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Antarctic threats: anthropogenic microfibers and plasticizers in the scallop *Adamussium colbecki* from the Ross Sea

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ABSTRACT

Antarctica is increasingly impacted by anthropogenic pollutants, including microplastics (MPs) and associated plastic additives. This study presents the first integrated assessment of microfibres (MFs) and plasticizers in the Antarctic scallop *Adamussium colbecki*, a key benthic species in the Ross Sea. A total of 54 MFs were isolated from 12 samples, with an average of 4.5 ± 2.7 MFs/individual, higher than previously reported for bivalves in the region. Characterized MFs were natural (cotton, wool, cellulose), semi-synthetic, or polyethylene terephthalate (PET), showing strong similarities with textiles used by personnel at the nearby scientific research station. Tissue-specific differences suggest distinct exposure pathways, particularly via filtration in scallop's gills and accumulation in the mantle. Co-occurrence of plastic additives like phthalates (e.g., DEHP, DiBP/DnBP), organophosphate esters (e.g., TPHP) and non-phthalate plasticizers (e.g., DEHA), was confirmed in both scallops and outdoor technical clothing, reinforcing the hypothesis that textile-derived MFs act as chemical vectors. Notably, several additive concentrations in *A.colbecki* exceeded levels reported for bivalves in other global regions, indicating a potential combination of local and long-range sources of contamination. These findings highlight the susceptibility of Antarctic coastal ecosystems to emerging contaminants coming from both local sources and long-range transport and underscore the urgent need for improved wastewater management practices and mitigation strategies within polar research infrastructures.

1. Introduction

Antarctica, long considered a symbol of pristine wilderness, is increasingly affected by human activity and pollution, including plastics. Contaminants released at lower latitudes reach the continent via long-range atmospheric transport and ocean currents (Lacerda et al., 2019; Xie et al., 2022; Cunningham et al., 2022; Da Silva et al., 2023). Additionally, local anthropogenic sources of pollution are increasing. Tourism, fisheries and the increasing number of scientific stations all contribute to the introduction of disturbances, including pollution (Caruso et al., 2024; Da Silva et al., 2023; Balakrishna et al., 2023). In the 2022-2023 season, more than 104,000 tourists visited Antarctica, representing a 40% increase compared to the 2019-2020 season (Bastmeijer et al., 2023). Plastic debris has been observed in Antarctic waters since the 1980s (Barrows et al., 2018) and microplastics (MPs) are now detected even in remote regions like East Antarctica and the Ross Sea (Convey et al., 2002; Barnes et al., 2010; Do Sul et al., 2011; Waller et al., 2017; Cincinelli et al., 2017; Reed et al., 2018; Cunningham et al., 2020; Kelly et al., 2020; Alurralde et al., 2022; Rota et al., 2022). The urgent issue of plastic contamination in Antarctica has been further underscored by the latest Resolution, "Towards ending plastic pollution in the Antarctic Treaty area," which was adopted on July 3, 2025, by the ATCM (Resolution 5 (2025)-ATCM 47, CEP 27, Milan).

The exact pathways through which MPs enter the region, whether via surface and sub-surface ocean currents or atmospheric transport across the Antarctic Circumpolar Current, remain uncertain (Bergmann

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et al., 2019; Lacerda et al., 2019; Marsh and van Sebille, 2021; Alurralde et al., 2022). Point sources of plastic pollution have been identified in terrestrial and marine environments near scientific base stations, with the highest concentrations observed in the Western Peninsula and the Ross Sea (Waller et al., 2017; Reed et al., 2018; Bergami et al., 2020b). Microfibers (MFs) are a major category of MPs, often representing about half of those found in air, sediments and waters (Koelmans et al., 2019). Millions of tonnes are released annually into aquatic environments, mainly from textile production and domestic laundry (Napper and Thompson, 2016; Carney Almroth et al., 2018). To enhance their performance, textiles are often treated with various additives, including plasticizers, flame retardants, heavy metals, organophosphate esters (OPEs), and bisphenols (Licina et al., 2019; Chen et al., 2022). These additives can leach from the fibers during washing, abrasion, or environmental transport, making MFs not only physical pollutants but also vectors of chemical contaminants (Hahladakis et al., 2018). In remote environments like Antarctica, scientific stations and maritime traffic, including research and tourist vessels, have been identified as notable sources of MFs pollution in surrounding marine ecosystems (Barnes et al., 2010; Reed et al., 2018; Alurralde et al., 2022). MFs have been identified in almost all abiotic matrices of the Antarctic marine environment such as sediments from Fildes Bay (Perfetti-Bolaño et al., 2022), sea ice cores sampled in the Weddell Sea (Cunningham et al., 2022) and surface waters from various locations in the Southern Ocean (Cincinelli et al., 2017). This contamination does not exclude marine biota. Bottari et al. (2022) found 37 anthropogenic microparticles (95% MFs) in six emerald rockcod *Trematomus bernacchii* from Terra Nova Bay, Ross Sea. In spiny plunderfish *Harpagifer antarcticus* from King George Island, 89% of gastrointestinal tracts contained MFs (Ergas et al., 2023). MFs were also detected in Gentoo penguin scats from Bird and Signy Islands, indicating their spread through the food web (Bessa et al., 2019). Additional records include Antarctic krill (Wilkie Johnston et al., 2023; Zhu et al., 2023; Lv et al., 2024; Simmons et al., 2025), sponges (Corti et al., 2023; Simmons et al., 2025), gastropods (Bergami et al., 2022; Sfriso et al., 2020) and bivalves (González-Aravena et al., 2024; Gonzalez-Pineda et al., 2024). MFs account for less than 80% in krill and sponges, while bivalves exhibit an amount of MFs higher than 90%. Compared to the other classes, bivalves have shown the highest MPs content per individual on average in invertebrates from the Ross Sea (1.9 items/individual across several species, Sfriso et al., 2020). Despite its geographic isolation and protection within the world's largest Marine Protected Area (MPA), several studies addressed MPs pollution in the Ross Sea finding occurrence in all environmental compartments including biota. Still a significant knowledge gap remains regarding MFs in benthic marine species, which may be more exposed to MFs, associated plastic additives, and other anthropogenic contaminants. Therefore, it's important to further assess the status of emerging pollution in the Ross Sea region as an MPA. In this framework, *A. colbecki* represent a key benthic species in Antarctic marine food webs. This scallop plays a central role in the benthic ecosystem, forming dense shell beds that provide habitat for a wide range of associated organisms (Schiaparelli and Aliani, 2019). As a filter-feeder and prey for Antarctic fishes, it occupies an important position in the trophic chain, acting both as a primary consumer and a prey for higher trophic level. *A. colbecki* is already recognized as a valuable sentinel species and bioindicator for climate change and chemical pollution, including plastics (Regoli et al., 2002; Magi et al., 2004; Grotti et al., 2016). For these reasons, this study investigates, for the first time, the occurrence of MFs along with plastic additives, such as OPEs, phthalates (PAEs), and alternative non-phthalate plasticizers (NPPs), in the soft tissues of the Antarctic scallop *Adamussium colbecki*, collected from the coastal waters of the Ross Sea. Furthermore, to investigate potential local sources of contamination, textiles polymeric composition and plastic additive content were also measured in outdoor technical clothing supplied to researchers and logistics personnel at the scientific station "Mario Zucchelli" (MZS) in the Ross Sea.

2. Materials and methods

2.1. Sampling

Adult specimens of *A.colbecki* were collected using a nylon-based bottom trawl at 150 m depth in the Gerlache Inlet at Terra Nova Bay (74° 38 '03"S 164° 01′ 43"E), near MZS (Fig. 1). The number of specimens collected followed the recommendations of the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) in terms of ethical and sustainable issues, particularly Article II, which states that scientific research should be conducted in a manner that preserves ecological relationships and minimizes impacts on Antarctic marine living resources (CCAMLR Conservation Measures, 2016).

In order to minimize disturbance to benthic communities, and specifically for this species which is abundant in the seabed ecosystems, the number of specimens collected was intentionally limited. Despite the small size, the sample remained representative of the study area. Similar studies conducted in the same region have utilized comparable or even smaller sample sizes, in accordance with the critical requirements of sampling within a MPA (Sfriso et al., 2020; Bottari et al., 2022). Sampling was conducted during two Italian Antarctic expeditions in the austral summers of December 2004 to January 2005 and December 2019 to January 2020. Following sorting and taxonomic identification, specimens with intact shells were placed in sealed sterile plastic bags and stored at − 80 °C for shipping to Italy and kept at the same temperature until analyses.

2.2. MFs analysis

MFs isolation, characterization, and quantification in *A.colbecki* ($n = 12$) were carried out following the protocol established in our previous study on the Antarctic whelk *Neobuccinum eatoni* (Bergami et al., 2023). Six scallops collected during the 2004 and six during 2019 expedition were processed using a thermo-oxidative digestion method. Of 12 scallops, three were used to assess digestion efficiency and three to evaluate recovery rates. Details on morphometric data are described in Supporting information (Tables S1–S2). After shell removal, whole body of each scallop were rinsed with ultrapure Milli-Q water and the mantle and gills dissected, transferred into a 500 mL glass flask containing 200 mL of 30% hydrogen peroxide (H_2O_2) solution for tissue digestion. Samples were incubated at 60 °C for 24 h with gentle agitation. Following digestion, saturated NaCl solution (250 mL, density = 1.2 g cm^{-3}) was added, shaken, and left to settle overnight. The solution was then filtered using a vacuum glass filtration apparatus (Supelco® 58061) equipped with Whatman filter papers (Grade 41, pore size 20 μm, diameter 47 mm). Each filter was first examined under a stereomicroscope to screen the entire surface, followed by detailed observation under an optical microscope (Olympus BX51 with DP50 camera) for microplastic identification and quantification. MFs were counted, imaged, and classified based on their shape and color following the method described by Germanov et al. (2018) in the BASEMAN final report. MFs size was then measured using ImageJ software (https://imagej.nih.gov/ij/). The description of polymeric characterization using μ-Raman and ATR-FTIR spectroscopy of Antarctic equipment and MFs isolated from samples is reported in Supporting informations (paragraph 1.3).

2.3. QA/QC procedures

Airborne contamination during sampling and transport was limited as scallops with intact shells were chosen for the analysis and immediately stored at − 80 °C in sealed sterile plastic bags. To prevent contamination during MFs isolation and quantification from scallops soft tissues, QA/QC procedures were adopted following guidelines from previous studies (Li et al., 2015; Bråte et al., 2018; von Friesen et al., 2019; Dawson et al., 2020; Cho et al., 2021). MFs extraction was carried

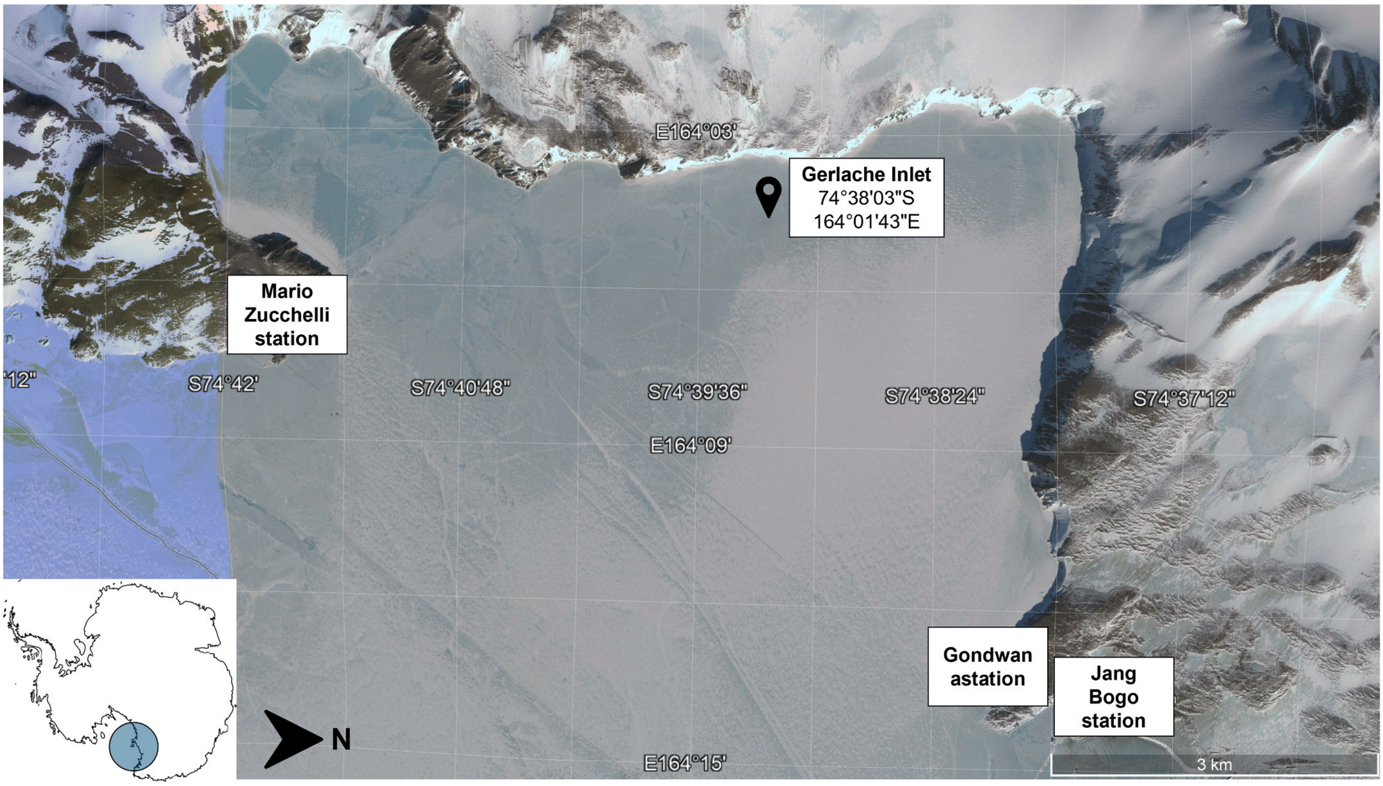

Fig. 1. Sampling site location (Gerlache Inlet 74° 38 '03"S 164° 01′ 43"E) of A.colbecki.

out under a chemical fume hood, in a closed and clean environment, with limited access to a few operators only wearing a white lab coat (100% cotton) and latex gloves. Before and after analysis, all work surfaces were cleaned with ethanol and laboratory paper towels. All the solutions (Milli-Q water, H_2O_2, NaCl) were filtered at 0.2 μm (Whatman cellulose nitrate membrane filters, diameter of 47 mm) using a vacuum glass filtration apparatus kit before use. For the analysis, only glassware or metal containers (i.e., flasks, beakers, petri dishes, filtration apparatus) were used and washed in filtered Milli-Q water before use. To establish any procedural contamination, analytical blanks were processed for each biological sample (24 blanks for 12 samples). Laboratory blanks were essential to normalize the number of MFs extracted from the scallops. Normalization was performed by subtracting from each sample the MFs detected in procedural blanks with matching shape, color, and polymer composition, ensuring a highly conservative estimate of MF content in Antarctic scallops. The extraction method was validated by assessing the digestion efficiency (%) and the recovery rate (%) of red polyethylene terephthalate (PET) microfibers (von Friesen et al., 2019; Dawson et al., 2020). MFs limits of detection (LOD) and quantification (LOQ) were calculated based on Bråte et al. (2018) using 24 laboratory blanks. QA/QC procedures are further described in the Supplementary material (paragraph 1.3).

2.4. *Plastic additives analysis*

2.4.1. *Standards and reagents*

Nineteen OPEs, eleven PAEs, and four NPPs were analyzed in the present study. Eighteen internal standards (IS) were used for quantification. Details of the standards used are reported in the Supplementary Materials (Tables S8–S9). Acetone and hexane were obtained from J.T. Baker (Centre Valley, PA, USA). Methanol, LC-grade water, ammonium acetate, and formic acid were purchased from Merck (Darmstadt, Germany). Glass wool was provided by Panreac AppliChem (Barcelona, Spain).

2.4.2. *Sample preparation*

Samples analyzed include six Antarctic scallops and six types of textiles from outdoor technical clothing used by researchers and logistics personnel at MZS (see Fig. S2 for textiles used to analyse plasticizers and Table S5 for the polymeric characterization of all technical equipment). The methodology applied for sample extraction and quantification was already validated and fully described in Fernandéz-Arribas et al. (2024). The whole body of Antarctic scallops (n = 6) was lyophilized for 24 h. After grinding the samples with a pestle, 1 g dry weight (dw) was extracted by sonication using 15 mL of hexane:acetone (1:1) for 15 min. The extraction was carried out twice and both extracts were combined. Then, the extract was evaporated under a gentle nitrogen stream to change the solvent, and it was reconstituted in 5 mL of hexane: methanol (1:3). The solution was centrifuged for 5 min at 4000 rpm, 20 °C and an aliquot of 200 μL was collected for analysis and spiked with 10 ng of IS mixture. The aliquot was analyzed by turbulent flow chromatography – high-pressure liquid chromatography – tandem mass spectrometry (TFC-HPLC-MS-MS), using the system Thermo Scientific TurboFlow™ (Waltham, MA, USA). Lipid weight (lw) was determined gravimetrically from the remaining 4.8 mL of sample, after evaporating the solvent using a nitrogen stream and drying the sample in an oven at 90 °C until a constant weight was reached. Clothes samples were first cut into small equal pieces and weighed 1 g for each sample, which included: winter suit (outer and inner parts), leggings, winter boots (black and red parts) (Table S5, Fig. S2). After that, 50 ng of OPEs, PAEs and NPPs labelled standards were added and extraction was carried out by sonication for 15 min using hexane:acetone (1:1) until pieces of tissue were covered. The extract was filtered using glass wool and then transferred to another glass tube. Finally, the extract was evaporated under a gentle nitrogen stream and reconstituted in 500 μL of MeOH for subsequent analysis. Instrumental analyses are further described in Supporting information (paragraph 2.1).

2.4.3. *QA/QC procedures*

All the quality parameters of the analytical method for plastic additives such as recoveries, LODs and LOQs have been previously described in the manuscript of Fernandéz-Arribas et al. (2024). To assess any potential background contamination during the plastic additives analysis procedure blanks were included in each batch of samples. These blanks account for any potential contamination from the extraction methodology together with those from the instrumental analysis. The blanks consisted of pre-cleaned cotton T-shirts (3 days of Soxhlet extraction with ethyl acetate) for textile samples and pre-cleaned hydromatrix (ultrasound extraction for 30 min with hexane: acetone (1:1)) for biota samples. For those compounds that were detected in blanks, if its area represented over 20% of the area of the corresponding sample, this sample was considered "not detected", as this concentration could potentially come from the background. Otherwise, in all those cases where the blank area was below this 20% threshold, the quantification of the sample was considered correct. The area of the compounds that were detected in blanks was subtracted from the one of the corresponding samples to correct any potential contamination that was associated with the sample extraction and/or instrumental analysis. Matrix spikes were used to support quantification and verify method performance in the studied matrices.

2.5. Statistical analysis

MFs counts are expressed as mean ± standard deviation (SD), as well as the concentrations of plastic additives, reported as average ± SD of OPEs, PAEs, and NPPs for scallops and textile items (Table 1). Statistical analyses were performed to compare the number of MFs between tissues (mantle vs. gills) and sampling years (2004 vs. 2019). The normality of data distribution was assessed using the Shapiro-Wilk test; since the assumptions for parametric tests were not met, the non-parametric Mann–Whitney *U* test was applied (Fig. 2). For plastic additives, values falling outside the calibration curve (highlighted in red in Table 1) were considered as semi-quantitative; however, they were retained in the dataset and included in the calculation of average concentrations, as they were consistently detected across both biological and textile matrices. Furthermore, a Spearman correlation test was used to evaluate the relationship between common plasticizers found in scallops and textile items. Prior to the analysis, data were log10-transformed when necessary to reduce the influence of extreme values and high variability (e.g., among PAEs and NPPs), which could disproportionately affect rank ordering. For all analyses, the significance level was set at $p < 0.05$. Where applicable, Bonferroni correction was employed to account for multiple comparisons. All statistical analyses were performed using GraphPad Prism version 8.0.1.

3. Results

3.1. MFs isolation

The results of QA/QC procedures are reported in Supplementary Material (Tables S3–S4).

After blank normalization, a total of 54 MFs were isolated in 12 Antarctic scallops, with an average of 4.50 ± 2.74 MFs per individual; 33 MFs were isolated from the mantle and 21 from the gills. In scallops collected in 2004, 19 MFs were found in the mantle and 11 in the gills (with an average number respectively of 3.16 ± 4.02 and 1.83 ± 4.02), whereas in those from 2019, respectively, 14 MFs in the mantle (average 2.33 ± 2.07) and 10 in the gills (1.67 ± 1.63 MFs) (Fig. 2). No statistical difference was found after comparison between the average number of MFs found in scallop's mantle and gills (Mann-Whitney U = 0, p-value = 0.33) and between years of sampling (Mann-Whitney U = 1, p-value = 0.66). The average length of MFs was 2.10 ± 1.15 mm, with a median of 2.16 mm ranging from 0.13 to 4.17 mm. The main colors were respectively black (41%) followed by blue (39%) and red (20%) (Figs. 3 and 4). Between tissues, colors differ slightly with blue fibers predominating in the mantle (52%), followed by black (30%) and red (18%). In the gills the most frequent color was black (57%), followed by red (24%) and blue (19%).

μ-FTIR and μ-Raman analyses conducted on representative MFs, selected by color and isolated from both gill and mantle tissues of scallops sampled in both years, as well as from analytical blanks (n = 13), revealed a predominance of natural polymers such as cotton, wool, and cellulose (46%). These were followed by semi-synthetic fibers (39%) and polyethylene terephthalate (PET; 15%) (Fig. 4 and S1; Table S6). Similarly, ATR-FTIR analysis of textile items from outdoor technical clothing and equipment used in the field (n = 10) identified a wide variety of polymers, including cellulose-based materials (7%) and composites (46%), polyamide (PA 15%), and PET (23%) (Table S5). The details of the polymer characterization are described in the Supporting Informations (paragraph 1.3).

3.2. Plastic additives

Concentration levels of OPEs, PAEs, and NPPs are summarized in Table 1 (ng/g ww and ng/g). Supplementary Table S10–S13 report individual scallop and textile sample concentrations.

In scallops, 13 OPEs were detected. RDP was the most abundant (39%; 43.1 ± 57.8 ng/g ww), followed by T2IPPP (15%; 16.8 ± 22.9 ng/g ww), TEHP (14%; 45.5 ng/g ww), TPHP (12%; 6.76 ± 3.6 ng/g ww), THP (11%; 36.1 ng/g ww), B4IPPPP (3.1%; 3.42 ± 4.34 ng/g ww), 4IPPDPP (2.4%; 2.04 ± 3.15 ng/g ww), DCP (1.4%; 0.81 ± 1.0 ng/g ww), and 2IPPDPP (1.1%; 0.66 ± 1.3 ng/g ww). TCEP, TPPO, TEP, and TPrP contributed <0.5% (Fig. 5).

Among PAEs, DEHP dominated (84%; 265 ± 122 ng/g ww), followed by DiBP-DnBP (16%; 48.9 ± 30.1 ng/g ww). For NPPs, DEHA accounted for 99% of total, with one value out of the calibration curve (highlighted in red) (Table 1). In textiles, 15 OPEs were detected. TBOEP was most abundant (90%), followed by TCEP (7.88%) and TCIPP (1.2%). The rest of the compounds accounted for a small percentage of the total, since several concentrations, mainly in the bootstraps, fell outside the calibration curve. Those results should be considered as semiquantitative values and are potentially overestimated due to the plateau effect of the calibration curve. Highest OPE levels were found in Outer Leggings (PP-blended, 86.64 μg/g), Inner Leggings (17.61 μg/g), Inner Jacket (PET, 14.63 μg/g), and Outer Jacket (PET-blended, 3.01 μg/g). Nine PAEs and three NPPs were found in textiles. DEHP dominated (55%), followed by DiBP-DnBP (37%) and BBzP (5.3%). DEHA represented 99.9% of NPPs. As observed for OPEs, the bootstraps showed the highest levels of contamination, with 304.01 and 255 μg/g tissue for PAEs, and 5287.56 and 3686.1 μg/g for NPPs, respectively (Table S13). Outer Leggings also had high levels: PAEs 191.4 μg/g; NPPs 3381.1 μg/g. Jackets contained 973.4 and 877.7 μg/g NPPs.

Among OPEs, TPHP and 2IPPDPP exhibited the highest mean concentrations in textiles, whereas TEHP, TPHP, and 4IPPDPP were the most abundant in scallops. For PAEs, DEHP was the dominant compound in both scallops and textiles. DiBP-DnBP was also detected in both matrices, while DCHP occurred at lower levels. Among NPPs, DEHA displayed the highest concentrations in both textiles and scallops, and ATBC was likewise detected in both matrices. To assess a potential relationship between plasticizers found in both matrices, a Spearman rank correlation analysis was performed. Due to the presence of several high values (among PAEs and NPPs for example), a log10 transformation of the data was applied. Indeed, the Spearman correlation increased (Correlation coefficient (r) = 0.7862, confident interval (CI) = 95%, *p-value* = 0.0023 (two-tailed) n = 13), indicating a stronger and more robust positive association between variables.

4. Discussion

4.1. MFs

MFs occurrence has been reported in benthic marine fauna from the

Table 1
Average ± SD of OPEs, PAEs and NPPs, expressed in ng/g ww for 6 scallops and ng/g textiles for 6 textile items (the higher values in textile items and scallops are expressed in μg/g to improve clarity and facilitate comparison). nd = not detected, red value = outside the calibration curve.

OPEs	SCALLOPS (ng/g ww)	TEXTILES (ng/g)
	Average ± SD	
TEP	0.06 ± 0.02	8.04 ± 14.1
TCEP	0.29 ± 0.29	37.51 ± 77.33 (μg/g)
TPPO	0.58 ± 0.80	3.73 ± 3.69
TCIPP	nd	5.72 ± 8.71 (μg/g)
TPrP	nd	67.7 ± 111
TDCIPP	nd	20.1 ± 46.7
TPHP	6.76 ± 3.62	692 ± 870
TNBP	nd	362 ± 574
DCP	0.81 ± 0.98	337 ± 558
TBOEP	nd	426.67 ± 625.27 (μg/g)
2IPPDPP	0.66 ± 1.28	472 ± 381
RDP	43.05 ± 57.8	nd
4IPPDPP	2.04 ± 3.15	28.03 ± 24.7
TCP	nd	341 ± 474
EHDPP	nd	3.31 ± 5.6 (μg/g)
B4IPPPP	3.42 ± 4.34	nd
T2IPPP	16.8 ± 22.9	nd
THP	36.2	nd
TEHP	45.5	399 ± 346
PAEs		
DMP	nd	67.8 ± 27.8
DEP	nd	1.94 ± 3.14 (μg/g)
DiBP-DnBP	48.9 ± 30.1	53.95 ± 61.45 (μg/g)
BBzP	nd	11.44 ± 9.02 (μg/g)
DnBP	nd	853 ± 513
DCHP	nd	126 ± 101
DEHP	265 ± 122	79.74 ± 69.77 (μg/g)
DnOP	nd	264.5 ± 110
DiDP	nd	1.02 ± 0.32
NPPs		
ATBC	1.11 ± 0.90	262 ± 325
DEHA	16.64 ± 6.2 (μg/g)	2389 ± 2022.8 (μg/g)
DINCH	nd	69.02 ± 52.1

Ross Sea, yet data on bivalves remain scarce and no studies have examined the potential co-occurrence of plasticizers linked to synthetic textiles used in technical clothing of research station personnel. Textile fibers represented the most abundant MPs category in surface waters and sediments of the Ross Sea (Terra Nova Bay) (Cincinelli et al., 2017; Munari et al., 2017) and globally are among the most commonly detected MPs in marine fauna, widely documented in marine mollusks, especially bivalves, from commercial markets and coastal monitoring studies (Cho et al., 2021; Stefanelli-Silva et al., 2024), including Arctic specimens (Fang et al., 2018). Only three studies reported MPs in Antarctic bivalves: *Cyclocardia astartoides* from the Antarctic Peninsula averaged 0.5 ± 0.7 items/individual (90% MFs) while *Laternula elliptica* from King George Island contained 42.9 ± 25.4 items/individual (94% MFs) (González-Aravena et al., 2024), with the 58 % of items of cellulose and 22 % considered plastic. In the Ross Sea, Sfriso et al. (2020) found an average of 1.9 items/individual across several species (*Aequiyoldia eightsii, Cyamiocardium denticulatum, Thyasira debilis, Yoldiella antarctica*) collected from Mario Zucchelli Station, Adelie Cove, and Icarus Camp locations, with the majority of synthetic polymers In comparison, *A. colbecki* contained 4.5 ± 2.7 MFs/individual, indicating greater exposure or accumulation potential. Variability in MPs number, shape, and size across species likely reflects pollution sources and species-specific traits. For instance, feeding strategies can influence MFs uptake, biodistribution and retention inside the body. *A.colbecki* is an epibenthic suspension feeder, actively resuspending bottom sediments to access organic matter and potentially more exposed to sediment-associated MFs. Body size may also contribute to interspecies variation in MPs' burden. When MPs abundance is normalized by grams of tissue, bivalves from the Antarctic Peninsula (*L. elliptica* (González-Aravena et al., 2024), exhibited higher MFs content compared to *A.colbecki* (1.82 items/g ww vs 0.36 ± 0.28 MF/g ww respectively). However, in the present study, whole scallops wet weight was estimated using an allometric approach (see paragraph 1.2 of Supplementary materials), and therefore the calculation of MPs/g could be an artefact. Despite this, body sizes offers useful information on particle distribution, facilitating comparisons with other studies MFs found in *A.colbecki* were longer (2.10 ± 1.5 mm on average) than those reported in Ross Sea bivalves (0.03–1.00 mm; Sfriso et al., 2020) and the Antarctic Peninsula (0.5–2 mm; Gonzalez-Pineda et al., 2024), possibly due to environmental exposure, buoyancy or species-specific ingestion mechanisms. In our study, MFs were specifically analyzed in gills and mantle, two of the most exposed tissues during filter-feeding in scallops and bivalves. As these tissues are in direct contact with the surrounding water, they are more likely to retain suspended particulate matter, including MPs and MFs. According to the literature, feeding mode appears to influence MPs occurrence more than body size (Fang et al., 2021; Porter et al., 2023). Anatomical features, such as the buccal cavity and digestive tract, also play a key role in mediating MPs exposure. These traits, shared across benthic taxa like bivalves, crustaceans, and nematodes (Fueser et al., 2019; Ward et al., 2019; Carreras-Colom et al., 2024; Pantó et al., 2024), influence not only MPs uptake but also the size of ingested particles, which often correlates with body mass (Jâms et al., 2020; Andrade et al., 2025). Volgare et al. (2022) observed that mussels (*M.galloprovincialis*) can accumulate MFs on their gill filaments, highlighting how the shape of fibers influences their entrapment in gills, where they are not easily expelled. Fibers smaller than 100 μm can penetrate gill tissues and be ingested or translocated into internal organs. Similarly, Auguste et al. (2023) reported that in *M.galloprovincialis,* short fibers (<150 μm) are primarily retained, with gills accumulating more than the digestive gland. Furthermore, MF exposure stimulated extracellular immune responses, indicating induction of immune/inflammatory processes and antioxidant enzyme activities, suggesting oxidative stress conditions and histopathological changes, even at lower concentrations (Auguste et al., 2023). Mladinich et al. (2022) further demonstrated that mussels and oysters tend to reject a higher proportion of longer fibers compared to shorter ones, due to their filtering, feeding, and excretion mechanisms. In this study, mantle tissue exhibits a higher number of MFs than gills (even though not significant), regardless of the sampling period (2004 or 2019). Anatomical differences may account for this pattern. Scallops possess an unfused mantle, a muscular velum, and numerous sensory tentacles that regulate water flow and support swimming (Shumway and Parsons, 2016). Their large, paired gills occupy much of the mantle cavity and serve dual functions, respiration and filter feeding, via ciliary currents capable of trapping suspended particles, including MFs. Unlike mussels and oysters, scallops have mantle tissues that are more directly exposed to the external environment. Moreover, scallops are active swimmers and exhibit selective feeding behavior: their labial palps can reject unwanted particles as pseudofaeces, which are expelled from the mantle cavity, often by vigorous valve clapping, as observed in the Antarctic scallop *A. colbecki* (Bailey et al., 2005). Therefore, the relatively large and open mantle cavity may contribute to greater retention of suspended particles. Importantly, *A.colbecki* is one of the most abundant bivalve species in Antarctic benthic ecosystems and plays a key role as a filter feeder, contributing to particle cycling and energy transfer within the food web (Schiaparelli and Aliani, 2019). As such, the accumulation of anthropogenic MFs in this species may have broader ecological implications, potentially facilitating their transfer to higher trophic levels.These findings underscore the importance of conducting tissue-specific analyses across different taxa to improve the current knowledge on MFs uptake and biodistribution and potential trophic transfer along food chains.

4.2. Plastic additives

Numerous studies confirm that plasticizers and flame-retardant additives have reached polar regions (Fu et al., 2020; Xie et al., 2022). Despite the geographic isolation of the Antarctic region and the strict regulations imposed by the Antarctic Treaty System, increasing human activity and long-range transport have been recognized as a significant source of “emerging organic contaminants” (EOC) (Gao et al., 2018; Fu et al., 2020; Kim et al., 2021; Xie et al., 2022). In Antarctica, plasticizers have likewise been found near research stations. Gao et al. (2018) identified flame retardants and plasticizers like TPHP, TEHP and BBP within 1.3 and 44.4 ng/L in lake and coastal waters of Fildes Peninsula, tracing them to local sources (station wastewaters, aircraft, sewage). Fu et al. (2020) reported several OPEs, including TCEP, TCIPP and TPhP in Antarctic marine organisms from various trophic levels, highlighting bioavailability and trophic magnification patterns (TMFs>1), particularly for chlorinated OPEs. Concentrations measured in fish and benthic invertebrates ranged from 0.02 to 5.4 ng/g ww. Han and Liu (2018) detected DEHA (126,000 ng/g) in Antarctic krill, showing NPPs incorporation into basal food webs. Further research from the Fildes Peninsula reported 11 OPEs at 0.87–15.7 ng/g dw in soils and 9.8–113 ng/g dw in moss and lichens, indicating local anthropogenic inputs (Cheng et al., 2025).

Direct measurements of PAEs and OPEs in the remote Ross Sea remain scarce. Yet legacy POPs (PCBs, HCB, DDTs, and PAHs) have been reported in sponges (Pala et al., 2023) and the bivalve *A.colbecki* (Grotti et al., 2016; Grotti et al., 2016 analyzed archived tissues from *A.colbecki* collected between 1996 and 2009 in Terra Nova Bay, reporting PCB concentrations of ~100–300 ng/g lw and total PAHs ranging from 200 to 500 ng/g lw. Although no clear temporal trends emerged, a spike during the 1997–98 austral summer was attributed to local human activity, confirming the species' value as a sentinel for hydrophobic pollutants in polar benthic ecosystems (Krasnobaev et al., 2020).

The PAEs and OPEs detected here in *A.colbecki,* for some compounds, appear higher than previously reported levels of legacy POP levels in the same region (see Table S12). Newly reported NPPs also indicate ongoing lipophilic contaminant accumulation and possible amplification via MPs ingestion. Laboratory studies support this: *M.galloprovincialis* accumulated 2.7-9.5 MPs/individual (Vega-Herrera et al., 2024) and showed 25–55% higher PFAS bioaccessibility when co-ingesting MPs. Similarly,

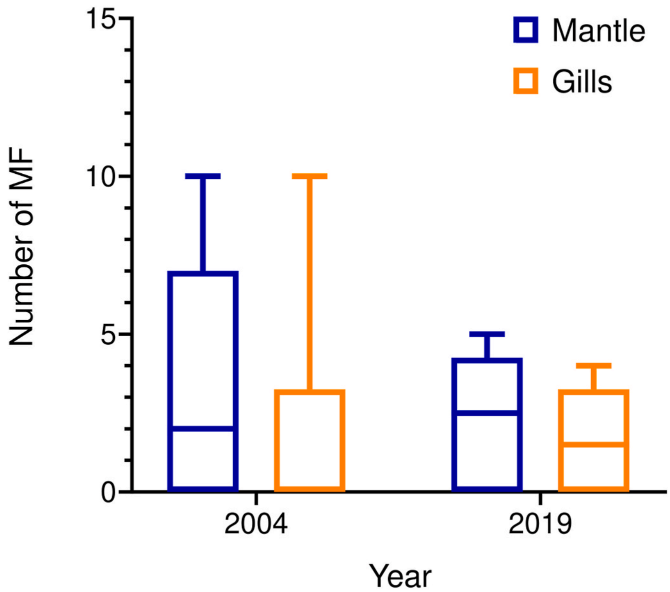

Fig. 2. Comparison of the average number of MFs found in mantle and gill tissues of A.colbecki across the two sampling years (2004 and 2019). No statistical differences were found between tissues or years (non-parametric Mann–Whitney test).

Álvarez-Ruiz et al. (2021) found chronic PE MPs exposure increased diclofenac and PFOS accumulation with slower depuration in *M. galloprovincialis.*

4.3. *Potential source of contamination*

MFs retrieved from scallop tissues were predominantly black and blue (~80%), in agreement with patterns observed in other Antarctic mollusks (González-Aravena et al., 2024; Gonzalez-Pineda et al., 2024; Bergami et al., 2023).

Polymeric characterization showed MFs were primarily natural (46%) and semi-synthetic (39%), while synthetic fibers, such as PET, accounted for 15%, aligning with clothing and equipment at MZS and previous Antarctic and global findings (Suaria et al., 2020; González-Aravena et al., 2024; Gonzalez-Pineda et al., 2024). Several studies have documented MPs contamination in the Ross Sea (Sfriso et al., 2020; Aves et al., 2022; Bergami et al., 2023). Cincinelli et al. (2017) reported MPs of PE and PP (57.1%), followed by PET and other polymers near-shore Road Bay and Tethys Bay areas and offshore of the Ross Sea coast. Similarly, Zhang et al. (2022) identified PET MFs as the most abundant in surface and subsurface waters across 28 stations in the Ross Sea, Dumont d'Urville Sea, Davis Sea, and Prydz Bay. Munari et al. (2017) reported fibers and fragments in sediments, with concentrations increasing closer to MZS. MFs found in Antarctic snow near McMurdo Station in the Ross Sea (Aves et al., 2022) and nanoplastics recovered

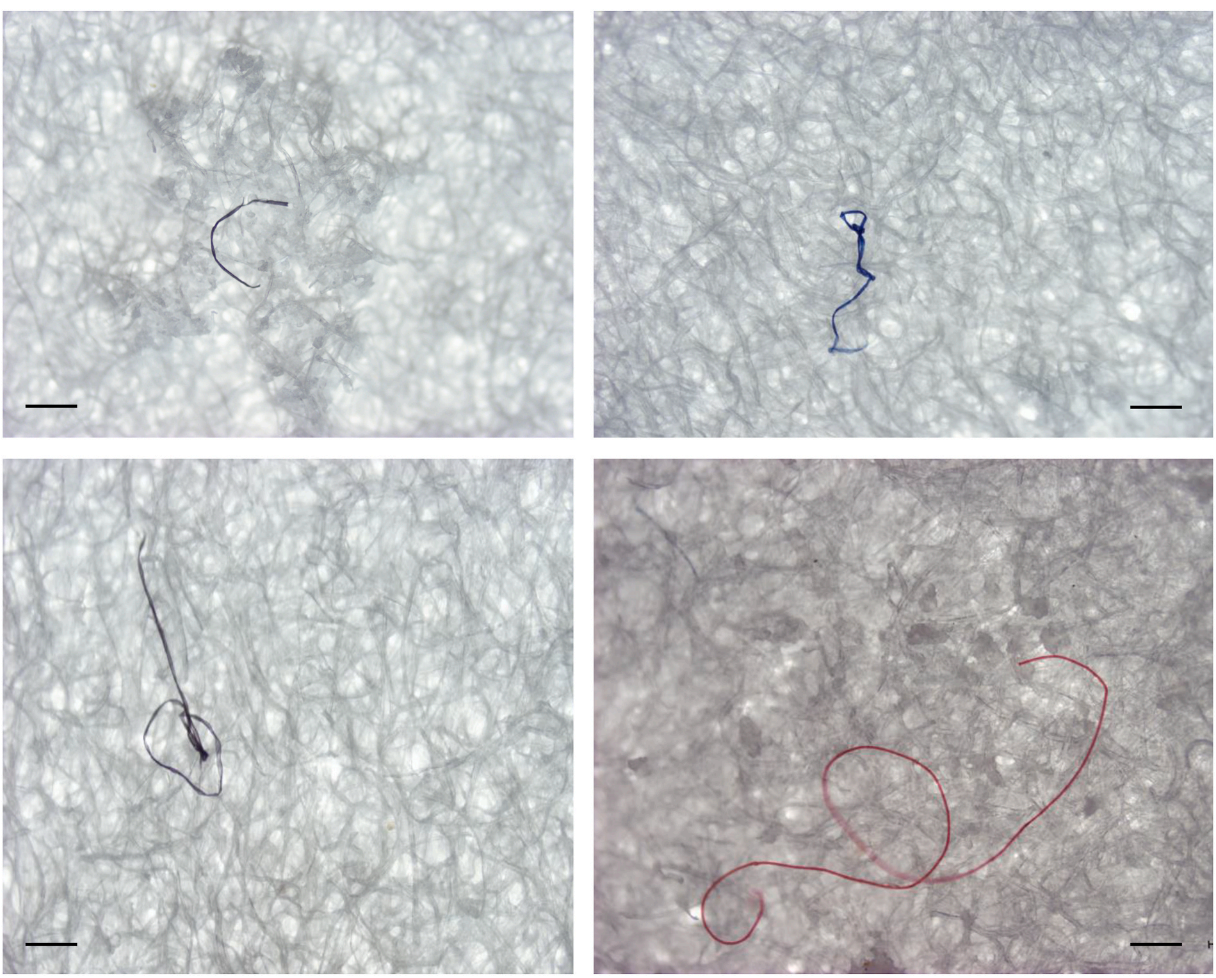

Fig. 3. Representative MFs retrieved from A.colbecki samples. Scale bar: 500 μm.

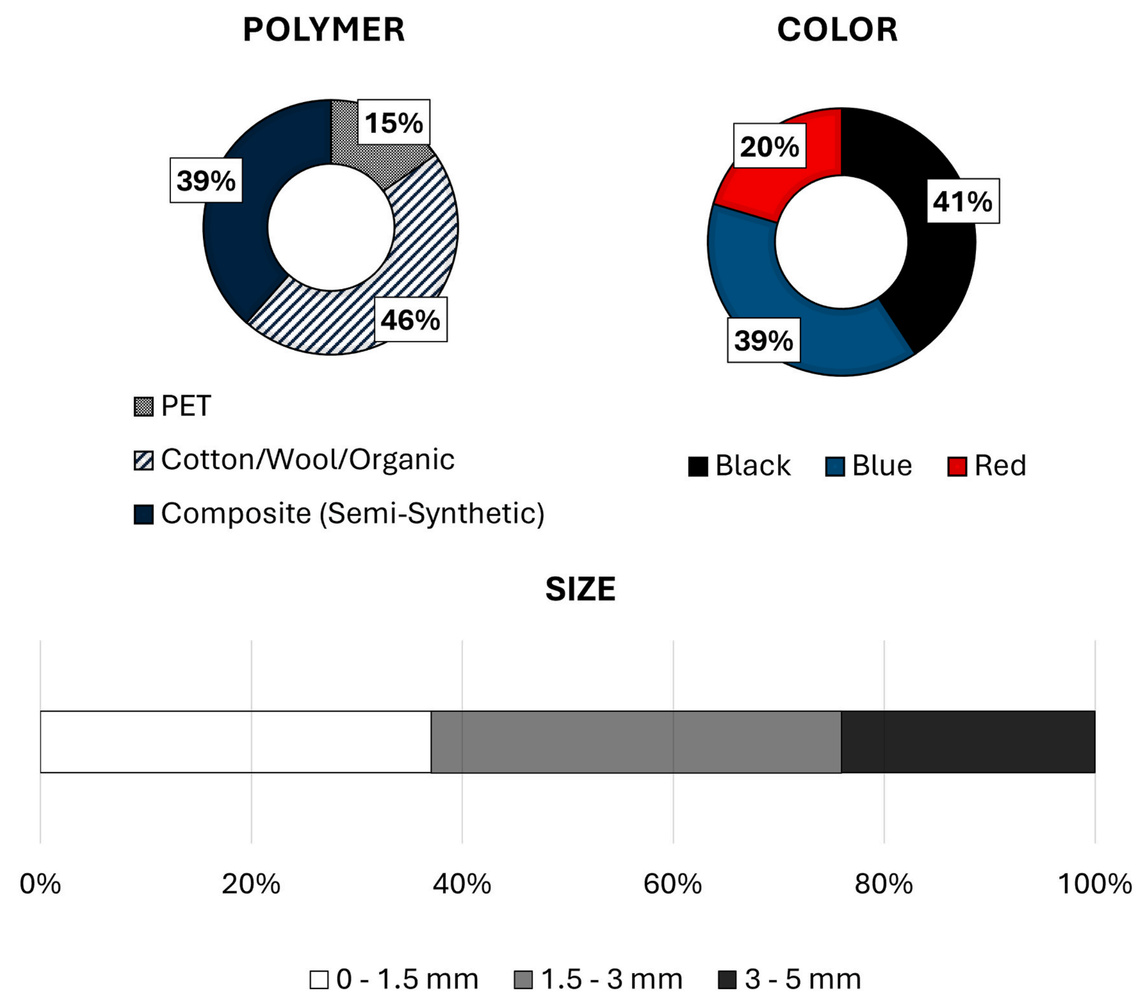

Fig. 4. Features of the MFs isolated from the soft tissues (mantle and gill) of the Antarctic scallop A.colbecki: polymers, colors and size.

from sea ice cores (Materić et al., 2022) further support the hypothesis of local contamination linked to human activities and the use of synthetic textile materials.

Sewage discharges from laundry have long been recognized as a major pathway for MFs release into coastal waters (Rochman, 2015; Napper and Thompson, 2016), and wastewater treatment plants are often inefficient in retaining fibers (Gröndahl et al., 2009; Stark et al., 2016). Moreover, although natural textile fibers such as cotton or rayon are often considered biodegradable, their environmental persistence may be underestimated, especially when chemically treated (Suaria et al., 2020). The co-occurrence of plasticizers in textiles and scallops may suggest a likely release pathway via leaching and washing, followed by bioaccumulation. Filter-feeding bivalves can ingest MFs carrying additives, and their cold, lipid-rich physiology enhances retention. Polar species' slow metabolism and high lipid content amplify uptake of lipophilic contaminants (Chapman and Riddle, 2005; Germanov et al., 2018; Borgå et al., 2022), so even low environmental concentrations can lead to substantial tissue burdens. In this context, the positive association observed between common plasticizers in scallops and textiles (Table 1) is supported by a significant Spearman correlation ($r = 0.7862$, *p-value* $= 0.0023$). The strengthening of the correlation following data transformation in log (10) moreover suggests that the relationship is robust and not driven solely by high-magnitude values. However, this pattern should be interpreted with caution, as correlation alone does not demonstrate causation or a direct transfer pathway. Instead, it may indicate that both matrices are influenced by similar environmental inputs or sources of contamination. Potential contributors could include local anthropogenic activities or more diffuse inputs, although identifying specific sources was beyond the scope of this study. While total OPEs may not consistently biomagnify, some compounds (TEP, TBOEP, T2IPPP) do, whereas others ((TCEP, log Kow 1.4; TPHP, log Kow ~4.6; TEHP, log Kow 9.5)) show species-specific patterns (Sala et al., 2024). The presence of TPHP, TEHP, 2IPPDPP, and 4IPPDPP in *A.colbecki* and textiles indicates dual exposure: direct release from materials and transfer through benthic and pelagic food webs, with high log Kow compounds persisting in lipid-rich tissues but exhibiting variable metabolic fates across species. DEHP reached a mean value of ~265 ng/g ww (~1131 ng/g dw), exceeding levels observed in shellfish from the Pearl River Delta, China, (averaging ~226 ng/g dw) (Gu et al., 2021). In oysters from Florida, ΣPAEs reached up to ~1021 ng/g ww (Lemos et al., 2024), with DEHP reaching 70.37 ng/g ww. Moreover, the presence of this compound has been documented in marine organisms from protected areas, including bivalves, fish and holothurians from a Mediterranean MPA, with bivalves having the highest values in terms of DEHP (mean 2.58 ± 0.55 μg/g ww) (Rios-Fuster et al., 2022). It was also found associated with MFs in the Mediterranean Sea urchin *Paracentrotus lividus* (Raguso et al., 2022). DEHP is a widespread plasticizer used in flexible PVC, flooring and coatings (Zolfaghari et al., 2014; Aldegunde-Louzao et al., 2024). Since it is not chemically bound to polymers, it can easily leach during use or laundering, with urban wastewater treatment plants being a primary source into aquatic environments (Zolfaghari et al., 2014). Our analysis revealed high concentrations of DEHP in textile items (mean:79.74 ± 69.77 μg/g), particularly in PP-blended leggings and bootstraps (Table S13). Also, DiBP-DnBP phthalate esters, extensively used as plasticizers in polymers (Lorber and Koch, 2013), were detected at relevant concentrations both

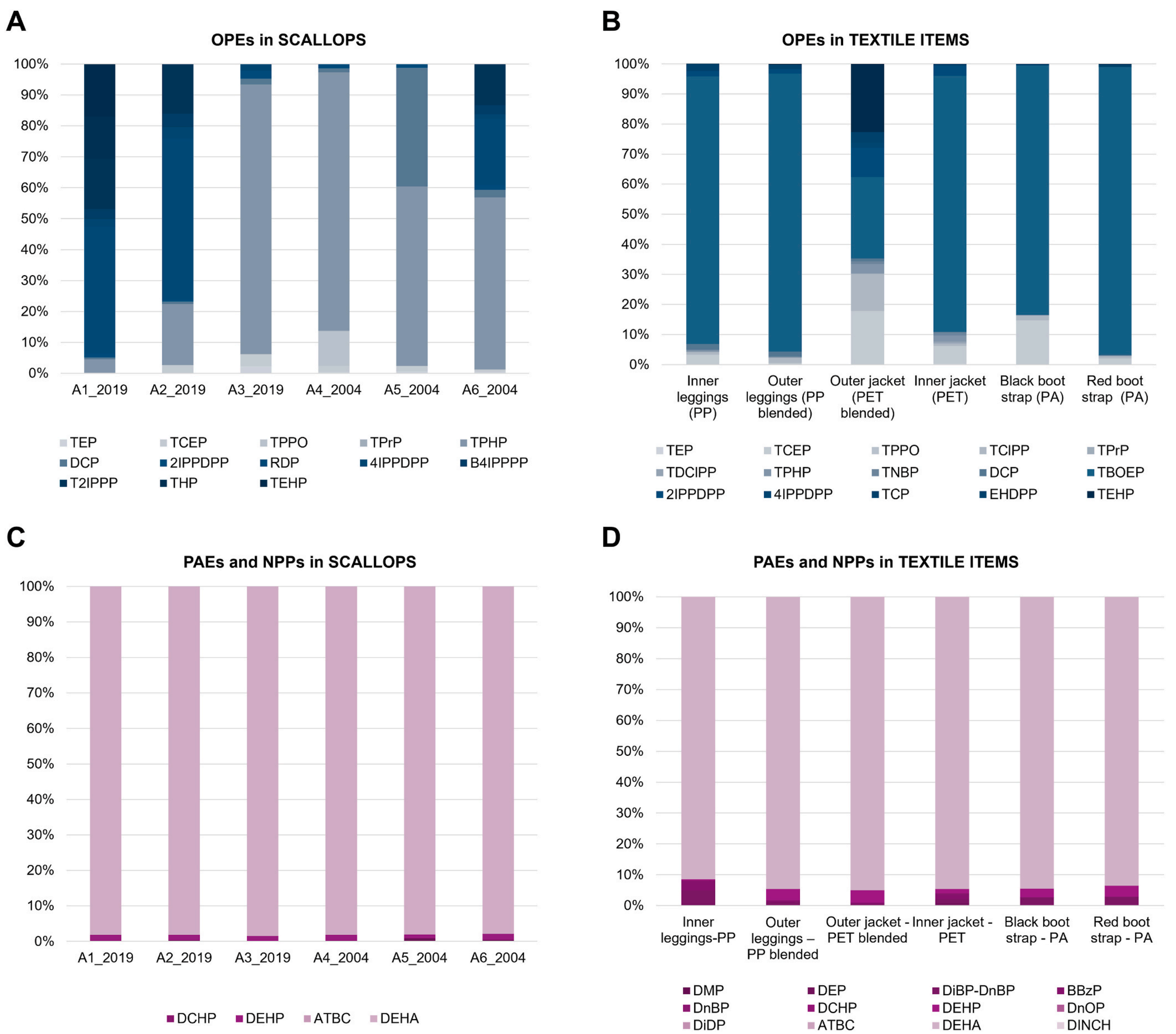

Fig. 5. 100% stacked columns for OPEs percentage in each scallop (A) and textile item (B), and for PAEs and NPPs (C, D), illustrating the relative contribution of each compound to the total profile within each matrix.

in scallops (48.9 ± 30.1 ng/g ww) and textiles (53.95 ± 61.45 μg/g) (Table 1), indicating a broader spectrum of potentially leachable additives present in these items. OPEs such as TPHP and TEHP, commonly used as flame retardants and plasticizers in PVC, synthetic fibers, and polyurethane foams (Wang et al., 2025), were among the most abundant compounds detected in *A.colbecki*. TPHP reach a mean concentration of 6.76 ± 3.6 ng/g ww (29.8 ± 15.7 ng/g dw), while TEHP reached 45.5 ng/g ww (210 ng/g dw in one sample) (see Tables S10–S11). These values exceeded those reported for *M.galloprovincialis* from the Bay of Marseille (2.4 ng/g dw; Castro-Jiménez and Ratola, 2020) and fall within the range observed for Spanish mussels (11–291 ng/g dw; Castro et al., 2020). TPHP was also highly concentrated in textile items (692 ± 870 ng/g; Table 1), especially in red bootstraps (2298 ng/g), while TEHP showed lower mean levels (399 ± 346 ng/g) but remained consistently present. Although both compounds tend to biodegrade relatively quickly in the aquatic environment and do not meet the criteria for persistence (van der Veen and de Boer, 2012), their widespread use leads to frequent detection. Among Isopropylated Triaryl Phosphates (ITPs), 2IPPDPP and 4IPPDPP were consistently found in scallops and textiles (Table 1). 2IPPDPP averaged 0.656 ± 1.3 ng/g ww in scallop tissue and 472 ± 381 ng/g in textiles, peaking at 1222 ng/g in PP-blended leggings, while 4IPPDPP showed higher levels in biota (2.04 ± 3.15 ng/g ww) but lower concentrations in the same textile (28.03 ± 24.7 ng/g) (Table S13). Data on bivalves are limited, but these compounds have also been detected in fish, turtles, and marine mammals, linking them to ingestion of plastic debris or prey (Sala et al., 2021, 2022; Dettoto et al., 2024). TCEP, a widespread OPE in textiles (mean 37.51 ± 77.33 μg/g), was especially high in the PET lining of a jacket (863 ng/g) and exceeded quantification limits in the bootstraps. In scallops, it was lower (0.29 ± 0.3 ng/g ww; 1.14 ± 0.3 ng/g dw) but detected in all specimens, indicating consistent exposure. Historically used in polyurethane foams, polyester resins, and fabrics, TCEP leaches easily due to non-covalent bonding and is frequently reported in marine organisms (Sala et al., 2022; Aminot et al., 2023). For example, Choi et al. (2020) found wide variability in OPE concentrations in oysters (6.12–206 ng/g dw, mean 40.9 ng/g dw), with highest levels near Weizhou Island, where $\sum$11OPEs reached 309 ± 60 ng/g dw, including TCIPP, TBOEP, TDCIPP, and TCEP.

RDP, a newer flame retardant increasingly used as a substitute for regulated OPEs like TPHP and TCIPP (Huang et al., 2020), was the most abundant compound in *A.colbecki* (mean 43.1 ± 57.8 ng/g ww; 190 ± 274 ng/g dw). Widely used in electronics, resins, and plastics, it is ubiquitous in the environment and highly toxic to aquatic organisms (van der Veen and de Boer, 2012). Few studies have reported its presence in marine biota (Castro et al., 2020), and it was not detected in textiles here, suggesting alternative inputs such as long-range transport or trophic bioaccumulation. In this study, higher RDP levels have been observed in scallops sampled in 2019 (Table S10) compared to 2004. However, RDP shows moderate bioaccumulation potential and at the same time its environmental persistence is limited (e.g. relatively short half-lives in water), and also the limited sample size complicates the interpretation. Thus, increased RDP concentration in 2019 could reflect both enhanced long-range input and trophic accumulation. Similarly, B4IPPP, T2IPPP, and THP were detected only in scallops (Table 1), indicating the need for further research on their sources and behaviour in polar food webs.

Among NPPs, DEHA showed exceptionally high concentrations in *A. colbecki* (16.64 ± 6.2 μg/g ww; 72.3 ± 23 μg/g dw) and in textiles (average 2389 ± 2022.8 μg/g) (Table 1), highlighting the need for further investigation. Widely used to enhance flexibility in plastics as a safer alternative to DEHP (Xu et al., 2019), DEHA is released during production, use, and disposal, and is ubiquitous in air, dust, and aquatic environments (Boor et al., 2015; Fromme et al., 2016; Behairy et al., 2021). Although generally low-toxicity, high-dose exposure can affect hepatic, reproductive, and developmental endpoints (Wato et al., 2009; Behairy et al., 2021). Another NPP, ATBC, was detected in scallops (1.11 ± 0.90 ng/g ww) and textiles (262 ± 325 ng/g). Commonly used in food packaging, cosmetics, adhesives, and inks (Johnson, 2002; Bui et al., 2016), it may have endocrine-disrupting and metabolic effects (Sheikh and Beg, 2019; Zhang et al., 2023) and has been reported in wild fish and environmental matrices (Dettoto et al., 2024; Fernández-Arribas et al., 2024). Overall, evidence from measured experiments and regulatory reviews indicates low bioaccumulation for both NPPs. Despite the high log Kow of DEHA (>6.11 to 8.39) (OECD, 2000), most authors attribute a low bioaccumulation due to rapid metabolism/biotransformation in aquatic species. However, there is some evidence suggesting DEHA may persist and cause adverse effects in certain invertebrates, and further research is needed in this area (California Office of Environmental Health Hazard Assessment, 2003). There's some evidence ATBC can appear in animal tissues, but the literature is mixed. Modelled BCFs range from low-moderate to high depending on assumptions, while few experimental studies indicate rapid absorption and metabolism with limited long-term retention (Dettoto et al., 2024).

Despite the 2019 Antarctic Treaty recommendations to limit the use of personal care products containing microbeads, the role of wastewater treatment facilities as sources of MFs pollution in Antarctic coastal environments remains largely overlooked. The widespread detection of plasticizers in *A.colbecki* underscores the risk of combined exposure. Target actions are urgently needed to quantify and mitigate inputs of MFs and leachable additives from research stations. Given the ecological importance of the Antarctic scallop *A.colbecki* and the need to preserve the unique biodiversity of the Ross Sea Marine Protected Area, further monitoring efforts across diverse benthic taxa and trophic levels are essential to assess the long-term ecological consequences of this contamination.

4.4. Study limitations

Some limitations of this study should be considered. Although relevant for the MPA, more studies on scallops from other Antarctic marine coastal areas will reinforce the hypothesis that bivalves are potential targets of MFs and chemical additives. In addition, polymer and additive characterization was performed on a subset of the observed particles following standard QA/QC practices in microplastic studies, which may have led to an underestimation of the total diversity of polymer present. Therefore, the results should be interpreted as a preliminary assessment, highlighting the need for further investigations with larger sample sizes and broader spatial coverage.

5. Conclusions

This study provides the first evidence of MFs and associated plasticizers in gills and mantle of *A. colbecki* from the Ross Sea. MF abundance exceeded previous records for local bivalves and was comparable to levels found in specimens from the Antarctic Peninsula. Tissue-specific patterns suggest different exposure routes, while polymer analysis revealed cellulose, semi-synthetic, and PET fibers closely matching textiles used at MZS, pointing to research stations as local MF sources. The co-occurrence of plastic additives, particularly phthalates (DEHP, DiBP/DnBP), OPEs (TPHP), and alternatives such as DEHA, in scallops and textiles, together with the positive Spearman correlation observed between the two matrices, suggests a possible environmental link between these contaminants and textile-derived MFs. Notably, concentrations of several plasticizers detected in *A.colbecki* were higher than those reported in marine bivalves from other regions worldwide, which may indicate the influence of local sources in addition to long-range transport. However, the limited number of samples analyzed for plasticizer quantification constrains the ability to robustly identify contamination sources and patterns. Further analyses of plasticizers in surrounding water and sediments will help clarify exposure pathways and the potential vector role of MFs.

Overall, these findings highlight the need for further investigation into the dynamics of MF uptake, clearance and retention at the tissue level, as well as their potential physiological effects. In this context, our results contribute to the growing body of evidence supporting the objectives of ATCM Resolution 5 (2021), which calls for improved monitoring and mitigation of microplastic pollution in Antarctica. Strengthening wastewater management and implementing targeted mitigation strategies at research stations may therefore represent important steps toward reducing local inputs and supporting ongoing conservation efforts in Antarctic ecosystems.

CRediT authorship contribution statement

Emma Ferrari: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Maria Vittoria Barbieri:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis. **Roberta Russo:** Writing – original draft, Methodology, Formal analysis. **Aleix Balasch García:** Writing – review & editing, Methodology, Formal analysis. **Valentina Ferrari:** Formal analysis. **Silvia Simonetti:** Writing – review & editing. **Giovanni Birarda:** Formal analysis. **Lisa Vaccari:** Writing – review & editing, Formal analysis. **Elisa Bergami:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Investigation, Data curation, Conceptualization. **Ethel Eljarrat:** Writing – review & editing, Methodology, Investigation, Data curation, Conceptualization. **Ilaria Corsi:** Writing – review & editing, Writing – original draft, Visualization, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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The graphical abstract was designed by Tatiana Rusconi.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2026.124412.

Data availability

Data will be made available on request.

References

Aldegunde-Louzao, N., Lolo-Aira, M., Herrero-Latorre, C., 2024. Fast phthalate detection in textile samples: a LC-MS/MS screening method using precursor ion scans. Chem. Proc. 16, 1. https://doi.org/10.3390/ecsoc-28-20150.

Alurralde, G., Isla, E., Fuentes, V., Olariaga, A., Maggioni, T., Rimondino, G., Tatián, M., 2022. Anthropogenic microfibres flux in an Antarctic coastal ecosystem: the tip of an iceberg? Mar. Pollut. Bull. 175, 113388. https://doi.org/10.1016/j.marpolbul.2022.113388.

Álvarez-Ruiz, R., Picó, Y., Campo, J., 2021. Bioaccumulation of emerging contaminants in mussel (*Mytilus galloprovincialis*): Influence of microplastics. Sci. Total Environ. 796, 149006. https://doi.org/10.1016/j.scitotenv.2021.149006.

Aminot, Y., Tao, L., Héas-Moisan, K., Pollono, C., O'Loghlin, M., Munschy, C., 2023. Organophosphate esters (OPEs) in the marine environment: spatial distribution and profiles in French coastal bivalves. Chemosphere 330, 138702. https://doi.org/10.1016/j.chemosphere.2023.138702.

Andrade, C., Sepúlveda, T., Pinto, B., Rivera, C., Aldea, C., Urbina, M., 2025. The feeding mode effect: influence on particle ingestion by four invertebrates from Sub-Antarctic and Antarctic waters. Environ. Sci. Pollut. Res. 32, 8318–8339. https://doi.org/10.1007/s11356-025-36144-6.

Antarctic Treaty database - Resolution 5 (2025)- ATCM 47 - CEP 27, Milan). https://www.ats.aq/devAS/Meetings/Measure/853.

Auguste, M., Leonessi, M., Bozzo, M., Risso, B., Cutroneo, L., Prandi, S., Kokalj, A.J., Drobne, D., Canesi, L., 2023. Multiple responses of *Mytilus galloprovincialis* to plastic microfibers. Sci. Total Environ. 890, 164318. https://doi.org/10.1016/j.scitotenv.2023.164318.

Aves, A.R., Revell, L.E., Gaw, S., Ruffell, H., Schuddeboom, A., Wotherspoon, N.E., LaRue, M., McDonald, A.J., 2022. First evidence of microplastics in Antarctic snow. Cryosphere 16, 2127–2145. https://doi.org/10.5194/tc-16-2127-2022.

Bailey, D.M., Johnston, I.A., Peck, L.S., 2005. Invertebrate muscle performance at high latitude: swimming activity in the Antarctic scallop, *Adamussium colbecki*. Polar Biol. 28, 464–469. https://doi.org/10.1007/s00300-004-0699-9.

Balakrishna, K., Praveenkumarreddy, Y., Nishitha, D., Gopal, C.M., Shenoy, J.K., Bhat, K., Khare, N., Dhangar, K., Kumar, M., 2023. Occurrences of UV filters, endocrine disruptive chemicals, alkyl phenolic compounds, fragrances, and hormones in the wastewater and coastal waters of Antarctica. Environ. Res. 222, 115327. https://doi.org/10.1016/j.envres.2023.115327.

Barnes, D.K., Walters, A., Gonçalves, L., 2010. Macroplastics at sea around Antarctica. Mar. Environ. Res. 70, 250–252. https://doi.org/10.1016/j.marenvres.2010.05.006.

Barrows, A.P.W., Cathey, S.E., Petersen, C.W., 2018. Marine environment microfiber contamination: global patterns and the diversity of microparticle origins. Environ. Pollut. 237, 275–284. https://doi.org/10.1016/j.envpol.2018.02.062.

Bastmeijer, K., Shibata, A., Steinhage, I., Ferrada, L.V., Bloom, E.T., 2023. Regulating Antarctic Tourism: The challenge of consensus-based decision making. Am. J. Int. Law 117, 651–676. https://doi.org/10.1017/ajil.2023.34.

Behairy, A., Abd El-Rahman, G.I., Aly, S.S.H., Fahmy, E.M., Abd-Elhakim, Y.M., 2021. Di (2-ethylhexyl) adipate plasticizer triggers hepatic, brain, and cardiac injury in rats: mitigating effect of *Peganum harmala* oil. Ecotoxicol. Environ. Saf. 208, 111620. https://doi.org/10.1016/j.ecoenv.2020.111620.

Bergami, E., Manno, C., Cappello, S., Vannuccini, M.L., Corsi, I., 2020. Nanoplastics affect moulting and faecal pellet sinking in Antarctic krill (*Euphausia superba*) juveniles. Environ. Int. 143, 105999. https://doi.org/10.1016/j.envint.2020.105999.

Bergami, E., Emerenciano, A.K., Pinto, L.P., Joviano, W.R., Font, A., de Godoy, T.A., Silva, J., González-Aravena, M., Corsi, I., 2022. Behavioural, physiological and molecular responses of the Antarctic fairy shrimp *Branchinecta gaini* (Daday, 1910) to polystyrene nanoplastics. NanoImpact 28, 100437. https://doi.org/10.1016/j.impact.2022.100437.

Bergami, E., Ferrari, E., Löder, M.G., Birarda, G., Laforsch, C., Vaccari, L., Corsi, I., 2023. Textile microfibers in wild Antarctic whelk *Neobuccinum eatoni* (Smith, 1875) from Terra Nova Bay (Ross Sea, Antarctica). Environ. Res. 216, 114487. https://doi.org/10.1016/j.envres.2022.114487.

Bergmann, M., Mützel, S., Primpke, S., Tekman, M.B., Trachsel, J., Gerdts, G., 2019. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. Sci. Adv. 5. https://doi.org/10.1126/sciadv.aax1157 eaax1157.

Bessa, F., Ratcliffe, N., Otero, V., Sobral, P., Marques, J.C., Waluda, C.M., Trathan, P.N., Xavier, J.C., 2019. Microplastics in gentoo penguins from the Antarctic region. Sci. Rep. 9, 14191. https://doi.org/10.1038/s41598-019-50621-2.

Boor, B.E., Liang, Y., Crain, N.E., Järnström, H., Novoselac, A., Xu, Y., 2015. Identification of phthalate and alternative plasticizers, flame retardants, and unreacted isocyanates in infant Crib Mattress covers and foam. Environ. Sci. Technol. Lett. 2, 89–94. https://doi.org/10.1021/acs.estlett.5b00039.

Borgå, K., McKinney, M.A., Routti, H., Fernie, K.J., Giebichenstein, J., Hallanger, I., Muir, D.C.G., 2022. The influence of global climate change on accumulation and toxicity of persistent organic pollutants and chemicals of emerging concern in Arctic food webs. Environ. Sci. Process. Impacts 24, 1544–1576. https://doi.org/10.1039/D1EM00469G.

Bottari, T., Nibali, V.C., Branca, C., Grotti, M., Savoca, S., Romeo, T., Spanò, N., Azzaro, M., Greco, S., D'Angelo, G., 2022. Anthropogenic microparticles in the emerald rockcod *Trematomus bernacchii* (Nototheniidae) from the Antarctic. Sci. Rep. 12, 17214. https://doi.org/10.1038/s41598-022-21670-x.

Bråte, I.L.N., Hurley, R., Iversen, K., Beyer, J., Thomas, K.V., Steindal, C.C., Green, N.W., Olsen, M., Lusher, A., 2018. *Mytilus* spp. as sentinels for monitoring microplastic pollution in Norwegian coastal waters: a qualitative and quantitative study. Environ. Pollut. 243, 383–393. https://doi.org/10.1016/j.envpol.2018.08.077.

Bui, T.T., Giovanoulis, G., Cousins, A.P., Magnér, J., Cousins, I.T., de Wit, C.A., 2016. Human exposure, hazard and risk of alternative plasticizers to phthalate esters. Sci. Total Environ. 541, 451–467. https://doi.org/10.1016/j.scitotenv.2015.09.036.

California Office of Environmental Health Hazard Assessment, 2003. DEHA Responses. https://oehha.ca.gov/sites/default/files/media/downloads/water/chemicals/deharesponses.pdf.

Carney Almroth, B.M., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles: a source of microplastics released into the environment. Environ. Sci. Pollut. Control Ser. 25, 1191–1199. https://doi.org/10.1007/s11356-017-0528-7.

Carreras-Colom, E., Follesa, M.C., Carugati, L., Mulas, A., Bellodi, A., Cau, A., 2024. Marine macro-litter mass outweighs biomass in trawl catches along abyssal seafloors of Sardinia channel (Italy). Environ. Sci. Pollut. Control Ser. 31, 43405–43416. https://doi.org/10.1007/s11356-024-33909-3.

Caruso, G., Azzaro, M., Dell'Acqua, O., Papale, M., Lo Giudice, A., Laganà, P., 2024. Plastic polymers and antibiotic resistance in an Antarctic environment (Ross Sea): are we revealing the tip of an iceberg? Microorganisms 12, 2083. https://doi.org/10.3390/microorganisms12102083.

Castro, V., Montes, R., Quintana, J.B., Rodil, R., Cela, R., 2020. Determination of 18 organophosphorus flame retardants/plasticizers in mussel samples by matrix solid-phase dispersion combined with liquid chromatography-tandem mass spectrometry. Talanta 208, 120470. https://doi.org/10.1016/j.talanta.2019.120470.

Castro-Jiménez, J., Ratola, N., 2020. An innovative approach for the simultaneous quantitative screening of organic plastic additives in complex matrices in marine coastal areas. Environ. Sci. Pollut. Control Ser. 27, 11450–11457. https://doi.org/10.1007/s11356-020-08069-9.

91-05 CCAMLR Conservation Measures, 2016 [WWW Document], n.d. URL. https://cm.ccamlr.org/en/measure-91-05-2016.

Chapman, P.M., Riddle, M.J., 2005. Polar marine toxicology—future research needs. Mar. Pollut. Bull. 50, 905–908. https://doi.org/10.1016/j.marpolbul.2005.06.001.

Chen, Y., Chen, Q., Zhang, Q., Zuo, C., Shi, H., 2022. An overview of chemical additives on (Micro)Plastic fibers: occurrence, release, and health risks. Rev. Environ. Contam. (formerly: Residue Rev. 260. https://doi.org/10.1007/s44169-022-00023-9.

Cheng, G., Li, R., Xu, Y., Hou, C., Jia, X., Li, B., Gao, H., Jin, S., Kong, L., Na, G., 2025. Occurrence and fugacity model simulation of organophosphate esters in atmosphere-soil-vegetation, Fildes peninsula, Antarctica. J. Environ. Sci. 157, 330–339. https://doi.org/10.1016/j.jes.2024.05.043.

Cho, Y., Shim, W.J., Jang, M., Han, G.M., Hong, S.H., 2021. Nationwide monitoring of microplastics in bivalves from the coastal environment of Korea. Environ. Pollut. 270, 116175. https://doi.org/10.1016/j.envpol.2020.116175.

Choi, W., Lee, S., Lee, H.-K., Moon, H.-B., 2020. Organophosphate flame retardants and plasticizers in sediment and bivalves along the Korean coast: occurrence, geographical distribution, and a potential for bioaccumulation. Mar. Pollut. Bull. 156, 111275. https://doi.org/10.1016/j.marpolbul.2020.111275.

Cincinelli, A., Scopetani, C., Chelazzi, D., Lombardini, E., Martellini, T., Katsoyiannis, A., Fossi, M.C., Corsolini, S., 2017. Microplastics in the surface waters of the Ross Sea (Antarctica): occurrence, distribution and characterization by FTIR. Chemosphere 175, 391–400. https://doi.org/10.1016/j.chemosphere.2017.02.024.

Convey, P., Barnes, D., Morton, A., 2002. Debris accumulation on oceanic island shores of the Scotia Arc, Antarctica. Polar Biol. 25, 612–617. https://doi.org/10.1007/s00300-002-0391-x.

Corti, A., Pagano, G., Giudice, A.L., Papale, M., Rizzo, C., Azzaro, M., Vinciguerra, V., Castelvetro, V., Giannarelli, S., 2023. Marine sponges as bioindicators of pollution by synthetic microfibers in Antarctica. Sci. Total Environ. 902, 166043. https://doi.org/10.1016/j.scitotenv.2023.166043.

Cunningham, E.M., Ehlers, S.M., Dick, J.T.A., Sigwart, J.D., Linse, K., Dick, J.J., Kiriakoulakis, K., 2020. High abundances of microplastic pollution in deep-sea sediments: evidence from Antarctica and the Southern Ocean. Environ. Sci. Technol. 54, 13661–13671. https://doi.org/10.1021/acs.est.0c03441.

Cunningham, E.M., Rico Seijo, N., Altieri, K.E., Audh, R.R., Burger, J.M., Bornman, T.G., Fawcett, S., Gwinnett, C.M., Osborne, A.O., Woodall, L.C., 2022. The transport and fate of microplastic fibres in the Antarctic: the role of multiple global processes. Front. Mar. Sci. 9, 1056081. https://doi.org/10.3389/fmars.2022.1056081.

da Silva, J., Taniguchi, S., Colabuono, F.I., Leonel, J., Dalla Rosa, L., Secchi, E.R., Borges, J.C.G., Siciliano, S., Acevedo, J., Aguayo-Lobo, A., 2023. Mobilization of persistent organic pollutants in humpback whales: insights from feeding areas in the

Antarctic Peninsula and Strait of Magellan to migration, breeding, and calving grounds along the Brazilian coast. Mar. Pollut. Bull. 194, 115448. https://doi.org/10.1016/j.marpolbul.2023.115448.
Dawson, A.L., Motti, C.A., Kroon, F.J., 2020. Solving a sticky situation: microplastic analysis of lipid-rich tissue. Front. Environ. Sci. 8. https://doi.org/10.3389/fenvs.2020.563565.
Dettoto, C., Maccantelli, A., Barbieri, M.V., Baini, M., Fernández-Arribas, J., Panti, C., Giani, D., Galli, M., Eljarrat, E., Fossi, M.C., 2024. Plasticizer levels in four fish species from the Ligurian Sea and Central Adriatic Sea (Mediterranean Sea) and potential risk for human consumption. Sci. Total Environ. 954, 176442. https://doi.org/10.1016/j.scitotenv.2024.176442.
do Sul, J.A.I., Barnes, D.K., Costa, M.F., Convey, P., Costa, E.S., Campos, L.S., 2011. Plastics in the Antarctic environment: are we looking only at the tip of the iceberg? Oecologia Australis 15, 150–170. https://doi.org/10.4257/oeco.2011.1501.11.
Ergas, M., Figueroa, D., Paschke, K., Urbina, M.A., Navarro, J.M., Vargas-Chacoff, L., 2023. Cellulosic and microplastic fibers in the Antarctic fish *Harpagifer antarcticus* and Sub-Antarctic *Harpagifer bispinis*. Mar. Pollut. Bull. 194, 115380. https://doi.org/10.1016/j.marpolbul.2023.115380.
Fang, C., Zheng, R., Zhang, Y., Hong, F., Mu, J., Chen, M., Song, P., Lin, L., Lin, H., Le, F., Bo, J., 2018. Microplastic contamination in benthic organisms from the Arctic and sub-Arctic regions. Chemosphere 209, 298–306. https://doi.org/10.1016/j.chemosphere.2018.06.101.
Fang, C., Zheng, R., Hong, F., Jiang, Y., Chen, J., Lin, H., Lin, L., Lei, R., Bailey, C., Bo, J., 2021. Microplastics in three typical benthic species from the Arctic: occurrence, characteristics, sources, and environmental implications. Environ. Res. 192, 110326. https://doi.org/10.1016/j.envres.2020.110326.
Fernández-Arribas, J., Callejas-Martos, S., Balasch, A., Moreno, T., Eljarrat, E., 2024. Simultaneous analysis of several plasticizer classes in different matrices by on-line turbulent flow chromatography-LC-MS–MS/MS. Anal. Bioanal. Chem. 416, 6957–6972. https://doi.org/10.1007/s00216-024-05593-2.
Fromme, H., Schütze, A., Lahrz, T., Kraft, M., Fembacher, L., Siewering, S., Burkardt, R., Dietrich, S., Koch, H.M., Völkel, W., 2016. Non-phthalate plasticizers in German daycare centers and human biomonitoring of DINCH metabolites in children attending the centers (LUPE 3). Int. J. Hyg Environ. Health 219, 33–39. https://doi.org/10.1016/j.ijheh.2015.08.002.
Fu, Jie, Fu, K., Gao, K., Li, H., Xue, Q., Chen, Y., Wang, L., Shi, J., Fu, Jianjie, Zhang, Q., Zhang, A., Jiang, G., 2020. Occurrence and trophic magnification of organophosphate esters in an antarctic ecosystem: insights into the shift from legacy to emerging pollutants. J. Hazard Mater. 396, 122742. https://doi.org/10.1016/j.jhazmat.2020.122742.
Fueser, H., Mueller, M.-T., Weiss, L., Höss, S., Traunspurger, W., 2019. Ingestion of microplastics by nematodes depends on feeding strategy and buccal cavity size. Environ. Pollut. 255, 113227. https://doi.org/10.1016/j.envpol.2019.113227.
Gao, X., Huang, C., Rao, K., Xu, Y., Huang, Q., Wang, F., Ma, M., Wang, Z., 2018. Occurrences, sources, and transport of hydrophobic organic contaminants in the waters of Fildes Peninsula, Antarctica. Environ. Pollut. 241, 950–958. https://doi.org/10.1016/j.envpol.2018.06.025.
Germanov, E.S., Marshall, A.D., Bejder, L., Fossi, M.C., Loneragan, N.R., 2018. Microplastics: no small problem for filter-feeding megafauna. Trends Ecol. Evol. 33, 227–232. https://doi.org/10.1016/j.tree.2018.01.005.
González-Aravena, M., Rotunno, C., Cárdenas, C.A., Torres, M., Morley, S.A., Hurley, J., Caro-Lara, L., Pozo, K., Galban, C., Rondon, R., 2024. Detection of plastic, cellulosic micro-fragments and microfibers in *Laternula elliptica* from King George Island (Maritime Antarctica). Mar. Pollut. Bull. 201, 116257. https://doi.org/10.1016/j.marpolbul.2024.116257.
Gonzalez-Pineda, M., Salvadó, H., Avila, C., 2024. Do antarctic bivalves present microdebris? The case of Livingston Island. Environ. Pollut. 351, 124086. https://doi.org/10.1016/j.envpol.2024.124086.
Gröndahl, F., Sidenmark, J., Thomsen, A., 2009. Survey of wastewater disposal practices at Antarctic research stations. Polar Res. 28, 298–306. https://doi.org/10.1111/j.1751-8369.2008.00056.x.
Grotti, M., Pizzini, S., Abelmoschi, M.L., Cozzi, G., Piazza, R., Soggia, F., 2016. Retrospective biomonitoring of chemical contamination in the marine coastal environment of Terra Nova Bay (Ross Sea, Antarctica) by environmental specimen banking. Chemosphere 165, 418–426. https://doi.org/10.1016/j.chemosphere.2016.09.049.
Gu, Y.-Y., Wei, Q., Wang, L.-Y., Zhang, Z.-M., Zhang, X.-Q., Sun, A.-L., Chen, J., Shi, X.-Z., 2021. A comprehensive study of the effects of phthalates on marine mussels: bioconcentration, enzymatic activities and metabolomics. Mar. Pollut. Bull. 168, 112393. https://doi.org/10.1016/j.marpolbul.2021.112393.
Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E., Purnell, P., 2018. An overview of chemical additives present in plastics: migration, release, fate and environmental impact during their use, disposal and recycling. J. Hazard Mater. 344, 179–199. https://doi.org/10.1016/j.jhazmat.2017.10.014.
Han, X., Liu, D., 2018. Di(2-ethylhexyl) adipate (DEHA) detection in Antarctic krill (Euphasia superba Dana). Polar Res. 37, 1457395. https://doi.org/10.1080/17518369.2018.1457395.
Huang, Y., Tan, H., Li, L., Yang, L., Sun, F., Li, J., Gong, X., Chen, D., 2020. A broad range of organophosphate tri- and di-esters in house dust from Adelaide, South Australia: concentrations, compositions, and human exposure risks. Environ. Int. 142, 105872. https://doi.org/10.1016/j.envint.2020.105872.
Jâms, I.B., Windsor, F.M., Poudevigne-Durance, T., Ormerod, S.J., Durance, I., 2020. Estimating the size distribution of plastics ingested by animals. Nat. Commun. 11, 1594. https://doi.org/10.1038/s41467-020-15406-6.
Johnson, W., 2002. Final report on the safety assessment of acetyl triethyl citrate, acetyl tributyl citrate, acetyl trihexyl citrate, and acetyl trioctyl citrate. Int. J. Toxicol. 21 (2), 1–17. https://doi.org/10.1080/10915810290096504.
Kelly, A., Lannuzel, D., Rodemann, T., Meiners, K.M., Auman, H.J., 2020. Microplastic contamination in east Antarctic sea ice. Mar. Pollut. Bull. 154, 111130. https://doi.org/10.1016/j.marpolbul.2020.111130.
Kim, J.-T., Choi, Y.-J., Barghi, M., Kim, J.-H., Jung, J.-W., Kim, K., Kang, J.-H., Lammel, G., Chang, Y.-S., 2021. Occurrence, distribution, and bioaccumulation of new and legacy persistent organic pollutants in an ecosystem on King George Island, maritime Antarctica. J. Hazard Mater. 405, 124141. https://doi.org/10.1016/j.jhazmat.2020.124141.
Koelmans, B., Henderson, L., Group, S.W., 2019. A Scientific Perspective on Microplastics in Nature and Society. https://doi.org/10.26356/microplastics.
Krasnobaev, A., ten Dam, G., Boerrigter-Eenling, R., Peng, F., van Leeuwen, S.P.J., Morley, S.A., Peck, L.S., van den Brink, N.W., 2020. Legacy and emerging persistent organic pollutants in antarctic benthic invertebrates near rothera point, Western antarctic peninsula. Environ. Sci. Technol. 54, 2763–2771. https://doi.org/10.1021/acs.est.9b06622.
Lacerda, A.L. d F., Rodrigues, L., dos, S., Van Sebille, E., Rodrigues, F.L., Ribeiro, L., Secchi, E.R., Kessler, F., Proietti, M.C., 2019. Plastics in sea surface waters around the Antarctic Peninsula. Sci. Rep. 9, 3977. https://doi.org/10.1038/s41598-019-40311-4.
Lemos, L.S., Di Perna, A.C., Steinman, K.J., Robeck, T.R., Quinete, N.S., 2024. Assessment of phthalate esters and physiological biomarkers in bottlenose dolphins (*Tursiops truncatus*) and killer whales (*Orcinus orca*). Animals 14, 1488. https://doi.org/10.3390/ani14101488.
Li, J., Yang, D., Li, L., Jabeen, K., Shi, H., 2015. Microplastics in commercial bivalves from China. Environ. Pollut. 207, 190–195. https://doi.org/10.1016/j.envpol.2015.09.018.
Licina, D., Morrison, G.C., Bekö, G., Weschler, C.J., Nazaroff, W.W., 2019. Clothing-mediated exposures to chemicals and particles. Environ. Sci. Technol. 53, 5559–5575. https://doi.org/10.1021/acs.est.9b00272.
Lorber, M., Koch, H.M., 2013. Development and application of simple pharmacokinetic models to study human exposure to di-n-butyl phthalate (DnBP) and diisobutyl phthalate (DiBP). Environ. Int. 59, 469–477. https://doi.org/10.1016/j.envint.2013.07.010.
Lv, L., Feng, W., Cai, J., Zhang, Y., Jiang, J., Liao, D., Yan, C., Sui, Y., Dong, X., 2024. Enrichment characteristics of microplastics in antarctic benthic and pelagic fish and krill near the Antarctic Peninsula. Sci. Total Environ. 951, 175582. https://doi.org/10.1016/j.scitotenv.2024.175582.
Magi, E., Di Carro, M., Rivaro, P., 2004. Analysis of butyltin compounds by gas chromatography–mass spectrometry: an application to the Antarctic bivalve *Adamussium colbecki*. Appl. Organomet. Chem. 18, 646–652. https://doi.org/10.1002/aoc.701.
Marsh, R., van Sebille, E., 2021. Chapter 8 - from the Southern Ocean to Antarctica and its changing ice shelves. In: Marsh, R., van Sebille, E. (Eds.), Ocean Currents. Elsevier, pp. 303–373. https://doi.org/10.1016/B978-0-12-816059-6.00006-1.
Materić, D., Kjær, H.A., Vallelonga, P., Tison, J.-L., Röckmann, T., Holzinger, R., 2022. Nanoplastics measurements in Northern and Southern polar ice. Environ. Res. 208, 112741. https://doi.org/10.1016/j.envres.2022.112741.
Mladinich, K., Holohan, B.A., Shumway, S.E., Brown, K., Ward, J.E., 2022. Determining the properties that govern selective ingestion and egestion of microplastics by the blue mussel (*Mytilus edulis*) and Eastern oyster (*Crassostrea virginica*). Environ. Sci. Technol. 56, 15770–15779. https://doi.org/10.1021/acs.est.2c06402.
Munari, C., Infantini, V., Scoponi, M., Rastelli, E., Corinaldesi, C., Mistri, M., 2017. Microplastics in the sediments of Terra Nova Bay (Ross Sea, Antarctica). Mar. Pollut. Bull. 122, 161–165. https://doi.org/10.1016/j.marpolbul.2017.06.039.
Napper, I.E., Thompson, R.C., 2016. Release of synthetic microplastic plastic fibres from domestic washing machines: effects of fabric type and washing conditions. Mar. Pollut. Bull. 112, 39–45. https://doi.org/10.1016/j.marpolbul.2016.09.025.
Pala, N., Jiménez, B., Roscales, J.L., Bertolino, M., Baroni, D., Figuerola, B., Avila, C., Corsolini, S., 2023. First evidence of legacy chlorinated POPs bioaccumulation in Antarctic sponges from the Ross sea and the South Shetland Islands. Environ. Pollut. 329, 121661. https://doi.org/10.1016/j.envpol.2023.121661.
Pantó, G., Aguilera Dal Grande, P., Vanreusel, A., Van Colen, C., 2024. Fauna – Microplastics interactions: empirical insights from benthos community exposure to marine plastic waste. Mar. Environ. Res. 200, 106664. https://doi.org/10.1016/j.marenvres.2024.106664.
Perfetti-Bolaño, A., Araneda, A., Muñoz, K., Barra, R.O., 2022. Occurrence and distribution of microplastics in soils and intertidal sediments at fildes Bay, maritime Antarctica. Front. Mar. Sci. 8. https://doi.org/10.3389/fmars.2021.774055.
Porter, A., Godbold, J.A., Lewis, C.N., Savage, G., Solan, M., Galloway, T.S., 2023. Microplastic burden in marine benthic invertebrates depends on species traits and feeding ecology within biogeographical provinces. Nat. Commun. 14, 8023. https://doi.org/10.1038/s41467-023-43788-w.
Raguso, C., Grech, D., Becchi, A., Ubaldi, P.G., Lasagni, M., Guala, I., Saliu, F., 2022. Detection of microplastics and phthalic acid esters in sea urchins from Sardinia (Western Mediterranean Sea). Mar. Pollut. Bull. 185, 114328. https://doi.org/10.1016/j.marpolbul.2022.114328.
Reed, S., Clark, M., Thompson, R., Hughes, K.A., 2018. Microplastics in marine sediments near Rothera research station, Antarctica. Mar. Pollut. Bull. 133, 460–463. https://doi.org/10.1016/j.marpolbul.2018.05.068.
Regoli, F., Nigro, M., Chiantore, M., Winston, G.W., 2002. Seasonal variations of susceptibility to oxidative stress in *Adamussium colbecki*, a key bioindicator species for the Antarctic marine environment. Sci. Total Environ. 289, 205–211. https://doi.org/10.1016/S0048-9697(01)01047-6.

Rios-Fuster, B., Alomar, C., Paniagua González, G., Garcinuño Martínez, R.M., Soliz Rojas, D.L., Fernández Hernando, P., Deudero, S., 2022. Assessing microplastic ingestion and occurrence of bisphenols and phthalates in bivalves, fish and holothurians from a Mediterranean marine protected area. Environ. Res. 214, 114034. https://doi.org/10.1016/j.envres.2022.114034.

Rochman, C.M., 2015. The complex mixture, fate and toxicity of chemicals associated with plastic debris in the marine environment. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 117–140. https://doi.org/10.1007/978-3-319-16510-3_5.

Rota, E., Bergami, E., Corsi, I., Bargagli, R., 2022. Macro-and microplastics in the Antarctic environment: ongoing assessment and perspectives. Environments 9, 93. https://doi.org/10.3390/environments9070093.

Sala, B., Balasch, A., Eljarrat, E., Cardona, L., 2021. First study on the presence of plastic additives in loggerhead sea turtles (*Caretta caretta*) from the Mediterranean Sea. Environ. Pollut. 283, 117108. https://doi.org/10.1016/j.envpol.2021.117108.

Sala, B., Giménez, J., Fernández-Arribas, J., Bravo, C., Lloret-Lloret, E., Esteban, A., Bellido, J.M., Coll, M., Eljarrat, E., 2022. Organophosphate ester plasticizers in edible fish from the Mediterranean Sea: marine pollution and human exposure. Environ. Pollut. 292, 118377. https://doi.org/10.1016/j.envpol.2021.118377.

Sala, B., Garcia-Garin, O., Eljarrat, E., 2024. From the depths to the apex: tracing the organophosphate ester journey through marine food webs. Sci. Total Environ. 955, 177228. https://doi.org/10.1016/j.scitotenv.2024.177228.

Schiaparelli, S., Aliani, S., 2019. Oceanographic moorings as year-round laboratories for investigating growth performance and settlement dynamics in the Antarctic scallop *Adamussium colbecki* (EA Smith, 1902). PeerJ 7, e6373. https://doi.org/10.7717/peerj.6373.

Sfriso, A.A., Tomio, Y., Rosso, B., Gambaro, A., Sfriso, A., Corami, F., Rastelli, E., Corinaldesi, C., Mistri, M., Munari, C., 2020. Microplastic accumulation in benthic invertebrates in Terra Nova bay (Ross Sea, Antarctica). Environ. Int. 137, 105587. https://doi.org/10.1016/j.envint.2020.105587.

Sheikh, I.A., Beg, M.A., 2019. Structural characterization of potential endocrine disrupting activity of alternate plasticizers di-(2-ethylhexyl) adipate (DEHA), acetyl tributyl citrate (ATBC) and 2,2,4-trimethyl 1,3-pentanediol diisobutyrate (TPIB) with human sex hormone-binding globulin. Reprod. Toxicol. 83, 46–53. https://doi.org/10.1016/j.reprotox.2018.11.003.

Shumway, S.E., Parsons, G.J., 2016. Scallops: Biology, Ecology, Aquaculture, and Fisheries. Elsevier.

Simmons, N.E., Barnes, D.K.A., Scourse, J.D., Whitaker, J.M., Garza, T.N., Janosik, A.M., 2025. Quantifying microplastics concentration of invertebrates from three Antarctic fjords. Mar. Pollut. Bull. 212, 117503. https://doi.org/10.1016/j.marpolbul.2024.117503.

Stark, J.S., Corbett, P.A., Dunshea, G., Johnstone, G., King, C., Mondon, J.A., Power, M. L., Samuel, A., Snape, I., Riddle, M., 2016. The environmental impact of sewage and wastewater outfalls in Antarctica: an example from Davis station, East Antarctica. Water Res. 105, 602–614. https://doi.org/10.1016/j.watres.2016.09.026.

Stefanelli-Silva, G., Friedemann, P., Rocha de Moraes, B., Ando, R.A., Campos, L. de Siqueira, Petti, M. Angélica Varella, Smith, C.R., Sumida, P.Y.G., 2024. Bottom-Feeders eat their fiber: ingestion of anthropogenic microdebris by antarctic deep-sea invertebrates depends on feeding ecology. Environ. Sci. Technol. 58, 22355–22367. https://doi.org/10.1021/acs.est.4c09487.

Suaria, G., Perold, V., Lee, J.R., Lebouard, F., Aliani, S., Ryan, P.G., 2020. Floating macro- and microplastics around the Southern Ocean: results from the Antarctic Circumnavigation Expedition. Environ. Int. 136, 105494. https://doi.org/10.1016/j.envint.2020.105494.

van der Veen, I., de Boer, J., 2012. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere 88, 1119–1153. https://doi.org/10.1016/j.chemosphere.2012.03.067.

Vega-Herrera, A., Savva, K., Lacoma, P., Santos, L.H.M.L.M., Hernández, A., Marmelo, I., Marques, A., Llorca, M., Farré, M., 2024. Bioaccumulation and dietary bioaccessibility of microplastics composition and cocontaminants in Mediterranean mussels. Chemosphere 363, 142934. https://doi.org/10.1016/j.chemosphere.2024.142934.

Volgare, M., Santonicola, S., Cocca, M., Avolio, R., Castaldo, R., Errico, M.E., Gentile, G., Raimo, G., Gasperi, M., Colavita, G., 2022. A versatile approach to evaluate the occurrence of microfibers in mussels *Mytilus galloprovincialis*. Sci. Rep. 12, 21827. https://doi.org/10.1038/s41598-022-25631-2.

von Friesen, L.W., Granberg, M.E., Hassellöv, M., Gabrielsen, G.W., Magnusson, K., 2019. An efficient and gentle enzymatic digestion protocol for the extraction of microplastics from bivalve tissue. Mar. Pollut. Bull. 142, 129–134. https://doi.org/10.1016/j.marpolbul.2019.03.016.

Waller, C.L., Griffiths, H.J., Waluda, C.M., Thorpe, S.E., Loaiza, I., Moreno, B., Pacherres, C.O., Hughes, K.A., 2017. Microplastics in the Antarctic marine system: an emerging area of research. Sci. Total Environ. 598, 220–227. https://doi.org/10.1016/j.scitotenv.2017.03.283.

Wang, X., Xue, Y., Zhang, X., Wang, J., Xia, K., Liu, W., Xie, Z., Liu, R., Liu, Q., 2025. Secondary organophosphate esters: a review of environmental source, occurrence, and human exposure. Crit. Rev. Environ. Sci. Technol. 55, 241–263. https://doi.org/10.1080/10643389.2024.2399968.

Ward, J.E., Rosa, M., Shumway, S.E., 2019. Capture, ingestion, and egestion of microplastics by suspension-feeding bivalves: a 40-year history. Anthropocene Coasts 2, 39–49. https://doi.org/10.1139/anc-2018-0027.

Wato, E., Asahiyama, M., Suzuki, A., Funyu, S., Amano, Y., 2009. Collaborative work on evaluation of ovarian toxicity 9) effects of 2- or 4-week repeated dose studies and fertility study of di(2-ethylhexyl)adipate (DEHA) in female rats. J. Toxicol. Sci. 34, SP101–SP109. https://doi.org/10.2131/jts.34.S101.

Wilkie Johnston, L., Bergami, E., Rowlands, E., Manno, C., 2023. Organic or junk food? Microplastic contamination in Antarctic krill and salps. R. Soc. Open Sci. 10, 221421. https://doi.org/10.1098/rsos.221421.

Xie, Z., Wang, P., Wang, X., Castro-Jiménez, J., Kallenborn, R., Liao, C., Mi, W., Lohmann, R., Vila-Costa, M., Dachs, J., 2022. Organophosphate ester pollution in the oceans. Nat. Rev. Earth Environ. 3, 309–322. https://doi.org/10.1038/s43017-022-00277-w.

Xu, H., Musi, B., Wang, Z., Zhou, T., Huang, Q., Liu, J., Li, T., Jiang, Z., Liao, S., Jill, G., Koo, E., 2019. Systemic toxicity of di (2-ethylhexyl) adipate (DEHA) in rats following 28-day intravenous exposure. Regul. Toxicol. Pharmacol. 104, 50–55. https://doi.org/10.1016/j.yrtph.2019.02.016.

Zhang, S., Zhang, W., Ju, M., Qu, L., Chu, X., Huo, C., Wang, J., 2022. Distribution characteristics of microplastics in surface and subsurface Antarctic seawater. Sci. Total Environ. 838, 156051. https://doi.org/10.1016/j.scitotenv.2022.156051.

Zhang, Y., Zhang, Z., Zhu, S., Liu, L., Liu, X., Yang, X., 2023. Acetyl tributyl citrate exposure at seemingly safe concentrations induces adverse effects in different genders of type 2 diabetes mice, especially brain tissue. Toxics 11, 877. https://doi.org/10.3390/toxics11100877.

Zhu, W., Liu, W., Chen, Y., Liao, K., Yu, W., Jin, H., 2023. Microplastics in Antarctic krill (*Euphausia superba*) from Antarctic region. Sci. Total Environ. 870, 161880. https://doi.org/10.1016/j.scitotenv.2023.161880.

Zolfaghari, M., Drogui, P., Seyhi, B., Brar, S.K., Buelna, G., Dubé, R., 2014. Occurrence, fate and effects of Di (2-ethylhexyl) phthalate in wastewater treatment plants: a review. Environ. Pollut. 194, 281–293. https://doi.org/10.1016/j.envpol.2014.07.014.