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Assessing microplastic contamination and polymer hazard in preserved clams marketed in Italy and Spain

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ABSTRACT

This study investigated microplastics in commercially preserved bivalves from Italy and Spain, including glass-jarred clams, canned clams, and canned cockles, analysing both edible tissue and preservation medium to reflect actual consumer exposure. Microplastics were found in all samples, with glass-jarred clams showing a load ten times higher (1.64 ± 0.27 MP g^{-1}) than canned cockles (0.11 ± 0.02 MP g^{-1}) and canned clams (0.09 ± 0.02 MP g^{-1}). Fibres were the most common type, mainly 5–500 μm in size and predominantly blue or transparent. Glass-jarred clams contained polyurethane, blue paint residues, polyamide, ABS, PVC, and epoxy resins, while canned products contained polyamide-based resins and polyacrylic acid. Polymer Hazard Indexes ranged from 217964 to 2324522, with glass-jarred clams having the highest rating. These findings indicate that contamination may occur during packaging and processing, with species-specific factors influencing uptake. The presence of high-hazard polymers emphasises the importance of standardised analytical methods, improved packaging, and updated dietary exposure data.

1. Introduction

Plastic materials are widely used in manufacturing, packaging, and consumer products due to their durability, affordability, and versatility. Plastic's ubiquity has resulted in a wide distribution of plastic debris in the environment, including microplastics (MPs), which are defined as plastic particles smaller than 5 mm and found in marine and freshwater systems, marine organisms, food, and human tissues (Smith et al., 2018). The estimated annual human intake of MPs ranges from 10000 to 1000000 particles, depending on dietary habits and geographical region (Kannan & Vimalkumar, 2021). Seafood, in particular, is a key component of many diets and provides essential nutrients such as high-quality protein, long-chain omega-3 polyunsaturated fatty acids (EPA and DHA), iodine, selenium, and vitamin D, all of which contribute to cardiovascular, neurological, and developmental health; however, they represent the foods most susceptible to microplastic contamination (FAO/WHO, 2021; Fred-Ahmadu et al., 2024; Tocher et al., 2019; Unuofin & Igwaran, 2023). In particular, filter-feeding bivalves in marine ecosystems accumulate microplastics from both natural and anthropogenic sources, such as packaging and processing materials (Mercogliano et al., 2020; Smith et al., 2018). Nevertheless, considerable uncertainties about the sources of MP contamination remain, including exposure to machinery, processing steps, and product type (Unuofin & Igwaran, 2023). For instance, pressurised steam sterilisation in metal tins may contribute to the release of MPs from polymer linings, whereas mechanical handling of raw shellfish may introduce plastic debris from conveyor belts, filters, or protective equipment. Furthermore, shell removal may expose the flesh, altering the distribution and retention of ingested MPs, influencing the levels detected in the final product (Vdovchenko & Resmini, 2024; EFSA et al., 2025). These variables are extremely important to consumers, regulators, and industry stakeholders but have received little attention. Indeed, few authors have focused on preserved products like jarred or canned bivalves (Hantoro et al., 2019; Mercogliano et al., 2020). Coatings, adhesives, sealants, and processing equipment have been identified as potential sources of MPs that may leach into food matrices or preservation liquids. As a result,

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when quantifying MP-related product contamination, both edible tissue and preservation liquids must be considered (Silva et al., 2024).

Furthermore, MPs are not chemically inert but rather active participants in microbial ecology, which has important implications for food safety and public health. Indeed, they may transport adsorbed pollutants, leach monomers or additives, and host microbial biofilms (the plastisphere), which may contain various microorganisms, including potential pathogens and antibiotic-resistant bacteria (Kannan & Vimalkumar, 2021; Wu et al., 2024). The complexities of MP contamination define it as a connection of environmental pollution, food safety, and public health, representing the One Health approach.

The European Food Safety Authority (EFSA) stated that current data on MPs and nanoplastics (NPs) in food are insufficient to establish a tolerable daily intake. On the other hand, the presence of particles smaller than 100 μm, which may penetrate the intestinal barrier, raises concerns and needs further investigation (EFSA, 2021). The European Union has laws on materials that come into contact with food, including Regulation (EC) 1935/2004, Regulation (EC) 2023/2006, and Regulation (EU) 10/2011. These laws primarily address chemical migration, but they do not include MPs. The present study aims at investigating the occurrence and polymeric composition of MPs in three types of commercially available preserved bivalves from Italy and Spain, comparing different packagings (glass jars and metal cans) and species (clams and cockles) to identify potential contamination pathways relevant to food safety and public health. Both the edible portion and the preservation medium were analysed, providing an assessment of potential contamination pathways. This study aims at creating a foundation for consumer risk assessments by measuring MP concentrations, determining polymer types, and using a Polymer Hazard Index (PHI) approach (Adam et al., 2019; Lithner et al., 2011). It may also contribute to international efforts for the development of evidence-based data for MP monitoring and regulatory policies that support the One Health approach.

2. Materials and methods

2.1. Sampling

Before sampling, the top 3 leading GDO chains in Spain and Italy were detected. For each GDO, a market research was performed aimed at selecting the brands of bivalve preserves sold in all 3 detected GDOs in each region, ensuring a representative variety of popular products currently available in Italian and Spanish retail markets. Forty-five samples of clam preserves were bought, in particular fifteen Italy-produced glass-jarred clams (GJC) labelled as "*vongole al naturale*" contained in glass jars with aluminium lids with preservation medium, weighing 130 g and 190 g; fifteen samples of canned cockles (CCo) labelled as "*berberechos natural*" produced in Spain, contained in metal tins with preservation medium, each weighing 111 g; fifteen samples of canned clams (CCl) labelled as "*almejas natural*", preserved in metal tins with a liquid content and weighing 111 g, produced in Spain. Only six of these samples reported the FAO fishing area 27.

None of the labels specified the scientific species name, with identification limited to the commercial denomination. From acquisition, the samples were stored at room temperature.

2.2. Extraction and identification of microplastics

Microplastics were extracted employing oxidative digestion followed by density separation, adapting the methods by Li et al. (2015) and Capuozzo et al. (2025). Each sample was divided into three replicates, and aliquots of 10 g (7 g of edible tissue and 3 g of preserving medium) for each replicate were digested for 24 h in 30% hydrogen peroxide (H_2O_2) (1:20 w/v) at 60 °C in an oscillating incubator at 80 rpm (ASAL s.r.l. Cernusco, Milano, Italy), followed by an additional 24 h at room temperature without oscillation, to optimise the digestion of the biological matrix. Density separation was performed by adding NaCl to distilled water (1.2 g mL^{-1}); the solution was incubated at room temperature overnight and then filtered before being added to the sample digestates (≈800 mL for each replicate) in 1-L cylinders; then they were rapidly covered with aluminium foil and maintained at room temperature overnight. The supernatants were filtered with a metal vacuum pump system (KNF Flodos AG, Sursee, Switzerland) through nitrocellulose filter membranes (1.2 μm pore size and 47 mm diameter) (Axiva Sichem Biotech, Delhi, India). Each filter was subsequently transferred in covered glass Petri dishes (Fischer Scientific, Segrate, Milano, Italy) to allow drying at room temperature. Procedural blanks were performed with the chemical solution alone to assess microplastic airborne contamination. After drying, each sample and procedural blank filter was observed under a stereomicroscope (Nikon, Moncalieri, Torino, Italy) to detect and count potential MPs. Sample particles similar to blank particles were excluded from results.

Particles were classified by morphologies (fibres, fragments, films, granules), dimension categories (5–500 μm, 501–1000 μm, 1001–5000 μm, >5000 μm), and colours following the guidelines by Hidalgo-Ruz et al. (2012). Images were acquired with a digital camera (Nikon X_Entry, Tokyo, Japan).

2.3. Polymer identification of microplastics

Particle chemical identification was performed employing Nicolet™ iS50R FT-IR Advanced KBr Gold Spectrometer (Thermo Fisher Scientific™, Segrate, Milano, Italy) with attenuated total reflectance (ATR) (High Energy Transmission Single Reflection AR-coated Diamond Crystal) accessory. Each FT-IR spectrum included 32 scans, measured within the wavelength range of 400 to 4000 cm^{-1} with a resolution of 4 cm^{-1}. All the suspected microplastics were first detected under the stereomicroscope and then analysed with FT-IR.

Only spectra with a correlation score ≥0.70 (Cho et al., 2021; Nalbone et al., 2024; Tang et al., 2022), obtained from the comparisons of sample polymers and reference polymers from the spectral library (Thermo Fisher Scientific™) were considered and included in the results, ensuring a robust identification approach.

In addition to the microplastic particles isolated from the samples, the packaging materials of each sample were also analysed using FT-IR to assess the polymer composition of the food-contact packaging.

During the FT-IR ATR analyses, in order to minimise the risk of laboratory contamination, procedural blanks were performed when samples were manipulated by exposing glass Petri dishes with empty filters to the laboratory environment.

2.4. QA/QC measures

To minimise contamination, strict precautions were implemented during the entire experiment.

All staff members wore a white 100% cotton lab coat and nitrile gloves all the time; the laboratory door remained closed, and movements in and out of the room were kept as minimal as possible. All laboratory equipment was made of glass or metal and rinsed three times with filtered distilled water before and after use and covered with aluminium foil to prevent airborne MP contamination. Work surfaces were kept clean using 100% cotton towels soaked in 90% filtered ethanol.

All fluids employed during the experiment (distilled water, saline solution, and hydrogen peroxide) were filtered before use with membrane filters with a pore size of 0.45 μm and a diameter of 47 mm (Axiva Sichem Biotech, Delhi, India) and kept in pre-rinsed glass flasks covered with aluminium foil.

2.5. Statistical analysis

Data were analysed with RStudio software (version 2025.09.2 +

418). To evaluate the distribution and compare microplastic contamination across sample types, several statistical tests were applied on confirmed microplastics. The Shapiro-Wilk test was used to assess the normality of the data for both the number of microplastics (n.MP) and microplastic concentration per gram (MP/g). This test determines whether the dataset deviates significantly from a normal distribution, which influences the subsequent statistical methods. Therefore, considering the relative small sample size, multiple non-parametric tests were applied ensuring a conservative strategy.

The Kruskal–Wallis test was conducted to compare the number of microplastics across the procedural blanks and the three types of bivalve samples (CCo, CCl and GJC). Post-hoc pairwise comparisons using Dunn's test with Bonferroni correction were employed to assess the significance of the difference between samples and blanks.

For comparisons among the three sample types (CCo, CCl, and GJC), the Kruskal-Wallis test was applied. When significant differences were detected, post-hoc pairwise comparisons were conducted using Dunn's test. The Bonferroni correction was applied to adjust the significance threshold according to the number of pairwise tests. To further investigate these pairwise differences, Mann–Whitney U tests were conducted for each comparison.

The relative abundance of microplastic morphologies (fibres, fragments, films, and granules), size classes (5–500 μm, 501–1000 μm, 1001–5000 μm, >5001 μm) and colour groups was compared within each sample type using the Friedman test. When significant differences were detected, post-hoc pairwise comparisons were performed using the Nemenyi test to identify which categories differed significantly. All statistical tests were conducted at a significance level of $\alpha = 0.05$, and exact p-values are reported to provide detailed evidence of statistical significance.

The homogeneity of dispersion was assessed using the Coefficient of Variation (CV), whereas multivariate differences in microplastic characteristics among sample groups (CCo, CCl, and GJC) were assessed using PERMANOVA based on Bray-Curtis dissimilarity of numerical microplastic categories (morphotypes, size classes, and colours), with 999 permutations used to test significance.

The polymer composition was determined using compositional data analysis (CoDA). PERMANOVA was used to assess differences between sample types.

2.6. Polymer Hazard Index (PHI)

To assess the potential hazards of microplastics (MPs) in the samples, both their concentrations and chemical identities were considered (Adam et al., 2019). The chemical toxicity of each polymer type, based on hazard scores from Lithner et al. (2011), was used to estimate the potential health risks associated with MPs detected in canned clams, canned cockles, and glass-jarred clams. The polymer hazard index (PHI) of MPs was calculated using the following formula:

$$PHI = \sum P_n \times S_n$$

where P_n is the percentage of each polymer type identified in the samples, and S_n is the corresponding hazard score from Lithner et al. (2011). To compare potential differences among the three sample types, the non-parametric Kruskal–Wallis test was applied.

3. Results

The analysis of microplastic contamination in GJC, CCo and CCl samples revealed significant differences across clam species, particle morphotypes, sizes, and colours.

For all product types (CCo, CCl, and GJC), microplastic concentrations in the samples were significantly higher than in their corresponding procedural blanks (Kruskal–Wallis, $p \approx 0.005$ for all comparisons), which exhibited very low microplastic levels (0.008 ± 1.53 MP/g for CCo, 0 MP/g for CCl, and 0.08 ± 2.38 MP/g for GJC).

Dunn's post-hoc tests confirmed the pairwise differences between blanks and samples (adjusted $p < 0.01$), indicating that the observed microplastic loads originate from the samples themselves rather than from background laboratory contamination.

Across all samples, microplastic counts (n. MP) and concentrations (MP/g wet weight) varied markedly among product types (Table 1) (Fig. 1).

Canned cockles (CCo) showed generally low microplastic contamination. MP concentrations across the 15 individual samples ranged from 0.027 MP/g to 0.274 MP/g, with particle counts per sample between 3 and 20 MPs (Table 1 of Supplementary Files). The highest contamination was found in sample CCo2, which contained 0.274 MP/g and 20 MPs, while the lowest was observed in CCo3, with 0.041 MP/g and 3 MPs (Table 1 of Supplementary Files). Across all CCo samples, the mean particle count was 7.67 ± 1.72 MPs, and the mean concentration was 0.11 ± 0.02 MP/g (Table 1).

Canned clams (CCl) exhibited similarly low microplastic levels, with concentrations between 0.027 MP/g and 0.216 MP/g and particle counts ranging from 1 to 8 MPs per sample. The highest contamination was recorded in sample CCl2 (0.216 MP/g, 8 MPs), whereas CCl5 had the lowest values (0.027 MP/g, 1 MP) (Table 1 of Supplementary Files). On average, canned clams contained 3.38 ± 0.84 MPs per sample and 0.09 ± 0.02 MP/g (Table 1).

In contrast, glass-jarred clams (GJC) were heavily contaminated compared to the canned samples. MP concentrations in GJC samples ranged from 1.36 MP/g to 2.12 MP/g, with particle counts from 59 to 119 MPs per sample. The highest concentration was found in GJC13 (2.12 MP/g, 92 MPs), while the highest particle count was observed in GJC2 (1.88 MP/g, 119 MPs). The lowest concentration among GJC samples was in GJC6 (1.36 MP/g, 59 MPs) (Table 1 of Supplementary Files). On average, GJC samples contained 74.62 ± 10.92 MPs per sample and 1.64 ± 0.27 MP/g (Table 1).

Shapiro–Wilk tests indicated that microplastic counts and concentrations were approximately normally distributed for both canned cockles (n.MP: $W = 0.959$, $p = 0.112$; MP/g: $W = 0.959$, $p = 0.112$) and canned clams (n.MP: $W = 0.962$, $p = 0.147$; MP/g: $W = 0.962$, $p = 0.147$). For glass-jarred clams, MP counts followed a normal distribution ($W = 0.982$, $p = 0.684$), whereas MP/g showed a slight deviation from normality ($W = 0.947$, $p = 0.040$).

Given this partial deviation from normality, non-parametric tests were applied. Kruskal–Wallis analyses revealed significant differences among sample types for both MP counts ($H = 123.51$, $p < 0.001$) and MP concentrations ($H = 9.50$, $p < 0.001$). Post-hoc Dunn's tests with Bonferroni correction showed that GJC samples contained significantly higher MP counts and concentrations than both CCo and CCl ($p < 0.001$), while no significant differences were observed between canned cockles and canned clams (Table 2).

Microplastic morphologies (fibres, fragments, films, and granules) differed significantly within each sample type (Figs. 2 and 3).

Friedman tests revealed highly significant differences in the relative abundance of fibres, fragments, films, and granules (CCo: $Q = 99.97$, $p = 1.57 \times 10^{-21}$, $W = 0.85$; CCl: $Q = 110.16$, $p = 1.01 \times 10^{-23}$, $W = 0.85$; GJC: $Q = 125.61$, $p = 4.78 \times 10^{-27}$, $W = 0.77$). Post-hoc Nemenyi tests showed that fibres were the most abundant morphotype in all sample types. For example, in CCo, fibres were significantly more abundant

Table 1
Microplastic counts (n.MP) and concentrations (MP/g wet weight) in canned cockles, canned clams, and glass-jarred clams. Glass-jarred clams contained markedly higher levels.

Sample	n.MP	MP/g wet weight
Canned cockles	7.67 ± 1.72	0.11 ± 0.02
Canned clams	3.38 ± 0.84	0.09 ± 0.02
Glass-jarred clams	74.62 ± 10.92	1.64 ± 0.27

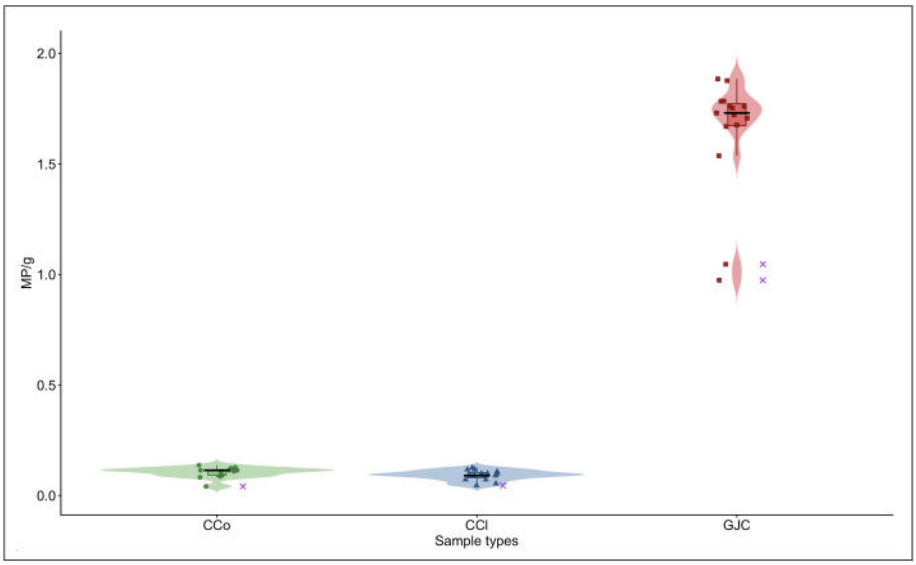

Fig. 1. Violin plots showed markedly higher microplastic concentrations in glass-jarred clams compared to canned cockles and clams, which displayed overlapping distributions and similar medians. Outliers (×) were identified using the 1.5 × IQR criterion.

Table 2
Post hoc Dunn's significant differences.

Variable	Sample Type	Median	Dunn Post-hoc Significant Differences
n.MP	CCo	7	GJC > CCo, CCo ≈ CCl
	CCl	3	GJC > CCl, CCl ≈ CCo
	GJC	74	GJC > CCo, GJC > CCl
MP/g	CCo	0.096	GJC > CCo, CCo ≈ CCl
	CCl	0.081	GJC > CCl, CCl ≈ CCo
	GJC	1.660	GJC > CCo, GJC > CCl

Mann–Whitney U tests confirmed these differences, showing strong effects for GJC comparisons (GJC vs. CCo: r = 1; GJC vs. CCl: r = 0.91) and moderate effects for CCo–CCl (r = 0.44).

than fragments ($p = 0.0010$), films ($p < 1 \times 10^{-12}$), and granules ($p < 1 \times 10^{-11}$). Fragments were more common than films and granules, while films and granules did not differ significantly. Similar patterns were observed in CCl and GJC, with GJC samples showing the highest fibre counts.

Particle size distribution also varied significantly (Friedman test, $p < 0.001$). In all sample types, small particles (5–500 μm) were the most abundant, while the largest particles (>5001 μm) were consistently the least abundant. Intermediate size classes (501–1000 μm and 1001–5000 μm) showed variable abundances depending on the sample type, with statistically significant differences in most pairwise comparisons. For instance, in CCo, 5–500 μm particles were significantly more abundant than 501–1000 μm, 1001–5000 μm, and >5001 μm ($p < 0.01$) (Fig. 4).

Microplastic colour also differed significantly among samples (Friedman test, $p < 0.001$). Transparent and blue particles were generally the most common across all sample types. In CCl, blue and transparent dominated, while other colours (black, red, grey, white, and green) were rare. In CCo, transparent particles were most abundant, followed by blue, and in GJC, transparent particles overwhelmingly dominated the counts. Post-hoc tests confirmed that transparent and blue microplastics were significantly more abundant than other colours, while red, white, and green were consistently the least frequent (Fig. 5).

Dispersion homogeneity was confirmed by CV values (16.5 - 22.2%). Notably, glass-jarred clams showed the highest but most uniform contamination (16.5%). Conversely, canned products exhibited lower but more incidental microplastic loads (CCo: 18.2%; CCl: 22.2%). Finally, PERMANOVA analysis confirmed a strong effect of sample type on microplastic composition ($F = 126.46$, $p = 0.001$), explaining 66.2%

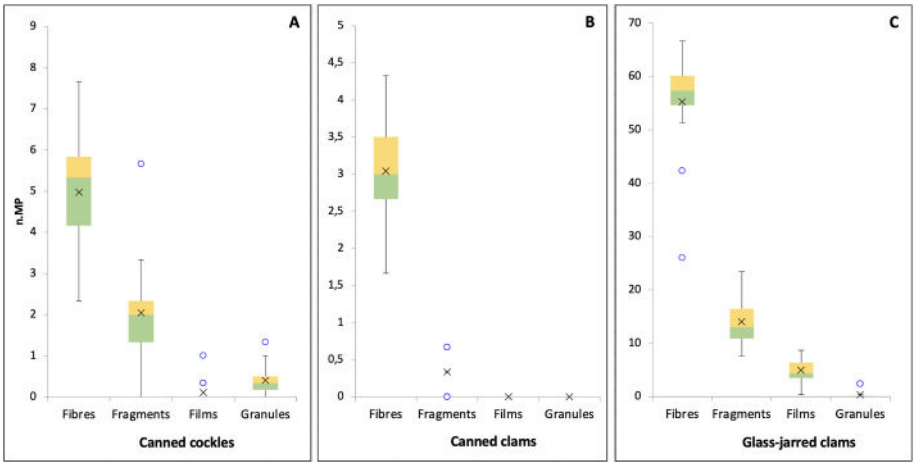

Fig. 2. Boxplots (A–C) show higher microplastic counts in glass-jarred clams, especially fibres and fragments, compared with canned cockles and clams, which had similar distributions and medians. Outliers (○) were identified using the 1.5 × IQR criterion.

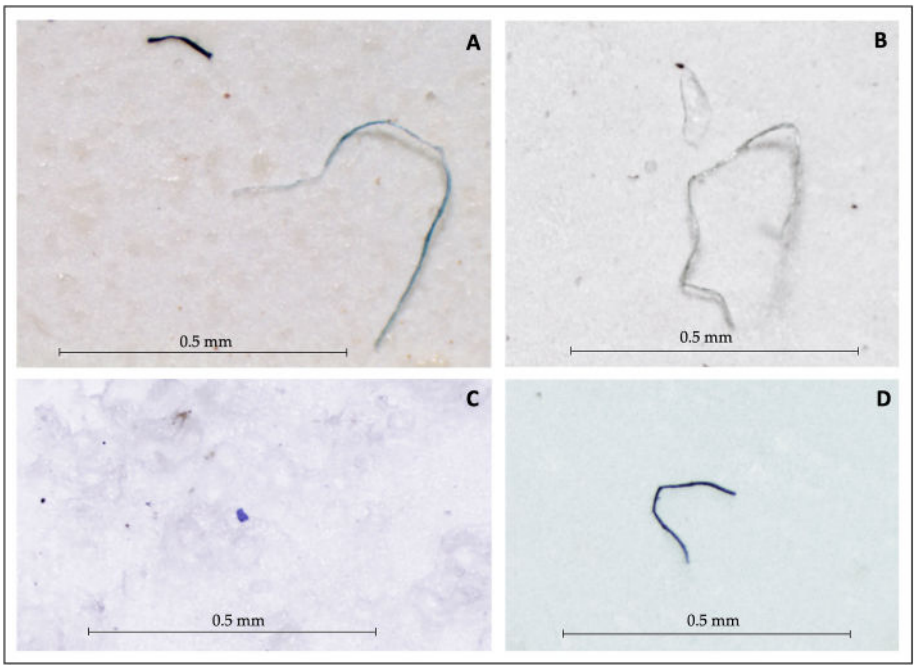

Fig. 3. Digital camera photos of: A) black and blue fibres (CCo8); B) transparent fibre and transparent film (GJC10); C) blue fragment (GJC13); D) blue fibre (CCl7). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

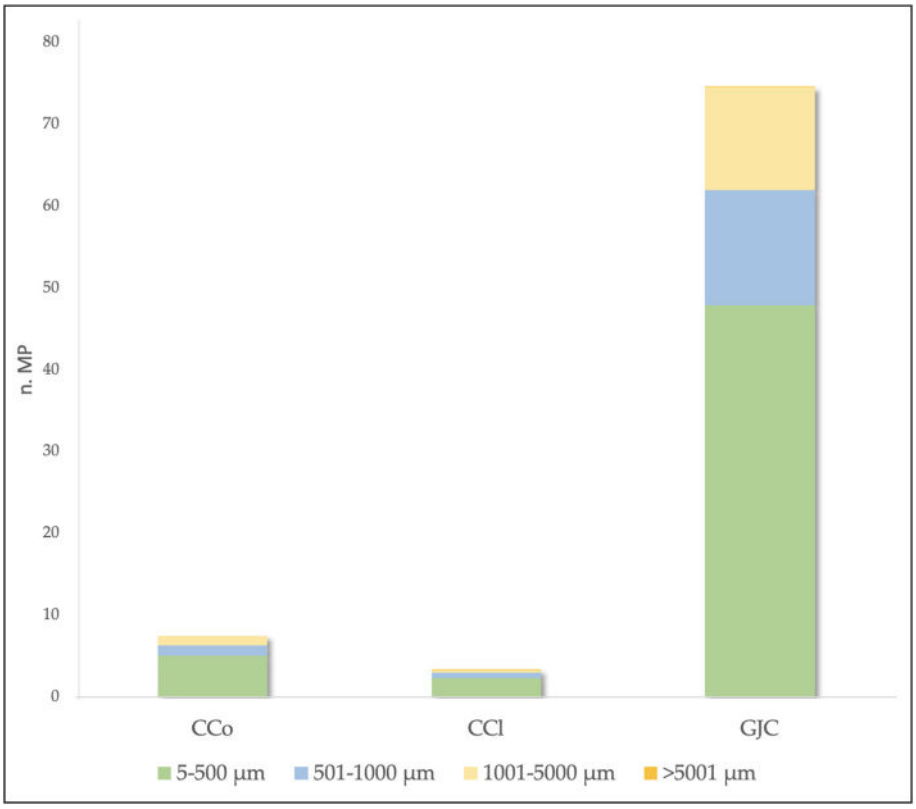

Fig. 4. Stacked bar chart of microplastic size distribution in CCo, CCl, and GJC. Most particles were 5–500 µm, with GJC showing markedly higher counts across all sizes.

of the variation, with the remaining 33.8% due to variation among individual samples. This shows that microplastic morphotype, size, and colour differ substantially between canned cockles, canned clams, and glass-jarred clams, highlighting distinct contamination patterns in these food matrices.

FT-IR analysis of jar closures revealed PET and polyester on the inner surface of the lid and polyvinyl stearate and a polyurethane-based component on the inner rim, whereas tinplate can analysis revealed polyacrylonitrile-co-ethyl acrylate-co-methacrylic acid and phenolic resins in the internal lacquer. PERMANOVA indicated significant

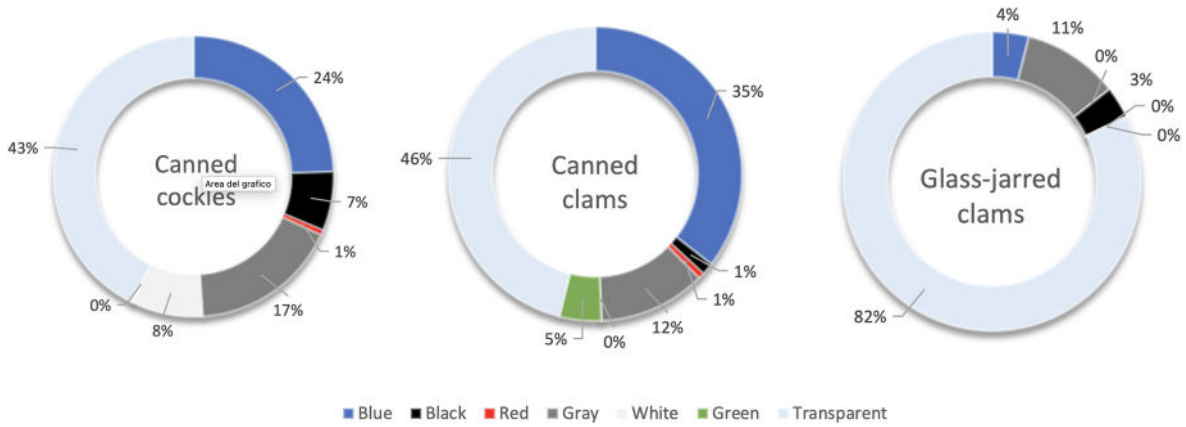

Fig. 5. Doughnut charts of microplastic colours in CCo, CCl, and GJC. GJC was dominated by transparent and grey particles; CCo and CCl had more evenly distributed colour. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

differences in microplastic chemical composition across sample types ($p = 0.001$).

FT-IR analysis identified polymers with a correlation score greater than 0.70 in 66%, 70%, and 73% of the visually sorted items for CCo, CCl, and GJC samples, respectively. Different polymer types were observed across the sample types. GJC samples had the highest difference, marked by polyurethane (PU, 23.0%), blue paint (19.1%), polyamide (PA, 16.4%), epoxy resin (9.7%), and polyester (PES, 8.2%), with minor contributions from acrylonitrile butadiene styrene (ABS, 2.2%), polyethylene (PE, 2.2%), polyvinyl alcohol (PVA, 7.7%), and cellulose acetate (3.4%). Polyacrylic acid, PA-based resins, and polydimethylsiloxane (PDMS) were absent.

PA (33.4%) and PA-based resins (25.3%), followed by polyacrylic acid (10.5%), acrylic + PA blends (7.3%), and epoxy resin (6.7%; $p < 0.05$), were predominant in CCl samples. Chlorinated PE, polyacrylamide, phenolic resin, and polymethylsiloxane were detected in trace amounts (<5%), whereas PU, ABS, PVA, and blue paint residues were not found.

CCo samples had an intermediate composition, dominated by acrylic + PA blends (26.7%), PA (23.0%), polyacrylic acid (13.9%), and cellophane (11.8%). Polydimethylsiloxane (PDMS, 7.7%) was identified only in CCo, along with minor amounts of PA 6 (12.3%), phenolic resin (6.3%), and PA-based resins (13.3%) (Fig. 6).

The Polymer Hazard Index (PHI) ranged from 304137 to 364570 for CCl, from 217964 to 232529 for CCo, and from 419718 to 1924522 for GJC. Kruskal–Wallis tests indicated no significant difference among sample types ($\chi^2 = 4.5714$, $p = 0.1017$); thus, the hypothesis of equal PHI distributions could not be rejected. Nevertheless, GJC samples had

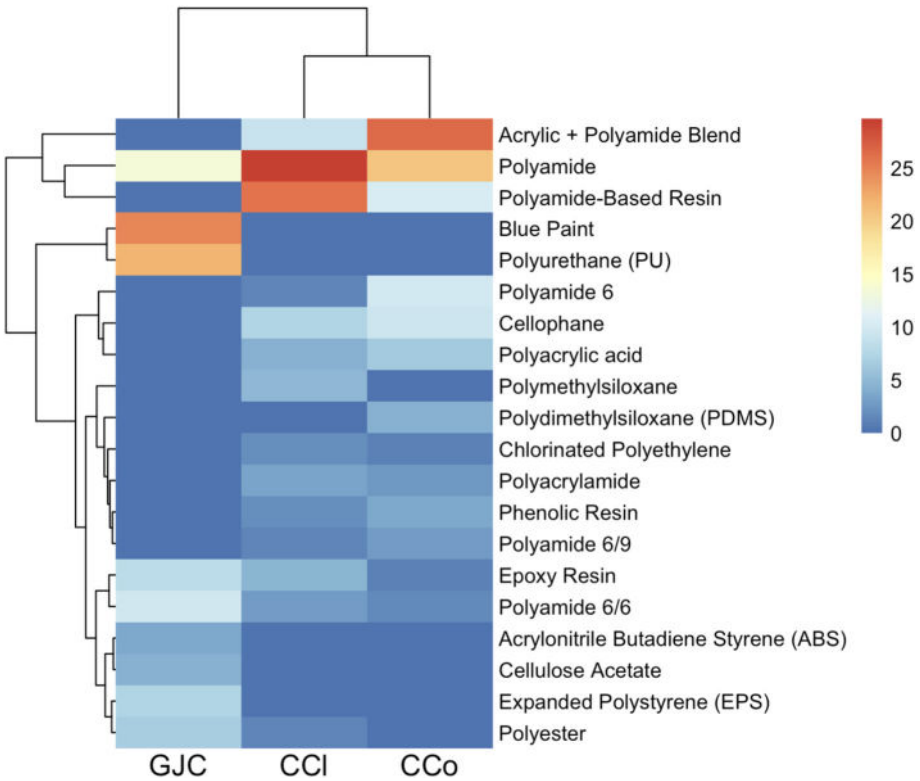

Fig. 6. Relative abundance (%) of the 20 most variable polymers in canned clams (CCl), canned cockles (CCo), and glass-jarred clams (GJC). Columns represent sample types expressed in percentage composition, while rows correspond to individual polymers. Hierarchical clustering separated the sample types, showing higher similarity among canned products compared to glass-jarred samples.

higher PHI values than canned samples.

4. Discussions

This study assessed the presence, composition, and potential risk of MPs in the main 3 brands of preserved bivalve products (i.e., glass-jarred clams, canned clams, and canned cockles) commercially available in the top 3 leading GDO chains in Italy and Spain. This comparative approach enabled an evaluation of microplastic contamination across different food types and packaging materials, focusing on products from retail markets and thus providing valuable knowledge for public health and food safety. The analysis of this research included both edible tissue and the preservation medium, offering a wider understanding of the substances to which consumers are exposed. The absence of detectable polymers in procedural blanks confirmed that MP counts and polymer compositions reflect actual contamination from the samples (marine environmental intake, processing, and packaging) rather than analytical artefacts. From a food safety and public health perspective, this makes the data reliable for supporting regulatory decisions and for developing mitigation strategies.

4.1. Packaging and processing impact on MP contamination

Microplastics were detected across all sample types, corroborating previous evidence of widespread MP pollution in seafood (Smith et al., 2018; Dambrosio et al., 2023; Quaglia et al., 2023; Unuofin & Igwaran, 2023).

Clams in glass jars showed approximately ten times higher MP levels compared to canned clams and cockles (MP levels were 1.64 ± 0.27 MP/g in GJC, 0.11 ± 0.02 MP/g in CCo, and 0.09 ± 0.02 MP/g in CCl), suggesting that packaging materials may impact MP release into edible tissue (Siddiqui et al., 2023). Glass-jarred products are generally preserved in heat-resistant glass jars with metal lids (aluminium or tin-plated steel) sealed to maintain vacuum integrity. The inner lid surface is coated with polymeric food-contact lacquers, which may include polyester, PET, epoxy-based resins, or similar coatings, while sealing compounds such as polyurethane-based gaskets or adhesives provide an airtight seal and maintain jar integrity during sterilisation and storage. Since the glass is chemically inert, the primary sources of polymer migration are the lid, gasket, and internal coating layers, which may degrade or release microparticles under thermal and mechanical stress (Epoxy Resin Committee, 2014).

Canned seafood, in contrast, is generally packaged in tinplate or aluminium cans designed to resist sterilisation. The interior surfaces are coated with protective lacquers, commonly epoxy resins, phenolic resins, polyesters, acrylics, or other polymeric films, to prevent corrosion and preserve product safety during storage. Cans, seams, and lids that incorporate sealants, adhesives, or multilayer polymeric films are exposed to high pressures, temperature fluctuations, and occasionally saline or acidic food matrices, which challenge coating integrity (Food Packaging Forum, 2017).

The levels observed in glass-jarred clams in this study exceed levels reported in previous studies on different seafood types (fresh, frozen, smoked, canned, and marinated), where a median of 0.9 MP/g was reported across all products, with canned fish reaching ~2.4 MP/g, while marinated and fresh products contained much lower amounts (Süssmann et al., 2025). Similarly, Nalbone et al. (2024) found 0.13 ± 0.17 MP/g in fresh clams and 0.19 ± 0.21 MP/g in processed clams (in brine and vacuum frozen), substantially below the 1.64 MP/g observed in our glass-jarred samples.

From a mechanistic standpoint, surface-associated polymers can infiltrate food matrices via various pathways, including washing, depuration, deshelling, sterilisation, mechanical abrasion (during capping, conveyor movement, or sealing), and chemical degradation of coatings. These processes can produce or redistribute MPs between tissue and the preservation medium, while certain particles may aggregate or degrade, diminishing detectability (Paul et al., 2023; Süssmann et al., 2025).

Metal cans typically incorporate polymeric lacquers, but hermetically sealed packaging and high-temperature sterilisation may limit additional contamination, explaining the lower MP counts in canned products (Muncke et al., 2020; Zimmermann et al., 2025). In contrast, glass jars may release microparticles during sealing and storage, particularly when lids contain multilayer coatings or painted surfaces (Silva et al., 2024). Therefore, analyses that exclude the preservation medium or overlook processing-related contributions may underestimate the actual MP intake by consumers.

4.2. Biological factors, size and morphotype distributions

Friedman analyses of morphotypes, size classes, and colour categories indicate that differences across products cannot be attributed only to post-harvest processing or packaging.

Data from this study showed a higher representation of fibrous morphotypes and of the smallest (<500 μm) and largest (>5000 μm) size classes in glass-jarred clams compared to CCo and CCl; intermediate size ranges were comparatively lower and relatively similar across the different products. Fibres tend to lodge in gill structures and mucous layers, whereas smaller fragments may translocate more readily into tissues or preservation liquid (Liang et al., 2023; McIlwraith et al., 2021). This selective retention and egestion behaviour is consistent with previous findings on bivalve feeding and MP retention, where smaller particles were shown to accumulate in digestive tissues and larger ones to be more efficiently expelled (Van Cauwenberghe & Janssen, 2014; Prata et al., 2020). Differences between the product types in this study suggest that species-specific biological characteristics, including feeding ecology, filtration rate, gill morphology, digestive transit, and retention capacity, influence MP uptake and retention, shaping contamination profiles after processing (Zhang et al., 2020). Nevertheless, in the samples of this study species-level identification was limited on the commercial labelling provided at the point of sale.

The frequent occurrence of grey and transparent particles in jarred products suggests that potential sources may include handling surfaces, marine biofouling, pigmented coatings or paints employed in industrial and naval applications (Gondikas et al., 2023). This confirms that the influence of packaging is not independent from environmental sources; indeed, bivalves naturally accumulate environmental MPs during filtration. However, the detection of packaging-specific polymers in the analysed samples indicates that post-harvest and packaging stages may add a secondary contamination load, susceptible to technological and regulatory control.

This inter-product and inter-species variability has direct implications for dietary exposure assessments, introducing substantial heterogeneity in microplastic intake estimates depending on the type of seafood consumed (Gündoğdu & Köşker, 2023).

Biological factors interplay with technological processing; thus, understanding these interactions is critical for accurate dietary exposure assessment, risk evaluation, and the development of targeted mitigation strategies. Integrating ecological, biological, and technological perspectives is therefore essential for evaluating microplastic contamination in seafood products, as the One Health approach promotes (Cox et al., 2019; Vethaak & Leslie, 2016).

4.3. Polymer identity, hazard potential, and regulatory context

A permutational multivariate analysis of variance (PERMANOVA) revealed significant differences in the chemical composition of polymers between the three seafood product categories ($p = 0.001$). Across all samples, 36 different polymers were identified (Table 3).

The polymeric profiles identified in the edible tissues of the processed clam products analysed in this study clearly reflect a combination of pre-harvest environmental exposure and post-harvest inputs occurring throughout industrial handling, processing, and packaging.

Table 3
Polymers chemically identified in this study with respective hazard classes and indexes according to Lithner et al., 2011. Polymers assigned as N.C. were not classified.

Polymers	Sample	Hazard class	Hazard score	Applications
Acrylic + Polyamide Blend	CCl - CCo	III + II-III	230 + 50 – 63[a]	Textile industry, including home furnishings (e.g., blankets and curtains).
Acrylonitrile Butadiene Styrene (ABS)	GJC	V	6552 – 6957[a]	Automobile and electronic components, military and aerospace applications.
Blue Paint	GJC	N.C.	N.C.	Coatings for ships, boats, offshore structures, and other marine vessels.
Cellophane	CCl - CCo	N.C.	N.C.	Food packaging, gift wrapping, arts and crafts.
Cellulose Acetate	GJC	N.C.	N.C.	Textile fibres, food packaging, cigarette filters, and films for LCD screens.
Chlorinated Polyethylene	CCl - CCo	N.C.	N.C.	Wire and cable insulation, hoses, construction materials (e.g., window and door profiles and pipes).
Dimethylmethacrylamide	CCl	N.C.	N.C.	Biomedical fields, adhesives, coatings, and water treatment.
Epoxy Resin	CCl - CCo - GJC	V	4226 – 7139[a]	Aerospace and automotive industry, construction, and electronics for insulation and shielding.
Expanded Polystyrene (EPS)	GJC	III	44	Packaging for cushioning fragile goods and insulating food, protective gear and decorative moldings.
Phenolic Resin	CCl - CCo	IV	1500	Coatings for metal packaging and industrial applications, electrical components, engineered wood, aerospace, and offshore industries.
Polyacrylamide	CCl - CCo	V	12379	Water treatment, mineral processing, construction grouts and soil stabilisers, textile and biomedical fields.
Polyacrylic acid	CCl - CCo	III	230	Diapers, cosmetics and paints, water treatment, pharmaceuticals.
Polyamide	CCl - CCo - GJC	N.C.	N.C.	Textile, automotive, electronic industries and packaging.
Polyamide 6	CCl - CCo	II	50	Engine components and gears, electrical components, packaging and textiles.
Polyamide 6/12	CCl	N.C.	N.C.	Automotive fuel lines, gears, electrical connectors, toothbrush bristles, precision seals, electronics.
Polyamide 6/6	CCl - CCo - GJC	III	63	Automotive and aerospace gears and fuel lines, electrical components, and industrial machinery parts.
Polyamide 6/9	CCl - CCo	N.C.	N.C.	Mechanical, electrical components and orthopaedic implants.
Polyamide 7	CCl - CCo	N.C.	N.C.	Powdered products.
Polyamide 6 + Polyamide 6/6 blend	CCo	II + III	50 + 63	Technical geotextiles, apparel, and industrial applications.
Polyamide 11	CCl	I	1	Oil and gas, aerospace, automotive, textiles, electronics and sports equipment, medical devices.
Polyamide 12	CCl	I	2	Fuel and hydraulic lines, pneumatic tubing, wire coatings, medical field.
Polyamide-based resin	CCl - CCo	N.C.	N.C.	Automotive components, high-performance coatings, printing inks, flexible packaging.
Polycaprolactone (PCL)	GJC	N.C.	N.C.	Biodegradable sutures, drug delivery systems, tissue engineering scaffolds, packaging, and 3D printing filament.
Polydimethyl siloxane (PDMS)	CCo	N.C.	N.C.	Medical devices, cosmetics, and electronics.
Polyester	GJC - CCl	IV	1117 – 1414[a]	Clothing and home furnishings, food packaging, industrial and automotive products.
Polyethylene (PE)	CCl	II	11	Food packaging, toys, pipes and tubing, medical devices like

(*continued on next page*)

Table 3 (*continued*)

Polymers	Sample	Hazard class	Hazard score	Applications
				syringes and implants.
Polyethylene + propylene blend	CCo	II + I	11 + 1	Automotive parts and pipes, recycled materials.
Polyethylene Terephthalate (PET)	CCl - CCo	II	4	Packaging, textiles, and automotive and electrical components.
EPDM Rubber	CCl - CCo	N.C.	N.C.	Automotive and construction industry, electrical insulation.
Polyfluoromaleic Anhydride:Methyl Methacrylate Copolymer	CCo	N.C.	N.C.	Biomedical fields, industrial and material science.
Polyhydroxybutyrate (PHB)	GJC	N.C.	N.C.	Bioplastics for packaging, medical devices, agriculture and nanotechnology.
Polymethylsiloxane	CCl	N.C.	N.C.	Sealants, adhesives, food processing (e.g., antifoaming agents), medical and biomedical fields.
Polytetrafluoroethylene (PTFE)	CCl	I	1	Kitchenware, medical devices, chemical industries, electrical insulation, aerospace components, industrial seals and bearings.
Polyurethane (PUR)	GJC	IV-V	556 – 13844[a]	Medical applications, automotive, furniture, construction, thermal insulation, footwear.
Polyvinyl Alcohol (PVA)	GJC	N.C.	N.C.	Food packaging, biomedicine, and wastewater treatment.
Polyvinylchloride (PVC)	GJC	V	5001-10551[a]	Packaging, film and sheet, carpet backing, electrical boxes, cables.
Polyvinylpirrolidone	CCl	N.C.	N.C.	Batteries, ceramics, fiberglass, inks, pharmaceuticals, biomedicals, and nutraceutical fields.
Viscose	CCo	N.C.	N.C.	Textiles and furnishings, industrial and technical applications.

[a] Values separated by hyphens correspond to the minimum and maximum hazard scores of the correspondent polymer, according to Lithner et al., 2011.

In GJC, several polymers typically associated with aquatic environmental contamination, most notably PA (16.35%), PE (2.17%), and PET (0.77%), match with the dominant polymers found in fresh clams by Jankauskas et al. (2024), who reported PA (29.5%), PE (19.8%), and PET (9.7%) as principal contaminants in fresh clams (*Amarilladesma mactroides*). Therefore, these polymers may be consistent with pre-harvest uptake. However, the GJC samples also contained a broad set of polymers directly accountable to post-harvest handling and packaging. FT-IR analysis of jar closures revealed PET and polyester on the inner surface of the lid and polyvinyl stearate and a polyurethane-based component on the inner rim. These materials were also detected in the edible tissue, where polyester (8.17%) and PU (22.99%) were among the most abundant polymers. Additional packaging-related polymers, including ABS (2.15%), epoxy resin (9.67%), EPS (6.99%), and blue paint (19.13%) from lid coatings or lettering, further support mechanical abrasion and thermal stress during sterilisation as post-harvest sources (Caponigro et al., 2025; Nikova et al., 2023).

The canned samples were characterised by a similar combination of pre- and post-harvest sources. Commonly encountered polymers, including PA and its derivatives, constituted 33.35% in canned clams (CCl) and 22.99% in canned cockles (CCo), mirroring the trends identified in fresh bivalves by Jankauskas et al. (2024). At the same time, several polymers present in the edible tissue could be directly linked to the packaging materials. Tinplate cans analysis in this study revealed polyacrylonitrile-co-ethyl acrylate-co-methacrylic acid and phenolic resins in the internal lacquer. Phenolic resins were detected both in CCl and CCo with 2.15% and 6.33%, respectively. CCl also showed the presence of polyacrylamide (5.01%), polyacrylic acid (10.47%), and cellophane (8.27%), while CCo exhibited polyacrylic acid (13.90%) and cellophane (11.81%). Additional polymers characteristic of processing and sealing technologies, including PDMS (7.73%) in CCo, acrylic + PA blends (26.70%), and minor components such as EPDM rubber (0.60%), PTFE (0.77%), and methacrylate-based copolymers (0.94%), are consistent with lubricants, gaskets, anti-foaming agents, multilayer films, and adhesives used during thermal processing and packaging operations (Kwon et al., 2017; Li et al., 2023; Muncke et al., 2020; Zimmermann et al., 2025; Čurlej et al., 2023).

The occurrence of PA, PE, and PET may indicate a pre-harvest origin, corroborating the findings of Jankauskas et al. (2024), while the variety of packaging-related polymers corresponds with the observations of Gündoğdu and Köşker (2023), who identified polyolefins, polyacrylonitrile, methacrylate derivatives, PA, PET, and PP in canned fish, most likely due to post-harvest processes and can-lining abrasion. A consistent pattern is observed in our dataset. The significant polymeric diversity in processed clams may be attributed to the cumulative effects of environmental contamination prior to harvesting and various post-harvest factors, including handling, cleaning, thermal processing, and the direct abrasion or degradation of packaging materials. Therefore, a match rate between polymers detected in packaging materials and those in the edible tissues is not feasible.

Although Polymer Hazard Index (PHI) values did not differ statistically across sample types ($\chi^2 = 4.5714$, $p = 0.1017$), glass-jarred clams consistently exhibited substantially higher PHI with different polymers classified as high hazard (IV-V hazard classes), including ABS, PVC, epoxy resins, PU, and polyacrylamide, up to tenfold greater than canned counterparts, while showing lower overall synthetic polymer content. This pattern aligns with observations in bivalves from other regions. Ding et al. (2022) found global hazard scores in seafood reaching risk category IV, dominated by polymers such as PET, PE, PP, polyester, PA, rayon, and cellophane; whereas Li et al. (2023) reported lower PHI levels in Daya Bay bivalves falling within risk categories II–III. Polymers in hazard categories IV–V, recognised as endocrine disruptors and potential carcinogens, may release monomers or additives such as aromatic amines, styrene, acrylonitrile, bisphenol derivatives, and chlorinated compounds, especially under heat or mechanical stress (Andrady, 2017; Rist et al., 2018; ECHA, 2025). Within the European Union, such substances are regulated under Regulation (EC) 1907/2006 and SMI, (REACH), which governs the registration, evaluation, authorisation, and restriction of chemicals to ensure safe use, and under Regulation (EC) 1272/2008 and SMI, on the classification, labelling, and

packaging of substances and mixtures, which classifies and labels substances according to their hazards. Specific migration limits for certain monomers and additives in food-contact materials are established by the consolidated Commission Regulation (EU) 10/2011, whereas Regulation (EU) 2024/3190 further extends these regulations by banning bisphenol A (BPA) for the manufacture of plastic materials intended for food contact, including polycarbonate infant feeding bottles and drinking cups. Furthermore, Regulation (EU) 2023/2055 addresses the intentional addition of synthetic polymers in all types of products, explicitly including microplastics, highlighting the growing recognition of microplastic exposure as a relevant concern. Although not specifically targeting food, it reflects a broader EU strategy to limit microplastic release into the environment, indirectly mitigating contamination pathways in food (Table 4).

Even though binding limits for microplastic content in foods are not yet established, the regulation highlights the relevance of controlling microparticle contamination and supports the development of preventive strategies. The results of this study, showing the occurrence of small-sized, high-hazard microplastics in preserved seafood, provide evidence that microplastics represent a potential food safety risk and lay the scientific basis for their formal recognition as a consumer health concern.

4.4. Dietary exposure

The Polymer Hazard Index (PHI) provides an indicator for the hazard potential of MPs, reflecting the contribution of polymer identity and additives. Although not a direct measure of exposure, elevated PHI values, especially when associated with hazardous polymers or high particle counts, merit consideration for their possible implications on consumer health (Fred-Ahmadu et al., 2024; Siddique et al., 2024).

Converting MP counts and PHI data into exposure estimates requires reliable consumption statistics. However, current European and international databases (e.g., EUMOFA, FAOSTAT) do not disaggregate preserved bivalve products, limiting the calculation of Estimated Average Daily Intake (EADI) values for canned or jarred seafood. The EADI indicator offers a useful basis for such estimation, but it relies on ingestion rate (IR) data representative of national dietary habits (Heo, Moon, & Kim, 2025). In the absence of detailed statistics on preserved seafood consumption, such as the samples of this study, exposure assessments remain uncertain.

This data gap limits the evaluation of cumulative and chronic exposure risks, particularly for vulnerable groups such as frequent seafood consumers, pregnant women, children, and older adults (Heo et al., 2025; Yuan et al., 2022).

Further contributing to this issue, the COVID-19 pandemic led to notable changes in consumer behaviour, with consumption of preserved and overall processed seafood products rising during lockdowns and the early recovery period. These shifts were driven by supply chain interruptions, mobility restrictions, and an increased preference for long-shelf-life foods (Pititto et al., 2021; Carpenter et al., 2023). If this trend persists, preserved seafood could represent an increasingly relevant part of diets and microplastic exposure. Without updated consumption surveys reflecting post-pandemic consumption patterns, current public health risk assessments may underestimate exposure to MPs (van Raamsdonk et al., 2020).

Table 4
Regulatory framework relevant to microplastics.

Regulation	Scope	Key provisions relevant to microplastics/polymers
Regulation (EC) No 1907/2006 and SMI (REACH)	Registration, evaluation, authorisation, and restriction of chemicals in the EU	Governs safe use of chemicals, including monomers and additives that may migrate from food-contact materials; indirectly relevant for polymers that release hazardous monomers
Regulation (EC) No 1272/2008 and SMI (CLP)(European Parliament & Council, 2008/2024)	Classification, labelling, and packaging of substances and mixtures	Classifies hazardous substances (including monomers and additives from plastics) according to physical, health, and environmental hazards
Commission Regulation (EU) 10/2011, and SMI(European Commission, 2011)	Plastics intended to come into contact with food	Sets specific migration limits (SMLs) for monomers and additives, e.g., styrene, acrylonitrile, bisphenol A; ensures compliance of plastic FCMs
Regulation (EU) 2024/3190(European Parliament & Council, 2004)	Food-contact materials	Bans the use of bisphenol A (BPA) in plastic food-contact articles, including polycarbonate bottles and drinking cups
Regulation (EU) 2023/2055(European Union, 2023)	Intentional addition of microplastic particles	Explicitly restricts solid, persistent synthetic polymer microparticles, including microplastics; reinforces mitigation of secondary contamination pathways, relevant for food-contact materials and environmental exposure

5. Conclusion

Microplastic contamination in preserved bivalves is a multifaceted issue with significant implications for food safety (Unuofin & Igwaran, 2023). Beyond primary microplastic contamination occurring during pre-harvest stages, the results of this study indicate that secondary contamination can arise in post-harvest operations, including processing (for example, salt and water used in the preservation medium) and packaging (Di Fiore et al., 2023; Mercogliano et al., 2020). In the absence of complete toxicological evidence to establish exposure limits, a precautionary approach is recommended, prioritising the minimisation of polymer contamination during processing and packaging and reinforcing monitoring activities within the One Health framework (Unuofin & Igwaran, 2023; Vitali et al., 2023).

The findings of this study show that packaging materials may migrate to edible tissue, although other contamination sources may be considered for the same polymers identified in the packagings. Therefore, considering that quantifying how packaging materials may contribute to MP contamination is not reasonable due to the great variability of involved factors, further studies should be conducted on every processing phase to assess the critical stages that contribute the most to the contamination. At the production stage, packaging and sealing materials should be critically assessed, with high-risk polymers in lids, gaskets, sealants, and coatings being replaced with safer alternatives, in line with requirements for food contact materials. Moreover, it is essential to identify and replace worn processing equipment and machinery that are more susceptible to releasing microplastics in order to mitigate contamination risk. Integrating microplastic mitigation measures into Good Manufacturing Practice (GMP) procedures offers a practical strategy to decrease consumer exposure and enhance overall food safety and risk management (Reg. EC 2023/2006) (European Commission, 2006). Addressing this issue also depends on robust analytical methods for isolating and characterising microplastics, including their classification by size, morphology, and polymer composition (Aloia et al., 2025; Di Fiore et al., 2023). Ultimately, controlling microplastic contamination is essential to protect public health and inform future regulatory policies (Unuofin & Igwaran, 2023).

CRediT authorship contribution statement

Flavia Capuozzo: Writing – original draft, Methodology,

Investigation, Formal analysis. **Nicoletta C. Quaglia:** Writing – review & editing, Methodology, Conceptualization. **Salud Deudero:** Resources. **Michele De Rosa:** Investigation. **Federica Ioanna:** Investigation. **Angela Dambrosio:** Writing – review & editing, Resources, Methodology, Conceptualization.

Informed consent statement

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Declaration of competing interest

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foodcont.2026.112269.

Data availability

The original contributions presented in the study are included in the article; further inquiries can be directed to the corresponding author.

References

Adam, V., Yang, T., & Nowack, B. (2019). Toward an ecotoxicological risk assessment of microplastics: Comparison of available hazard and exposure data in freshwaters. *Environmental Toxicology and Chemistry, 38*(2), 436–447. https://doi.org/10.1002/etc.4323

Aloia, R., Annunziata, L., Di Giacinto, F., De Simone, S., Profico, C., Fanelli, I., Di Francesco, G., Mussi, V., & Scortichini, G. (2025). Application of Raman spectroscopy for the analysis of microplastics in food and beverages: A comprehensive review. *Applied Spectroscopy Reviews*, 1–35. https://doi.org/10.1080/05704928.2025.2512771

Andrady, A. L. (2017). The plastic in microplastics: A review. *Marine Pollution Bulletin, 119*(1), 12–22. https://doi.org/10.1016/j.marpolbul.2017.01.082

EFSA (European Food Safety Authority), Barthélémy, E., Cariou, R., Castle, L., Crebelli, R., Di Consiglio, E., Hemy Dumas, T., Franz, R., Grog, K., Lambré, C., Lampi, E., Milana, M. R., Munoz Guajardo, I., Pronk, M., Rivière, G., da Silva, M., Tietz, T., Tsochatzis, E., & Van Hoeck, E. (2025). Literature review on micro- and nanoplastic release from food contact materials during their use. *EFSA Supporting Publication 2025: EN-9733*, 53. https://doi.org/10.2903/sp.efsa.2025.EN-9733DigitalObjectIdentifier

Caponigro, V., Di Fiore, C., Carriera, F., Iannone, A., Malinconico, A., Campiglia, P., Crescenzi, C., & Avino, P. (2025). Evaluating microplastic emission from takeaway containers: A micro-Raman approach across diverse exposure scenarios. *Food Chemistry, 464*, Article 141716. https://doi.org/10.1016/j.foodchem.2024.141716

Capuozzo, F., Quaglia, N. C., Di Pinto, A., De Rosa, M., Ioanna, F., & Dambrosio, A. (2025). Evaluation of digestion methods in microplastic recovery from mussels (*Mytilus galloprovincialis*) for a standardised microplastic isolation protocol. *Foods, 14*(22), 3853. https://doi.org/10.3390/foods14223853

Carpenter, G., Carvalho, N., Guillen, J., Prellezo, R., Villasante, S., Andersen, J. L., Mravlje, E. A., Berkenhagen, J., Brigaudeau, C., Burke, B., Santos, A. C., Cano, S., Contini, F., Da-Rocha, J. M., Davidjuka, I., Fernández Martínez, F. M., Fontaneda-López, I., Gambino, M., Garcia Caballero, E., … Zhelev, K. (2023). The economic performance of the EU fishing fleet during the COVID-19 pandemic. *Aquatic Living Resources, 36*, 2. https://doi.org/10.1186/s43591-022-00039-x

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. *Environmental Pollution, 270*, Article 116175.

Cox, K. D., Covernton, G. A., Davies, H. L., Dower, J. F., Juanes, F., & Dudas, S. E. (2019). Human consumption of microplastics. *Environmental Science & Technology, 53*(12), 7068–7074. https://doi.org/10.1021/acs.est.9b01517

Čurlej, J., Zajác, P., Čapla, J., & Hleba, L. (2023). Safety issues of microplastics released from food contact materials. *Journal of Microbiology, Biotechnology and Food Sciences, 13*(2), Article e10317. https://doi.org/10.55251/jmbfs.10317

Dambrosio, A., Cometa, S., Capuozzo, F., Ceci, E., Derosa, M., & Quaglia, N. C. (2023). Occurrence and characterization of microplastics in commercial mussels (*Mytilus galloprovincialis*) from Apulia region (Italy). *Foods, 12*(7), 1495. https://doi.org/10.3390/foods12071495

Di Fiore, C., Carriera, F., Russo, M. V., & Avino, P. (2023). Are microplastics a macro issue? A review on the sources of contamination, analytical challenges and impact on human health of microplastics in food. *Foods, 12*(21), 3915. https://doi.org/10.3390/foods12213915

Ding, J., Sun, Y., He, C., Li, J., & Li, F. (2022). Towards risk assessments of microplastics in bivalve mollusks globally. *Journal of Marine Science and Engineering, 10*(2), 288. https://doi.org/10.3390/jmse10020288

Epoxy Resin Committee. (2014). *Food packaging: Factsheet*. European Council for Epoxy Resin Manufacturers. https://epoxy-europe.eu/wp-content/uploads/2014/06/Epoxy_Resin_Committee_Factsheet_Food_Packaging.pdf.

European Chemicals Agency (ECHA). (2025). *Candidate list of substances of very high concern for authorization*. ECHA.

European Commission. (2006). *Commission Regulation (EC) 2023/2006 of 22 December 2006 on good manufacturing practice for materials and articles intended to come into contact with food*. EUR-Lex (Consolidated version 16 March 2025) https://eur-lex.europa.eu/eli/reg/2006/2023/oj.

European Commission. (2011). *Commission Regulation (EU) 10/2011 of 14 January 2011 on plastic materials and articles intended to come into contact with food (Consolidated version 16 March 2025)*. EUR-Lex. https://eur-lex.europa.eu/eli/reg/2011/10/oj.

European Food Safety Authority (EFSA). (2021). A coordinated approach to assess the human health risks from microplastics — Progress and recommendations. *EFSA Supporting Publication, 18*. https://doi.org/10.2903/sp.efsa.2021.EN-6815

European Parliament & Council. (2004). *Regulation (EC) 1935/2004 of the European parliament and of the council of 27 October 2004 on materials and articles intended to come into contact with food*. EUR-Lex (Consolidated version 27 March 2021) https://eur-lex.europa.eu/eli/reg/2004/1935/oj.

European Parliament & Council. (2008/2024). *Regulation (EC) 1272/2008 of 16 December 2008 on classification, labelling and packaging of substances and mixtures (CLP)*. EUR-Lex (Consolidated version 01 February 2025) https://eur-lex.europa.eu/eli/reg/2008/1272/eng.

European Union. (2023). Regulation (EU) 2023/2055 of the European parliament and of the council of 28 November 2023 on the restriction of intentionally added microplastics and amending annex XVII to regulation (EC) no 1907/2006 (REACH). *Official Journal of the European Union L, 293*, 1–24. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32023R2055.

FAO/WHO. (2021). Microplastics in food safety: Assessment and risk management considerations. *FAO/WHO Technical Report*. https://doi.org/10.4060/cb5860en

Food Packaging Forum. (2017). Can coatings [Dossier]. https://foodpackagingforum.org/wp-content/uploads/2017/02/BirgitGeueke_FPFwebinar170207_CanCoatings.pdf.

Fred-Ahmadu, O. H., Ahmadu, F. O., Adedapo, A. E., Oghenovo, I., Ogunmodede, O. T., & Benson, N. U. (2024). Microplastics and chemical contamination in aquaculture ecosystems: The role of climate change and implications for food safety—a review. *Environmental Sciences Europe, 36*(1), 181. https://doi.org/10.1186/s12302-024-00995-6

Gondikas, A., Mattsson, K., & Hassellöv, M. (2023). Methods for the detection and characterization of boat paint microplastics in the marine environment. *Frontiers in Environmental Chemistry, 4*, Article 1090704. https://doi.org/10.3389/fenvc.2023.1090704

Gündoğdu, S., & Köşker, A. R. (2023). Microplastic contamination in canned fish sold in Türkiye. *PeerJ, 11*, Article e14627. https://doi.org/10.7717/peerj.14627

Hantoro, I., Löhr, A. J., Van Belleghem, F. G., Widianarko, B., & Ragas, A. M. (2019). Microplastics in coastal areas and seafood: Implications for food safety. *Food Additives & Contaminants: Part A, 36*(5), 674–711. https://doi.org/10.1080/19440049.2019.1585581

Heo, S. J., Moon, N., & Kim, J. H. (2025). A systematic review and quality assessment of estimated daily intake of microplastics through food. *Reviews on Environmental Health, 40*(2), 371–392. https://doi.org/10.1515/reveh-2024-0111

Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2012). Microplastics in the marine environment: a review of the methods used for identification and quantification. *Environmental Science & Technology, 46*(6), 3060–3075.

Jankauskas, L., Pinho, G. L. L., Sanz-Lazaro, C., Casado-Coy, N., Rangel, D. F., Ribeiro, V. V., & Castro, Í. B. (2024). Microplastic in clams: An extensive spatial assessment in south Brazil. *Marine Pollution Bulletin, 201*, Article 116203. https://doi.org/10.1016/j.marpolbul.2024.116203

Kannan, K., & Vimalkumar, K. (2021). A review of human exposure to microplastics and insights into microplastics as obesogens. *Frontiers in Endocrinology, 12*, Article 724989. https://doi.org/10.3389/fendo.2021.724989

Kwon, J. H., Chang, S., Hong, S. H., & Shim, W. J. (2017). Microplastics as a vector of hydrophobic contaminants: Importance of hydrophobic additives. *Integrated Environmental Assessment and Management, 13*(3), 494–499. https://doi.org/10.1002/ieam.1906

Li, H. X., Shi, M., Tian, F., Lin, L., Liu, S., Hou, R., Peng, J.-P., & Xu, X. R. (2023). Microplastics contamination in bivalves from the Daya Bay: Species variability and spatio-temporal distribution and human health risks. *Science of the Total Environment, 841*, Article 156749. https://doi.org/10.1016/j.scitotenv.2022.156749

Li, J., Yang, D., Li, L., Jabeen, K., & Shi, H. (2015). Microplastics in commercial bivalves from China. *Environmental Pollution, 207*, 190–195. https://doi.org/10.1016/j.envpol.2015.09.018

Liang, W., Li, B., Jong, M. C., Ma, C., Zuo, C., Chen, Q., & Shi, H. (2023). Process-oriented impacts of microplastic fibers on behavior and histology of fish. *Journal of Hazardous Materials, 448*, Article 130856. https://doi.org/10.1016/j.jhazmat.2023.130856

Lithner, D., Larsson, Å., & Dave, G. (2011). Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Science of the Total Environment, 409*(18), 3309–3324. https://doi.org/10.1016/j.scitotenv.2011.04.038

McIlwraith, H. K., Kim, J., Helm, P., Bhavsar, S. P., Metzger, J. S., & Rochman, C. M. (2021). Evidence of microplastic translocation in wild-caught fish and implications for microplastic accumulation dynamics in food webs. *Environmental Science & Technology, 55*(18), 12372–12382. https://doi.org/10.1021/acs.est.1c02922

Mercogliano, R., Avio, C. G., Regoli, F., Anastasio, A., Colavita, G., & Santonicola, S. (2020). Occurrence of microplastics in commercial seafood under the perspective of the human food chain. A review. *Journal of Agricultural and Food Chemistry, 68*(19), 5296–5301. https://doi.org/10.1021/acs.jafc.0c01209

Muncke, J., Andersson, A.-M., Backhaus, T., Boucher, J. M., Carney Almroth, B., Castillo, A. C., Chevrier, J., Demeneix, B. A., Emmanuel, J. A., Fini, J.-B., Gee, D., Geueke, B., Groh, K., Heindel, J. J., Houlihan, J., Kassotis, C. D., Kwiatkowski, C. F., Lefferts, L. Y., Maffini, M. V., … Vom Saal, F. S. (2020). Impacts of food contact chemicals on human health: A consensus statement. *Environmental Health, 19*(1), 25. https://doi.org/10.1186/s12940-020-0572-5

Nalbone, L., Giarratana, F., Genovese, M., & Panebianco, A. (2024). Occurrence of microplastics in store-bought fresh and processed clams in Italy. *Marine Pollution Bulletin, 206*, Article 116739. https://doi.org/10.1016/j.marpolbul.2024.116739

Nikova, E. V., Temkov, M., & Rocha, J. M. (2023). Occurrence of meso/micro/nano plastics and plastic additives in food from food packaging. In *Advances in food and nutrition research, 103* pp. 41–99). Academic Press. https://doi.org/10.1016/bs.afnr.2022.08.001.

Paul, A. T., Hannon, C., Švonja, M., Connellan, I., & Frias, J. (2023). Efficacy of microplastic depuration on two commercial oyster species from the west coast of Ireland. *Journal of the World Aquaculture Society, 54*(5), 1217–1234. https://doi.org/10.1111/jwas.12989

Pititto, A., Rainone, D., Sannino, V., Chever, T., Herry, L., Parant, S., Souidi, S., Ballesteros, M., Chapela, R., & Santiago, J. L. (2021). Research for PECH committee–impacts of the COVID-19 pandemic on EU fisheries and aquaculture. European parliament. *Policy Department for Structural and Cohesion Policies, Brussels*. http://icsfarchives.net/id/eprint/20044.

Prata, J. C., Da Costa, J. P., Lopes, I., Duarte, A. C., & Rocha-Santos, T. (2020). Environmental exposure to microplastics: An overview on possible human health effects. *Science of the Total Environment, 702*, Article 134455. https://doi.org/10.1016/j.scitotenv.2019.134455

Quaglia, N. C., Capuozzo, F., Ceci, E., Cometa, S., Di Pinto, A., Mottola, A., Piredda, R., & Dambrosio, A. (2023). Preliminary survey on the occurrence of microplastics in bivalve mollusks marketed in Apulian fish markets. *Italian Journal of Food Safety, 12*, Article 10906. https://doi.org/10.4081/ijfs.2023.10906

Rist, S., Almroth, B. C., Hartmann, N. B., & Karlsson, T. M. (2018). A critical perspective on early communications concerning human health aspects of microplastics. *Science of the Total Environment, 626*, 720–726. https://doi.org/10.1016/j.scitotenv.2018.01.092

Siddique, M. A. M., Hossain, I., Sunji, M. M. R., Tahsin, T., Walker, T. R., & Rahman, M. S. (2024). Characterization, source identification and hazard index assessment of ingested microplastics in farmed tilapia Oreochromis niloticus. *Ecological Indicators, 158*, 111334.

Siddiqui, S. A., Bahmid, N. A., Salman, S. H. M., Nawaz, A., Walayat, N., Shekhawat, G. K., Gvozdenko, A. A., Blinov, A. V., & Nagdalian, A. A. (2023). Migration of microplastics from plastic packaging into foods and its potential threats on human health. In *Advances in food and nutrition research, 103* pp. 313–359). Academic Press. https://doi.org/10.1016/bs.afnr.2022.07.002.

Silva, D. M., Almeida, C. M. R., Guardiola, F. A., Pereira, R., Rodrigues, S. M., & Ramos, S. (2024). Uncovering microplastics contamination in canned seafood. *Food Chemistry, 448*, Article 139049. https://doi.org/10.1016/j.foodchem.2024.139049

Smith, M., Love, D. C., Rochman, C. M., & Neff, R. A. (2018). Microplastics in seafood and the implications for human health. *Current Environmental Health Reports, 5*(3), 375–386. https://doi.org/10.1007/s40572-018-0206-z

Süssmann, J., Krause, T., Fischer, E. K., Walz, E., Greiner, R., Rohn, S., & Fritsche, J. (2025). Microplastics in fresh and processed seafood–A survey of products sold in Germany. *Food Control.* , Article 111565. https://doi.org/10.1016/j.foodcont.2025.111565

Tang, R., Zhang, T., Song, K., Sun, Y., Chen, Y., Huang, W., & Feng, Z. (2022). Microplastics in commercial clams from the intertidal zone of the South Yellow Sea, China. *Frontiers in Marine Science, 9*, Article 905923.

Tocher, D. R., Betancor, M. B., Sprague, M., Olsen, R. E., & Napier, J. A. (2019). Omega-3 long-chain polyunsaturated fatty acids, EPA and DHA: Bridging the gap between supply and demand. *Nutrients, 11*(1), 89. https://doi.org/10.3390/nu11010089

Unuofin, J. O., & Igwaran, A. (2023). Microplastics in seafood: Implications for food security, safety, and human health. *Journal of Sea Research, 194*, Article 102410. https://doi.org/10.1016/j.seares.2023.102410

Van Cauwenberghe, L., & Janssen, C. R. (2014). Microplastics in bivalves cultured for human consumption. *Environmental Pollution, 193*, 65–70. https://doi.org/10.1016/j.envpol.2014.06.010

van Raamsdonk, L. W., van der Zande, M., Koelmans, A. A., Hoogenboom, R. L., Peters, R. J., Groot, M. J., Peijnenburg, A. A. C. M., & Weesepoel, Y. J. (2020). Current insights into monitoring, bioaccumulation, and potential health effects of microplastics present in the food chain. *Foods, 9*(1), 72. https://doi.org/10.3390/foods9010072

Vdovchenko, A., & Resmini, M. (2024). Mapping microplastics in humans: Analysis of polymer types, and shapes in food and drinking water—a systematic review. *International Journal of Molecular Sciences, 25*(13), 7074. https://doi.org/10.3390/ijms25137074

Vethaak, A. D., & Leslie, H. A. (2016). Plastic debris is a human health issue. *Environmental Science and Technology, 50*(13), 6825–6826.

Vitali, C., Peters, R. J., Janssen, H. G., & Nielen, M. W. (2023). Microplastics and nanoplastics in food, water, and beverages; part I. Occurrence. *TrAC, Trends in Analytical Chemistry, 159*, Article 116670. https://doi.org/10.1016/j.trac.2022.116670

Wu, L., Dong, J., Shen, Z., & Zhou, Y. (2024). Microplastics as vectors for antibiotic resistance: Role of pathogens, heavy metals, and pharmaceuticals and personal care products. *Journal of Water Process Engineering, 67*, Article 106124. https://doi.org/10.1016/j.jwpe.2024.106124

Yuan, Z., Nag, R., & Cummins, E. (2022). Ranking of potential hazards from microplastics polymers in the marine environment. *Journal of Hazardous Materials, 429*, Article 128399.

Zhang, F., Man, Y. B., Mo, W. Y., Man, K. Y., & Wong, M. H. (2020). Direct and indirect effects of microplastics on bivalves, with a focus on edible species: A mini-review. *Critical Reviews in Environmental Science and Technology, 50*(20), 2109–2143. https://doi.org/10.1016/j.jhazmat.2022.12839

Zimmermann, L., Geueke, B., Parkinson, L. V., Schür, C., Wagner, M., & Muncke, J. (2025). Food contact articles as source of micro-and nanoplastics: A systematic evidence map. *Npj Science of Food, 9*(1), 111. https://doi.org/10.1038/s41538-025-00470-3