.2-6, Section 0, Annexes) and 18 (***toxicity;*** Table 7.2-7, Section A.2.8, Annexes) papers for review. During the review-stage, papers were further omitted if data presented was not of relevance to the European environment, only dealt with individual chemicals without a direct link to TW, or if full-texts were not available in English language, leaving 27 and 16 papers. Where reviews were included in the datasets, data was (where possible) retrieved from the original publications.

A.2.3. Concentrations of TWPs in the (marine) environment

Environmental monitoring data for TWPs is limited compared to other types of MPs and chemical pollutants, mainly due to analytical challenges related to quantification. Only in recent years have methods that directly identify TWPs based on spectroscopic or mass spectrometry-based identification of NR/SBR been developed and reported[25–27]. Earlier works have typically used the elemental composition fingerprint, focusing on the large content of Zn and sometimes sulphur or organic carbon, to distinguish 'TWP' contribution to particle loads[34–39]. Even today, there are challenges with the identification of TWPs in environmental matrices (waters, soils, sediments, biota), as rubber marker compounds are non-specific to vehicle tyres and are also often found in analytically challenging low concentrations[40–43]. As can be seen from the overview in Figure 7.2-1, the reviewed papers demonstrate a lack of harmonization in methodologies for quantification of TWP. In an effort to achieve harmonization, it is first necessary to agree on an unambiguous definition and characterization of TWPs – which is acknowledged as difficult[44]. Furthermore, each method and method category offers individual challenges, as recently reviewed by Khan et al[44].

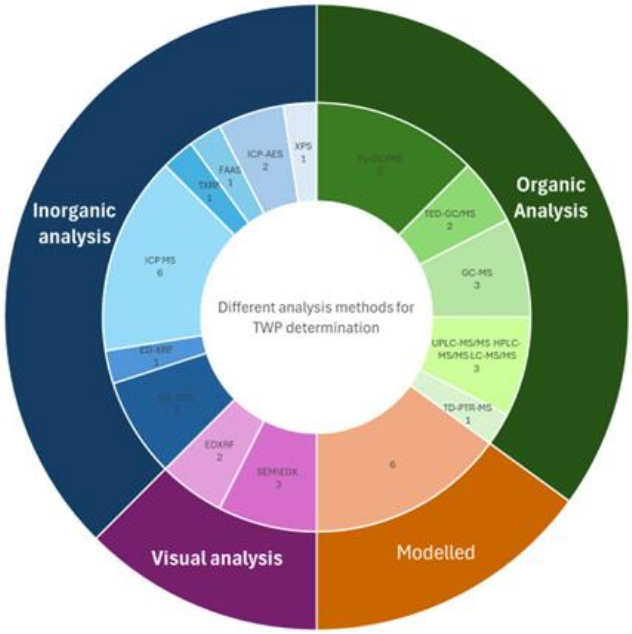

Figure 7.2-1. Overview of applied analytical techniques for the analysis of TWPs in the reviewed studies.

A total of 27 studies from the current literature review addressing ***concentrations*** were selected for supposedly reporting concentrations of TWP and/or TWC measured in environmental matrices sampled across Europe. Three of the studies were reviews and/or modelling studies. Of the 24 quantitative studies, nine attempted to quantify TWPs, ten analysed various TWC, and five analysed both. However, most (including reviews) reported concentrations in road dust, soil, air, stormwater, wash water and sweeps and collected in direct proximity to roads or urban centres and serve only to underline the generalized observations regarding spreading of TWPs being determined by size, where smaller particles tend to travel further both by air and water, while larger particles are deposited close to the source. No datapoints from the European marine environment was identified. As a curiosity, Materic et al 2022[45] report quantified nano-sized TW from a Greenland ice core (average concentration 3.2 ng/mL). Another recent review reported three aqueous environment datapoints in European matrices. However, upon inspection of the original paper, no environmental lake sediment data was actually reported in Klökner et al.[46], which was a study focused on method development and otherwise only reported data in settling and sedimentation basins. Concentrations in European river sediments found in two other papers, were also misreported in the review paper. Unice et al[47]. reported concentrations of TWP in sediments from the river Seine in the range 62-11,600 mg/kg dry weight (quantified by pyrolysis GC-MS), with higher concentrations generally found in closer vicinity to urbanized or high traffic-flow areas. In contrast, concentrations up to 0.072 mg/kg dry weight TWP were found in river sediments sampled in close vicinity to a highway or city in Germany[48].

A.2.4. Methodology for exposure assessment

Due to the lack of a pre-existing dataset for TWP exposure, we utilised a population-based model to estimate TWP production across Europe and Non-European countries sharing a regional sea. We categorised TWP production per capita into three scenarios:

- LOW: 0.8 kg/person[20]
- MEDIUM: 1.3 kg/person[49]
- HIGH: 2.5 kg/person[50]

For each TWP production level, we modelled three additional emission scenarios regarding the release of TWPs into surface waters:

- LOW: 25% reaching surface waters
- MEDIUM: 50% reaching surface waters
- HIGH: 75% reaching surface waters

We aggregated TWP emissions across catchments for each of these nine scenarios to calculate the TWP loads per basin. Finally, we normalised the TWP loads by river discharge to estimate concentrations in particles per m3 from each watershed.

The TWP concentration per catchment was thus calculated as follows for each of the following 9 scenarios:

- [TWP] = Population*0.8(low production scenario)*0.25(low emission scenario 25%)/annual discharge in m^3
- [TWP] = Population*0.8(low production scenario)*0.50(medium emission scenario 50%)/annual discharge in m^3
- [TWP] = Population*0.8(low production scenario)*0.75(high emission scenario 75%)/annual discharge in m^3
- [TWP] = Population*1.3(medium production scenario)*0.25(low emission scenario 25%)/annual discharge in m^3
- [TWP] = Population*1.3(medium production scenario)*0.50(medium emission scenario 50%)/annual discharge in m^3
- [TWP] = Population*1.3(medium production scenario)*0.75(high emission scenario 75%)/annual discharge in m^3
- [TWP] = Population*2.5(max production scenario)*0.25(low scenario emission 25%)/annual discharge in m^3
- [TWP] = Population*2.5(max production scenario)*0.50(medium scenario emission 50%)/annual discharge in m^3
- [TWP] = Population*2.5(max production scenario)*0.75(high scenario emission 75%)/annual discharge in m^3

The annual discharge per catchment: https://data.hydrosheds.org/file/technical-documentation/HydroRIVERS_TechDoc_v10.pdf

Discharge per year has been calculated as follows: DIS_AV_CMS*31,536,000 seconds.

The model outputs of the annual TWP emissions (number of TWP/year) and concentrations (TWP/m^3 per year) across the 4 European regional seas and 9 scenarios are presented in Table 7.2-5.

Table 7.2-5. Yearly TWP emissions (#TWP/year) and concentrations (#TWP/m³ per year) under different scenarios across the four regional seas of Europe.

NAME	Population	Yearly TWP emissions(2,5 scenario)	Yearly TWP emissions(1,3 scenario)	Yearly TWP emissions(0,8 scenario)	MEAN_assessment25_P25	MEAN_assessment25_P50	MEAN_assessment25_P75	MEAN_assessment13_P25	MEAN_assessment13_P50	MEAN_assessment13_P75	MEAN_assessment08_P25	MEAN_assessment08_P50	MEAN_assessment08_P75
Mediterranean	515173550.6	1287933876	669725615.7	412138840.4	0.00541439	0.01082878	0.01624317	0.002815483	0.005630966	0.008446448	0.001732605	0.00346521	0.005197814
Baltic Sea	102661527.3	256653818.3	133459985.5	82129221.87	0.000445367	0.000890734	0.001336101	0.000231591	0.000463182	0.000694773	0.000142517	0.000285035	0.000427552
Black Sea	193361425.2	483403563	251369852.8	154689140.2	0.001363156	0.002726311	0.004089467	0.000708841	0.001417682	0.002126523	0.00043621	0.00087242	0.001308629
Atlantic	260933803.6	652334508.9	339213944.6	208747042.8	0.000338973	0.000677946	0.001016919	0.000176266	0.000352532	0.000528798	0.000108471	0.000216943	0.000325414

A.2.5. Overview of TWP and TWC exposure data sources

Table 7.2-6. Overview of environmental concentration papers included after primary screening of title and abstract.

Reference	Included/Excluded	Type of article
Zhang et al 2023[51]	Excluded	-
Ding et al 2023[52]	Included	Review
Xu et al 2022[53]	Excluded	-
Thomas et al 2022[54]	Excluded	-
Rodland et al 2022[55]	Included	Method development and quantitative study
Obermaier and Pistocchi 2022[56]	Included	Review and model
Muller et al 2022[57]	Included	Method and quantitative study
Materic et al 2022[45]	Included	Quantitative study
Johannessen et al 2022[58]	Included	Semiquantitative study
Johannessen et al 2022[59]	Included	Review
Jarlskog et al 2022[60]	Included	Quantitative study
Jandacka et al 2022[61]	Included	Quantitative and qualitative study.
Gossmann et al 2022[27]	Included	Quantitative study
Gao et al 2022[40]	Included	Method and quantitative study
Youn et al 2021[62]	Excluded	-
Saifur and Gardner 2021[63]	Included	Review
Martins et al 2021[64]	Included	Quantitative study
Luo et al 2021[65]	Included	Review
Jarlskog et al 2021[66]	Included	Quantitative study
Gossmann et al 2021[48]	Included	Quantitative study
Farahani et al 2021[67]	Excluded	-
Amdal et al 2021[68]	Excluded	-
Rauert et al 2020[69]	Excluded	-
Parker et al 2020[70]	Excluded	-
Evangeliou et al 2020[71]	Included	Modelling study
Baensch-Baltruschat et al 2020[7]	Included	Review
Panko et al 2019[72]	Included	Quantitative study
Leads and Weinstein 2019[73]	Excluded	-
Klockner et al 2019[74]	Included	Method development and quantitative study
Galon-Negru et al 2019[75]	Included	Quantitative study
Zhang et al 2018[76]	Included	Quantitative study
Peter et al 2018[77]	Excluded	-
Fomba et al 2018[34]	Included	Quantitative and qualitative study.
Budai and Clement 2018[78]	Included	Quantitative study
Alves et al 2018[79]	Included	Quantitative study
Valotto et al 2015[80]	Included	Quantitative study
Panko et al 2013[81]	Included	Quantitative study
Ducret-Stich et al 2013[82]	Included	Modelling study

A.2.6. Methodology for hazard assessment

To assess the environmental risk of TWP chemicals and PFAS in the Atlantic Ocean, Mediterranean Sea, and the Black Sea, water quality benchmarks are needed for these chemicals. Such a benchmark, often termed the Predicted No Effect Concentration (PNEC), represents the concentration of a chemical expected to cause no or negligible impacts for a given ecosystem. Derivation of PNECs rely heavily on laboratory data from ecotoxicity testing of the chemicals. In ecotoxicity testing, biological variables (most often survival, reproduction, or growth) responsive to chemical exposure are measured in individual

species. Endpoints typically collected from ecotoxicity testing are LC_x (lethal concentration for x% of the test organisms), EC_x (effective concentration causing a biological response in x% of the test organisms) and NOEC (no observed effect concentration) from acute, chronic, and sub-lethal ecotoxicity tests.

By using these ecotoxicity data, PNECs can be derived by using assessment factors (AF) or Species Sensitivity Distributions (SSDs)[83–85]. These methods are used to account for the confidence in extrapolating the toxicity endpoint data to an entire ecosystem. PNEC can be derived by dividing the lowest value of ecotoxicity data by an appropriate AF, which is based on the number of taxa tested and what tests are performed (acute or chronic tests)[83]. When larger amounts of ecotoxicity data are available for multiple species, PNECs can be derived using SSDs[84]. SSD is a statistical approach that estimates either the concentration of a chemical that is toxic to no more than x% of all species (the HC_x) or the potential fraction of species affected by a given exposure concentration of a chemical[2,86]. The SSD approach is increasingly used in environmental risk assessment[2,87]. One of the main advantages of the SSD approach compared to the AF approach is that SSDs can be used to develop community-level benchmarks, while the AF approach relies solely on the data for the most sensitive species tested[87].

The environmental risk is generally considered acceptable when the PNEC is lower than the predicted environmental concentration (PEC)[83]. If the concentration of a given chemical in the environment is known (measured or modelled), a risk quotient (RQ) can be estimated by dividing the PEC with the PNEC (SSD-generated or AF generated)[83,88]. Given that different chemicals display additive toxicity, the RQs can be summarized for multiple chemicals.

For the SSD approach to provide robust PNECs for use in assessing the environmental risk of specific chemicals in specific ecosystems, there are some important aspects that need to be considered[88]. The **data sets used to perform SSDs need to be of high quality, transparent, and available**. For input data collection, the source and underlying data sets should be available. The **quality criteria** used for data filtration should be clear and transparent. Filtration of data from large ecotoxicity databases may be necessary to ensure data used are relevant in terms of endpoints and that they are comparable. The **chosen level of protection must also be defined** (the X in HC_x). Here, typically, the fifth percentile (HC_5) is used in regulatory frameworks for the purposes of water quality criteria[87].

The precision, robustness and reliability of an SSD depends on the **number and the community relevance of the species used to fit the distribution**. The **chosen endpoint should have ecological relevance**, typically acute or chronic effects on survival, growth, and reproduction. As toxicity occurs as a function of exposure time as well as exposure concentration, **exposure time** used in the ecotoxicity tests should be considered. Acute aquatic toxicity tests are generally 4 days or less, and chronic aquatic toxicity tests can be weeks to years, depending on the life span of the species and the biological response monitored[89]. An example of criteria for ecotoxicity data to be regarded as chronic for

various species is Postuma et al.,[2] where >12 h for algae and bacteria, >24 h for unicellular organisms, >48 h for crustaceans and >7 days for molluscs, worms and fish were classified as chronic ecotoxicity data.

If SSD-generated PNECs are to be used to assess a specific region or community, **ecologically relevant species should be included**. SSDs generally do not represent any specific community, but are often interpreted as if they do[88]. Using single species ecotoxicity data as input to SSDs assumes that all species included are equally ecologically relevant as they are weighted equally. Thus, the approach does not consider the functional importance of the species included nor how complex ecological interactions between species in a community may be affected by chemical exposure. If the data allows, SSDs can provide information about the risk of chemicals impacting species composition but makes no connection to underlying community and ecosystem processes[88].

Within SOS-ZEROPOL2023, we have attempted to assess the feasibility of establishing PNECs for PFAS and TWP chemicals using AFs and SSDs to be used in ecological risk assessment for these chemicals for communities inhabiting the Atlantic Ocean, Mediterranean Sea, and the Black Sea. PNECs can be derived in different manners: a deterministic approach based on the use of coefficients called assessment factors (AFs) and a statistical approach based on the so-called species sensitivity distribution (SSD). The most common method involves using assessment factors (AFs), where threshold exposure concentrations determined in laboratory tests for individual species are adjusted to apply to populations in real-world ecosystems. This is done by dividing the measured values by AFs, which vary based on the quantity and quality of available toxicity data. When sufficient ecotoxicological data is available across different species, the species sensitivity distribution (SSD) method is preferred. SSD is a cumulative probability distribution applied to a set of toxicity thresholds for individual species, based on the assumption that acceptable effect levels follow a specific distribution pattern relative to chemical concentration and that the tested species represent a random sample of the ecosystem.

To do this we i) acquired a list of relevant chemicals and ii) collected ecotoxicity data from the largest online ecotoxicity database using chemical CAS information. iii) The available data was filtered for relevance (only toxicological endpoints and seawater tests) and iv) we attempted to construct SSDs and water quality benchmarks (HC_5) were estimated. v) We discuss the relevancy of the obtained HC_5s as input to ecological risk assessment for the three regions of interest, and identify data gaps, and finally, vi) propose recommendations for a future framework for ecological risk assessment for the selected regions.

The main question asked was: **Can an SSD-based approach be used to derive water quality benchmarks for PFAS and TWP-associated chemicals in different marine regions?**

To ensure transparency of data collected for this work, we retrieved aquatic toxicity data by downloading the ECOTOXicology Knowledgebase, U.S. Environmental Protection

Agency[2] (version dated 6th March 2024) and selected ecotoxicity data of interest using CAS numbers. ECOTOX is a comprehensive database of environmental toxicity data on single chemicals for aquatic as well as terrestrial species, and the data are derived predominantly from the peer-reviewed literature. All data are openly available and downloadable from the provided link.

For SSD construction we selected only data from seawater tests because these were relevant for the marine regions selected for this work. We also focussed exclusively on the endpoint LC_{50}. The main reason for this is that EC_{50} and NOEC data are generated from tests assessing different biological variables in response to exposure, whereas LC_{50} is only assessing one (mortality). Filtering out EC_{50} data was done to avoid excessive time-consuming manual assessment and grouping of biological variables. EC_{50} data are based on any biological variable that can be measured (from molecular to ecologically relevant variables) and would need substantial curation before being used. Including EC_{50} and NOEC data will inevitably involve deeper curation of the data set, possibly involving a thorough review of all original literature where data have been collected. Doing this will enrich the datasets and improve the SSDs, particularly for algae studies where EC_{50}s are exclusively used.

Our data filtration process, particularly selecting only data from seawater tests, reduced the data available for SSD construction significantly. We further assessed how many SSDs we were able to construct as a function of number of species included (Figure 7.2-2). By including 4 species, we could construct SSDs for 11 out of the 332 chemicals (PFAS and TWCs combined) included in this survey. If the species cut-off is increased, even fewer SSDs would result.

[2] http:/www.epa.gov/ecotox/

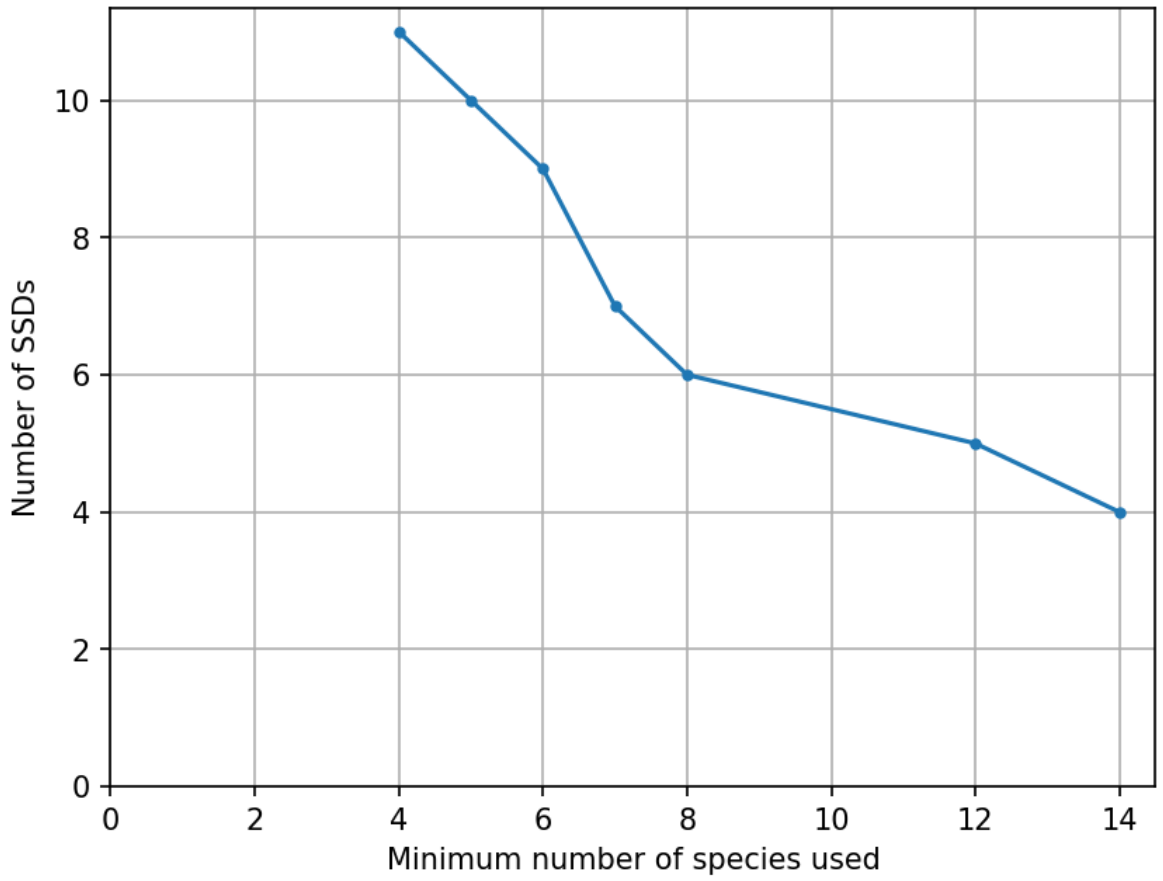

Figure 7.2-2. Number of chemicals where SSDs can be constructed as a function of number of seawater species with available LC_{50} data for any given PFAS and TWP-associated chemical.

There is a need to consider that we based the SSDs in this work on available acute toxicity data (LC_{50}), and the level of protection (HC_5) chosen can (at best) provide protection of 5% of the species against acute toxic effects. Chronic toxicity occurs at lower concentrations, and for many of the chemicals included in this survey, particularly PFAS, display other modes of action than narcosis, which is the biological variable monitored in acute toxicity tests. Thus, an HC_5 for chronic toxicity will be well below the HC_5s generated for acute toxicity here. This is probably also the reason for much higher number of EC_{50} and NOEC entries than LC_{50}s for PFAS chemicals in the ECOTOX database. It would be of value to re-assess and curate the available data for chronic effects, which probably will warrant a more robust HC5 (more data) and more protective benchmarks (ECs are lower than LCs). Conversion of acute SSDs to chronic SSDs through AFs is an option[90]. Furthermore, adding PFAS and TWC toxicity data (LC_{50}, EC_{50}, NO_{EC}) from additional sources (e.g. grey literature) and/or through modelling-based methods will enrich data and SSD coverage[2,87,90]. Section A.2.7 contains a detailed overview of the work conducted by Posthuma et al. (2019) using a large, curated dataset comprised of 30 806 records (3445 substances, 1556 different taxa, 2513 chronic NOEC values, 28 293 acute EC50 values), from which they derived SSDs for 12 386 chemicals.

From the ECOTOX database, some information about test species is available, like taxonomy. However, to identify their geographical distribution is not straight forward. Connecting the Latin names to geographical distribution can to some extent be automatically using online and freely available resources (e.g. WoRMS or OBIS databases), but this will need additional efforts to set up for defined regions like Atlantic Ocean, Mediterranean Sea, and Black Sea. It is also possible to conduct this exercise manually, and a manual survey of the WoRMS database for the 77 species where PFOA toxicity data was collected, showed that 55 were freshwater, 28 were found in seawater and 19 were found in brackish water. Of these species 18, 19 and 12 species have recorded presence in the Northern Atlantic Ocean, Mediterranean Sea, and Black Sea, respectively. Unfortunately, data filtration removed too many of these tests to do a region-specific SSD, so more data are certainly needed for region-specific SSD construction to be feasible. It is debatable if separation/filtration/generation of data to enable regional distribution will provide robust regional SSDs. Evidence of sensitivity differences between freshwater and seawater species to organic pollutants exists (e.g., (Hutchinson et al., 1998; Sverdrup et al., 2002, 2001),[91–93] however, unless serious data enrichment is achieved, it is recommended combining all available ecotoxicity data into 'global' SSDs which will have higher robustness and statistical strength.

A.2.7. Extended information about SSD approaches

Adding data from additional sources (e.g. grey literature) and/or through modelling-based methods will enrich data and SSD coverage[2,87,90]. Using a large curated data set that comprised of 30 806 records (3445 substances, 1556 different taxa, 2513 chronic NOEC values, 28 293 acute EC50 values), Posthuma and co-workers [2] derived SSDs for 12 386 chemicals. This was a curated dataset which consisted of data from a range of different sources, including ECETOX, published and grey literature, industry, regulatory agencies, and modelling approaches. Although the curated database itself is not available, the log-normal SSDs for individual chemicals (including CAS) were reported by the median and the standard deviation of log-transformed ecotoxicity data (HC50) and by a quality score. Within the list of 12386 chemicals, we managed to identify 198 chemicals from the original lists we used as a basis for our survey and estimated HC5s for acute toxicity for these chemicals. Importantly, however, the underlying ecotoxicity data used to generate these SSDs and benchmarks are not readily available and thus cannot be evaluated. The data were used to perform ecological risk assessments for freshwater recipients throughout Europe. Thus, Posthuma and coworkers have included freshwater data in their assessment. It is not explicitly stated that they included seawater ecotoxicity data as part of their SSDs, but it is highly likely that no filtration was done between freshwater and marine data was performed. In their ecological risk assessment of freshwater recipients, Posthuma and co-workers did not consider region-specific SSDs for the chemicals.

A.2.8. Overview of TWP, TWC and TWL aquatic environmental hazard data sources

Table 7.2-7. Overview of aquatic environmental hazard papers included after primary screening of title and abstract.

Reference	Type of study	Comments
Wang et al 2023[94]	Modelling study	
Wang et al 2023[95]	Modelling study	
Maji et al 2023[96]	Only single compounds – excluded	
Li et al 2023[97]	Toxicity study – leachates	No reported thresholds
Yang et al 2022[98]	Toxicity study – particles and	
Shin et al 2022[99]	Toxicity study – leachates	
Page et al 2022[100]	Toxicity study - leachates	
Liu et al 2022[101]	Toxicity study– particles and	No reported thresholds
LaPlaca et al 2022[102]	Toxicity study	No reported thresholds
Khan et al 2022[103]	Review	
Hiki and Yamamoto	Only single compounds – excluded	
Garrard et al 2022[105]	Toxicity study – particles in	No reported thresholds
Di et al 2022[106]	Only single compounds – excluded	
McIntyre et al 2021[107]	Toxicity study - leachates	No reported thresholds
Luo et al 2021[65]	Review	
Koski et al 2021[108]	Toxicity study – with and without particles	No reported thresholds
LaPlaca and van den Hurk 2020[109]	Toxicity study – with particles, but crumb rubber	No reported thresholds
Halsband et al 2020[110]	Toxicity study – crumb rubber leachates	No reported thresholds

A.2.9. Overview of hazard data available for TWCs in freshwater and seawater

Table 7.2-8. Overview of the number of data entries, number of species and taxonomic classes (based on taxonomical class) tested for individual TWCs from the ECOTOX database. The data are also separated into freshwater (FW) and seawater (SW) data.

		All			FW			SW		
CAS	**Chemical name**	**# of entries**	**# of species**	**# of taxonomic classes**	**# of entries**	**# of species**	**# of taxonomic classes**	**# of entries**	**# of species**	**# of taxonomic classes**
50-00-0	Formaldehyde	4996	542	58	1917	156	27	623	93	19
80-05-7	Bisphenol A	4586	164	32	2578	74	15	995	41	16
117-81-7	bis(2-ethylhexyl) benzene-1,2-dicarboxylate	4186	171	33	1948	79	19	490	33	9
84-74-2	Dibutyl phthalate	3481	173	34	1439	51	12	240	37	14
104-40-5	4-Nonylphenol	2713	186	25	1822	136	19	434	37	5
330-54-1	3-(3,4-Dichlorophenyl)-1,1-dimethylurea	2578	505	52	1318	196	29	585	104	22
67-68-5	Methylsulfinylmethane	2290	173	40	1388	59	16	310	50	19
115-86-6	['Triphenyl phosphate', 'hydroxy-tris(phenoxy)phosphanium']	1565	78	15	1037	28	8	8	8	2
137-26-8	dimethylcarbamothioylsulfanyl dimethylaminomethanedithioate	1387	220	34	259	40	15	28	11	8
1478-61-1	4-[1,1,1,3,3,3-hexafluoro-2-(4-hydroxyphenyl)propan-2-yl]phenol	1021	16	8	434	5	4	106	2	1
84-66-2	Diethyl phthalate	861	57	20	574	30	12	97	10	6
1163-19-5	1,2,3,4,5-pentabromo-6-(2,3,4,5,6-pentabromophenoxy)benzene	758	36	14	218	11	6	22	7	5
57-13-6	UREA	675	89	21	232	50	13	31	6	5
140-66-9	4-(2,4,4-trimethylpentan-2-yl)phenol	567	42	13	299	17	6	162	13	4
62-53-3	Aniline	504	114	28	422	82	20	9	8	6
131-11-3	Dimethyl phthalate	495	56	18	214	24	7	85	12	8
64-19-7	acetic acid	380	95	29	154	41	17	48	16	8
123-31-9	Benzene-1,4-diol	356	83	25	230	56	14	12	9	5
28553-12-0	bis(7-methyloctyl) benzene-1,2-dicarboxylate	352	23	10	243	18	6	85	4	3
88-99-3	phthalic acid	352	27	10	46	8	5	0	0	0
133-07-3	Folpet	327	73	25	132	28	11	19	9	4
107-13-1	Acrylonitrile	323	65	21	140	26	11	73	11	6
69-72-7	2-Hydroxybenzoic acid	278	49	13	134	17	5	1	1	1
107-21-1	Ethane-1,2-diol	261	48	20	208	29	15	38	13	6
144-62-7	oxalic acid	242	37	15	22	11	6	0	0	0
50-81-7	2-(1,2-dihydroxyethyl)-4,5-dihydroxyfuran-3-one	213	29	12	66	8	7	10	4	3
108-46-3	benzene-1,3-diol	213	47	21	74	31	14	10	6	5

149-30-4	3H-1,3-benzothiazole-2-thione	212	17	6	90	10	3	0	0	0
111-42-2	2-(2-Hydroxyethylamino)ethanol	198	32	16	174	26	13	16	5	5
79-10-7	Acrylic acid	193	17	13	32	11	9	4	3	2
65-85-0	Benzoic acid	177	63	18	100	34	12	2	1	1
71-36-3	1-Butanol	154	44	14	79	21	11	9	6	4
60-00-4	Ethylenedinitrilotetraacetic acid	148	37	15	52	21	10	29	6	4
1806-26-4	4-OCTYLPHENOL	141	12	5	135	10	4	4	1	1
122-39-4	Diphenylamine	131	21	12	107	13	6	4	1	1
75-56-9	2-methyloxirane	131	19	6	6	4	2	2	1	1
124-07-2	Octanoic acid	123	36	11	9	3	3	31	7	4
84-75-3	dihexyl benzene-1,2-dicarboxylate	120	18	9	84	12	6	18	3	3
131-55-5	Bis(2,4-dihydroxyphenyl)methanone	115	10	6	64	5	3	49	2	2
123-54-6	Pentane-2,4-dione	111	27	15	99	24	12	8	1	1
131-17-9	diprop-2-enyl benzene-1,2-dicarboxylate	102	16	13	62	10	7	37	4	4
110-91-8	Morpholine	99	26	10	43	17	9	5	5	1
131-56-6	(2,4-dihydroxyphenyl)-phenylmethanone	95	5	4	79	4	3	16	1	1
80-62-6	methyl 2-methylprop-2-enoate	92	16	8	90	14	7	1	1	1
64-18-6	formic acid	90	25	9	14	9	5	1	2	2
75-21-8	['Oxirane', '1-oxacycloprop-2-yne']	89	37	6	12	3	2	7	2	2
75-01-4	Chloroethylene	89	24	9	51	16	7	0	0	0
108-05-4	Vinyl acetate	89	21	11	84	15	8	3	5	4
107-15-3	['ethane-1,2-diamine', 'ethyne-1,2-diamine', 'Oxalonitrile']	88	26	11	68	22	8	5	2	2
75-35-4	1,1-Dichloroethene	88	30	9	41	7	3	18	4	3
84-61-7	dicyclohexyl benzene-1,2-dicarboxylate	87	5	4	49	2	2	0	0	0
123-30-8	4-aminophenol	85	19	8	73	18	7	0	0	0
111-46-6	2-(2-Hydroxyethoxy)ethanol	79	21	9	70	19	9	1	1	1
57-14-7	1,1-Dimethylhydrazine	78	14	7	60	11	5	6	1	1
106-89-8	2-(Chloromethyl)oxirane	75	18	8	65	15	8	3	2	1
117-84-0	dioctyl benzene-1,2-dicarboxylate	74	18	9	34	10	5	9	5	5
84-69-5	Diisobutyl phthalate	72	10	7	37	2	1	1	2	2
141-43-5	2-aminoethanol	67	19	11	60	17	9	1	1	1
92-87-5	4-(4-aminophenyl)aniline	65	30	11	55	23	7	4	1	1
2437-79-8	2,4-dichloro-1-(2,4-dichlorophenyl)benzene	59	22	13	38	11	7	10	9	7
102-71-6	['2-(bis(2-hydroxyethynyl)amino)ethynol', '2-(bis(2-hydroxyethyl)amino)ethanol']	59	22	11	49	15	7	3	3	3
97-77-8	diethylcarbamothioylsulfanyl diethylaminomethanedithioate	57	10	7	43	7	5	0	0	0
149-57-5	2-Ethylhexanoic acid	56	6	5	4	3	3	0	0	0
80-15-9	2-hydroperoxypropan-2-ylbenzene	56	14	8	55	14	8	0	0	0

109-73-9	butan-1-amine	55	19	9	52	17	8	1	1	1
126-72-7	['tris(2,3-dibromopropoxy)-hydroxyphosphanium', 'Tris(2,3-dibromopropyl) phosphate']	54	14	5	51	11	3	0	0	0
100-21-0	Terephthalic acid	51	9	6	48	6	4	0	0	0
25013-16-5	tert-butyl-4-methoxyphenol	48	16	7	17	8	3	0	0	0
86-74-8	Carbazole	45	13	6	29	11	5	9	1	1
78-32-0	tris(4-methylphenyl) phosphate	45	9	4	41	6	2	0	0	0
108-91-8	Cyclohexylamine	40	15	8	39	14	7	0	0	0
95-14-7	2H-Benzotriazole	39	18	8	17	10	5	0	0	0
128-37-0	2,6-ditert-butyl-4-methylphenol	38	13	7	11	6	3	1	1	1
583-39-1	1,3-dihydrobenzimidazole-2-thione	37	4	3	1	1	1	0	0	0
140-88-5	ethyl prop-2-enoate	35	10	7	15	5	3	1	1	1
75-31-0	propan-2-amine	31	15	5	31	15	5	0	0	0
103-23-1	Bis(2-ethylhexyl) hexanedioate	28	10	7	14	6	3	0	0	0
105-55-5	1,3-Diethylthiourea	28	7	4	14	4	2	0	0	0
110-43-0	Heptan-2-one	28	22	6	3	4	3	0	0	0
95-80-7	4-methylbenzene-1,3-diamine	26	5	4	19	4	3	0	0	0
143-07-7	Lauric acid	25	9	5	2	1	1	0	0	0
85-44-9	2-benzofuran-1,3-dione	24	8	5	16	5	3	0	0	0
75-12-7	FORMAMIDE	24	7	5	18	1	1	0	0	0
3147-75-9	2-(benzotriazol-2-yl)-4-(2,4,4-trimethylpentan-2-yl)phenol	22	1	1	22	1	1	0	0	0
110-15-6	butanedioic acid	22	18	11	9	6	4	0	4	3
2440-22-4	2-(benzotriazol-2-yl)-4-methylphenol	21	1	1	21	1	1	0	0	0
100-97-0	Methenamine	20	12	5	13	7	3	3	3	2
57-10-3	Hexadecanoic acid	20	11	7	3	3	2	0	1	1
7673-09-8	Trichloromelamine	20	5	3	12	3	2	0	0	0
105-60-2	azepan-2-one	19	7	5	10	5	4	0	0	0
91-53-2	6-ethoxy-2,2,4-trimethyl-1H-quinoline	18	7	6	5	3	3	0	0	0
85-41-6	Phthalimide	17	6	4	16	5	3	0	0	0
108-31-6	furan-2,5-dione	16	10	4	12	9	3	0	0	0
1948-33-0	2-tert-butylbenzene-1,4-diol	13	6	3	5	4	2	1	1	1
111-92-2	N-butylbutan-1-amine	12	5	4	6	4	3	0	0	0
109-46-6	1,3-Dibutylthiourea	12	4	2	12	4	2	0	0	0
1987-50-4	4-HEPTYLPHENOL	12	2	2	0	0	0	12	2	2
106-28-5	['3,7,11-trimethyldodeca-2,6,10-trien-1-ol', '(2E,6E)-3,7,11-trimethyldodeca-2,6,10-trien-1-ol']	11	7	2	0	0	0	0	0	0
77-99-6	2-ethyl-2-(hydroxymethyl)propane-1,3-diol	10	7	5	6	4	4	4	3	2
97-88-1	butyl 2-methylprop-2-enoate	9	6	4	9	6	4	0	0	0

115-77-5	2,2-bis(hydroxymethyl)propane-1,3-diol	9	5	3	8	4	3	1	1	1
110-17-8	But-2-enedioic acid	9	13	9	6	5	3	0	4	3
57-11-4	octadecanoic acid	8	5	4	3	2	1	0	0	0
79-41-4	2-methylprop-2-enoic acid	8	5	3	8	5	3	0	0	0
306-52-5	2,2,2-Trichloroethyl dihydrogen phosphate	8	5	4	7	4	3	0	0	0
111-55-7	2-acetyloxyethyl acetate	8	4	3	7	3	3	1	1	1
101-14-4	4-[(4-amino-3-chlorophenyl)methyl]-2-chloroaniline	8	2	2	0	0	0	0	0	0
101-72-4	N'-phenyl-N-propan-2-ylbenzene-1,4-diamine	7	2	2	0	0	0	0	0	0
59-89-2	4-Nitrosomorpholine	7	5	4	2	3	2	0	0	0
102-82-9	N,N-dibutylbutan-1-amine	7	4	4	7	4	4	0	0	0
103-49-1	1-phenyl-N-(phenylmethyl)methanamine	6	3	1	6	3	1	0	0	0
112-24-3	N,N'-bis(2-aminoethyl)ethane-1,2-diamine	6	3	3	4	2	2	0	0	0
141-28-6	Diethyl hexanedioate	6	1	1	6	1	1	0	0	0
2998-04-1		6	4	2	6	4	2	0	0	0
29878-31-7	4-methyl-2H-benzotriazole	6	2	2	6	2	2	0	0	0
111-40-0	N-(2-aminoethyl)ethane-1,2-diamine	6	6	3	5	5	3	1	1	1
126-99-8	2-Chlorobuta-1,3-diene	6	1	1	0	0	0	0	0	0
102-06-7	1,2-diphenylguanidine	5	3	2	5	3	2	0	0	0
101-67-7	4-Octyl-N-(4-octylphenyl)aniline	5	5	3	4	4	2	0	0	0
80-07-9		5	6	3	5	6	3	0	0	0
75-64-9	2-methylpropan-2-amine	4	3	3	4	3	3	0	0	0
84-76-4	dinonyl benzene-1,2-dicarboxylate	4	2	2	3	1	1	1	1	1
100-64-1	N-cyclohexylidenehydroxylamine	4	4	2	4	4	2	0	0	0
101-83-7	N-Cyclohexylcyclohexanamine	4	2	2	4	2	2	0	0	0
124-30-1	octadecan-1-amine	4	3	3	4	3	3	0	0	0
102-77-2	4-(1,3-benzothiazol-2-ylsulfanyl)morpholine	3	3	1	3	3	1	0	0	0
120-78-5	2-(1,3-benzothiazol-2-yldisulfanyl)-1,3-benzothiazole	3	3	1	3	3	1	0	0	0
147-47-7	2,2,4-trimethyl-1H-quinoline	3	1	1	0	0	0	0	0	0
930-55-2	1-Nitrosopyrrolidine	3	1	1	0	0	0	0	0	0
100-75-4	1-Nitrosopiperidine	3	1	1	0	0	0	0	0	0
123-79-5	Dioctyl hexanedioate	3	1	0	0	0	0	0	0	0
95-16-9	1,3-benzothiazole	2	3	2	1	1	1	0	0	0
101-54-2	N-phenylbenzene-1,4-diamine	2	1	1	2	1	1	0	0	0
2382-96-9	3H-1,3-benzoxazole-2-thione	2	2	2	0	0	0	0	0	0
131-91-9	1-nitrosonaphthalen-2-ol	2	1	1	2	1	1	0	0	0
112-61-8	Methyl octadecanoate	2	1	1	0	0	0	0	0	0
136-95-8		1	1	1	0	0	0	0	0	0

80-43-3	2-(2-phenylpropan-2-ylperoxy)propan-2-ylbenzene	1	1	1	1	1	1	0	0	0
94-36-0	benzoyl benzenecarboperoxoate	1	1	1	0	0	0	1	1	1
101-77-9	4-[(4-aminophenyl)methyl]aniline	1	2	2	0	1	1	0	0	0
104-15-4	4-Methylbenzenesulfonic acid	1	1	1	1	1	1	0	0	0
298-06-6		1	1	1	1	1	1	0	0	0
109-43-3	Dibutyl decanedioate	1	1	1	0	0	0	0	0	0
74-31-7	N,N'-di(phenyl)benzene-1,4-diamine	1	1	1	0	0	0	0	0	0
93-46-9	N,N'-di(naphthalen-2-yl)benzene-1,4-diamine	1	2	2	0	1	1	0	0	0
996-98-5	ethanedihydrazide	1	1	1	1	1	1	0	0	0
112-57-2	N'-(2-aminoethyl)-N-[2-(2-aminoethylamino)ethyl]ethane-1,2-diamine	1	1	1	0	0	0	0	0	0
106-45-6	4-Methylbenzenethiol	1	1	1	0	0	0	0	0	0
103-71-9	Isocyanatobenzene	1	1	1	0	0	0	1	1	1
615-22-5	2-methylsulfanyl-1,3-benzothiazole	0	0	0	0	0	0	0	0	0
121-00-6		0	0	0	0	0	0	0	0	0
3089-11-0	N2,N2,N4,N4,N6,N6-hexakis(methoxymethyl)-1,3,5-triazine-2,4,6-triamine	0	0	0	0	0	0	0	0	0
107-56-2	di(propan-2-yloxy)-sulfanyl-sulfanylidenephosphorane	0	1	1	0	1	1	0	0	0
97-90-5	2-(2-methylprop-2-enoyloxy)ethyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0
4979-32-2	N-(1,3-benzothiazol-2-ylsulfanyl)-N-cyclohexylcyclohexanamine	0	0	0	0	0	0	0	0	0
119-07-3		0	0	0	0	0	0	0	0	0
627-93-0	Dimethyl hexanedioate	0	1	1	0	1	1	0	0	0
3319-31-1	Tris(2-ethylhexyl) benzene-1,2,4-tricarboxylate	0	0	0	0	0	0	0	0	0
89-04-3		0	0	0	0	0	0	0	0	0
1843-05-6		0	0	0	0	0	0	0	0	0
101-25-7	3,7-dinitroso-1,3,5,7-tetrazabicyclo[3.3.1]nonane	0	0	0	0	0	0	0	0	0
1189-08-8		0	0	0	0	0	0	0	0	0
101-68-8	1-isocyanato-4-[(4-isocyanatophenyl)methyl]benzene	0	7	4	0	7	4	0	0	0
80-51-3	4-[4-(hydrazinesulfonyl)phenoxy]benzenesulfonohydrazide	0	0	0	0	0	0	0	0	0
1185-55-3	trimethoxy-methylsilane	0	0	0	0	0	0	0	0	0
2530-85-0	3-trimethoxysilylpropyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0
2530-83-8	trimethoxy-[3-(oxiran-2-ylmethoxy)propyl]silane	0	0	0	0	0	0	0	0	0
143-24-8	1-(2-methoxyethoxy)-2-[2-(2-methoxyethoxy)ethoxy]ethane	0	0	0	0	0	0	0	0	0

A.2.10. Overview of data distribution between seawater and freshwater and the three selected endpoints

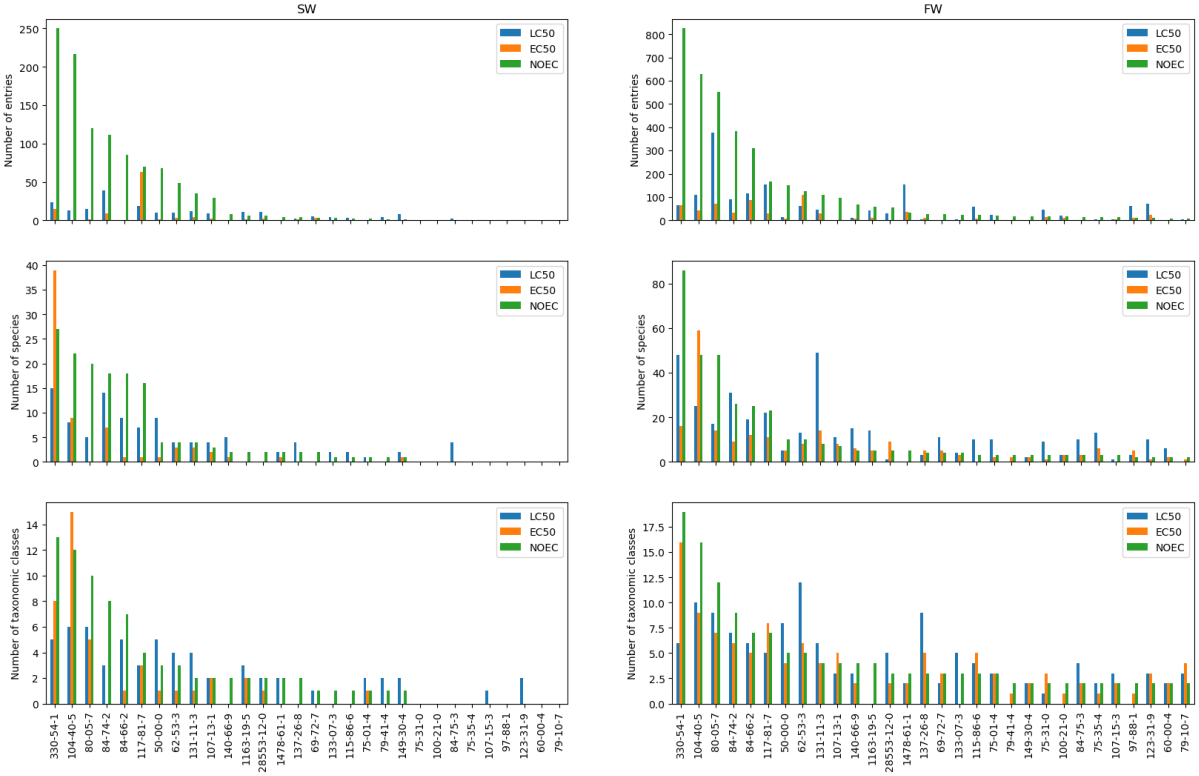

Figure 7.2-3. Number of entries of toxicity data in the ECOTOXicology Knowledgebase for seawater (left panels) and freshwater (right panels). Number of entries (top panels), number of species (middle panels) and number of taxonomic classes (bottom panels). Entries for LC50, EC50 and NOEC data are given in differently coloured bars for the 30 TWP chemicals with the highest number of NOEC entries. All data are available in Table 7.2-9.

Table 7.2-9. TWC data from the ECOTOX database filtered for toxicological relevant endpoints LC50, EC50 and NOEC distributed between seawater (SW) and freshwater (FW).

		SW									FW								
		LC_{50}			EC_{50}			NOEC			LC_{50}			EC_{50}			NOEC		
CAS	Chemical name	# of entries	# of species	# of taxonomic classes	# of entries	# of species	# of taxonomic classes	# of entries	# of species	# of taxonomic classes	# of entries	# of species	# of taxonomic classes	# of entries	# of species	# of taxonomic classes	# of entries	# of species	# of taxonomic classes
50-00-1	FORMALDEHYDE	39	14	6	9	7	5	111	18	10	154	31	8	30	9	4	168	26	5
80-05-7	Bisphenol A	23	8	5	15	9	8	250	22	13	66	17	9	64	14	7	825	48	12
330-54-1	3-(3,4-Dichlorophenyl)-1,1-dimethylurea	19	15	6	63	39	15	70	27	12	60	25	6	110	59	16	125	48	19
104-40-5	4-NONYLPHENOL	15	9	4	1	1	1	120	18	3	378	48	10	71	16	9	551	86	16
117-81-7	bis(2-ethylhexyl) benzene-1,2-dicarboxylate	13	5	3	0	0	0	217	20	8	109	22	5	42	11	8	629	23	7
67-68-5	Methylsulfinylmethane	12	4	3	4	3	3	35	4	4	10	5	3	3	3	3	0	0	0
131-11-3		11	5	4	1	1	1	6	2	2	43	14	6	12	5	4	57	5	4
84-66-2		11	4	3	2	2	2	6	3	2	46	13	6	29	8	5	110	10	7
84-74-2	Dibutyl phthalate	10	7	5	1	1	1	68	16	7	89	19	7	34	12	6	385	25	9
140-66-9	4-(2,4,4-trimethylpentan-2-yl)phenol	10	4	2	3	3	2	49	4	2	10	5	3	6	5	2	68	10	4
107-13-1	Acrylonitrile	9	9	5	2	1	1	29	4	3	44	11	3	15	5	5	16	4	4
57-13-6	UREA	8	2	2	0	0	0	1	1	1	44	12	4	4	2	2	1	1	1
133-07-3		6	3	2	0	0	0	0	0	0	14	10	5	0	0	0	5	3	3
75-21-8	['Oxirane', '1-oxacycloprop-2-yne']	6	1	1	0	0	0	0	0	0	12	3	2	0	0	0	0	0	0
111-42-2	2-(2-Hydroxyethylamino)ethanol	5	2	2	3	1	1	3	2	2	71	13	7	24	6	3	10	3	2
108-46-3	benzene-1,3-diol	5	2	1	0	0	0	0	0	0	27	13	6	2	2	1	0	0	0
122-39-4		4	1	1	0	0	0	0	0	0	8	3	2	25	5	4	1	2	2
77-99-6	2-ethyl-2-(hydroxymethyl)propane-1,3-diol	4	3	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
75-35-4	1,1-Dichloroethene	4	1	1	0	0	0	1	1	1	19	2	2	7	3	1	2	2	2
107-21-1	Ethane-1,2-diol	4	2	2	0	0	0	0	0	0	61	10	4	12	1	1	12	2	2
64-19-7	acetic acid	4	4	2	0	0	0	3	2	2	21	9	5	8	3	3	0	0	0
92-87-5	4-(4-aminophenyl)aniline	4	1	1	0	0	0	0	0	0	26	7	2	2	1	1	0	0	0
137-26-8	dimethylcarbamothioylsulfanyl dimethylaminomethanedithioate	3	2	2	3	1	1	0	0	0	58	15	9	8	6	5	22	5	3
100-97-0		3	3	2	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
71-36-3		3	2	2	0	0	0	2	1	1	22	6	3	4	2	2	1	1	1
115-86-6	['Triphenyl phosphate', 'hydroxy-tris(phenoxy)phosphanium']	3	2	2	0	0	0	0	0	0	115	11	4	88	8	5	311	7	3
28553-12-0	bis(7-methyloctyl) benzene-1,2-dicarboxylate	2	2	2	0	0	0	0	0	0	31	10	5	2	2	2	55	3	3
84-75-3	dihexyl benzene-1,2-dicarboxylate	2	2	2	1	1	1	4	1	1	23	6	4	2	2	2	21	2	2
123-31-9	Benzene-1,4-diol	2	2	2	2	1	1	0	0	0	19	10	3	9	3	3	16	3	2
131-56-6	(2,4-dihydroxyphenyl)-phenylmethanone	2	1	1	6	1	1	0	0	0	4	1	1	0	0	0	6	1	1
108-05-4		2	4	3	0	0	0	0	0	0	59	9	3	1	1	1	3	3	1
62-53-3	Aniline	1	1	1	1	1	1	0	0	0	155	49	12	35	14	6	32	8	5
84-69-5		1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
141-43-5	2-aminoethanol	1	1	1	0	0	0	0	0	0	15	7	3	9	3	2	0	0	0
84-76-4	dinonyl benzene-1,2-dicarboxylate	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1987-50-4	4-HEPTYLPHENOL	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
102-71-6	['2-(bis(2-hydroxyethynyl)amino)ethynol', '2-(bis(2-hydroxyethyl)amino)ethanol']	1	1	1	0	0	0	0	0	0	4	4	2	4	2	2	2	1	1
149-30-4	3H-1,3-benzothiazole-2-thione	0	0	0	0	0	0	0	0	0	22	4	2	5	3	2	5	4	2
95-16-9	1,3-benzothiazole	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
102-06-7	1,2-diphenylguanidine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
615-22-5	2-methylsulfanyl-1,3-benzothiazole	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
128-37-0	2,6-ditert-butyl-4-methylphenol	0	0	0	0	0	0	0	0	0	6	1	1	2	2	2	0	0	0
102-77-2	4-(1,3-benzothiazol-2-ylsulfanyl)morpholine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
120-78-5	2-(1,3-benzothiazol-2-yldisulfanyl)-1,3-benzothiazole	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

57-11-4	octadecanoic acid	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
121-00-6		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
25013-16-5		0	0	0	0	0	0	0	0	0	10	5	2	1	1	1	0	0	0
136-95-8		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
65-85-0		0	0	0	0	0	0	0	0	0	3	3	3	15	6	3	2	1	1
97-77-8	diethylcarbamothioylsulfanyl diethylaminomethanedithioate	0	0	0	0	0	0	0	0	0	5	4	2	6	3	2	2	1	1
108-91-8		0	0	0	0	0	0	0	0	0	6	4	2	3	3	2	0	0	0
117-84-0	dioctyl benzene-1,2-dicarboxylate	0	0	0	0	0	0	4	2	2	1	1	1	0	0	0	8	3	2
101-72-4	N'-phenyl-N-propan-2-ylbenzene-1,4-diamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
147-47-7	2,2,4-trimethyl-1H-quinoline	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
930-55-2	1-Nitrosopyrrolidine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
100-75-4	1-Nitrosopiperidine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
59-89-2	4-Nitrosomorpholine	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
80-43-3	2-(2-phenylpropan-2-ylperoxy)propan-2-ylbenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
94-36-0	benzoyl benzenecarboperoxoate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
79-10-7		0	0	0	0	0	0	0	0	0	3	3	3	4	4	4	2	2	2
3089-11-0	N2,N2,N4,N4,N6,N6-hexakis(methoxymethyl)-1,3,5-triazine-2,4,6-triamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
149-57-5	2-Ethylhexanoic acid	0	0	0	0	0	0	0	0	0	1	1	1	2	2	2	1	1	1
86-74-8		0	0	0	0	0	0	0	0	0	4	1	1	4	2	2	3	1	1
85-41-6		0	0	0	0	0	0	0	0	0	8	3	1	0	0	0	0	0	0
107-56-2	di(propan-2-yloxy)-sulfanyl-sulfanylidenephosphorane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
131-17-9	diprop-2-enyl benzene-1,2-dicarboxylate	0	0	0	0	0	0	2	1	1	2	2	2	8	2	2	4	1	1
84-61-7	dicyclohexyl benzene-1,2-dicarboxylate	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	14	1	1
103-23-1	Bis(2-ethylhexyl) hexanedioate	0	0	0	0	0	0	0	0	0	6	6	3	0	0	0	4	1	1
97-90-5	2-(2-methylprop-2-enoyloxy)ethyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
101-54-2	N-phenylbenzene-1,4-diamine	0	0	0	0	0	0	0	0	0	0	0	0	2	1	1	0	0	0
101-77-9	4-[(4-aminophenyl)methyl]aniline	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
79-41-4	2-methylprop-2-enoic acid	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	2	2	2
105-60-2	azepan-2-one	0	0	0	0	0	0	0	0	0	1	1	1	4	2	2	1	1	1
109-73-9	butan-1-amine	0	0	0	0	0	0	0	0	0	7	4	2	4	2	2	0	0	0
111-92-2	N-butylbutan-1-amine	0	0	0	0	0	0	0	0	0	2	1	1	1	1	1	0	0	0
103-49-1	1-phenyl-N-(phenylmethyl)methanamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
57-10-3	Hexadecanoic acid	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
104-15-4	4-Methylbenzenesulfonic acid	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
69-72-7	2-Hydroxybenzoic acid	0	0	0	0	0	0	0	0	0	4	3	2	12	5	3	27	4	3
2108798		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
108-31-6	furan-2,5-dione	0	0	0	0	0	0	0	0	0	2	2	1	3	2	2	0	0	0
85-44-9	2-benzofuran-1,3-dione	0	0	0	0	0	0	0	0	0	2	2	1	5	3	2	0	0	0
2382-96-9	3H-1,3-benzoxazole-2-thione	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
583-39-1	1,3-dihydrobenzimidazole-2-thione	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
4979-32-2	N-(1,3-benzothiazol-2-ylsulfanyl)-N-cyclohexylcyclohexanamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
105-55-5	1,3-Diethylthiourea	0	0	0	0	0	0	0	0	0	4	3	2	2	2	1	0	0	0
109-46-6	1,3-Dibutylthiourea	0	0	0	0	0	0	0	0	0	4	3	2	2	2	1	0	0	0
75-64-9	2-methylpropan-2-amine	0	0	0	0	0	0	0	0	0	2	1	1	1	1	1	0	0	0
112-24-3	N,N'-bis(2-aminoethyl)ethane-1,2-diamine	0	0	0	0	0	0	0	0	0	1	1	1	2	1	1	0	0	0
75-12-7	FORMAMIDE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
298-06-6		0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
1478-61-1	4-[1,1,1,3,3,3-hexafluoro-2-(4-hydroxyphenyl)propan-2-yl]phenol	0	0	0	0	0	0	85	2	1	13	2	2	7	2	2	151	3	3
88-99-3	phthalic acid	0	0	0	0	0	0	0	0	0	7	2	2	12	5	3	1	1	1
119-07-3		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
627-93-0	Dimethyl hexanedioate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
141-28-6	Diethyl hexanedioate	0	0	0	0	0	0	0	0	0	4	1	1	2	1	1	0	0	0
401129		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

123-79-5	Dioctyl hexanedioate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
109-43-3	Dibutyl decanedioate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3319-31-1	Tris(2-ethylhexyl) benzene-1,2,4-tricarboxylate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
89-04-3		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
306-52-5	2,2,2-Trichloroethyl dihydrogen phosphate	0	0	0	0	0	0	0	0	0	5	4	3	1	1	1	0	0	0
78-32-0	tris(4-methylphenyl) phosphate	0	0	0	0	0	0	0	0	0	4	2	1	2	1	1	0	0	0
143-07-7		0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	0	0	0
74-31-7	N,N'-di(phenyl)benzene-1,4-diamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
93-46-9	N,N'-di(naphthalen-2-yl)benzene-1,4-diamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
101-67-7	4-Octyl-N-(4-octylphenyl)aniline	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
91-53-2	6-ethoxy-2,2,4-trimethyl-1H-quinoline	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
123-30-8	4-aminophenol	0	0	0	0	0	0	0	0	0	2	2	1	5	2	2	2	1	1
95-80-7	4-methylbenzene-1,3-diamine	0	0	0	0	0	0	0	0	0	1	1	1	3	2	2	0	0	0
996-98-5	ethanedihydrazide	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
60-00-4		0	0	0	0	0	0	0	0	0	8	4	2	4	2	2	4	2	2
1163-19-5	1,2,3,4,5-pentabromo-6-(2,3,4,5,6-pentabromophenoxy)benzene	0	0	0	0	0	0	8	2	2	0	0	0	0	0	0	96	5	4
126-72-7	['tris(2,3-dibromopropoxy)-hydroxyphosphanium', 'Tris(2,3-dibromopropyl) phosphate']	0	0	0	0	0	0	0	0	0	13	2	1	14	4	2	0	0	0
131-55-5	Bis(2,4-dihydroxyphenyl)methanone	0	0	0	0	0	0	0	0	0	5	2	2	1	1	1	23	2	1
1843-05-6		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2440-22-4	2-(benzotriazol-2-yl)-4-methylphenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	17	1	1
3147-75-9	2-(benzotriazol-2-yl)-4-(2,4,4-trimethylpentan-2-yl)phenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	18	1	1
95-14-7	2H-Benzotriazole	0	0	0	0	0	0	0	0	0	6	3	3	0	0	0	0	0	0
29878-31-7	4-methyl-2H-benzotriazole	0	0	0	0	0	0	0	0	0	2	2	2	0	0	0	0	0	0
50-81-7	2-(1,2-dihydroxyethyl)-4,5-dihydroxyfuran-3-one	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
131-91-9	1-nitrosonaphthalen-2-ol	0	0	0	0	0	0	0	0	0	2	1	1	0	0	0	0	0	0
101-25-7	3,7-dinitroso-1,3,5,7-tetrazabicyclo[3.3.1]nonane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1948-33-0	2-tert-butylbenzene-1,4-diol	0	0	0	0	0	0	0	0	0	4	4	2	1	1	1	0	0	0
124-07-2		0	0	0	0	0	0	0	0	0	1	1	1	4	1	1	0	0	0
97-88-1	butyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1	2	2	2
1189-08-8		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
107-15-3	['ethane-1,2-diamine', 'ethyne-1,2-diamine', 'Oxalonitrile']	0	0	0	0	0	0	0	0	0	15	7	3	7	4	2	3	2	2
111-40-0	N-(2-aminoethyl)ethane-1,2-diamine	0	0	0	0	0	0	0	0	0	4	3	2	1	1	1	0	0	0
112-57-2	N'-(2-aminoethyl)-N-[2-(2-aminoethylamino)ethyl]ethane-1,2-diamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
111-46-6	2-(2-Hydroxyethoxy)ethanol	0	0	0	0	0	0	0	0	0	7	5	3	0	0	0	1	1	1
106-45-6	4-Methylbenzenethiol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
75-56-9		0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
106-89-8	2-(Chloromethyl)oxirane	0	0	0	0	0	0	0	0	0	36	5	2	1	1	1	0	0	0
75-01-4		0	0	0	0	0	0	0	0	0	3	3	3	3	3	3	13	3	3
126-99-8	2-Chlorobuta-1,3-diene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
140-88-5	ethyl prop-2-enoate	0	0	0	0	0	0	0	0	0	4	3	2	0	0	0	1	1	1
80-62-6	methyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0	67	7	2	1	1	1	0	0	0
101-68-8	1-isocyanato-4-[(4-isocyanatophenyl)methyl]benzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
100-21-0	Terephthalic acid	0	0	0	0	0	0	0	0	0	0	0	0	2	2	1	27	3	2
75-31-0	propan-2-amine	0	0	0	0	0	0	0	0	0	2	1	1	12	9	3	5	5	2
110-43-0	Heptan-2-one	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
110-91-8	Morpholine	0	0	0	0	0	0	0	0	0	7	5	3	3	3	2	0	0	0
100-64-1	N-cyclohexylidenehydroxylamine	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
101-83-7	N-Cyclohexylcyclohexanamine	0	0	0	0	0	0	0	0	0	0	0	0	3	2	2	0	0	0
2437-79-8	2,4-dichloro-1-(2,4-dichlorophenyl)benzene	0	0	0	0	0	0	0	0	0	5	4	2	2	1	1	0	0	0
103-71-9	Isocyanatobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
111-55-7	2-acetyloxyethyl acetate	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0

106-28-5	['3,7,11-trimethyldodeca-2,6,10-trien-1-ol', '(2E,6E)-3,7,11-trimethyldodeca-2,6,10-trien-1-ol']	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
144-62-7	oxalic acid	0	0	0	0	0	0	0	0	0	1	1	1	3	3	3	0	0	0
57-14-7	1,1-Dimethylhydrazine	0	0	0	0	0	0	0	0	0	26	8	4	6	2	2	0	0	0
80-51-3	4-[4-(hydrazinesulfonyl)phenoxy]benzenesulfonohydrazide	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
124-30-1	octadecan-1-amine	0	0	0	0	0	0	0	0	0	0	0	0	3	2	2	0	0	0
80-07-9		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
115-77-5	2,2-bis(hydroxymethyl)propane-1,3-diol	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
1806-26-4	4-OCTYLPHENOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
110-15-6	butanedioic acid	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
110-17-8	But-2-enedioic acid	0	0	0	0	0	0	0	0	0	0	0	0	2	2	2	0	0	0
1185-55-3	trimethoxy-methylsilane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2530-85-0	3-trimethoxysilylpropyl 2-methylprop-2-enoate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2530-83-8	trimethoxy-[3-(oxiran-2-ylmethoxy)propyl]silane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
123-54-6	Pentane-2,4-dione	0	0	0	6	1	1	0	0	0	42	12	6	8	4	2	0	0	0
80-15-9	2-hydroperoxypropan-2-ylbenzene	0	0	0	0	0	0	0	0	0	4	4	2	1	1	1	14	2	1
64-18-6	formic acid	0	1	1	0	0	0	0	0	0	2	2	2	4	2	2	0	0	0
101-14-4	4-[(4-amino-3-chlorophenyl)methyl]-2-chloroaniline	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
102-82-9	N,N-dibutylbutan-1-amine	0	0	0	0	0	0	0	0	0	0	0	0	3	2	2	0	0	0
143-24-8	1-(2-methoxyethoxy)-2-[2-(2-methoxyethoxy)ethoxy]ethane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
112-61-8	Methyl octadecanoate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

A.2.11. Impact of environmental parameters on TWL toxicity

Several environmental parameters, including temperature, salinity, pH, UV-exposure and turbulence are known to impact the leaching of chemicals from micronized rubber particles[3,4]. Increased temperature and duration of leaching have shown to cause increased toxicity of resulting leachates[3,111], which to some extent may be explained by increased concentrations of certain chemicals due, particularly Zn[110] and smaller/more polar toxic organic molecules[4] over time. Comparative studies have shown higher toxicity of TR leachates to freshwater than to marine species[112], and studies have also shown that leachates produced under increasing salinity have decreasing toxicity[113,114]. UV-exposure has shown to increase toxicity of resulting leachates[115], but not of leachates themselves[3,116], indicating that UV-exposing tyres or TWPs will lead to formation of UV-degradation products that leach and cause toxicity. One proposed such chemicals is 6PPD-quinone, a degradation product of 6PPD[117].

A.2.12. TWCs, not TWPs, are drivers of TW toxicity in marine organisms

Most of the literature obtained from the literature study herein did in fact investigate toxicity of TW leachates (TWL) rather than of particles themselves. Some studies focused only on individual chemical toxicity, but these were omitted from this part of the work. While there is limited available data comparing TWP and TWL toxicity, it is generally acknowledged that toxicity to aquatic organisms is driven by chemicals not particulate effects[118]. Furthermore, while evidence varies, it is commonly observed that chemical leachates of TWPs and/or chemicals attributed to TWP as origin, can be among the main contributors to toxicity from urban and road runoff[97,117]. Given the availability of data, we here focus on TWL rather than TWP toxicity.

As far as understanding the main toxicants out of the large number of chemicals possibly present in tyres and TWPs, this field of science remains in its infancy. Studies have indicated that rubber in itself is not toxic to aquatic organisms[97,119], which may be as expected due to the relatively insoluble nature of rubber, leaving the leachable organics and metals.

Many studies have attempted a toxicity identification evaluation (TIE) of TWLs, with fractionation into inorganic and organic fractions. Both fractions have proven toxic, with Zn proposed as a key toxicant in the inorganic fraction[98,119–121], and a wide variety of chemicals such as aniline[121], PAHs[99,100], benzothiazoles[110,112,122], diphenyl guanidine[122], diphenyl-phenylene-diamines (e.g. 6PPD)[121,122], organophosphate flame retardants[100], phenols and phthalates[110] identified as potentially hazard-driving substances in the organic fraction. Lately, a transformation product of the antioxidant diphenyl-phenylene-diamine 6PPD; 6PPD-quinone, has been identified as a possible main driver of toxicity towards certain salmonid species. This compound has also been proved to accumulate in blood of fish ingesting crumb rubber[123]. However, effects thresholds for 6PPD-quinone has been found to vary massively between species, even between salmonids[107,117,124,125].

There are disagreements about the main toxic fraction is the inorganics or the organics due to differing results from TIE approaches[126]. Zn was identified as the key toxicant leading to mortality of *T. japonicus* and the coexistence of benzothiazole in the leachates was proposed to have an antagonistic effect on Zn toxicity[98].

Based on the complex suite of chemicals that *may* be present in tyres and as such in TWPs, one might expect differences in resulting leachates. In comparative studies, variation between tyres[3] have been demonstrated, but taking into account the variety of chemicals[1] tyres *may* be composed of, there is not sufficient evidence to relay whether differences are on a basis of tyre type, manufacturer or even just random.

Table 7.2-10. Summary of the car tyre and TWL studies and associated toxicity data for aquatic organisms. Thresholds are given as a mass TWP/rubber per volume equivalent of the leachates (g/L).

DOI	First author	Year	Species	Latin	Life-stage	Media (seawater, freshwater, soil, sediment)	Starting material	Leachate details	Exp. time	End-point	EC50/LC50 /TC50	LOEC	NOEC
10.1016/j.envint.2005.02.001	Gualtieri	2005	Green algae	*Pseudokirchneriella subcapitata*		Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	50g/L leachate (100%, 50%,10%1%);	72h	growth inhibition	0.47		
				Pseudokirchneriella subcapitata		Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	100g/L (100%,50%, 10%,1%)	72h	growth inhibition	1.6		
				Pseudokirchneriella subcapitata		Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	100g/L with EDTA (100%,50%, 10%,1%)	72h	growth inhibition	14		
				Daphnia magna	embryo	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	50g/L leachate (100%, 50%,10%1%);	24h	death rate	29		
				Daphnia magna	embryo	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	50g/L leachate (100%, 50%,10%1%);	48h	death rate	27		
				Daphnia magna	embryo	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	100g/L (100%,50%, 10%,1%)	48h	death rate	53		
				Xenopus laevis	embryo-larval	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	50g/L leachate (100%, 50%,10%1%);	120h	Mortality	40		
				Xenopus laevis	embryo-larval	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	100g/L (100%,50%, 10%,1%)		Mortality	27		
				Xenopus laevis	embryo-larval	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	50g/L leachate (100%, 50%,10%1%);		Malformations	19		
-				*Xenopus laevis*	embryo-larval	Ultrapure water at pH 3 (simulated acid rain)	TRWP from simulator	100g/L (100%,50%, 10%,1%)		Malformations	71		
10.1016/j.chemosphere.2009.08.034	Wik	2009		*Pseudokirchneriella subcapitata*		Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Growth inhibition	0.05		
				Pseudokirchneriella subcapitata		Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Growth inhibition	0.57		
				Pseudokirchneriella subcapitata		Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Growth inhibition	0.77		
				Danio rerio		Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Lethality	0.55		

				Danio rerio	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Lethality	-
				Danio rerio	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Lethality	-
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	0.55
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	0.55
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	0.84
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	9d	Survival	0.05
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	9d	Survival	0.07
				Ceriodaphnia dubia	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	9d	Survival	0.11
				Daphnia magna	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	0.67
				Daphnia magna	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	3.1
-				*Daphnia magna*	Freshwater	Abraded tyre particles	0.01-10 g rubber/L MQ, 5 days, filtered	48h	Immobilization	1.2
10.1007/s10646-011-0750-x	Marwood	2010		*Pseudokirchneriella subcapitata*	Freshwater	TRWP from simulator	0.1-10 g/L sediment/freshwater concentrations, mixed 24 h, centrifuged	72h	Growth inhibition	-
				Daphnia magna	Freshwater	TRWP from simulator	0.1-10 g/L sediment/freshwater concentrations, mixed 24 h, centrifuged	48h	Immobilization	-
			Fathead minnow	Pimephales promelas	Freshwater	TRWP from simulator	0.1-10 g/L sediment/freshwater concentrations, mixed 24 h, centrifuged	96h	Survival	-
				Daphnia magna	Freshwater	TRWP from simulator	10 g/L TRWP, 72h at 44C, filtered, diluted	48h	Immobilization	4.36
				Daphnia magna	Freshwater	TRWP from simulator	10 g/L TRWP, 72h at 21C, filtered, diluted	48h	Immobilization	-

			Daphnia magna		Freshwater	TRWP from simulator	10 g/L TRWP, 72h at 44C, filtered, mixed with sediment 24 h, diluted	48h	Immobilization	5.08
-			*Daphnia magna*		Freshwater	TRWP from simulator	10 g/L TRWP, 72h at 21C, filtered, mixed with sediment 24 h, diluted	48h	Immobilization	-
10.1016/j.watres.2019.115270	Capolupo	2020	*Pseudokirchneriella subcapitata*		Freshwater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	72h	Growth inhibition	15.2
			Skeletonema costatum		Seawater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	72h	Growth inhibition	0.4
			M. galloprovincialis	gametes	Seawater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	? 30-60mins?	Gamete fertilization	29.1
			M. galloprovincialis	embryo-larval	Seawater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	48h	Development	1.776
			M. galloprovincialis	larvae	Seawater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	48h	Motility	15
-			*M. galloprovincialis*	larvae	Seawater	ELT crumb rubber	80 g/L CTR, 14 days at 25C, filtered, diluted	144h	Survival	47.5
10.1016/j.envint.2005.02.001	Gualtieri	2005	*Raphidocelis subcapitata*		Freshwater	Abraded tyre particles	100 g/L leachate, 24h, diluted	72h	Growth inhibition	1.64
			Raphidocelis subcapitata		Freshwater	Abraded tyre particles	50 g/L leachate, 24 h, diluted	72h	Growth inhibition	0.465
			D. magna		Freshwater	Abraded tyre particles	100 g/L leachate, 24h, diluted	48h	Survival	53.3
			D. magna		Freshwater	Abraded tyre particles	50 g/L leachate, 24 h, diluted	24h	Survival	29.15
-			*D. magna*		Freshwater	Abraded tyre particles	50 g/L leachate, 24 h, diluted	48h	Survival	26.75

10.1577/1548-8659(1998)127%3C0796:TOSTLI%3E2.0.CO;2	Hartwell	1998		sheepshead minnows *Cyprinodon variegatus*	larval	Saline water, 5%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Mortality	5	6.5	3
				sheepshead minnows *Cyprinodon variegatus*	larval	Saline water, 15%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Mortality	13	**12.5**	**6.5**
				daggerblade grass shrimp *Palaemonetes pugio*		Saline water, 5%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Mortality	31.5	25	12.5
				sheepshead minnows *Cyprinodon variegatus*	larval	Saline water, 5%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Growth		6.5	3
				sheepshead minnows *Cyprinodon variegatus*	larval	Saline water, 15%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Growth		6.5	3
-				daggerblade grass shrimp *Palaemonetes pugio*		Saline water, 5%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted	96h	Growth		25	12.5
10.1023/A:1005282201554	Hartwell	2000	Microtox	*Allivibrio fischeri*		Deionized water	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted				*12.5*	
			Microtox	*Allivibrio fischeri*		Saline water, 5%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted				*44*	
			Microtox	*Allivibrio fischeri*		Saline water, 15%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted				*60*	
			Microtox	*Allivibrio fischeri*		Saline water, 20%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted				*58*	
-			Microtox	*Allivibrio fischeri*		Saline water, 25%	Tyre chips/particles	50 g/L leachate, 7d, filtered and settled+decanted				*56*	
10.1016/j.chemosphere.2004.08.103	Wik	2005	Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	5.26		

Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.69
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	1.92
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.31
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	6.88
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	1.16
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	5.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	2.41
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	4.42
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.58
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	32
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	1.71
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	3.51
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.58
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	32
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	2.17
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	0.51

			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.125
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	0.29
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.0625
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	0.32
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.125
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	24h	Immobilization	0.71
-			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Grated tyre pieces	0.26-16 g/L, 72h, 20C,	48h	Immobilization	0.125
10.1016/j.chemosphere.2005.12.045	Wik	2006	Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	3.2
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	0.7
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	1.2
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	0.4
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	2.9
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.6
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	10.3
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.9
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	4.2

Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.7
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.9
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.8
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	8.6
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.6
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	8.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.6
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	5.7
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.8
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.3
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	5.8

Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.8
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	5.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	9.2
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.1
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.5
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	4.9
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	7
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	3.4
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	3.5
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.6
Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10

			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	2.9
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	3.6
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	1.7
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	3.2
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	6.5
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	24h	Immobilization	>10
			Planktonic crustacean	*Daphnia magna*	neonates	Freshwater	Abraded tyre particles	10 g/L, 44C, 72h, filtered, diluted	48h	Immobilization	>10
10.1016/j.chemosphere.2022.134099	Yang	2022	Copepod	*Tigriopus japonicus*	adult	Filtered artificial seawater (FASW)	Scrap tyres, crushed and filtered < 78 µm	10 g/L, stirred for 60 days, room temperature, filtered, diluted	96h	Survival	5.34
								With particles			0.7714
10.1016/j.jhazmat.2022.129417	Shin	2022	Rotifer	*Brachionus plicatilis*	Adult	Filtered artificial seawater	TRWP from simulator	1 g/L, 42C, 72h, filtered, diluted	24h	Mortality	0.601
10.1016/j.aquatox.2022.106299	Page	2022		*Rhodomonas salina*		Artificial seawater	Unused winter car tyre	1 g TWP/L, rotate at 1 rpm, 72h, 20C, filtered	72h	growth inhibition	0.64
				Thalassiosira weissflogii							0.73
				Heterocapsa steinii							0.23

10.1016/j.jhazmat.2024.134401	Du-carree	2024	Microalgae	Rhodomonas salina	Artificial seawater	Cryomilled tread from unused tyre	1 g TWP/L, rotate at 20 rpm, 72h, 21C, filtered	24 h	growth inhibition	0.4843
-							1 g TWP/L, rotate at 20 rpm, 72h, 21C, filtered, SPE HLB	24 h	growth inhibition	0.4682
10.1016/j.aquatox.2021.105769	Halle	2021		Hyalella azteca		Pristine TWP				0.19
-						Worn TWP				0.91

A.3. Existing value chain and technological actions and strategies

A.3.1. Relevance, approach and challenges

The main question to be addressed, by providing factual information, is the following: "What effects on pollution can be expected from which interventions?" To address this question, it is necessary to have information about many aspects. In the task covered by this report, the product chain and the potential technological actions for mitigating the pollution are put central.

A.3.2. Relevance of investigating the '*product chain*'

In the project proposal this aspect was called a value chain, a term often observed in reports such as this one. A value chain typically identifies where and how much value is added to a material or product in consecutive steps from raw material to the door of the consumer. This concept is very useful to understand the relations between companies in a sector and their interdependence. In the SOS-ZEROPOL project, however, economic aspects do not play a key role in the analysis while product and emission volumes are of high importance. Furthermore, usually a value chain is analysed from the viewpoint of a consumable end product (type), while the focus in this project is either on an intermediate chemical (in the case of PFAS) or on an end-of-life material (in the case of TWPs). To accommodate for this different focus, it was decided to refer to a 'product chain' instead of a value chain.

A.3.3. Relevance of investigating '*mitigating actions and alternatives*'

To support the discussions in the Living Labs, the scenarios that are built based on the outcomes of these discussions, as well as evaluations in other work packages, it is important to have a trustworthy and extensive overview of technological actions that can be taken to mitigate emissions of, specific for this report, PFAS. In the integrated assessment template, the measures were split between those that can currently be taken and those that will become available in the future, mostly for practical use in the discussions with stakeholders.

A.3.4. Challenges in this work

The research in this work package is based mostly on literature review (scientific articles and patents but also producer's websites and popular science). For some parts, semi-structured interviews with experts were performed to obtain the required information.

Diffuse emissions represent much higher volumes than point source emissions for TWPs. Even small quantities of the (persistent) TWP chemical substances may have adverse health effects, therefore low concentrations of the chemicals that have been spread widely throughout the environment form a serious issue.

Evaluating technological measures that can be taken to reduce emissions and thereby decrease future pollution can be done from a broad technical perspective, when using literature as the main information source. In practice, the feasibility to integrate technical

emission reduction measures into existing equipment, business processes, and policy will be different for each specific case. Many of the technical measures have not yet been developed at commercial scale for TWPs. It is very challenging to predict if and which emission mitigation technology will finally prove technically and commercially successful in the future.

A.4. Mapping current governance strategies/efforts/arrangements

A.4.1. Background information

A quick scan of governance efforts at the regional and level within these three regions showed that limited governance efforts exist at the regional and national levels. Governance arrangements that address TWPs within these regions are not yet institutionalised, also because much relevant EU legislation is currently being revised, meaning that the transposition and implementation of EU legislation has not yet begun. In response, we paid more attention to identifying overlap and disconnects between current EU regulatory developments around zero pollution of TWPs and regional, national and industry governance efforts to address production, use and end-of-pipe of TWPs in the Mediterranean Sea.

A.4.2. Methodology

Through a range of personal communications and semi-structured interviews with expert stakeholders, we gained a more in-depth understanding of the relations between the national-, regional- and EU-level regulatory developments, as well as which discourses and power relations influence the shaping of current regional and national regulation and policies related to the selected pollutant.

A.4.3. Scoping three regional case studies

The case study pollutant selection by regional sea was aligned with that of the living labs to optimise the utility of findings in both directions and to create the possibility of data triangulation. The case study pollutants for each regional sea were determined by the SOS-ZEROPOL2030 project team at the consortium meeting in Potsdam in October 2023. Criteria included the relevance to the overall project objectives, and the level of public awareness of the pollutant in the region.

The scoping in terms of a sectoral and/or life cycle stage focus was finalised at a meeting in Wageningen in January 2024 with project partners present from WP3 and WP4. Criteria for narrowing down on a sector included:

- Salience - covering a significant source of pollution emissions
- Geographical - having a tangible link to the region
- Life cycle stages - addressing interaction along the product chain
- Policy relevance - link to policy developments
- Living labs - the possibility for a constructive dialogue in the Living Labs

- Project relevance - the potential broader applicability of lessons from the case study to other pollutants

The medical sector in the Northeast Atlantic was selected because it would touch on policy debates of 'essential use', has a clear connection to the region and is specific enough to allow stakeholders to make the step from challenges to criteria for change to occur in the living labs. For the Black Sea the awareness level of PFAS is still very low, which makes a sectoral approach difficult. That is why we chose to focus on public administration and the process of implementing revisions of EU directives with a focus on monitoring and drinking water. For TWP pollution, there are two stages in the life cycle where significant gains can be made: Production of tyres (including the chemical ingredients) and the end-of-pipe collection through improved wastewater treatment facilities. These were both chosen to be able to cover the pollutant life cycle.

The preparatory work resulted in three regional case studies (Table 7.2-11) which cover not only the distinct geographical areas of the SOS-ZEROPOL2030 project and consortium, but also various thematic focuses and sectors as demonstrated in Table 7.2-11.

Table 7.2-11. Overview of the scoping results of the three regional case studies

Region in Focus	Pollutant	Regional Convention	National focuses	Sector
Northeast Atlantic Sea	PFAS	OSPAR Convention	Netherlands & France	Medical
Black Sea	PFAS	Bucharest Convention	Bulgaria & Romania	Public Administration (monitoring & drinking water)
Mediterranean Sea	TWP	Barcelona Convention	France & Italy	Tyre production and WWT

A.4.4. Data collection and analysis

The work involved a comprehensive overview of the academic literature on TWP in the Mediterranean Sea region. The desk study, along with the review work conducted in WP2, allowed for the foundation of the existing research on the governance of TWP in Europe (if any). Scholarly databases such as Web of Science and Google Scholar were queried using key works relevant to the research focus (i.e., Govern, Policy, Tyre, Tire, Pollution, North-Mediterranean). In addition, a search of grey literature, including policy documents, news articles, and industry press releases allowed for another layer of verification as to the state of knowledge.

To gain further insight into developments in the Mediterranean Sea region, we contacted various expert stakeholders (Table 7.2-12). Especially in the Mediterranean Sea region, the lack of governance efforts for TWPs resulted in difficulties in identifying relevant interviewees or information on the topic. As a result, we either had informal interviews or email-based correspondence where we put several concrete questions to the expert and therefore no audio recording was needed. We refer to this as 'personal communication' in Table 7.2-12.

We did two semi-structured interviews which were recorded for internal note taking purposes. All interviewees consented to take part in the interview and were provided full GDPR compliance information on the project before taking part.

The sampling of respondents for personal communications or semi-structured interviews was purposeful based, i.e. based on the stakeholder (organisation's) expertise, knowledge available on the topic, and region the interviewee was focused on. We also applied the snowball sampling method – that is gaining additional key contacts via previously conducted interviews.

Table 7.2-12 Organisations interviewed or contacted in relation to the governance of TWP in the Mediterranean Sea region.

Organisation contacted	Date	Type of contact
MedWaves	11.03.2024	Personal communication
UNEP/MAP	22.04.2024	Semi-structured interview
	26.04.2024	Semi-structured interview
University of Barcelona	20.03.2024	Personal communication
PlanBleu	14.03.2024	Personal communication
Zero Waste France	09.04.2024	Personal communication
ARPA Sicily	08.04.2024	Contacted, no response

B. List of abbreviations (in chronological order)

SO	SINTEF Ocean
WR	Stichting Wageningen Research
VLIZ	Vlaams Instituut voor de Zee
MIO-ECSDE	Mediterranean Information Office for the Environment, Culture and Sustainable Development
UCC	University College Cork
RIFS	Research Institute for Sustainability
GRIDA	GRID Arendal
WU	Wageningen University
NTNU	Norwegian University of Science and Technology
SOS-ZEROPOL2030	Source to Seas - Zero Pollution 2030
PFAS	Per- and polyfluoroalkyl substances
TWPs	Tyre wear particles
TWCs	Tyre wear chemicals
NIAS	Non-intentionally added substances
TWLs	Tyre wear leachates
EU	European Union
MPs	Microplastics
CEDR	Conference of European Directors of Roads
WWTP	Wastewater treatment plant
SSD	Species sensitivity distribution
LCx	Lethal concentration for x% of the test organisms
ECx	Effective concentration causing a biological response in x% of the test organisms
NOEC	No observed effect concentration
HCx	Hazardous concentration for x% of the test species
TIE	Toxicity identification evaluation
TRWPs	Tyre and road wear particles
PM2.5	Particulate matter below 2.5 µm
PM10	Particulate matter below 10 µm
UNECE	United Nations Economic Commission for Europe
UWWTD	Urban Wastewater Treatment Directive
WFD	Water Framework Directive
UN	United Nations
ETRMA	European Tyre and Rubber Manufacturers Association
ETRTO	European Tyre and Rim Technical Organisation
ACEA	Automobile Manufacturers Association
EUCAR	European Council for Automotive R&D
SBR	Styrene-butadiene rubber
BR	Butadiene rubber
NR	Natural rubber

CBS	N-cyclohexyl-2-benzothiazolsulphenamide
ZnO	Zinc oxide
TMQ	Trimethyl quinoline
DPPD	Phenylenediamines
PPDA	Paraphenylenediamines
PAHs	Polycyclic aromatic hydrocarbons
MES-oils	Mild extracted solvate oils
MBT	Mercaptobenzothiazole
TMTD	Tetramethyl thiuram disulfide
PCC	Precipitated calcium carbonate
PNEC	Predicted no-effect concentration
AF	Assessment factor
PEC	Predicted environmental concentration
RQ	Risk quotient
CAS	Chemical Abstracts Service

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