

RESEARCH

Open Access



Risk-based management framework for microplastics in aquatic ecosystems

Alvine C. Mehinto^{1*}, Scott Coffin², Albert A. Koelmans³, Susanne M. Brander⁴, Martin Wagner⁵, Leah M. Thornton Hampton¹, Allen G. Burton Jr⁶, Ezra Miller⁷, Todd Gouin⁸, Stephen B. Weisberg¹ and Chelsea M. Rochman⁹

Abstract

Microplastic particles (MPs) are ubiquitous across a wide range of aquatic habitats but determining an appropriate level of risk management is hindered by a poor understanding of environmental risk. Here, we introduce a risk management framework for aquatic ecosystems that identifies four critical management thresholds, ranging from low regulatory concern to the highest level of concern where pollution control measures could be introduced to mitigate environmental emissions. The four thresholds were derived using a species sensitivity distribution (SSD) approach and the best available data from the peer-reviewed literature. This included a total of 290 data points extracted from 21 peer-reviewed microplastic toxicity studies meeting a minimal set of pre-defined quality criteria. The meta-analysis resulted in the development of critical thresholds for two effects mechanisms: food dilution with thresholds ranging from ~0.5 to 35 particles/L, and tissue translocation with thresholds ranging from ~60 to 4100 particles/L. This project was completed within an expert working group, which assigned high confidence to the management framework and associated analytical approach for developing thresholds, and very low to high confidence in the numerical thresholds. Consequently, several research recommendations are presented, which would strengthen confidence in quantifying threshold values for use in risk assessment and management. These recommendations include a need for high quality toxicity tests, and for an improved understanding of the mechanisms of action to better establish links to ecologically relevant adverse effects.

Keywords Microplastics, Ecotoxicology, Risk thresholds, Risk management, Aquatic health

*Correspondence:

Alvine C. Mehinto
alvinam@sccwrp.org

¹ Southern California Coastal Water Research Project Authority, 3535 Harbor Blvd, Costa Mesa, CA 92626, USA

² California State Water Resources Control Board, Sacramento, California, USA

³ Aquatic Ecology and Water Quality Management Group, Wageningen University, Wageningen, The Netherlands

⁴ Department Fisheries, Wildlife, and Conservation Sciences, Coastal Oregon Marine Experiment Station, Oregon State University, Newport, Oregon, USA

⁵ Department of Biology, Norwegian University of Science and Technology (NTNU), Trondheim, Norway

⁶ School for Environment and Sustainability, University of Michigan, Ann Arbor, MI 48109, USA

⁷ San Francisco Estuary Institute, Richmond, California, USA

⁸ TG Environmental Research, Sharnbrook, Bedfordshire, UK

⁹ Department of Ecology and Evolutionary Biology, University of Toronto, Toronto, Ontario, Canada



© The Author(s) 2022. **Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

Introduction

Microplastic particles (MPs; <5000µm) are pervasive in the aquatic environment, including ocean surface waters [54], deep ocean trenches [1], wetlands [36], lakes [13, 14] and the Arctic [20]. Exposure to MPs has been associated with various types of biological responses, including disruption of feeding [61], decreases in growth [48], tissue inflammation [43], changes in gene expression [42, 66], and decreases in reproductive success [55]. The observed adverse effects have resulted in increasing concern regarding the environmental risks of MPs, thus requiring the development and application of appropriate risk assessment and management tools [21].

In order to appropriately inform MPs management decisions, there is a need to establish critical threshold values at which adverse effects are likely to occur (Coffin and Weisberg, [11]). However, the development and application of such risk management framework is currently limited because MPs are a diverse class of pollutants with varying properties (i.e., different sizes, shapes, polymer types, and chemical additives) that can influence the toxicity outcome in aquatic organisms [37, 49]. For instance, several laboratory-based studies have documented chemical effects of MPs from additives or other sorbed contaminants [17, 31, 38, 70]. Others have reported physical effects of MPs as a result of food dilution where the volume of ingested particles creates a false sense of satiation and reduces nutrient intake [12, 47, 53], or as a result of translocation of MPs across epithelial

barriers, leading to tissue inflammation and oxidative stress [39, 64].

Further complicating the development of thresholds are the discrepancies between laboratory-based exposures and environmentally relevant exposure scenarios. Most toxicity studies are using monodispersed MPs (e.g., single-sized virgin beads or fragments of one polymer type) [21, 32], while ambient exposures are to a mixture of particle sizes, shapes and chemistries [7], including a high proportion of microfibers [4], and other chemical contaminants from different parallel exposure pathways. Moreover, accurate estimates of environmental concentrations for smaller MPs (<300µm) remain sparse [9], and particles <1µm can be difficult to detect in environmental samples [50]. However, recent studies have proposed an alignment strategy to extrapolate laboratory results of varying levels of polydispersity and facilitate comparisons with the polydispersity of MPs in the environment [33, 34]. This represents a promising advancement to improve environmental risk assessments for MPs.

Here, we report the outcome of an expert workshop that aimed to develop and apply a risk management framework for MPs in aquatic ecosystems. The framework was populated using a meta-analysis of the peer-reviewed literature to estimate threshold values for two different effect mechanisms, food dilution and translocation. The workshop addresses the State of California legislative mandates for enhanced MPs management [63]

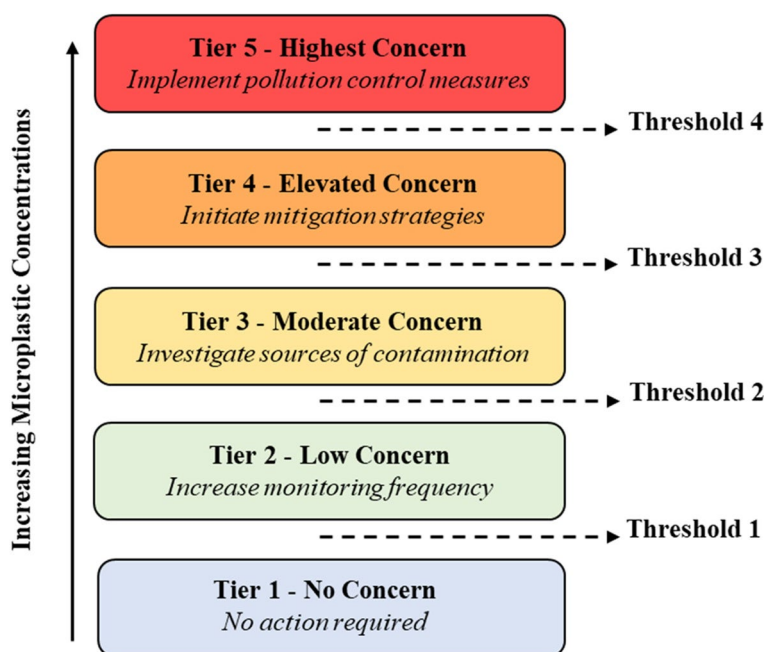


Fig. 1 Proposed tiered management framework to implement health-based thresholds for microplastics

and advances risk assessment analysis by introducing a tiered management framework that defines four thresholds based on varying levels of MPs in surface waters and corresponding to different levels of concern and management responses. Finally, the level of confidence in the framework and each of the threshold values was evaluated along with research needs to improve confidence in these thresholds in future iterations.

Proposed microplastics management framework

The proposed framework developed for water discharge management consists of five levels of risk management concern, with four hazard thresholds based on different levels of confidence that MPs can cause an adverse effect to aquatic life (Fig. 1). Threshold 1 defines the level below which managers should be confident in the absence of biological effects and is limited to decisions related to monitoring needs. In contrast, Threshold 4 corresponds to a level of concern where managers can be more confident that ecosystem level effects manifest. Thus, observations of MPs in aquatic systems in exceedance of Threshold 4 would require the implementation of stringent and potentially expensive management actions, such as limiting uses of a waterbody (e.g., fishing restrictions) or source controls (e.g., regulation of local point sources). Workshop participants recommended that Threshold 1 be implemented to establish a “protective” level below which environmental risks are low, minimizing Type I errors. The more “predictive” Threshold 4 is based on high quality and reliable toxicity data reporting significant effects of MPs exposures. This minimizes Type II error to ensure that the associated management actions are appropriately justified. Thresholds 2 and 3 represent a gradation in management decisions and associated thresholds between Thresholds 1 and 4. Although the development of a risk management framework fills a critical gap to address MP pollution in aquatic environments, other precautionary measures to reduce sources of MPs (e.g., product bans) should be conducted in parallel for effective MPs management.

Methods

Approach to derive health-based thresholds

The approach developed and applied here to calculate the four thresholds is based on the use of a species sensitivity distribution (SSD) [3, 51, 59]. This probabilistic approach, often used to develop water quality criteria, summarizes ecotoxicological data for various species and taxa in order to compare interspecies sensitivity to a specific contaminant [16, 44]. Because they integrate a large set of toxicity data for all species exposed to a contaminant of interest, SSDs and the derivation of hazard concentration (HC) levels protective of aquatic communities are considered a pragmatic approach that is not without limitations but the best available tool so far [16].

To establish threshold values reflective of the proposed “protective-to-predictive” tiered-management framework, workshop participants agreed on a suite of SSD parameters representing an increased level of confidence in ecologically relevant effects (Table 1). Parameters include the use of an appropriate data collapsing method, a percentage of species affected (HC), model point estimate, and the level of biological organization of the endpoints considered. To estimate the lower thresholds values (#1 and #2), experts recommended the inclusion of endpoints at all levels of biological organization and use of the 1st quartile to summarize the data for each species. Threshold 1 (Investigative monitoring) is based on the lower 95% confidence level of the HC5, while Threshold 2 (Discharge monitoring) is based on the median HC5. For the higher “predictive” thresholds, (i.e., Threshold 3 and 4), the recommended SSD parameters include the use of organismal and population level endpoints only, summarized using the median data collapsing method. Threshold 3 (Management planning) is set at the median HC5 and Threshold 4 (Source control measures) is set at the median HC10.

Analyses to populate the thresholds

Effect-concentration data were collected using the Toxicity of Microplastics Explorer (ToMEx) database, built to gather peer-reviewed literature on the biological effects of different shapes and polymer types of MPs ranging

Table 1 Species sensitivity distribution (SSD) parameters and data filters used to derive multiple thresholds as part of the tiered framework

Threshold description	Hazard concentration (HC)	Data collapsing method	Point estimate	Biological scale of endpoints
Threshold 1- “Investigative monitoring”	HC5	1st quartile	lower 95%	molecular to population
Threshold 2- “Discharge monitoring”	HC5	1st quartile	median	molecular to population
Threshold 3- “Management planning”	HC5	median	median	organismal and population
Threshold 4- “Source control measures”	HC10	median	median	organismal and population

between 0.001–5000 μm (Hampton et al. A, [23]). Due to uncertainties in environmental distributions for MPs smaller than 1 μm , only toxicity data between 1 and 5000 μm were included in our analysis. Each study was evaluated against a pre-defined set of 14 quality criteria for experimental set up, particle characterization and dose-response. Our criteria were adapted from those described in de Ruijter et al. [12], with some modifications (see Table S1A for more details). Out of the 162 peer-reviewed toxicity studies representing 5871 datapoints and 109 species included in the database, only 21 studies met our pre-defined set of quality criteria. A total of 290 datapoints was extracted using the following dose descriptors: no observed effect concentration (NOEC), lowest observed effect concentration (LOEC), lethal effect concentration (LCx), and percent effect concentration (ECx). Highest observed no effect concentrations (HONEC) were excluded from our analyses due to their limited reliability [2].

SSDs were constructed using the SSDTools package in R as described by Thorley and Schwarz [56]. The analyses were performed using NOECs for chronic exposures. To do so, effect metrics (i.e., LOECs and EC50/LC50) values were converted using assessment factors of 2 and 10, respectively. For the conversion of acute to chronic data, an assessment factor of 10 was applied [62]. The basis for the assessment factors applied is provided in Tables S1B and S1C. Raw data used in the SSDs are provided in Table S1D. Separate SSDs were developed for two different hypothesized mechanisms of toxicity. To identify the dataset relevant to food dilution, species-dependent ingestible size ranges based on mouth opening were used as the upper limits. Studies using algal species were excluded as ingestion is not a plausible mechanism. For tissue translocation, preliminary results of a binomial logistic regression model (using 27 studies for 19 species) suggested that particles shorter than 83 μm were most relevant to trigger this effect mechanism (supplemental information S2).

Prior to incorporation of data into the SSD, data were aligned and rescaled to 1–5000 μm for two size-related dose metrics, volume for food dilution and surface area for tissue translocation, following the methods by Koelmans et al. [33] and Kooi et al. [35]. This data alignment resolved an inconsistency in available toxicity data of different levels of polydispersity, including monodisperse (i.e., same size and/or shape versus polydisperse MPs). This approach also allowed correction for bioavailability of particles for food dilution and tissue translocation. A detailed description of the data alignment approach used can be found in the supplemental information S3.

Sensitivity analyses

Three types of sensitivity analyses were conducted, focused on the alignment procedure, endpoints selection, and studies selection. The first one examined the variability associated with various assumptions used in the alignment method proposed by Koelmans et al. [33]. Specifically, we assessed the influence of alpha values proposed in Kooi et al. [35] and used these to convert effects data from particles used in the laboratory to effects data for polydisperse particles as seen in the environment. We also examined the influence of the assumed limitation in bioavailability of particles (i.e., estimated range for width and length for each particle type) for food dilution and tissue translocation. The second type of sensitivity analysis examined the importance of endpoints selection (all endpoints vs. fitness vs. mortality) in the SSDs to calculate threshold values. The third type of analysis used the leave-one-out method to remove individual studies and assess the variability in the underlying data [57]. For all three types of sensitivity analyses, the specific parameter examined was modified one-at-a-time and their impact on the resulting SSDs was evaluated. Additional details on the different sensitivity analyses are provided in Supplemental Information S3.

Expert's confidence in the outcomes

Experts were asked to describe their relative confidence level in the decision framework and analytical process adopted. Workshop participants engaged in critical discussions on the suitability of a tiered-management construct with four thresholds. The relative level of confidence in the calculated threshold values was also evaluated based on the amount, quality, and consistency of data. As part of this process, experts conducted a detailed review of five studies that were driving the outcome of

Table 2 Proposed microplastics toxicity thresholds for food dilution, relevant for particle sizes between 1 and 5000 μm

Threshold	Particles/L (95% CI)	Mass (mg)/L (95% CI)
Threshold 1- "Investigative monitoring"	0.3 ^a	0.05 ^a
Threshold 2- "Discharge monitoring"	3 (0.3 to 66)	0.4 (0.05 to 11)
Threshold 3- "Management planning"	5 (0.4 to 219)	0.9 (0.07 to 36)
Threshold 4- "Source control measures"	34 (3 to 859)	6 (0.4 to 141)

^a Threshold 1 is the lower 95% CI of the HC5 calculated for Threshold 2, therefore confidence intervals cannot be reported for this threshold

Table 3 Proposed microplastics toxicity thresholds for tissue translocation, relevant for particle sizes between 1 and 83 μm

Threshold	Particles/L (95% CI)	Mass (mg)/L (95% CI)
Threshold 1- “Investigative monitoring”	60 ^a	10 ^a
Threshold 2- “Discharge monitoring”	312 (57 to 4680)	51 (10 to 770)
Threshold 3- “Management planning”	890 (118 to 19,000)	146 (19 to 3120)
Threshold 4- “Source control measures”	4110 (493 to 69,100)	676 (81 to 11,400)

^a Threshold 1 is the lower 95% CI of the HC5 calculated for Threshold 2, therefore confidence intervals cannot be reported for this threshold

the SSDs to ensure that data were correctly entered in the database and aligned. The confidence vote was achieved using a semi-quantitative approach similar to that used by the Intergovernmental Panel on Climate Change (IPCC [41]). Each expert anonymously rated their confidence level on a scale from 1 to 5 (very low, low, medium, high, very high). Once the votes were tallied, the experts discussed the outcome, and shared their perspectives on the appropriateness and quality of the data.

Results

Thresholds

SSD-derived thresholds were calculated using toxicity data for 14–16 species from 6 or 7 taxonomic groups, depending on the threshold level and effect mechanism (Figure S1). The estimated threshold values for food dilution ranged between 0.3 and 34 particles/L (mass equivalent of 0.05 to 6 mg/L), with 95% confidence interval (CI) spanning over two orders of magnitude (Table 2). Results of these SSDs identified the marine bivalve *Pinctada margaritifera* (black lip pearl oyster) and estuarine fish

Oryzias melastigma (marine medaka) as the two most sensitive species (Figure S1). For tissue translocation, estimated threshold values ranged between 60 and 4110 particles/L (mass equivalent between 10 and 676 mg/L), with wide 95% CI (Table 3). *P. margaritifera* deemed to be the most sensitive, was reported to have reduced assimilation efficiency and altered energy balance upon exposure to polystyrene microbeads (6–10 μm) for 2 months [18]. In Wang et al., [60], *O. melastigma* exposed to 10 μm polystyrene spheres for 2 months had altered antioxidant enzyme expression and lowered circulating concentrations of sex steroids. *O. melastigma*, was also identified as the most sensitive to potential tissue translocation-related effects, together with the freshwater crustacean *Ceriodaphnia dubia*. Studies included in the SSDs for *C. dubia* showed that acute exposure to MP polyethylene particles (2–5 μm) had a significant impact on reproduction and survival [27, 28, 69].

While the studies used for thresholds derivatization met all our pre-defined quality standards, it should be noted that none of them met all 20 quality criteria considered necessary for risk assessment by de Ruijter et al. [12]. For example, most studies used virgin MPs, did not verify background contamination, and did not confirm actual exposure concentrations (i.e., nominal concentrations reported). Moreover, the in-depth review of the studies driving the lower portion of the SSD curves also indicated that some of these studies poorly described MP sample preparation and may have used unreliable quantification methods.

Sensitivity analysis

The results of the sensitivity analyses revealed that assumed distribution values for polydisperse laboratory experiments were of negligible impact to the resulting thresholds (– 15 to + 5%; Table 4 and supplemental information S4). The assumed values used for bioaccessibility

Table 4 Sensitivity analyses to assess the impact of alignment method, endpoint selection and individual studies, on the threshold values calculated

Parameter	Food dilution		Tissue translocation	
	Max increase(% difference)	Max decrease(% difference)	Max increase(% difference)	Max decrease(% difference)
Polydisperse laboratory assumptions (mass, volume, surface area)	+ 5%	– 2%	+ 5%	– 15%
Bioaccessible assumptions (length, width)	+ 31%	– 53%	+ 32%	– 89%
Environmental MPs polydispersity distribution (length, volume, surface area)	NA	– 87%	+ 285%	– 31%
Endpoint selection (all vs. fitness vs. mortality)	+ 628,000%	NA	+ 2511%	– 38%
Study influence (leave-one-out method)	+ 300%	– 26%	+ 275%	– 47%

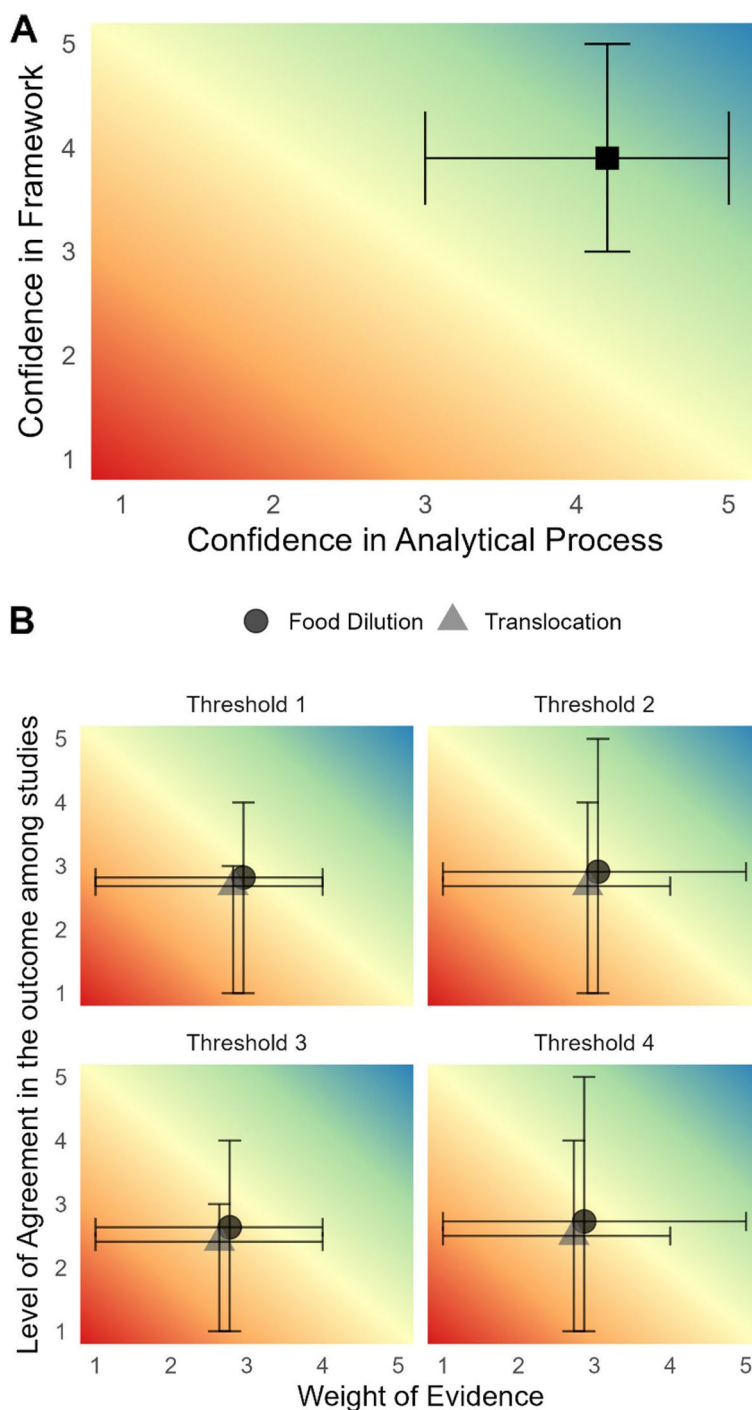


Fig. 2 Mean confidence scores for (A) the management framework and (B) threshold values. Whiskers represent the range of experts' votes. Scoring scale: 1- very low, 2- low, 3- medium, 4- high and 5- very high

based on shape and for estimating environmental MP polydispersity distribution had moderate impacts on the thresholds (− 89 to + 32% and − 87 to 289%, respectively). The results of the leave-one-out analysis implied the most sensitive study used to derive the SSD also had

a moderate influence on the thresholds calculated (− 47 to + 300% difference) (Figure S4). For both tissue translocation and food dilution-based thresholds, endpoint selection had the largest impact on the resulting thresholds (Table 4). When comparing SSD-derived thresholds

based on all endpoints to those calculated using fitness or mortality, thresholds values varied between -38 and $628,000\%$, with the influence largely due to using only mortality as an endpoint (supplemental information S4).

Expert's confidence in the outcomes

Overall, workshop participants expressed high confidence in the proposed multi-tiered management framework and the use of SSDs and data alignment calculations to derive hazard threshold values. The experts' scores ranged between 3.0 and 5.0 for both with mean score of 4.2 (high) for the framework and 3.9 (high) for analytical approach (Fig. 2A). The confidence level in the threshold values for food dilution and tissue translocation were highly variable among experts, with individual scores ranging from 1.0 (very low) to 4.0 (high). Thus, mean scores for individual thresholds ranged between 2.4 and 3.0, with slightly higher scores for food dilution-related thresholds (Fig. 2B).

Discussion

The tiered-management framework presented here is an enhancement of an approach used by the California State Water Resources Control Board to monitor other emerging contaminants [40]. The key to the framework is a recognition that there is not a single threshold that can be adopted to manage the potential environmental risks of MPs, but rather a need for multiple tiers representing varying levels of management decisions. Tiers 1–3 promote incremental information gathering needed to better understand the issue prior to implementing costly regulations in Tiers 4 and 5. The proposed strategy is focused on decisions regarding water quality management in aquatic habitats and is not intended to discourage upstream pollution control measures such as reduction. The same approach could also be applied to other types of MP management concerns, such as the presence of MPs in tissues and food.

The SSD approach used here has considerable precedent for toxicity threshold development and risk assessment for chemical contaminants [6, 16, 26, 45, 58]. This approach has also been applied to derive thresholds for non-chemical stressors including nanomaterials [10, 19], nanoplastics [65], and MPs (e.g., [2, 8, 15]). Coupled with Koelmans et al.'s [33] alignment approach, this probabilistic data integration approach proved particularly useful for MPs due to the large differences in particle size and shape used in toxicity studies [49]. A key decision in employing an SSD approach is the parameterization to produce four different thresholds that cover the gradation between Type I and Type II error regarding toxicity. The experts quickly reached consensus on the selection of HC values, point

estimates and data collapsing methods, but discussed at length the selection of endpoints. The consensus was that Thresholds 1 and 2 should include endpoints from all levels of biological organization, including effects at the molecular and cellular level. This is consistent with the evolution of toxicity testing and the recognition that sub-organismal biomarkers can serve as early indicators to prevent adverse toxicity effects [30]. For Thresholds 3 and 4, the consensus was to limit to organismal-level effects for greater confidence in ecologically relevant effects. However, the experts did not restrict the categories of endpoints to consider, thus including data for fitness, metabolism, and behavioral endpoints. Adverse outcome pathways associated with microplastics exposure are still in development [29] and currently available data are not sufficient to determine low or medium priority endpoints. While regulatory frameworks have favored the use of fitness endpoints (e.g., growth, development, mortality) for threshold development, other categories of endpoints such as immune or behavioral changes may also lead to impaired fitness. Moreover, the use of severe effects (e.g., mortality data only) would not be appropriate for a precautionary management approach aiming to prevent impacts to aquatic life. As more information is garnered on the toxicity pathways for MPs, data selection for SSD implementation should be reconsidered.

Application of the SSD approach yielded thresholds lower for food dilution than tissue translocation. Microplastics toxicity is known to be size-related, which may be predicted by volume and surface area of the exposed MPs ([5, 24]). Thus, it is possible that a small number of large particles will have a large volume in the gut leading to reduced food intake. It should be noted that our 8 threshold values generally fell within the range of SSD-derived thresholds previously reported by Everaert et al. [15] for marine species (HC5 of 33.3 particles/L, with a 95% confidence interval of 0.36–13,943 particles/L) using unaligned data, and by Koelmans et al. [33] for freshwater ecosystems (HC5 of 75.6 particles/L, with a 95% confidence interval of 11–521 particles) using an alignment-based method. Other studies that calculated predicted no-effect concentrations (PNECs) using unaligned data also reported values within the range of thresholds calculated in this study [2, 67]. However, all these studies proposed a single value, often applicable to a specific habitat (i.e., freshwater or marine). Here, we chose to merge all aquatic species (freshwater and marine) in the meta-analysis based on the assumption that effect mechanisms have a stronger influence on toxicity outcome than habitat, and the recognition that some species live across salinity gradients. This decision allowed the use of a larger dataset, although managers may choose

to repopulate this framework with taxa specific to their geography and habitat should sufficient data be available. Overall, food dilution thresholds were within the lower range of concentrations reported in aquatic habitats, while those developed for tissue translocation were much higher than reported in the environment [22, 68]. Our findings suggest food dilution will drive management responses and future studies should better define the dose-response relationship for this effect mechanism.

Evaluation of experts' uncertainties revealed a high level of confidence in the management framework and analytical approach for populating the thresholds, but relatively low confidence in the thresholds themselves. One area of concern was the lack of separation observed between Thresholds 2 and 3, which failed to reflect the increase in likelihood for impact and did not support the need for more stringent management decisions. This suggests that additional parameterization of the SSDs may be needed. The main area of uncertainties, however, stemmed from the limited data of sufficient quality and environmental relevance. Most studies included in the SSDs used spheres or fragments, despite evidence that fibers are one of the shapes most frequently detected in the environment [4]. Fibers are also believed to exert higher toxicity in comparison to other MP shapes [46, 52]. The alignment method partially addresses this concern, but a more representative underlying dataset would reduce some these uncertainties. In addition, over 90% of studies in the database did not meet all quality criteria described by de Ruijter et al. [12]. Instead, a reduced set of quality criteria was identified, focusing on select details of exposure conditions (e.g., polymer type, shape, size, nominal concentration) and a minimum of 3 MP concentrations (excluding controls). To ensure that sufficient data was included in the SSDs, other key quality criteria critical to the assessment of dose-response relationships such as verification of MP exposure concentrations or chemical composition of tested MPs were not considered. These shortcomings had an influence on the relative confidence in threshold values derived. Recommendations to improve data quality in future studies are expanded in Hampton et al. C [25]. Another concern was the absence of established toxicity pathways for MPs. While several studies reported gene expression changes, altered metabolism, oxidative stress or tissue inflammation following microplastic exposure, few provided clear relationships between these endpoints and more apical effects at the organism or population level. Studies on adverse outcome pathways that integrate responses across multiple biological scales would increase the weight of evidence and overall confidence in health-based thresholds for MPs.

Conclusions

We introduce a MPs risk management framework that identifies multiple levels of potential management action depending on MPs concentrations and associated biological effect thresholds. Included in that framework are four health-based thresholds that distinguish those management levels and a process for calculating thresholds. While this work was done to address a California legislative mandate, it has relevance to other jurisdictions globally. More importantly, the multi-tiered framework is adaptive to new management decisions and to incorporation of additional data as it becomes available. The most important data needed to improve confidence in thresholds produced by the framework are ones that enhance the knowledge on dose-response relationships and the effects of environmentally realistic (polydisperse) mixtures of diverse particles, and studies that better establish adverse outcome pathways.

Abbreviations

CI	Confidence interval
EC	Effect concentration
HONEC	Highest observed effect concentration
LC	Lethal effect concentration
LOEC	Lowest observed effect concentration
MP	Microplastic particle
NOEC	No observed effect concentration
PNEC	Predicted no-effect concentration
SSD	Species sensitivity distribution
ToMEX	Toxicity of Microplastics Explorer

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s43591-022-00033-3>.

Additional file 1.

Additional file 2: Supplementary table.

Acknowledgments

The authors acknowledge the participants of the California Health Effect Workshop.

Peer review

In keeping with research integrity principles, all SCCWRP workshop articles co-authored by Dr. Steve Weisberg, Dr. Albert Koelmans, and other *Microplastics and Nanoplastics* editors were handled by independent editors of the journal.

Authors' contributions

ACM, SC, SMB, GAB, EM, CMR and SBW developed the tiered management framework. ACM, SC, AAK, SMB, MW, LMTH, CMR, and SBW developed the analytical approach and SC, AAK, LMTH and ACM, led the meta-analysis for threshold values calculations. All co-authors participated in the confidence voting exercise and contributed to manuscript preparation. The author(s) read and approved the final manuscript.

Funding

Funding for this project was provided by the California State Water Resources Control Board.

Availability of data and materials

All data used in this study are included within this manuscript and the supplemental information. Toxicity data used in the analyses may also be downloaded using the online RShiny ToMEx App hosted at <https://microplast.ics.sccwrp.org/> and described in detail in Hampton et al. "A Living Tool for the Continued Exploration of Microplastics Toxicity" (submitted in this issue).

Declarations

Ethics approval and consent to participate

Not Applicable.

Consent for publication

The authors of this manuscript consent to Springer journal policies for publication.

Competing interests

The authors ACM, SC, GAB, LTH, EM and SBW declare having no known competing financial interests or professional relationships that could have appeared to influence the work reported in this paper. The following authors declare financial interests/professional relationships which may be considered as potential competing interests:

- AAK reports a relationship with World Health Organization, European Food Safety Authority, United Nations and SAPEA & European Commission Group of Chief Scientific Advisors that includes consulting or advisory, and financial support in the form of funding grants or travel reimbursement. AAK also acknowledges research funding from the European Chemical Industry Council, the Dutch Research Council, the Norwegian Research Council and Rijkswaterstaat Water Traffic and Living Environment. All research is carried out under the Dutch code of conduct for scientific integrity.
- SMB reports a relationship with Ocean Science Trust that includes consulting or advisory and acknowledges funding from the National Science Foundation under grant agreement No. 1935028.
- MW reports a relationship with The European Parliament, SAPEA and European Commission Group of Chief Scientific Advisors, The World Bank, the Food Packaging Forum, and Hold Norge Rent that includes travel reimbursement. MW reports a relationship with the Norwegian Committee for Food and the Environment that included travel reimbursement and consulting.
- TG reports a relationship with World Health Organization, European Centre for Ecotoxicology and Toxicology of Chemicals, Environment and Climate Change Canada, and American Chemistry Council Chemical Products and Technology Division that include consulting or advisory. TG reports a relationship with European Food Safety Authority and Horizon 2020 that include travel reimbursement or funding grant. TG reports a non-financial relationship with International Life Sciences Institute Europe ILSI.
- CMR reports a relationship with Ocean Conservancy as a Scientific Advisor.

Received: 30 November 2021 Accepted: 29 April 2022

Published online: 23 June 2022

References

1. Abel SM, Primpke S, Int-Veen I, Brandt A, Gerdt G. Systematic identification of microplastics in abyssal and hadal sediments of the Kuril Kamchatka trench. *Environ Pollut*. 2021;269:116095.
2. Adam V, von Wyl A, Nowack B. Probabilistic environmental risk assessment of microplastics in marine habitats. *Aquat Toxicol*. 2021;230:105689.
3. Aldenberg T, Slob W. Confidence limits for hazardous concentrations based on logistically distributed NOEC toxicity data. *Ecotoxicol Environ Saf*. 1993;25(1):48–63.
4. Athey SN, Erdle LM. Are we underestimating anthropogenic microfiber pollution? A critical review of occurrence, methods, and reporting. *Environ Toxicol Chem*. 2021.
5. Au SY, Lee CM, Weinstein JE, van den Hurk P, Klaine SJ. Trophic transfer of microplastics in aquatic ecosystems: identifying critical research needs. *Integr Environ Assess Manag*. 2017;13(3):505–9.
6. Belanger S, Barron M, Craig P, Dyer S, Galay-Burgos M, Hamer M, et al. Future needs and recommendations in the development of species sensitivity distributions: estimating toxicity thresholds for aquatic ecological communities and assessing impacts of chemical exposures. *Integr Environ Assess Manag*. 2017;13(4):664–74.
7. Bergmann M, Wirzberger V, Krumpal T, Lorenz C, Primpke S, Tekman MB, et al. High quantities of microplastic in Arctic Deep-Sea sediments from the HAUSGARTEN observatory. *Environ Sci Technol*. 2017;51(19):11000–10.
8. Besseling E, Redondo-Hasselerharm P, Foekema EM, Koelmans AA. Quantifying ecological risks of aquatic micro- and nanoplastic. *Crit Rev Environ Sci Technol*. 2019;49(1):32–80.
9. Brander SM, Renick VC, Foley MM, Steele C, Woo M, Lusher A, et al. Sampling and quality assurance and quality control: a guide for scientists investigating the occurrence of microplastics across matrices. *Appl Spectrosc*. 2020;74(9):1099–125.
10. Coll C, Notter D, Gottschalk F, Sun T, Som C, Nowack B. Probabilistic environmental risk assessment of five nanomaterials (nano-TiO₂, nano-Ag, nano-ZnO, CNT, and fullerenes). *Nanotoxicology*. 2016;10(4):436–44.
11. Coffin S, Weisberg SB. The importance of understanding health effects pathways and thresholds in support of microplastics management. *Microplastics and Nanoplastics This issue*.
12. de Ruijter VN, Redondo-Hasselerharm PE, Gouin T, Koelmans AA. Quality criteria for microplastic effect studies in the context of risk assessment: a critical review. *Environ Sci Technol*. 2020;54(19):11692–705.
13. Di Pippo F, Venezia C, Sighicelli M, Pietrelli L, Di Vito S, Nuglio S, et al. Microplastic-associated biofilms in lentic Italian ecosystems. *Water Res*. 2020;187:116429.
14. Earn A, Bucci K, Rochman CM. A systematic review of the literature on plastic pollution in the Laurentian Great Lakes and its effects on freshwater biota. *J Great Lakes Res*. 2021;47(1):120–33.
15. Everaert G, De Rijcke M, Lonzeville B, Janssen CR, Backhaus T, Mees J, et al. Risks of floating microplastic in the global ocean. *Environ Pollut*. 2020;267:115499.
16. Fox D, van Dam R, Fisher R, Batley G, Tillmanns A, Thorley J, et al. Recent developments in species sensitivity distribution modeling. *Environ Toxicol Chem*. 2021;40(2):293–308.
17. Gardon T, Morvan L, Huvet A, Quillien V, Soyez C, Le Moullac G, et al. Microplastics induce dose-specific transcriptomic disruptions in energy metabolism and immunity of the pearl oyster *Pinctada margaritifera*. *Environ Pollut*. 2020;266:115180.
18. Gardon T, Reisser C, Soyez C, Quillien V, Le Moullac G. Microplastics affect energy balance and gametogenesis in the pearl oyster *Pinctada margaritifera*. *Environ Sci Technol*. 2018;52(9):5277–86.
19. Garner KL, Suh S, Lenihan HS, Keller AA. Species sensitivity distributions for engineered Nanomaterials. *Environ Sci Technol*. 2015;49(9):5753–9.
20. González-Pleiter M, Edo C, Velázquez D, Casero-Chamorro MC, Leganés F, Quesada A, et al. First detection of microplastics in the freshwater of an Antarctic specially protected area. *Mar Pollut Bull*. 2020;161:111811.
21. Gouin T, Becker RA, Collot A-G, Davis JW, Howard B, Inawaka K, et al. Toward the development and application of an environmental risk assessment framework for microplastic. *Environ Toxicol Chem*. 2019;38(10):2087–100.
22. Grbić J, Helm P, Athey S, Rochman CM. Microplastics entering north-western Lake Ontario are diverse and linked to urban sources. *Water Res*. 2020;174:115623.
23. Hampton LMT, Lowman H, Coffin S, Darin E, De Frond H, Hermabessiere L, et al. A living tool for the continued exploration of microplastics toxicity. *Microplastics and Nanoplastics A This issue*. 2022a.
24. Hampton LMT, Brander S, Coffin S, Cole M, Hermabessiere L, Koelmans AA, Rochman CM. Characterizing microplastic hazards: which concentration metrics and particle characteristics are most informative for understanding toxicity in aquatic organisms? *Microplastics and Nanoplastics B. This issue*.
25. Hampton LMT, Bouwmeester H, Brander S, Coffin S, Cole M, Hermabessiere L, et al. Research recommendations to better understand the potential health impacts of microplastics to humans and aquatic ecosystems. *Microplastics and Nanoplastics C This issue*.
26. Hill KL, Breton RL, Manning GE, Teed RS, Capdevielle M, Slezak B. Deriving a water quality guideline for protection of aquatic communities exposed to triclosan in the Canadian environment. *Integr Environ Assess Manag*. 2018;14(4):437–41.

27. Jaikumar G, Baas J, Brun NR, Vijver MG, Bosker T. Acute sensitivity of three Cladoceran species to different types of microplastics in combination with thermal stress. *Environ Pollut.* 2018;239:733–40.
28. Jaikumar G, Brun NR, Vijver MG, Bosker T. Reproductive toxicity of primary and secondary microplastics to three cladocerans during chronic exposure. *Environ Pollut.* 2019;249:638–46.
29. Jeong J, Choi J. Adverse outcome pathways potentially related to hazard identification of microplastics based on toxicity mechanisms. *Chemosphere.* 2019;231:249–55.
30. Keller DA, Juberg DR, Catlin N, Farland WH, Hess FG, Wolf DC, et al. Identification and characterization of adverse effects in 21st century toxicology. *Toxicol Sci.* 2012;126(2):291–7.
31. Kim D, Chae Y, An Y-J. Mixture toxicity of nickel and microplastics with different functional groups on *Daphnia magna*. *Environ Sci Technol.* 2017;51(21):12852–8.
32. Koelmans AA, Besseling E, Foekema E, Kooi M, Mintenig S, Ossendorp BC, et al. Risks of plastic debris: Unravelling fact, opinion, perception, and belief. *Environ Sci Technol.* 2017;51(20):11513–9.
33. Koelmans AA, Redondo-Hasselerharm PE, Mohamed Nor NH, Kooi M. Solving the nonalignment of methods and approaches used in microplastic research to consistently characterize risk. *Environ Sci Technol.* 2020;54(19):12307–15.
34. Kooi M, Koelmans AA. Simplifying microplastic via continuous probability distributions for size, shape, and density. *Environ Sci Technol Lett.* 2019;6(9):551–7.
35. Kooi M, Primpke S, Mintenig SM, Lorenz C, Gerdtz G, Koelmans AA. Characterizing the multidimensionality of microplastics across environmental compartments. *Water Res.* 2021;202:117429.
36. Kumar R, Sharma P, Manna C, Jain M. Abundance, interaction, ingestion, ecological concerns, and mitigation policies of microplastic pollution in riverine ecosystem: a review. *Sci Total Environ.* 2021;782:146695.
37. Lambert S, Scherer C, Wagner M. Ecotoxicity testing of microplastics: considering the heterogeneity of physicochemical properties. *Integr Environ Assess Manag.* 2017;13(3):470–5.
38. Lei L, Liu M, Song Y, Lu S, Hu J, Cao C, et al. Polystyrene (nano) microplastics cause size-dependent neurotoxicity, oxidative damage and other adverse effects in *Caenorhabditis elegans*. *Environ Sci: Nano.* 2018;5(8):2009–20.
39. Limonta G, Mancía A, Benkhalqui A, Bertolucci C, Abelli L, Fossi MC, et al. Microplastics induce transcriptional changes, immune response and behavioral alterations in adult zebrafish. *Sci Rep.* 2019;9(1):15775.
40. Maruya KA, Dodder NG, Mehinto AC, Denslow ND, Schlenk D, Snyder SA, et al. A tiered, integrated biological and chemical monitoring framework for contaminants of emerging concern in aquatic ecosystems. *Integr Environ Assess Manag.* 2016;12(3):540–7.
41. Mastrandrea MD, Field CB, Stocker TF, Edenhofer O, Ebi KL, Frame DJ, et al. Guidance note for lead authors of the IPCC fifth assessment report on consistent treatment of uncertainties. 2010.
42. Paul-Pont I, Lacroix C, González Fernández C, Hégaret H, Lambert C, Le Goïc N, et al. Exposure of marine mussels *Mytilus* spp. to polystyrene microplastics: toxicity and influence on fluoranthene bioaccumulation. *Environ Pollut.* 2016;216:724–37.
43. Pirsabe M, Hossini H, Makhdoumi P. Review of microplastic occurrence and toxicological effects in marine environment: experimental evidence of inflammation. *Process Saf Environ Prot.* 2020;142:1–14.
44. Posthuma L, Suter II GW, Traas TP, editors. *Species Sensitivity Distributions in Ecotoxicology*. 1st Edition. CRC Press; 2001.
45. Posthuma L, van Gils J, Zijp MC, van de Meent D, de Zwart D. Species sensitivity distributions for use in environmental protection, assessment, and management of aquatic ecosystems for 12 386 chemicals. *Environ Toxicol Chem.* 2019;38(4):905–17.
46. Qiao R, Deng Y, Zhang S, Wolosker MB, Zhu Q, Ren H, et al. Accumulation of different shapes of microplastics initiates intestinal injury and gut microbiota dysbiosis in the gut of zebrafish. *Chemosphere.* 2019;236:124334.
47. Rauchschalbe M-T, Fueser H, Traunspurger W, Höss S. Bacterial consumption by nematodes is disturbed by the presence of polystyrene beads: the roles of food dilution and pharyngeal pumping. *Environ Pollut.* 2021;273:116471.
48. Redondo-Hasselerharm PE, Falahudin D, Peeters ETHM, Koelmans AA. Microplastic effect thresholds for freshwater benthic macroinvertebrates. *Environ Sci Technol.* 2018;52(4):2278–86.
49. Rochman CM, Brookson C, Bikker J, Djuric N, Earn A, Buccì K, et al. Rethinking microplastics as a diverse contaminant suite. *Environ Toxicol Chem.* 2019;38(4):703–11.
50. Shupe HJ, Boenisch KM, Harper BJ, Brander SM, Harper SL. Effect of Nanoplastic type and surface chemistry on particle agglomeration over a salinity gradient. *Environ Toxicol Chem.* 2021;40(7):1820–6.
51. Stephan CE, Mount DJ, Hansen DJ, Gentile JH, Chapman GA, Brungs WA. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. Washington, DC: US Environmental Protection Agency Report, Office of Research and Development; 1985.
52. Stienbarger CD, Joseph J, Athey SN, Monteleone B, Andrady AL, Watanabe WO, et al. Direct ingestion, trophic transfer, and physiological effects of microplastics in the early life stages of *Centropomus striata*, a commercially and recreationally valuable fishery species. *Environ Pollut.* 2021;285:117653.
53. Straub S, Hirsch PE, Burkhardt-Holm P. Biodegradable and petroleum-based microplastics do not differ in their ingestion and excretion but in their biological effects in a freshwater invertebrate *Gammarus fossarum*. *Int J Environ Res Public Health.* 2017;14(7):774.
54. Suaria G, Perold V, Lee JR, Lebouard F, Aliani S, Ryan PG. Floating macro- and microplastics around the Southern Ocean: results from the Antarctic circumnavigation expedition. *Environ Int.* 2020;136:105494.
55. Sussarellu R, Suquet M, Thomas Y, Lambert C, Fabioux C, Pernet MEJ, et al. Oyster reproduction is affected by exposure to polystyrene microplastics. *PNAS.* 2016;113(9):2430–5.
56. Thorley J, Schwarz C. *ssdtools: An R package to fit Species Sensitivity Distributions*. *J Open Source Software.* 2018;3(31):1082.
57. Viechtbauer W, Cheung M. Outlier and influence diagnostics for meta-analysis. *Res Synth Methods.* 2010.
58. VKM- Norwegian Scientific Committee for Food and Environment, Skårø JU, Alexander J, Haave M, Jakubowicz I, Knutsen HK, et al. Microplastics; occurrence, levels and implications for environment and human health related to food. Scientific opinion of the Scientific Steering Committee of the Norwegian Scientific Committee for Food and Environment, Oslo, Norway: Norwegian Scientific Committee for Food and Environment (VKM); 2019. VKM report 2019:16.
59. Wagner C. Estimation of ecotoxicological protection levels from NOEC toxicity data. *Water Res.* 1991;25(10):1237–42.
60. Wang J, Li Y, Lu L, Zheng M, Zhang X, Tian H, et al. Polystyrene microplastics cause tissue damages, sex-specific reproductive disruption and transgenerational effects in marine medaka (*Oryzias melastigma*). *Environ Pollut.* 2019;254:113024.
61. Wang X, Liu L, Zheng H, Wang M, Fu Y, Luo X, et al. Polystyrene microplastics impaired the feeding and swimming behavior of mysid shrimp *Neomysis japonica*. *Mar Pollut Bull.* 2020;150:110660.
62. Wigger H, Kawecki D, Nowack B, Adam V. Systematic consideration of parameter uncertainty and variability in probabilistic species sensitivity distributions. *Integr Environ Assess Manag.* 2020;16(2):211–22.
63. Wyer H, Polhemus D, Moore S, Weisberg S, Coffin S, Rochman C. Steps scientists can take to inform aquatic microplastics management: a perspective informed by the California experience. *Appl Spectrosc.* 2020;74:000370282094603.
64. Xia X, Sun M, Zhou M, Chang Z, Li L. Polyvinyl chloride microplastics induce growth inhibition and oxidative stress in *Cyprinus carpio* var larvae. *Science of The Total Environment* 2020;716:136479.
65. Yang T, Nowack B. A Meta-analysis of Ecotoxicological Hazard data for Nanoplastics in marine and freshwater systems. *Environ Toxicol Chem.* 2020;39(12):2588–98.
66. Zhang W, Liu Z, Tang S, Li D, Jiang Q, Zhang T. Transcriptional response provides insights into the effect of chronic polystyrene nanoplastic exposure on *Daphnia pulex*. *Chemosphere.* 2020a;238:124563.
67. Zhang X, Leng Y, Liu X, Huang K, Wang J. Microplastics' pollution and risk assessment in an Urban River: a case study in the Yongjiang River, Nanjing City. *South China Expo Health.* 2020b;12(2):141–51.
68. Zhu X, Munno K, Grbic J, Werbowski LM, Bikker J, Ho A, et al. Holistic assessment of microplastics and other anthropogenic microdebris

in an Urban Bay sheds light on their sources and fate. *ACS EST Water*. 2021;1(6):1401–10.

69. Ziajahromi S, Kumar A, Neale PA, Leusch FD. Impact of microplastic beads and fibers on waterflea (*Ceriodaphnia dubia*) survival, growth, and reproduction: implications of single and mixture exposures. *Environ Sci Technol*. 2017;51(22):13397–406.
70. Zimmermann L, Göttlich S, Oehlmann J, Wagner M, Völker C. What are the drivers of microplastic toxicity? Comparing the toxicity of plastic chemicals and particles to *Daphnia magna*. *Environmental Pollution*. 2020;267:115392.

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Submit your manuscript to a SpringerOpen[®] journal and benefit from:

- ▶ Convenient online submission
- ▶ Rigorous peer review
- ▶ Open access: articles freely available online
- ▶ High visibility within the field
- ▶ Retaining the copyright to your article

Submit your next manuscript at ▶ [springeropen.com](https://www.springeropen.com)
