



Combined exposure of poly(lactic-co-glycolic acid) microplastics and global warming affects the population dynamics of a coastal copepod[☆]

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ABSTRACT

Microplastics induce ecotoxicological effects on coastal and estuarine organisms, but their combined effects with other environmental stressors such as climate change are poorly understood. The goal of the present work was to assess the combined effects of microplastics and water warming on the benthic copepod *Nitokra spinipes* at individual-level and to estimate population effects. *Nitokra spinipes* were exposed to Poly(lactic-co-glycolic-acid) microbeads (5 μm) at 0, 0.1, and 1% of food content, at 22 and 25 °C (+3 °C, based on IPCC-SSP5–8.5). The obtained filtration rates, a proxy for energy assimilation (assuming constant assimilation efficiency), were used in an individual-based model implementation of the dynamic energy budget theory (DEB-IBM) to extrapolate population-level effects under a warming scenario. Our results indicate a reduction in filtration rates by PLA microplastics at +3 °C, with 96-h EC₅₀ = 0.029% microplastics to food content, and the DEB-IBM model indicated a population-level EC₅₀ = 0.0098% microplastics to food content. Our results suggest that the combined exposure to elevated water temperatures and microplastics induces a decrease in energy assimilation on benthic copepods, with negative effects at population level.

1. Introduction

Anthropogenic stressors (Ostrowski et al., 2021), such as the emission of pollutants like microplastics, can affect coastal and estuarine ecosystems from local to global scales (SAPEA-Science Advice for Policy by European Academies, 2019; Smith et al., 2024). Microplastics (1 μm – 5 mm) are known to accumulate in estuaries (Lam et al., 2022; Roscher et al., 2021) where they can be trapped due to tidal and hydrological effects (Cardoso-Mohedano et al., 2023; Vibhatabandhu et al., 2024; Yang et al., 2024), and are of higher concentration in coastal ecosystems when compared to open seas (Amenábar et al., 2024; Wang et al., 2022). This accumulation is of concern, since estuaries and coastal areas are highly productive ecosystems that serve as nursery areas for a wide range of marine organisms, including invertebrates and commercially important fish (Barbier et al., 2011; McLusky and Elliott, 2004).

In addition to microplastics, the coastal and estuarine ecosystems are particularly vulnerable to anthropogenic stressors associated to climate change, such as global warming (Cooley et al., 2023; Smith et al., 2024). Current anthropogenic CO₂ accumulation in the atmosphere and ocean surface is projected to cause a global temperature increase of 1.5–4 °C by 2100 (Lee et al., 2023). Under a no-mitigation scenario (SSP5–8.5), surface water temperatures in the North Sea and Baltic Sea are expected to rise by 3 °C and 5 °C, respectively, by 2100 (Cael et al., 2024; Kristiansen et al., 2024; Lee et al., 2023). At a local scale, this projection is supported by historical seawater temperature data in European coasts (Desmit et al., 2020; Kristiansen et al., 2024), and a 3 °C increase of water temperature has also been observed in marine heatwave events in the North Sea (Semmoury et al., 2022) and the Baltic Sea (Safonova et al., 2024). The impact of warmer sea surface waters has already led to visible adverse outcomes in the North Sea, such as a 15–80% population

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decline of key copepod species, one of the most abundant invertebrate groups in aquatic ecosystems, from the year 2010 to 2020 (Semmour et al., 2023), which can lead to cascading food web effects. In this context, there is the need to assess the combined effects of microplastics and warming on copepods, from individual- to population-levels, which act as a main link between primary producers and higher trophic levels in food webs (Kwok et al., 2015).

In the environment, aquatic organisms simultaneously exposed to microplastics and other stressors, which can have additive, synergistic or subtractive physiological effects (Catarino et al., 2022). These effects can be of particular concern when affecting organisms at the base of the food web, such as primary consumers (Malinowski et al., 2023), since microplastics (both petroleum and bio-based) can be released through weathering processes, reaching ingestible sizes (Niu et al., 2024). For example, exposure to elevated water temperatures can increase the metabolic demands of copepods (Vigliano Relva et al., 2024) and their ingestion rates as a response (Li et al., 2015; Saiz et al., 2022), leading to a higher ingestion of microplastics. The inhibition on the proportional nutrient intake of food items can result in a lower capacity for energy uptake and leads to an energy (mis)-use (Sussarellu et al., 2016), since organisms could waste their energetic reserves by trying to digest indigestible microplastics (Cole et al., 2015). In long-term exposures, some laboratory-based population experiments on anomopods (e.g. *Daphnia* spp.) have confirmed that microplastic ingestion caused a decrease in the number of reproducing adults leading to a decrease in population density (Lyu et al., 2021; Zebrowski et al., 2022). Furthermore, microplastics are also known to induce negative effects on pelagic copepods, even at current environmentally relevant concentrations, with expected negative effects at population level (Everaert et al., 2022). These findings underscore the need to address the combined effects of microplastics and environmental stressors on key organisms such as copepods at the population level in coastal and marine food webs (Hansen et al., 2024; Kwok et al., 2015). Such information will enable us to quantify the impact of microplastics on the functioning of coastal and marine ecosystems (Zhou et al., 2023).

The goal of the present study was to assess the combined effects of microplastics, of a bio-based polymer, Poly(lactic-co-glycolic-acid) microbeads (5 μm), and warming water temperatures (+3 $^{\circ}\text{C}$) on the ingestion rate and the population dynamics of a brackish water benthic copepod *Nitokra spinipes*. Despite being a benthic species, *N. spinipes* efficiently ingests and assimilates suspended microalgae (De Troch et al., 2005), i.e. phytoplankton. Consequently, this species can be exposed not only to particles that accumulate in sediments, where microplastics tend to concentrate (Trusler et al., 2024; Xu et al., 2020), but also to microplastics suspended alongside phytoplankton. This feeding behaviour increases the likelihood of *N. spinipes* ingesting microplastics, thereby enhancing their exposure to these contaminants. The effects on *N. spinipes* individuals were assessed by quantifying shifts in the filtration rate (the capacity of the copepods to filter food from surrounding water), a proxy for energy assimilation (assuming constant assimilation efficiency, Liu et al., 2022), in a concentration–response relation (0, 0.1, and 1% microplastics to food content). Then, based on the empirical EC_{50} in ingestion rates, their population dynamics was simulated under the projected warming scenario (25 $^{\circ}\text{C}$), while exposed to microplastics, using an individual-based model implementation of the dynamic energy budget theory (DEB-IBM model). The DEB theory is a mechanistic framework that models energy acquisition and allocation within organisms, and when coupled with individual-based models, it allows extrapolating sublethal effects on individuals to impacts on population-level dynamics (Koch and De Schampelaere, 2023; Kooijman, 2010; Detailed description of the DEB-IBM model is available in the supporting information).

2. Materials and methods

2.1. Test materials

In this work we used fluorescent poly(lactic-co-glycolic acid) (PLGA) microbeads (reference LG5000, Phosphorex®, USA) with a density 1.3 g/mL, and a nominal diameter of 5 μm . Prior to use, 0.4 mg PLGA microbeads were pre-weighed based on volume-to-density-ratio calculations (Niu et al., 2020) and added into 2 mL of 0.2 μm filtered brackish water (salinity = 7 PSU, prepared as per Davies et al. (2023)) to produce an initial stock suspension of 1×10^9 particles / L. To do so, particles were first dispersed in the test vials by adding 0.01% Tween80 (Cospheric, USA), at a Tween80 concentration known to induce no effect on both microalgae and copepods (Niu et al., 2020; Yuan et al., 2024), followed by five minutes of vortexing. We assessed the actual concentration and length of the PLGA microbeads in the initial stock using fluorescent microscopy and image analysis (Niu et al., 2024) in ImageJ (Schindelin et al., 2012). The mean PLGA microbeads concentration and diameter was quantified as $1.1 \pm 0.13 \times 10^9$ particles / L ($N = 7$ images) and $6.7 \pm 3.2 \mu\text{m}$ ($N = 763$ particles) respectively. Then, a further dilution of the stock suspension was prepared in 0.2 μm pre-filtered brackish water (1×10^8 particles / L).

2.2. Model species

The strain of *Nitokra spinipes* used in this work was obtained from Ghent university (BE) (Koch et al., 2017) in 2020 and has been in continuous culture at the Marine Station Ostend (MSO) of the Flanders Marine Institute (VLIZ, Ostend, Belgium) in darkness with acclimated cultures both at $21.8 \pm 0.0 \text{ }^{\circ}\text{C}$ ($n = 11$, Table S1) and $25.3 \pm 0.1 \text{ }^{\circ}\text{C}$ ($n = 16$, Table S1). The culture medium was prepared by diluting natural seawater (salinity: 7 PSU; pH 8.0 ± 0.2) according to the ISO 18220 protocol (ISO, 2016), filtered through a 0.22 μm sterile filter and conditioned over night at the corresponding culture temperature. The *N. spinipes* cultures were maintained in darkness under static-renewal conditions and fed weekly with a commercial salmon feed (Astrawos, Södertälje, Sweden). A month prior to experimental work, three to four-week-old cultures at both temperatures were selected and gradually adapted to algal-ingestion. The microalga *Rhodomonas salina* (5–20 μm in cell length) was used as the food source and was provided twice a week at a final concentration of 2.5×10^5 cells/mL (Koch et al., 2017).

2.3. Experimental design

To investigate the combined effect of microplastics exposure and global warming on the food ingestion rate of *N. spinipes*, we assessed the filtration rate on copepods exposed to PLGA microbeads at 22 $^{\circ}\text{C}$ (control) and 25 $^{\circ}\text{C}$, an +3 $^{\circ}\text{C}$ warming scenario according to SSP5–8.5 (Lee et al., 2023). The control temperature reflects typical summer surface seawater conditions in the Baltic Sea, the origin of the tested *N. spinipes* strain (Koch et al., 2017; Suursaar, 2020), where the tested strain of the harpacticoid copepod *N. spinipes* is originally from (Koch et al., 2017). The dietary exposure of copepods was performed according to ISO 14669 (ISO, 1999), with a few modifications. In brief, three- to four-week-old copepods were exposed in darkness for 96 h to PLGA microbeads at concentrations of 0 (control), 0.1%, and 1% relative to food content (1×10^5 *R. salina* cells / mL, vortexed for homogenization), corresponding to 0, 1×10^5 , and 1×10^6 particles / L, respectively. For each treatment of the PLGA microbeads concentration and water temperature, 65 organisms were randomly selected regardless of gender, but avoiding egg-carrying females, which were separated into five replicates of fifteen organisms. Prior to exposure, all organisms were incubated overnight in control medium without food, to empty their gastrointestinal tracts. On the exposure day, organisms were transferred to pre-sterilized glass vessels of 25 mL containing 10 mL test medium. During the exposure, test vessels were incubated in the dark on an orbital

shaker offset at 20 rpm and permanently kept at the control (22 °C) or the warming (25 °C) scenarios, according to the allocated treatments.

We assessed their filtration rate which indicates the capacity of the copepods to filter food from surrounding water. The filtration rate was calculated using the formula of Rigler (Eq. 1) as described in Peters (Rh, 1984):

$$F = \frac{[\ln C_1 - \ln C_2] \times V}{N \times \Delta t} \quad (1)$$

where C_1 and C_2 are the cell densities (cells/mL) of *R. salina* at test beginning and test end, V is the volume of the test medium (10^4 μ L), N is the number of *N. spinipes* (15 individuals), and Δt is the exposure duration (5760 mins). The effect of microplastics (P_{Fi} , %) on the filtration rate F [μ L / (ind * min)] was calculated as:

$$P_{Fi} = \frac{F_c - F_i}{F_c} \times 100\% \quad (2)$$

where F_c and F_i are the mean filtration rate [μ L / (ind * min)] of control (no microplastics) and microplastics exposure i per temperature.

2.4. Ecological model to assess populational effects

To tailor the DEB-IBM for application to *N. spinipes*, three key modifications were implemented to align with its biological and ecological traits. Firstly, the model was adjusted to incorporate life stage-specific energy allocation, adhering to the standard DEB framework (sbp type). This structure tracks energy flow from birth to puberty, where growth halts as individuals transition to the adult stage and redirect energy toward reproduction (parameterization detailed in Table S2).

Secondly, temperature-dependence in metabolic rates and life cycle progression of *N. spinipes* was accounted for by integrating an Arrhenius correction factor from literature (Koch and De Schampelaere, 2023; Kooijman, 2010). Following the approach by Koch and De Schampelaere (2019), the metabolic parameter values were adjusted as a function of temperature (T , in Kelvin) using the formula:

$$k(T) = k_1 \times \exp\left(\frac{T_A}{T_1} - \frac{T_A}{T}\right) \quad (3)$$

where $k(T)$ is the parameter value at temperature T (K), k_1 is the parameter value at the reference temperature T_1 (K), and T_A is the species-specific Arrhenius temperature parameter (both values were based on literature, Table S2).

Lastly, the effects of microplastic ingestion on the filtration rate of *N. spinipes* was modelled. Building on prior findings regarding population-level effects of microplastics in copepods (e.g. Everaert et al., 2022), a dose-response relationship was established between the microplastic concentration in food (% food content) and the surface-specific ingestion rate J_{XAm} [J / (day * cm^2)]. The relationship was defined as:

$$J_{XAm, microplastics} = J_{XAm} \times \left\{ 1 - [\sin(a \times \ln C + b)]^2 \right\} \quad (4)$$

where C (% food content) represents the environmental microplastics concentration, a and b are the slope and intercept of the observed dose-response relationship (section *statistical analysis*) between the microplastic concentrations and the surface-specific ingestion rate (section *Experimental design*). To determine the population-level effects, the parameters a and b derived from the ingestion experiment under the warming water scenario (25 °C, as no effect on the filtration rate was observed at the control scenario, Fig. 1) was incorporated into the DEB model. Eq. 4 therefore links the experimental data obtained with the DEB-IBM model. We then extrapolated the population level effect over 2000 days (Fig. S4; Everaert et al., 2022) by simulating a concentration series of microplastics ranging from 0 and 1% of the available food content (Table S3).

2.5. Statistical analysis

The dataset containing the experimental results of this work are accessible via: doi:10.14284/722 (Niu et al., 2025). The data of this work is both open access and FAIR, archived in the Marine Data Archive

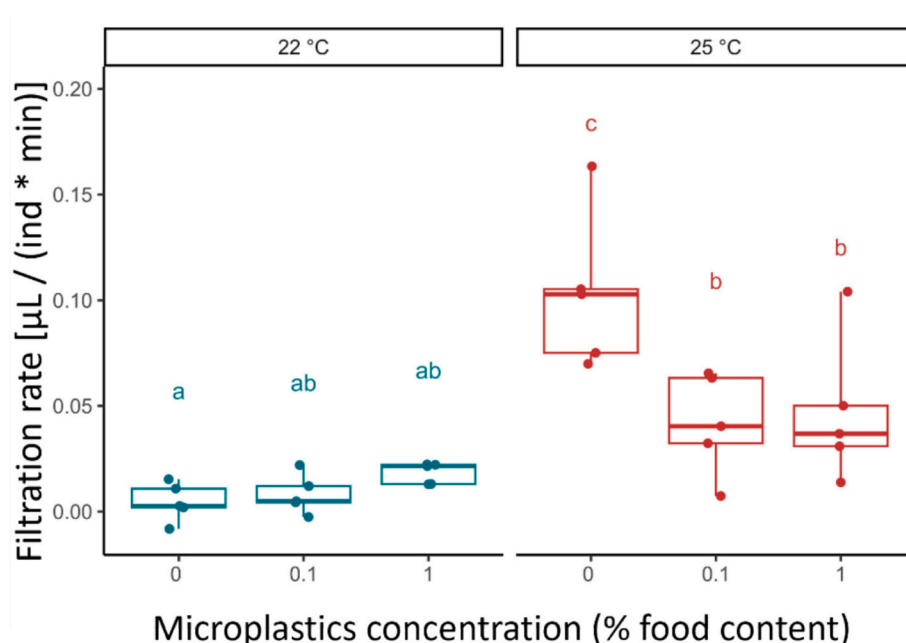


Fig. 1. Boxplots of the filtration rate of *N. spinipes* at 22 °C (left panel) and 25 °C (right panel), with median filtration rate [μ L / (ind * min)] against PLGA microbeads proportional concentration (% food content). Exposure was done at 0 (control), 0.1 and 1% particles to food content, corresponding to concentrations of 0, 1×10^5 and 1×10^6 particles / L. The colour of box represents copepods incubated at control (22 °C, in green) and warming (25 °C, in dark red) water temperature scenarios. Each dot represents a test vessel unit containing 15 copepods ($n = 5$). Different letters indicate statistically significant differences between groups ($p_{\text{Tukey's HSD}} < 0.05$). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

- MDA (VLIZ) [<https://mda.vliz.be/introduction.php>], and accessible via the Integrated Marine Information System - IMIS (VLIZ) [<https://marineinfo.org/about>]. All statistical analysis were performed in R v4.2.1 (R Core Team, 2019). An analysis of Variance (ANOVA) was performed to check if the filtration rate of the copepods was dependent on the concentration of PLGA microbeads and water temperature (Niu et al., 2025). Deviations from normality and homoscedasticity were checked using the Shapiro-Wilk and Levene's test, respectively, and statistical significance was accepted for $p < 0.05$. When the assumption of normality was not met ($p_{\text{Shapiro-Wilk}} < 0.05$ and skewness > 0), data were square-root transformed. Tukey's honestly significant difference (Tukey's HSD) test was applied to compare the variance in the filtration rate among groups. Results are expressed in mean values \pm standard deviation (SD). Due to the limited tested concentrations of microplastic content to food (in total three concentrations), the 96-h EC_{50} of the filtration rate, the slope, and the intercept of the concentration-response relationship was quantified using linear regression after an arcsine square-root transformation (EPA, 2002; Newman, 2012). The DEB-IBM in this work was developed in NetLogo 6.2 (Wilensky, 1999). Using concentrations using the drc package in R (Ritz et al., 2015), to obtain the corresponding EC_{50} at the population level (Niu et al., 2025). All plots were done using the R package ggplot2 (Wickham, 2011).

2.6. Quality assurance and quality control

Quality Criteria and Quality Control (QA/QC) measures were implemented during the experimental procedures to prevent contamination from airborne microplastics and other particles, as well as cross-contamination between samples (De Ruijter et al., 2020). All solvents used were of analytical grade. All glassware was pre-cleaned using diluted decon-90 detergent and rinsed thoroughly with Milli-Q water and acetone, and glassware was sterilized in an autoclave at 134 °C for one hour. All manipulations, including the preparation of algal food stocks and PLGA microbead suspensions, were carried in a clean flow cabinet, where contamination has been reported to be minimal (Meyers et al., 2022; Niu et al., 2024).

3. Results

3.1. Individual-level effects of PLGA microbeads and warming

Under the warming scenario (25 °C), exposure to Poly(lactic-co-glycolic-acid) microbeads (PLGA) 5 μm microbeads reduced the filtration rate of the adult *N. spinipes* (Fig. 1). In the absence of PLGA microbeads, adult *N. spinipes* exposed to warming temperatures exhibited a higher filtration rate compared to those maintained at control temperature (22 °C). Specifically, in the control treatment without PLGA microbeads, the mean (\pm standard deviation) filtration rate increased from $0.005 \pm 0.009 \mu\text{L} / (\text{ind}\cdot\text{min})$ at 22 °C to $0.103 \pm 0.037 \mu\text{L} / (\text{ind}\cdot\text{min})$ at 25 °C ($p_{\text{ANOVA}} < 0.05$). When exposed to PLGA microbeads (5 μm) under the warming scenario (25 °C), a decrease in filtration rate was observed. The rate dropped to $0.042 \pm 0.024 \mu\text{L} / (\text{ind}\cdot\text{min})$ at 0.1% food content ($p_{\text{Tukey-HSD}} = 0.02$) and to $0.047 \pm 0.034 \mu\text{L} / (\text{ind}\cdot\text{min})$ at 1% food content ($p_{\text{Tukey-HSD}} = 0.04$). The 96-h EC_{50} of PLGA microbeads under warming was estimated at 0.029% food content, equivalent to 2.9×10^4 particles / L. In contrast, at the control temperature (22 °C), exposure to PLGA microbeads did not significantly affect filtration rate ($p_{\text{ANOVA}} > 0.68$). Moreover, the filtration rate under warming and PLGA microplastics exposure was 5.4 times lower compared to the control scenario without PLGA exposure ($p_{\text{Tukey-HSD}} < 0.001$).

3.2. Population dynamics and impact of PLGA microbeads and warming

Exposure to PLGA microbeads under the warming scenario (+3 °C) significantly disrupt the population dynamics of *N. spinipes*, resulting in decreased equilibrium population densities (Fig. 2). In the absence of microplastics, the model predicted a mean (\pm standard deviation) equilibrium population density of $8626 \pm 124 \text{ ind} / \text{m}^3$. At a concentration of 0.01% food content, this value declined to $5165 \pm 158 \text{ ind} / \text{m}^3$, and at concentrations above 0.05%, the equilibrium density dropped to 0 ind / m^3 (Fig. 2). The EC_{50} for the equilibrium population density under the warming scenario was estimated to be 0.0098% food content (95% confidential interval 0.0095% – 0.0102%, equivalent to

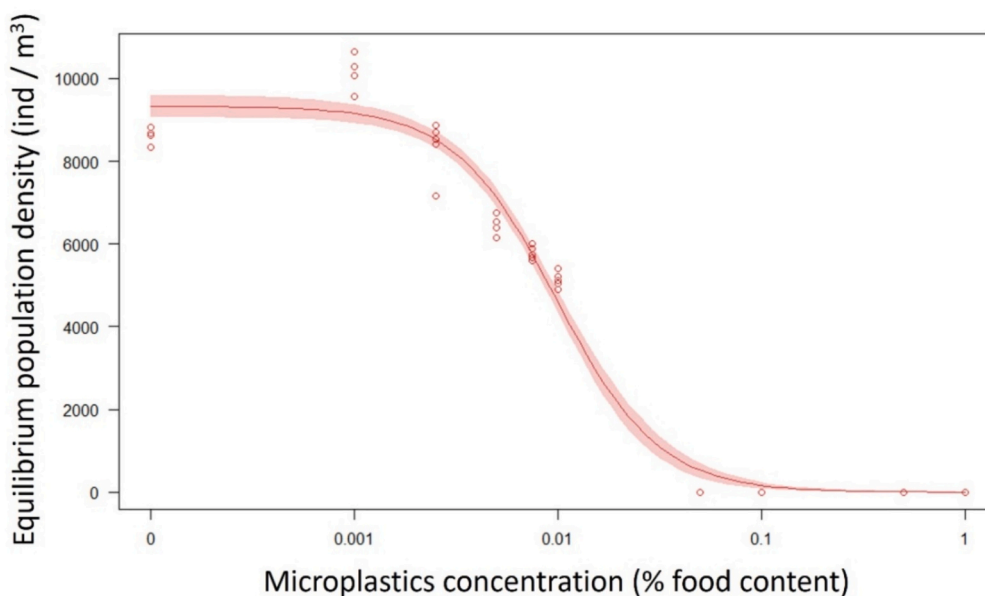


Fig. 2. Concentration-response relationship between PLGA microplastics proportional concentration and the estimated equilibrium population density of *N. spinipes* under the warming scenario (+3 °C), using the drc package (Ritz et al., 2015) in R (R Core Team, 2019). Each dot represents the equilibrium population density from a single simulation iteration. The red curve represents the fitted three-parameter log-logistic model (Ritz et al., 2015) based on the mean value of 10 iterations. The shaded red area indicates the 95% confidence interval of the fitted curve. Details of the model simulation and individual concentration-response curves from each iteration are provided in the supplementary information (SI, Model simulation and Fig. S4). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

33 PLGA microbeads / L).

4. Discussion

In the present study, a 96-h exposure to 0.01% microplastics to food content of poly(lactic-co-glycolic acid) (PLGA) microplastics under a warming scenario (+3 °C) resulted in a 60% reduction in the filtration rate of the adult harpacticoid copepod *Nitokra spinipes* (Fig. 1). These findings suggest that the ingestion of microplastics, particularly when misidentified as food, can impair feeding performance, consistent with prior observations in copepods (Coppock et al., 2019; Everaert et al., 2022) and other aquatic invertebrates such as bivalves (Gardon et al., 2018). Similarly, other copepods such as *Temora longicornis* have also exhibited a 50% reduction in filtration rate following a 48-h exposure to polyethylene (PE) microplastics (10–45 µm) at a concentration of 0.002% microplastics to food content (1956 particles / L) (Everaert et al., 2022). In the harpacticoid copepod *Calanus helgolandicus*, the exposure to PE microplastics (10–30 µm) impaired their clearance rate at a concentration of 32% microplastics to food content (1×10^5 particles / L) (Coppock et al., 2019).

The observed feeding inhibition may result from physical blockage, such as clogging of ingestion appendages or the digestive tract (Ziajahromi et al., 2018), or from (mis-)recognition. The microplastics used in our study were most likely not subject to selection behaviours by the copepods, as this behaviour usually takes place when the organisms are exposed to fibres, not spheres (Coppock et al., 2019). Therefore, the 5 µm PLGA microbeads underwent mis-ingestion, as their size and morphology resembled that of the microalgal prey, *Rhodomonas salina*. It should be noted that the mass concentration of PLGA microbeads was low, even at the highest exposure level (81.8 µg / L, based on 1×10^6 particles / L), indicating that potential ecotoxicological effects from leachates were likely negligible (Gunaalan et al., 2020). When microplastics are (mis-)ingested, the energy is diverted to processing these non-nutritive particles (Traboni et al., 2023), reducing actual nutrient uptake (Park et al., 2024). Previous studies have suggested that long-term exposure (≥ 7 days) to warming (i.e. elevated water temperature) and very high concentration of microplastics ($\geq 10^9$ particles / L) can impair the quality of prey microalgae for aquatic crustacean (Hiltunen et al., 2021; Zhang et al., 2022). However, in our case, the quality of the fed microalgae *Rhodomonas salina* is unlikely to be affected by microplastics (1×10^6 particles / L) and warming under 96-h incubation in complete darkness. Thus, the energy assimilation efficiency of *N. spinipes* likely remained stable across treatments. Nevertheless, the observed decline in feeding activity could induce an additional decrease on energy acquisition and assimilation rates (Park et al., 2024), potentially impairing critical biological processes such as growth, reproduction, and development (Cole et al., 2015; Redondo-Hasse-lerharm et al., 2018; Ziajahromi et al., 2018).

In the warming exposure scenario (+3 °C), copepods exhibited higher filtration rates than at control treatments (Fig. 1), consistent with increased feeding rates for the harpacticoid *Tigriopus japonicus* (Li et al., 2015) and the calanoid *Paracartia grani* (Saiz et al., 2022), when exposed to higher temperature. The fact that the exposure to +3 °C led to an effect in the feeding rate, this means that the likelihood of (mis-)ingestion of PLGA microplastics, thereby enhancing the vulnerability of the copepod to microplastics exposure. The increased filtration and subsequent ingestion rates are likely driven by elevated metabolic demands at higher temperatures, requiring greater energy intake and mobilization of energetic reserves (Vigliano Relva et al., 2024). As the assimilation efficiency is likely to remain constant at higher temperatures (Liu et al., 2022), the inhibition of the feeding rate by microplastics can result in a reduced energy assimilation rate, leading to a shift of the energy budget from reproduction to structural growth and elevated maintenance costs (Sussarellu et al., 2016). In the long term, organisms may experience reduced fecundity (Haberle et al., 2025; Reyserhove et al., 2017) or increased mortality (Lyu et al., 2022) if the energetic demands for

individual maintenance are not met. In this context, the observed copepod effects at individual-level (Fig. 1) can lead to long term effects at population levels, with broader consequences for food web dynamics, such as reduced energy transfer to higher trophic levels (Barneche et al., 2021; Ockenden et al., 2024).

The single exposure of PLGA microplastics at concentrations up to 1% microplastics to food content (1×10^6 particles / L), at the control temperature scenario, did not induce any significant differences in the filtration rate of *N. spinipes* compared to the no PLGA treatment (Fig. 1). This suggests that the tested bio-based microplastics had no acute effects on ingestion rates at environmentally relevant microplastic concentrations, when the organisms were exposed to a single stressor. In the present work, the microplastics exposure concentrations (0.1 and 1% microplastics to food content, equivalent to 1×10^5 and 1×10^6 particles / L, respectively) were below the higher end of absolute microplastic concentrations reported in estuarine systems ($< 10^5$ particles / L for particles 1–5000 µm) and below relative concentration estimates such as 1.1% food content in Jinhae Bay, assuming a microalgae density of 336 cells / mL (Kim et al., 2019; Song et al., 2015). These findings indicate that PLGA microplastics may pose a limited ecotoxicological risk to marine organisms when exposure occurs under controlled environmental conditions (Everaert et al., 2020), particularly for spherical, regular-shaped particles, contrasting with potential effects induced by irregular microplastics such as fibres, which have been shown to induce more pronounced adverse effects (Cole et al., 2019; Schwarzer et al., 2022). Future research should therefore investigate how particle shape of microplastics modulates ingestion behaviour, physiological stress, and consequent population dynamics to better inform ecological risk assessments. However, when the copepods are subjected to additional stressors, the cumulative effects can induce a population decline. Our results indicate that assessing the effects of microplastics under multiple-stressors scenarios provide a more comprehensive understanding of the complex interactions between plastic pollution and the environment.

Our study indicates that the equilibrium population density of *N. spinipes* would decrease by 50% at a PLGA microplastic concentration of 0.0098% microplastics to food content under warming conditions (+3 °C) (Fig. 2), suggesting that environmentally relevant microplastics concentrations may impair copepod population dynamics under combined stressor scenarios. The negative effects of microplastics on population dynamics have been demonstrated by laboratory-based studies on pelagic copepods (Shore et al., 2021) and other aquatic crustaceans such as cladocerans (Lyu et al., 2022; Schür et al., 2023; Zebrowski et al., 2022), though at concentrations exceeding current environmental levels, $> 10^7$ particles / L and 20% of microplastics to food content (Koelmans et al., 2022). The population decline observed in these studies (Lyu et al., 2022; Shore et al., 2021) resulted from the reduced survival of reproducing adults when their energetic demands of individual maintenance would not be met, which in our model simulation could be one of the major causes that induces the estimated population decline.

As microplastic concentrations are expected to continue increasing in marine environments under a business-as-usual scenario for both plastic pollution and greenhouse gas emissions (Wei et al., 2024), our results suggest that copepod populations could experience a 50% decline within the upcoming 100 years (EC₅₀ of equilibrium population density: 33 PLGA microbeads / L; Fig. 2). For example, surface microplastic concentrations in the Mediterranean Sea are projected to reach 52.2 particles / L by 2100 under this scenario (Everaert et al., 2020). Furthermore, copepod populations can be more sensitive to microplastics exposure than what would be indicated by short-term individual sublethal effects, as the estimated EC₅₀ of the equilibrium population density (0.0098% food content) by PLGA microplastics at 25 °C was threefold lower than the EC₅₀ of the filtration rate (0.029% food content). A similar pattern was observed for *T. longicornis* individuals, where the EC₅₀ of the equilibrium population density (593 particles / L) was

fourfold lower than the EC₅₀ of the filtration rate (1956 particles / L) (Everaert et al., 2022). It is however unknown if a hormetic effect, i.e. a biphasic dose-response relationship (Hood et al., 2018; Mattson, 2008), could be observed in organisms exposed to cumulative stressors such as microplastics and climate change, as observed for copepods exposed to UV radiation (Heine et al., 2019), flame retardants (Sun et al., 2022), or cupric iron (Sharp and Stearns, 1997), or if this effect would be observed on their population dynamics. But in any case, our findings indicate that even at low dose exposure to microplastic and elevated temperatures, the cumulative long-term impacts, primarily through reduced energy uptake, can lead to a decreased survival of reproductive adults (Lyu et al., 2022). Given the ecological role of copepods as primary consumers in marine food webs (Gunaalan et al., 2023), a disruption to their population dynamics may have cascading consequences for higher trophic levels and ecosystem function. Overall, our results indicate that the combination of microplastics, including bio-based PLGA, with other anthropogenic stressors, such as global warming, can have adverse effects on harpacticoid copepod populations.

5. Conclusion

Our results indicate that under a warming scenario (+3 °C), poly (lactic-co-glycolic acid) (PLGA) microplastics at concentrations as low as 0.01% microplastics to food content reduced the filtration rate of *Nitokra spinipes*, with an estimated 96-h EC₅₀ of 0.029% microplastics to food content. No effects on filtration rate were observed under the control temperature scenario. Using a DEB-IBM model, we further estimated that equilibrium population density could decrease by 50% at 0.0098% food content under the warming scenario. The population-level EC₅₀ was threefold lower than the individual-level EC₅₀, indicating that *N. spinipes* populations may be more sensitive to microplastic exposure than suggested by short-term individual-level endpoints. These findings inform that the combined exposure to (bio-based) microplastics and warming may have adverse effects on harpacticoid copepod populations, with potential disruptions in the structure and functions of coastal benthic ecosystems. Future research should further investigate the combined and interactive effects between microplastics, from both petroleum-based and bio-based origins, as well as other anthropogenic stressors, to better understand their effects on key organisms within marine and coastal (including benthic) food webs. Such studies will be crucial for predicting risk and mitigating the broader ecological consequences of microplastic pollution in a multiple-stressor environment.

CRedit authorship contribution statement

Zhiyue Niu: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Conceptualization. **Jana Asselman:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Colin R. Janssen:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Charlotte P. Linley:** Writing – review & editing, Methodology, Investigation. **Karel De Schampelaere:** Writing – review & editing, Resources, Methodology. **Gert Everaert:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Ana I. Catarino:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Investigation, Conceptualization.

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Declaration of competing interest

The authors declare no conflict of financial interests/personal relationships which may be considered as potential competing interests.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.marpolbul.2025.118465>.

Data availability

The dataset produced and presented in the work can be found in the official data repository Marine Data Archive, doi:10.14284/722 (Niu et al., 2025).

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