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Microplastic contamination of packaged meat: Occurrence and associated risks

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25 MP-XPS. In addition to these MP-XPS, it should also be pointed out that fibres can also
26 contaminate meat.

27

28 *Keywords:*

29 Microplastics, extruded polystyrene, fibres, packaging, contaminant, human alimentation

30 **1. Introduction**

31 Between 1950 and 2015, 7,800 million tons of plastic were produced, half of which
32 between 2002 and 2015 (Geyer et al., 2017). The physical properties of plastic materials make
33 them essential in industrialized societies. In Europe, the packaging, construction, and
34 automotive sectors account for nearly 70% of the demand for plastic, with nearly 40% for
35 packaging alone (PlasticsEurope, 2018). In the food sector, the use of plastic packaging helps
36 in the storage, transport, protection, and preservation of products while reducing their waste
37 (Lange & Wyser, 2003; Mathlouthi, 2013; Piringer & Baner, 2008; PlasticsEurope, 2012).
38 Because some plastics such as expanded polystyrene (EPS) or extruded polystyrene (XPS)
39 provide a good protection barrier from oxygen, water vapour, and microorganisms, they
40 facilitate the preservation of food products at a desired temperature; hence, they are widely
41 used in food packaging. However, it has been recently reported that packaging may release
42 plastic particles and subsequently contaminate our food with plastic fragments (Oßmann et
43 al., 2018; Schymanski et al., 2018; Winkler et al., 2019).

44 The presence of small plastic particles in the natural environment has been known since the
45 early 1970s (Buchanan, 1971), but it is only in the early 2010s that their presence in food was
46 reported (L. G. A. Barboza et al., 2018). Among these, microplastics, namely plastic particles
47 smaller than 5 mm in size (GESAMP, 2019), were found. In 2013, a study carried out on
48 honey and sugar coming from different countries revealed contamination by fibres and
49 fragments (Liebezeit & Liebezeit, 2013). Although no chemical analysis was performed to

50 assess the chemical nature of these particles, a connection was established between the
51 morphology of these fragments and the plastic bags used by beekeepers to supply sugar to
52 bees. In 2014, the identification of microplastics in mussels (*Mytilus edulis*) grown for human
53 consumption suggests that humans ingest these particles (Van Cauwenberghe & Janssen,
54 2014). To date, several studies have reported the presence of plastic particles in seafood and
55 fish for human consumption (Azevedo-Santos et al., 2019; Barboza & Gimenez, 2015).
56 Microplastics have also been observed in salt (Gündoğdu, 2018; Iñiguez et al., 2017; Karami
57 et al., 2017; Yang et al., 2015), in beers (Kosuth et al., 2018; Liebezeit & Liebezeit, 2014),
58 and in water bottles (Mason et al., 2018). These microplastics, generally smaller than 100µm,
59 probably come partly from the packaging and/or bottling process (Mason et al., 2018). A
60 large-scale study of tap water showed that 81% of the water sampled was contaminated
61 mainly by fibres of which an unknown part is of synthetic origin (Kosuth et al., 2018). Fibres
62 can be defined as particles of equal thickness along their entire length, the difference between
63 artificial and natural fibres being based on the presence or absence of visible cellular or
64 organic structures (Hidalgo-Ruz et al., 2012). For the purposes of this publication, no
65 distinction will be made between artificial and natural fibres. The origin of the fibres observed
66 in food is not always clear, but airborne contamination is often thought of. The atmosphere is
67 an important vector of microplastics, as shown by the atmospheric deposition of synthetic
68 fibres in Paris (Boucher & Friot, 2017; Dris et al., 2016). These fibres are also omnipresent in
69 our indoor environment to the point of posing a significant risk of contamination of samples
70 (Dehaut et al., 2019). It is commonly accepted that synthetic textiles are the main source of
71 these fibres (Boucher & Friot, 2017; Prata, 2018).

72 This study is part of the OceanWise project, a European project supported by the European
73 funding program INTERREG Atlantic Area. OceanWise aims to jointly develop a set of long-
74 term measures to reduce the impact of expanded and extruded polystyrene (EPS/XPS)

75 products in the North-East Atlantic Ocean. Some economic activities, which generate
76 EPS/XPS waste, are particularly investigated in the OceanWise project: the fishing industry
77 (fisheries, aquaculture, sea-food), the food industry (supermarket chains, distribution of
78 vegetables, fish, meat, fruit), consumer goods, outdoor festivals, and tourism. In this context,
79 several observations of more or less free sub-millimetre particles inside food packaging with
80 an XPS bottom containing meat were made during a preliminary study for the OceanWise
81 project tests. These microparticles have the same colour as the XPS food packaging that
82 contained them. Faced with these elements, it was therefore decided to verify the following
83 hypotheses: Are these particles microplastics of extruded polystyrene (MP-XPS)? What is the
84 estimated mass of MP-XPS per mass of meat? How many fibres are present on the surface
85 and inside the food packaging? In an attempt to determine the origin of the particles and fibres
86 as well as the timing of the deposition, the distribution of MP-XPS was studied inside and
87 outside the tray. Thus, this preliminary study, limited to the case of France, seeks to answer
88 these questions and identify some elements of discussion regarding the possible origins of
89 these particles, as well as the possible consequences of ingesting them for human health.

90 **2. Materials and methods**

91 ***2.1. Sample collection and preparation***

92 This study focuses on meat products (chicken) packed in extruded polystyrene trays
93 (230x140x20mm). The mass of the pieces of meat was determined by reading the mass
94 displayed on the label on the plastic packaging. Samples of meat from brands B, C, D had a
95 roughly equivalent mass (on average close to 250g), while brand A meat samples were
96 slightly larger in mass (about 315g) (table 1). Products from four different brands (named A,
97 B, C, and D) were purchased in a local supermarket (n=3 by brand). No special conditions
98 were taken to transport (in few minutes) them to the laboratory.

99 Table 1. Sample characteristics.

| Brand | Colour of the tray | Meat | Mass of meat (g) |
|-------|--------------------|----------------------|------------------|
| A | Black | White chicken breast | 315±15 |
| B | Yellow | Turkey escalope | 239±47 |
| C | Yellow | Turkey escalope | 254±36 |
| D | Yellow | Turkey escalope | 240±47 |

100

101 It is necessary to take into consideration all the surfaces of the packaging in order to describe
102 the state of contamination of the tray and obtain information on the sources of this
103 contamination. Thus, the outside of the packaging (i.e. tray and plastic film) was first rinsed
104 with distilled water. This rinsing water was then analysed to determine the amounts of
105 particles and fibres on the outside of the packaging. In order to avoid generating MP-XPS
106 when tearing off the plastic film sealing the trays, the tray openings were made with a scalpel.
107 Then, after opening, the meat was rinsed thoroughly until the thin layer of fat covering the
108 surface of the samples was completely collected. Finally, the inside of the tray and plastic film
109 were rinsed with distilled water. The rinsing water was then vacuum filtered (Buchner JIPO,
110 62 mm) on a glass microfiber filter (pore diameter: 0.8µm, diameter 55 mm, Fisherbrand
111 MF300). This step allows the fat layer to be vacuumed with the water, while the particles
112 remain trapped on the surface of the filter. In order to limit external contamination by fibres,
113 rinsing the packaging and meat and filtering were carried out under a funnel hood. The
114 operators conducting the experiments wore a cotton gown and nitrile gloves to avoid
115 contamination by fibres from their clothing. The glassware used was systematically rinsed at
116 least three times with distilled water and one last time with ethanol. After the filtration step,
117 the filters were stored in glass petri dishes also in order to limit contamination by the fibres
118 present in the ambient environment.

119 ***2.2. Isolation, visual characterization, and identification of microplastics***

120 A dissection microscope (30X magnification) was used to count fibres and potential MP-XPS.
121 All fibres were counted, but their chemical nature was not identified. The spectra of all
122 recovered fragments were acquired using an Attenuated Total Reflection Fourier Transform
123 Infrared microspectrometer (ATR-FTIR Lumos, Bruker). All spectra were recorded in the
124 absorbance mode in the 4,000-600 cm⁻¹ region with 4 cm⁻¹ resolution and 16 scans. The
125 spectra were acquired and compared to reference spectra of polystyrene.

126 The maximum and minimum Feret diameters of the particles were measured using the camera
127 of the Lumos microscope associated with the OPUS software. These diameters were then
128 used to estimate the volume of the particles. For this calculation, the shape of the particle was
129 approximated to a spheroid, with the two smallest axes considered as equal.

130 ***2.3. Estimated per capita consumption of microplastics and statistical analysis***

131 Since the microplastics were too light to be weighted by a precision balance (readability:
132 0.1mg), the mass of the particles was calculated according to two density hypotheses. The low
133 hypothesis corresponds to the density of extruded polystyrene (40 kg.m⁻³), whereas the high
134 hypothesis corresponds to that of polystyrene (1040 kg.m⁻³) (Kedzierski, Le Tilly, César, et
135 al., 2017). The per capita consumption of MP-XPS was calculated using the weight of the
136 meat samples and the average meat consumption rate of a French person (Tavoularis &
137 Sauvage, 2018). The following equation was used:

$$138 \quad N_t = N_{MP}/m * m_t \quad (1)$$

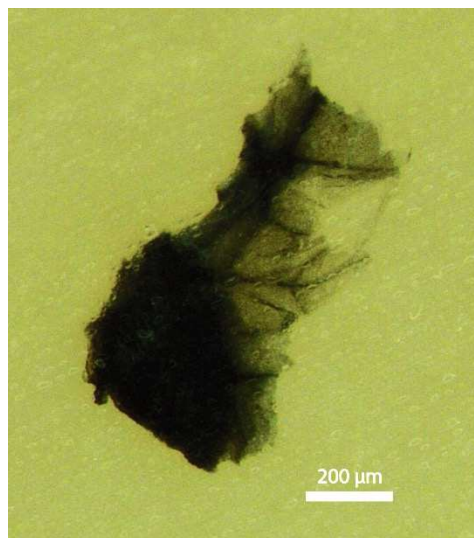
139 Where, N_t is the number of microplastics per person and per day, N_{MP} is the number of
140 microplastics per tray, m is the mass of meat in the tray, and m_t is the mass of meat
141 consumed per unit of time and per person (135 g/d/person).

142 The equality of group averages was then measured using the Excel Student and the Fisher
143 tests.

144 **3. Results**

145 **3.1. Particles**

146 The characteristics of the microparticles were the same as those of the XPS tray: presence of
147 vacuoles, identical colour, and identical chemical nature. There is therefore no doubt that they
148 came from the tray. FTIR spectra and the microscopic observations of the collected particles
149 showed the presence of MP-XPS microparticles inside the packaging as well as on its outer
150 surface. It should also be noted that some MP-XPS were found between the meat and the
151 plastic seal (Figure 1). The MP-XPS particles stuck on the surface of the meat were difficult
152 to recover during the rinsing phase. A thorough rinsing was necessary each time to recover
153 the particles, usually also by removing the layer of fat from the surface of the meat. The
154 average size of MP-XPS, measured along the main axis, was comprised between 300 and
155 450 μm . On the secondary axis, the average size of MP-XPS varied between 130 and 250 μm .
156 The variability in particle size was quite high within the packaging of the same brand. The
157 colour of the MP-XPS was generally the same as that of the XPS tray. The morphology of the
158 particles was variable. Some seemed to belong to porosity walls, others were more or less
159 compressed blocks of XPS. Thus, we used the density of XPS (40 kg/m^3) and polystyrene
160 ($1,040 \text{ kg/m}^3$) to calculate the total mass of MP-XPS per sample (Kedzierski, Le Tilly, Cesar,
161 et al., 2017).

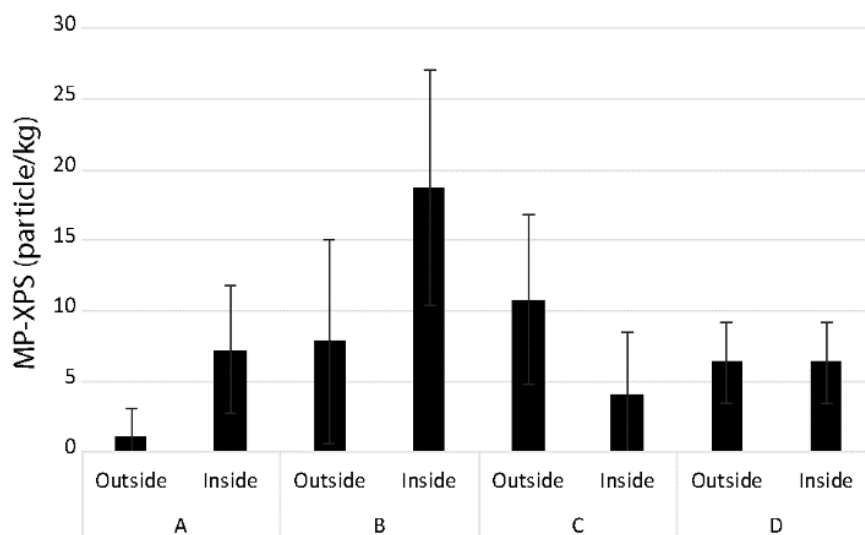


162

163 **Figure 1.** MP-XPS observed with a dissecting microscope (the Photoshop software was used
164 to improve the image brightness). The observed particle was trapped between the sealing film
165 and the meat. In this example, the particle was black because the tray itself was that colour. In
166 the yellow trays, the particles were yellow. In this example, the particle contrasts with the
167 colour of the meat, making it very easy to observed. The structure of the particle corresponds,
168 for the thinnest parts, to the walls of the porosity of the XPS, and for the thickest part, to a
169 fragment of XPS. Therefore, the density of these MP-XPS varies extensively, as noted above.

170

171 The average quantities of MP-XPS observed on the external surface of the food packaging
172 varied, depending on the brands, between 1.1 ± 1.9 and 10.8 ± 6.0 MP-XPS/kg of packaged
173 meat (Figure 2). Variations were large for the packaging samples of the same brand. The
174 amounts of MP-XPS recovered inside the packaging were on average higher than those found
175 outside for brands A and B whereas the opposite was observed for brand C. The average
176 quantities of MP-XPS observed on the internal surface of food packaging varied, depending
177 on the brands, between 4.0 ± 4.5 and 18.7 ± 8.3 MP-XPS/kg of packaged meat. The amounts
178 recovered inside and outside were similar for brand D. Due to the large variability between
179 samples of the same brand, there was no statistically significant difference between the
180 quantities of MP-XPS inside and outside the packaging. For all the brands under
181 investigation, the number of MP-XPS was generally below 20. Although the amounts of MP-
182 XPS found on the surface of packaging A were on average lower than those of other brands,
183 there was also no statistically significant difference from one brand to another. Similarly,
184 although the quantities of particles found inside tray B were on average much higher than
185 those observed in the other trays, no statistical differences were clearly identified. However,
186 the *p*-values were very close to 0.05. There is therefore a weak statistical presumption that
187 brand B meat is more contaminated than other brands.



188

189 **Figure 2.** Number of MP-XPS per kilogram of meat observed at the surface of and inside the
 190 food packaging for the different brands.

191 The masses of MP-XPS were calculated per meat mass according to the two extreme particle
 192 densities (40 and 1,040 kg/m³) (Table 1). In the first case, the masses of MP-XPS present on
 193 the outer surface of the packaging ranged from 5 µg/kg of packaged meat (brand A) to
 194 93 µg/kg (brand B). Within the packaging, the masses of MP-XPS varied between 2 µg/kg
 195 (brand D) and 402 µg/kg (brand B). In the second case, the masses of MP-XPS present on the
 196 outer surface of the packages varied between 0.14 mg/kg of packaged meat (brand A) and
 197 2.4 mg/kg (brand B). Inside the packaging, the masses of MP-XPS varied between 54 µg/kg
 198 (brand D) and 10.5 mg/kg (brand B).

199 **Table 1.** Mass of MP-XPS per kilogram of meat for the different brands estimated from each
 200 density hypothesis.

| Brand | | Hypothesis 1: 40 kg/m ³ mass of MP-XPS in mg/kg | Hypothesis 2: 1,040 kg/m ³ mass of MP-XPS in mg/kg |
|-------|---------|---|--|
| A | Outside | 0.005 ± 0.009 | 0.14 ± 0.24 |
| | Inside | 0.09 ± 0.12 | 2.2 ± 3.1 |
| B | Outside | 0.09 ± 0.16 | 2.4 ± 4.2 |

| | | | |
|---|---------|-------------------|-----------------|
| | Inside | 0.4 ± 0.5 | 10.5 ± 13.7 |
| C | Outside | 0.02 ± 0.03 | 0.5 ± 0.8 |
| | Inside | 0.2 ± 0.2 | 4.5 ± 4.5 |
| D | Outside | 0.05 ± 0.08 | 1.2 ± 2.0 |
| | Inside | 0.002 ± 0.001 | 0.05 ± 0.03 |

201

202 Based on the average French daily consumption of meat, the mass of XPS observed inside the

203 packaging and potentially ingested per day and per year was calculated (Table 2). If the

204 particle density was 40 kg/m^3 (low hypothesis), the average mass of XPS ingested per day

205 could range from $0.1 \text{ }\mu\text{g/d}$ (brand D) to $54 \text{ }\mu\text{g/d}$ (brand B). Per year, this would vary between

206 0.04 mg/y (brand D) and 19.7 mg/y (brand B). For a particle density of $1,040 \text{ kg/m}^3$ (high

207 hypothesis), the average mass of XPS ingested per day could vary between $7 \text{ }\mu\text{g/d}$ (brand D)

208 and 1.4 mg/d (brand B). Per year, this could range from 2.6 mg/y (brand D) to 511 mg/y

209 (brand B).

210 **Table 2.** Consumption of MP-XPS per person based on a meat consumption of

211 135 g/d/person .

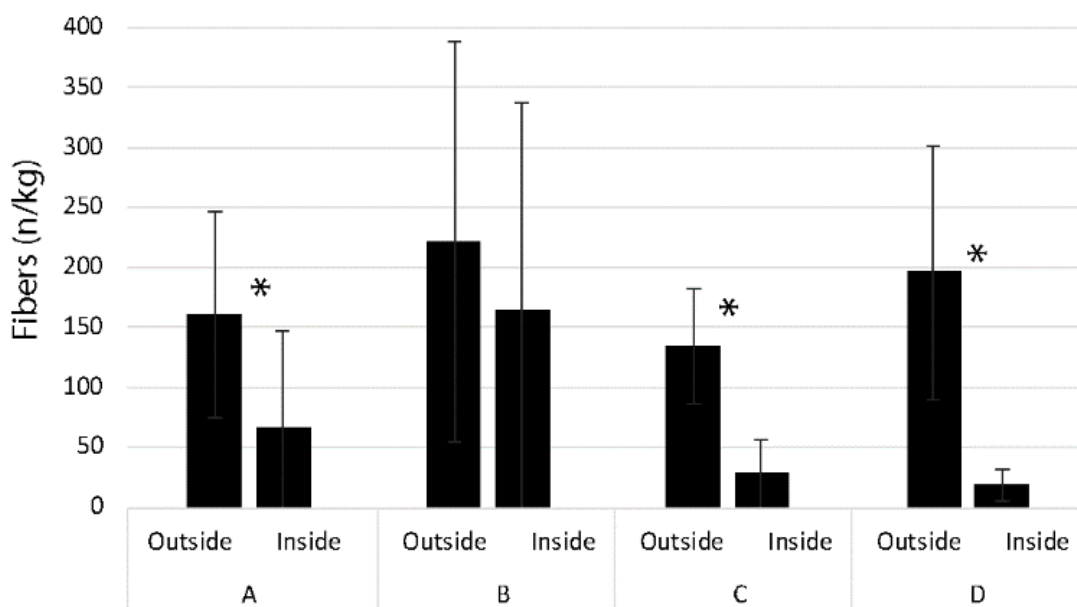
| Brand | MP-XPS density | Daily consumption (mg/d) | Annual consumption (mg/y) |
|-------|--------------------------------------|-----------------------------|------------------------------|
| A | Hypothesis 1: 40 kg/m^3 | 0.01 ± 0.02 | 4.2 ± 6.0 |
| | Hypothesis 2: $1,040 \text{ kg/m}^3$ | 0.3 ± 0.4 | 110.1 ± 154.9 |
| B | Hypothesis 1: 40 kg/m^3 | 0.5 ± 0.07 | 19.8 ± 26.0 |
| | Hypothesis 2: $1,040 \text{ kg/m}^3$ | 1.4 ± 1.9 | 515.2 ± 675.3 |
| C | Hypothesis 1: 40 kg/m^3 | 0.02 ± 0.02 | 8.6 ± 8.4 |
| | Hypothesis 2: $1,040 \text{ kg/m}^3$ | 0.6 ± 0.6 | 223.2 ± 219.4 |
| D | Hypothesis 1: 40 kg/m^3 | 0.0003 ± 0.0001 | 0.1 ± 0.05 |

| | | | |
|--|---------------------------------------|---------------|-----------|
| | Hypothesis 2: 1,040 kg/m ³ | 0.007 ± 0.003 | 2.7 ± 1.3 |
|--|---------------------------------------|---------------|-----------|

212

213 **3.2. Fibres**

214 On average, the number of fibres observed on the surface of the food packaging was between
 215 134 fibres/kg (brand C) and 221 fibres/kg (brand B) (Figure 3). For most of the brands tested,
 216 the number of fibres inside the packaging was between 18 fibres/kg (brand D) and
 217 164 fibres/kg (brand B). For brands A, C, and D, the quantities of fibres were therefore
 218 higher outside than inside. In the case of brand D, the amounts of fibres were about 11 times
 219 lower on the inside of the package than on the outside. For brands A and C, the amounts were
 220 between 2 and 5 times lower. However, this was not true for brand B, where the number of
 221 fibres inside and outside of the packaging was quite similar. Thus, the differences between the
 222 amounts observed inside and outside the packaging were significant for brands A, C, and D.
 223 For fibres on the external food packaging, there was no statistical difference between the
 224 brands, nor was there any statistical difference between brands for the fibres present inside the
 225 packaging. This was due in particular to the variability observed from one sample to another
 226 for the same brand.



227

228 **Figure 3.** Amounts of microfibrils per kilogram of packaged meat observed on the surface and
229 inside of the food packaging examined. * Results that show statistical differences (p -value <
230 0.05).

231 **4. Discussion**

232 ***4.1. Contamination of the surface of meat products with microplastics***

233 *4.1.1. Sources of MP-XPS*

234 To our knowledge, this study is the first one to draw attention to the microplastics found on
235 the surface of meat products sealed in their packaging. This contamination necessarily raises
236 the question of the origin of the microplastics. The observation of the presence of MP-XPS
237 inside and outside the tray, as well as between the tray and the meat, and between the meat
238 and the protective film, leads to the hypothesis that the deposition of particles starts before the
239 meat was deposited and lasts at least until the film was closed. It is possible that the
240 packaging may be contaminated by XPS dust suspended in the air of the production buildings.
241 The MP-XPS are easily airborne because of their low size and mass and because their
242 electrostatic properties make them sticky. It should also be noted that the presence of
243 microplastics trapped between the meat and the plastic film that seals the tray implies that
244 MP-XPS are present, suspended in the air, in the food preparation areas.

245 *4.1.2. Areas for study improvement*

246 Feedback on the protocol proposed within the framework of this study has generated several
247 propositions for improvement. In particular, it would be desirable to work on more samples of
248 the same brand. When the protocol was designed, working on the basis of a triplicate seemed
249 relevant. However, due to the variability observed from one sample to another, it appears *a*
250 *posteriori* that this number could be increased. In the case of the study of microplastics
251 ingested by living organisms, a minimal number of 50 individuals per species is

252 recommended to obtain a representative sample (Dehaut et al., 2019). This recommendation
253 may also be relevant to the study of microplastic contamination of packaged food.

254 **4.2. Ingesting MP-XPS: A threat to human health?**

255 Regardless of the source of these MP-XPS, it should be noted that despite the thorough
256 rinsing of the surface of the meat examined, plastic particles often remained trapped on the
257 meat surface. Thus, whether the meat is rinsed or not, the MP-XPS are likely to be present
258 when the food is cooked, and some will be ingested.

259 *4.2.1. Potential impacts of polystyrene microplastics (MP-PS) on human health*

260 While it has sometimes been argued that certain species fished are potential microplastic
261 vectors for humans, extraction from the digestive system (containing microplastics) before
262 consumption greatly reduces this risk (Alomar & Deudero, 2017; Rummel et al., 2016;
263 Sanchez et al., 2014; L Van Cauwenberghe et al., 2015; Lisbeth Van Cauwenberghe &
264 Janssen, 2014; Vroom et al., 2016). Another source of contamination of human food comes
265 from the packaging (Schymanski et al., 2018). The masses of MP-XPS ingested per day and
266 per person could reach 1.4 mg based on the results obtained in this study, but there is no
267 reference or standard for determining whether or not this value is dangerous to human health.
268 In fact, the potential impact of MP-PS ingestion on human health remains very limited (Wang
269 et al., 2019). A study conducted on human epithelial (HeLa) and cerebral (T98G) cell lines
270 has shown that PS microspheres (diameter 10µm) caused oxidative stress (Schirinzi et al.,
271 2017). On the contrary, MP-PS (1, 4 and 10µm) do not seem to have an impact on some
272 human immune cells (macrophages) (Stock et al., 2019). In mice exposed to 0.5-50 µm
273 polystyrene particles, disruptions in energy and lipid metabolism, oxidative stress, neurotoxic
274 disorders, and microbiotic dysbiosis were observed that suggest potential risks to mammalian
275 health (Deng et al., 2017; Lu et al., 2018). However, these results are qualified by a recent

276 study suggesting that, under certain conditions, oral exposure to microplastic particles does
277 not pose an acute risk to mammalian health (Stock et al., 2019).

278 However, several limitations to the transferability of these hazardous effects to the human
279 case must be mentioned. First, the sensitivity to plastics may vary between animal species
280 (Walum, 1998). Second, it is necessary that the plastic quantities ingested and the exposure
281 duration of the animals be relevant to the quantities actually observed in their natural
282 environment or, as in this case, in human food (Bour et al., 2018; Lönnstedt & Eklöv, 2016).

283 In experiments conducted on mouse exposure to MP-PS, the dose tested was 0.1 mg/d, which
284 is in line with the amounts that could be ingested by humans (Deng et al., 2017). However,
285 this dose is much higher if we consider the difference in body weight between a mouse and a
286 human being. Thus, it is important to note that the polystyrene particles used in these studies
287 are one to two orders of magnitude smaller than those observed in the present study.

288 Nevertheless, it cannot be completely excluded that polystyrene particles from a few
289 micrometres to a few tens of micrometres in size were not effectively detected and extracted
290 from the surface of the samples. However, while it is possible that the total number of
291 microplastics may be underestimated, the very low mass of these fine particles should not
292 have too much influence on the total mass values of MP-XPS calculated in this study.

293 However, it should be noted, that it is essentially on this size range of small microplastics
294 (less than a few tens of microns) that the discussion on the toxicity of polystyrene particles is
295 based. Therefore, a better assessment of the amount of these small microplastics present on
296 the surface of foods appears to be important to better identify the associated risks. Finally, the
297 number of studies on the impact of MP-PS on mammals is very low, which makes it difficult
298 to accurately assess their potential impact on humans (Rainieri & Barranco, 2019).

299 *4.2.2. Styrene desorption*

300 In addition, for polystyrene, there is a risk of desorption of styrene monomers during the use
301 phase (Rist et al., 2018). Styrene is quickly absorbed by organisms through ingestion or
302 inhalation (Bonnard et al., 2016). It is stored in fatty tissues where it is rapidly metabolized
303 (half-life of about 6 hours). After metabolisation, it is mainly eliminated by urine. Styrene
304 therefore does not accumulate in organisms. In terms of chronic toxicity, in rats, ingestion of
305 high doses of styrene (1.0 g/kg/d, 5d/week, 28d) causes irritation of the gastrointestinal tract,
306 which is lethal. This dose corresponds, for a person weighing 70kg, to about 50,000 times the
307 maximum daily mass of MP-XPS potentially ingested per day and per person recorded in our
308 study (1.4 mg/d). Since a small fraction of styrene is desorbed from MP-XPS, the doses are
309 actually even lower.

310 At lower doses, ingestion induces changes in kidney and liver weight. Styrene is metabolized
311 by cytochrome P450 enzymes to styrene-7,8-oxide that undergoes hydration or conjugation
312 with glutathione (Shen, Li, Ding, & Zheng, 2014). In mice, styrene is involved in lung tumour
313 development, whereas no data indicate a similar effect in humans (Cruzan et al., 2018). The
314 derived no-effect level (DNEL) of styrene is 7.7 µg/kg bw/d (European Chemicals Agency,
315 2019e), which represents, in the framework of our study, about 0.4 times the maximum daily
316 mass of MP-XPS potentially ingested per day and per person weighing 70 kg. However,
317 again, the amounts of styrene released from MP-XPS are most likely much lower. Finally, in
318 humans, styrene absorption results in memory loss, difficulties in concentration and learning,
319 brain and liver damage, and cancer (Gibbs & Mulligan, 1997).

320 More generally, according to a recent study, the toxicity of polystyrene packaging can vary
321 greatly from one polystyrene sample to another (Zimmermann et al., 2019). Thus, some PS
322 samples were cytotoxic to the cells used in the AREc32 test at lower concentrations than the
323 other plastics tested. This study also highlights a toxicity linked to oestrogenic (Yeast
324 Estrogen Screen) and especially anti-androgenic (Yeast Anti-Androgen Screen) effects for

325 some samples. This implies that some additives used in the food packaging have endocrine
326 activity. Due to higher average meat consumption among men than women (Hercberg &
327 Tallec, 2000), there is a differential exposure to MP-XPS between the sexes. Similarly, in
328 France, the protein intake reaches its maximum around adolescence (Hercberg & Tallec,
329 2000), corresponding with a period of vulnerability to endocrine disruptors.

330 Thus, the ingestion of microplastics could be an additional route of contamination that would
331 add to the more traditional route of contamination via food. However, this contamination
332 would be several orders of magnitude lower than that passing through food products (Rist et
333 al., 2018).

334 *4.2.3. Cooking microplastics*

335 Cooking conditions can vary considerably from one cooking method to another. Thus, the
336 temperatures reached during cooking may vary between 100 and 230°C or much more for
337 open fire grills. When subjected to temperatures between 200 and 300°C, the molecular
338 weight of polystyrene decreases due to thermal degradation inducing random scission of the
339 carbon chain, and disproportionation of the macroradicals formed, leading to the release of
340 many degradation products (McNeill et al., 1990). At 200°C, the rate of volatile fragments
341 formed is low, but the amounts of degradation residues may be significantly higher. At around
342 300°C, gaseous styrene is mainly produced, but so are other compounds such as ethylbenzene,
343 methylbenzene (or toluene), benzene, phenylpropene, α -methylstyrene, 1-methylindene, and
344 naphthalene (McNeill et al., 1990). The toxicity of these molecules cannot be established for
345 the doses of MP-XPS quantified on the surface of the meat even if an endocrine activity of
346 these molecules cannot be completely excluded (Zimmermann et al., 2019). Furthermore,
347 based on the DNEL of styrene (7.7 $\mu\text{g}/\text{kg bw}/\text{d}$), low toxicity leading to irritation cannot be
348 completely ruled out (European Chemicals Agency, 2019).

349 *4.3. Microfibres: Another contaminant*

350 The number of fibres observed inside the trays is of the same order of magnitude as that
351 observed in previous studies on other food products; as for honey, sugar, or salt, the observed
352 quantities were generally lower than 1,000 fibres per kg of food (Gündoğdu, 2018; Iñiguez et
353 al., 2017; Karami et al., 2017; Liebezeit & Liebezeit, 2013). If, in the case of these studies, the
354 origin of the contamination is not always easy to determine (contamination in the natural
355 environment or during the manufacturing process of the product), in the case of our study, the
356 presence of fibres inside the tray implies the presence of these fibres in the production
357 environment of the food product. In the absence of analysis of procedural blank, it cannot be
358 totally excluded that a fraction of the fibres observed are derived from airborne
359 contamination. However, the differences observed, in particular between the inner and outer
360 surfaces of the trays, are statistically significant. It is highly unlikely that, under similar study
361 conditions such large variations can be due to contamination without affecting the variance
362 within the same brand.

363 The chemical nature of these fibres, outside the scope of the preliminary analysis, was not
364 analysed. Thus, it is not possible to specify whether these fibres are mineral or organic. The
365 impact of the inhalation of man-made mineral fibres is well known; as a function of different
366 parameters such as particle size and composition, they have the intrinsic potential to induce
367 oxidative stress, inflammation, genotoxicity, and carcinogenicity (Greim, 2004).

368 Similarly, the consequences of the inhalation of man-made organic fibres is known (Gasperi
369 et al., 2018; Greim, 2004). For example, various studies conducted on cohorts of textile
370 workers have shown no or a moderate impact on the respiratory tract (Warheit et al., 2001).

371 The impact of fibre ingestion is less well documented as this mode of contamination appears,
372 in the air, to be secondary to respiration. Very few studies report cases of airborne ingestion
373 of these fibres by living beings (Cook & Olson, 1979; Zhao et al., 2016). Hence, apart from
374 the particular case of certain mineral fibres (e.g. asbestos), which has been well documented

375 for several decades (Davis, 1981), the impact of fibre ingestion on human health is poorly
376 described in the scientific literature. Nevertheless, it should also be noted that the average
377 amount of fibres outside the tray is up to 10 times higher than inside the tray. It thus appears
378 that the packaging plays a protective role against microfibres.

379 **5. Conclusion**

380 Only few scientific studies report the presence of microplastics that contaminate food. Our
381 study, by assessing the presence of microplastics on the surface of meat products, extends the
382 state of knowledge on the contamination of human food by microplastics. These microplastics
383 are highly adherent to the meat surface and are likely to be eaten by consumers. Although
384 polystyrene trays are made of polystyrene suitable for food contact, the presence of MP-XPS
385 on the food surface is potentially problematic due to the toxicity of the microparticles, the
386 potential desorption of styrene, and the formation of degradation products during cooking.
387 Nevertheless, given current knowledge on the issue, it should be noted that the comparison of
388 the amounts of MP-XPS observed in the trays with the data available in the scientific
389 literature does not enable us to definitively conclude on the existence or not of potential risks
390 for humans. However, in view of the tray uses, no doubt should remain concerning the
391 contamination impacts for human health.

392 As a consequence, this preliminary investigation should be expanded and further developed
393 along several lines of investigation. First, it would be relevant to identify the sources of
394 contamination in association with possible distinct packaging processes. This step is
395 necessary to propose counter measures that could reduce exposition to MP-XPS. Second, the
396 risks associated with the regular ingestion of MP-XPS should be clarified. In addition, greater
397 knowledge of the molecules that can be desorbed by MP-XPS is necessary, especially if they
398 behave as endocrine disruptors.

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404 **References**

405 Alomar, C., & Deudero, S. (2017). Evidence of microplastic ingestion in the shark *Galeus*
406 *melastomus* Rafinesque, 1810 in the continental shelf off the western Mediterranean Sea.
407 *Environmental Pollution*, 223, 223–229.

408 <https://doi.org/https://doi.org/10.1016/j.envpol.2017.01.015>

409 Azevedo-Santos, V. M., Gonçalves, G. R. L., Manoel, P. S., Andrade, M. C., Lima, F. P., &
410 Pelicice, F. M. (2019). Plastic ingestion by fish: A global assessment. *Environmental*
411 *Pollution*, 255, 112994. <https://doi.org/https://doi.org/10.1016/j.envpol.2019.112994>

412 Barboza, L. G. A., Dick Vethaak, A., Lavorante, B. R. B. O., Lundebye, A.-K., &
413 Guilhermino, L. (2018). Marine microplastic debris: An emerging issue for food
414 security, food safety and human health. *Marine Pollution Bulletin*, 133, 336–348.

415 <https://doi.org/https://doi.org/10.1016/j.marpolbul.2018.05.047>

416 Barboza, L. G., & Gimenez, B. C. (2015). Microplastics in the marine environment: Current
417 trends and future perspectives. *Marine Pollution Bulletin*, 97(1–2), 5–12.

418 <https://doi.org/10.1016/j.marpolbul.2015.06.008>

419 Boucher, J., & Friot, D. (2017). Primary microplastics in the oceans: A global evaluation of
420 sources. In *Primary microplastics in the oceans: A global evaluation of sources*.

421 <https://doi.org/10.2305/iucn.ch.2017.01.en>

422 Bour, A., Haarr, A., Keiter, S., & Hylland, K. (2018). Environmentally relevant microplastic
423 exposure affects sediment-dwelling bivalves. *Environmental Pollution*, 236, 652–660.

424 <https://doi.org/https://doi.org/10.1016/j.envpol.2018.02.006>

425 Buchanan, J. B. (1971). Pollution by synthetic fibres. *Marine Pollution Bulletin*, 2, 23.

426 Cook, P. M., & Olson, G. F. (1979). Ingested mineral fibers: elimination in human urine.
427 *Science*, 204(4389), 195 LP – 198. <https://doi.org/10.1126/science.219478>

428 Davis, J. M. G. (1981). The biological effects of mineral fibres. *The Annals of Occupational*
429 *Hygiene*, 24(2), 227–234. <https://doi.org/10.1093/annhyg/24.2.227>

430 Dehaut, A., Hermabessiere, L., & Duflos, G. (2019). Current frontiers and recommendations
431 for the study of microplastics in seafood. *TrAC Trends in Analytical Chemistry*, 116,
432 346–359. <https://doi.org/https://doi.org/10.1016/j.trac.2018.11.011>

433 Deng, Y., Zhang, Y., Lemos, B., & Ren, H. (2017). Tissue accumulation of microplastics in
434 mice and biomarker responses suggest widespread health risks of exposure. *Scientific*
435 *Reports*, 7(1), 46687. <https://doi.org/10.1038/srep46687>

436 Dris, R., Gasperi, J., Saad, M., Mirande, C., & Tassin, B. (2016). Synthetic fibers in
437 atmospheric fallout: A source of microplastics in the environment? *Marine Pollution*
438 *Bulletin*, 104(1–2), 290–293. <https://doi.org/10.1016/j.marpolbul.2016.01.006>

439 European Chemicals Agency. (2019). *Styrene - Brief Profile*. [https://echa.europa.eu/brief-](https://echa.europa.eu/brief-profile/-/briefprofile/100.002.592)
440 [profile/-/briefprofile/100.002.592](https://echa.europa.eu/brief-profile/-/briefprofile/100.002.592)

441 Gasperi, J., Wright, S. L., Dris, R., Collard, F., Mandin, C., Guerrouache, M., Langlois, V.,
442 Kelly, F. J., & Tassin, B. (2018). Microplastics in air: Are we breathing it in? *Current*
443 *Opinion in Environmental Science & Health*, 1, 1–5.
444 <https://doi.org/https://doi.org/10.1016/j.coesh.2017.10.002>

445 Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever
446 made. *Science Advances*, 3(7), e1700782. <https://doi.org/10.1126/sciadv.1700782>

447 Gibbs, B. F., & Mulligan, C. N. (1997). Styrene Toxicity: An Ecotoxicological Assessment.
448 *Ecotoxicology and Environmental Safety*, 38(3), 181–194.
449 <https://doi.org/https://doi.org/10.1006/eesa.1997.1526>

450 Greim, H. A. (2004). Research needs to improve risk assessment of fiber toxicity. *Mutation*
451 *Research/Fundamental and Molecular Mechanisms of Mutagenesis*, 553(1), 11–22.
452 <https://doi.org/https://doi.org/10.1016/j.mrfmmm.2004.06.013>

453 Gündoğdu, S. (2018). Contamination of table salts from Turkey with microplastics. *Food*
454 *Additives & Contaminants: Part A*, 35(5), 1006–1014.
455 <https://doi.org/10.1080/19440049.2018.1447694>

456 Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2012). Microplastics in the
457 Marine Environment: A Review of the Methods Used for Identification and
458 Quantification. *Environmental Science & Technology*, 46(6), 3060–3075.
459 <https://doi.org/10.1021/es2031505>

460 Iñiguez, M. E., Conesa, J. A., & Fullana, A. (2017). Microplastics in Spanish Table Salt.
461 *Scientific Reports*, 7(1), 8620. <https://doi.org/10.1038/s41598-017-09128-x>

462 Karami, A., Golieskardi, A., Keong Choo, C., Larat, V., Galloway, T. S., & Salamatinia, B.
463 (2017). The presence of microplastics in commercial salts from different countries.
464 *Scientific Reports*, 7, 46173. <https://doi.org/10.1038/srep46173>

465 Kedzierski, M., Le Tilly, V., Cesar, G., Sire, O., & Bruzaud, S. (2017). Efficient microplastics
466 extraction from sand. A cost effective methodology based on sodium iodide recycling.
467 *Marine Pollution Bulletin*, 115(1–2), 120–129.
468 <https://doi.org/10.1016/j.marpolbul.2016.12.002>

469 Kedzierski, M., Le Tilly, V., César, G., Sire, O., & Bruzaud, S. (2017). Efficient microplastics
470 extraction from sand. A cost effective methodology based on sodium iodide recycling.
471 *Marine Pollution Bulletin*, 115(1–2). <https://doi.org/10.1016/j.marpolbul.2016.12.002>

472 Kosuth, M., Mason, S. A., & Wattenberg, E. V. (2018). Anthropogenic contamination of tap
473 water, beer, and sea salt. *PLOS ONE*, 13(4), e0194970.
474 <https://doi.org/10.1371/journal.pone.0194970>

475 Lange, J., & Wyser, Y. (2003). Recent innovations in barrier technologies for plastic
476 packaging—a review. *Packaging Technology and Science*, 16(4), 149–158.
477 <https://doi.org/10.1002/pts.621>

478 Liebezeit, G., & Liebezeit, E. (2013). Non-pollen particulates in honey and sugar. *Food*
479 *Additives & Contaminants: Part A*, 30(12), 2136–2140.
480 <https://doi.org/10.1080/19440049.2013.843025>

481 Liebezeit, G., & Liebezeit, E. (2014). Synthetic particles as contaminants in German beers.
482 *Food Additives & Contaminants: Part A*, 31(9), 1574–1578.
483 <https://doi.org/10.1080/19440049.2014.945099>

484 Lönnstedt, O. M., & Eklöv, P. (2016). Environmentally relevant concentrations of
485 microplastic particles influence larval fish ecology. *Science*, 352(6290), 1213 LP – 1216.
486 <https://doi.org/10.1126/science.aad8828>

487 Lu, L., Wan, Z., Luo, T., Fu, Z., & Jin, Y. (2018). Polystyrene microplastics induce gut
488 microbiota dysbiosis and hepatic lipid metabolism disorder in mice. *Science of The Total*
489 *Environment*, 631–632, 449–458.
490 <https://doi.org/https://doi.org/10.1016/j.scitotenv.2018.03.051>

491 Mason, S. A., Welch, V. G., & Neratko, J. (2018). Synthetic Polymer Contamination in
492 Bottled Water. *Frontiers in Chemistry*, 6, 407. <https://doi.org/10.3389/fchem.2018.00407>

493 Mathlouthi, M. (2013). *Food Packaging and Preservation*. Springer US.
494 <https://books.google.fr/books?id=2gHTBwAAQBAJ>

495 McNeill, I. C., Zulfiqar, M., & Kousar, T. (1990). A detailed investigation of the products of
496 the thermal degradation of polystyrene. *Polymer Degradation and Stability*, 28(2), 131–
497 151. [https://doi.org/https://doi.org/10.1016/0141-3910\(90\)90002-O](https://doi.org/https://doi.org/10.1016/0141-3910(90)90002-O)

498 Oßmann, B. E., Sarau, G., Holtmannspötter, H., Pischetsrieder, M., Christiansen, S. H., &
499 Dicke, W. (2018). Small-sized microplastics and pigmented particles in bottled mineral

500 water. *Water Research*, 141, 307–316.
501 <https://doi.org/https://doi.org/10.1016/j.watres.2018.05.027>

502 Piringer, O. G., & Baner, A. L. (2008). *Plastic Packaging Materials for Food: Barrier*
503 *Function, Mass Transport, Quality Assurance, and Legislation*. Wiley.
504 <https://books.google.fr/books?id=7Uz9XW-VihgC>

505 PlasticsEurope. (2012). *Plastic Packaging: Born to protect*.

506 PlasticsEurope. (2018). *Plastics - the Facts 2018*.

507 Prata, J. C. (2018). Airborne microplastics: Consequences to human health? *Environmental*
508 *Pollution*, 234, 115–126. <https://doi.org/10.1016/j.envpol.2017.11.043>

509 Rainieri, S., & Barranco, A. (2019). Microplastics, a food safety issue? *Trends in Food*
510 *Science & Technology*, 84, 55–57.
511 <https://doi.org/https://doi.org/10.1016/j.tifs.2018.12.009>

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

516 Rummel, C. D., Loder, M. G., Fricke, N. F., Lang, T., Griebeler, E. M., Janke, M., & Gerdts,
517 G. (2016). Plastic ingestion by pelagic and demersal fish from the North Sea and Baltic
518 Sea. *Marine Pollution Bulletin*, 102(1), 134–141.
519 <https://doi.org/10.1016/j.marpolbul.2015.11.043>

520 Sanchez, W., Bender, C., & Porcher, J.-M. (2014). Wild gudgeons (*Gobio gobio*) from French
521 rivers are contaminated by microplastics: Preliminary study and first evidence.
522 *Environmental Research*, 128, 98–100. <https://doi.org/10.1016/j.envres.2013.11.004>

523 Schirinzi, G. F., Pérez-Pomeda, I., Sanchís, J., Rossini, C., Farré, M., & Barceló, D. (2017).
524 Cytotoxic effects of commonly used nanomaterials and microplastics on cerebral and

525 epithelial human cells. *Environmental Research*, 159, 579–587.
526 <https://doi.org/https://doi.org/10.1016/j.envres.2017.08.043>

527 Schymanski, D., Goldbeck, C., Humpf, H.-U., & Fürst, P. (2018). Analysis of microplastics in
528 water by micro-Raman spectroscopy: Release of plastic particles from different
529 packaging into mineral water. *Water Research*, 129, 154–162.
530 <https://doi.org/https://doi.org/10.1016/j.watres.2017.11.011>

531 Stock, V., Böhmert, L., Lisicki, E., Block, R., Cara-Carmona, J., Pack, L. K., Selb, R.,
532 Lichtenstein, D., Voss, L., Henderson, C. J., Zabinsky, E., Sieg, H., Braeuning, A., &
533 Lampen, A. (2019). Uptake and effects of orally ingested polystyrene microplastic
534 particles in vitro and in vivo. *Archives of Toxicology*, 93(7), 1817–1833.
535 <https://doi.org/10.1007/s00204-019-02478-7>

536 Tavoularis, G., & Sauvage, É. (2018). Les nouvelles générations transforment la
537 consommation de viande. *CREDOC*, 4.

538 Van Cauwenberghe, L., Claessens, M., Vandegheuchte, M. B., & Janssen, C. R. (2015).
539 Microplastics are taken up by mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*)
540 living in natural habitats. *Environmental Pollution*, 199, 10–17.
541 <https://doi.org/10.1016/j.envpol.2015.01.008>

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

545 Vroom, R., Halsband, C., Besseling, E., & Koelmans, A. A. (2016). *Effects of microplastics*
546 *on zooplankton : Microplastic ingestion: the role of taste.*

547 Walum, E. (1998). Acute oral toxicity. *Environmental Health Perspectives*, 106(suppl 2),
548 497–503. <https://doi.org/10.1289/ehp.98106497>

549 Wang, W., Gao, H., Jin, S., Li, R., & Na, G. (2019). The ecotoxicological effects of

550 microplastics on aquatic food web, from primary producer to human: A review.
551 *Ecotoxicology and Environmental Safety*, 173, 110–117.
552 <https://doi.org/https://doi.org/10.1016/j.ecoenv.2019.01.113>

553 Warheit, D. B., Hart, G. A., Hesterberg, T. W., Collins, J. J., Dyer, W. M., Swaen, G. M. H.,
554 Castranova, V., Soiefer, A. I., & Kennedy, G. L. (2001). Potential Pulmonary Effects of
555 Man-Made Organic Fiber (MMOF) Dusts. *Critical Reviews in Toxicology*, 31(6), 697–
556 736. <https://doi.org/10.1080/20014091111965>

557 Winkler, A., Santo, N., Ortenzi, M. A., Bolzoni, E., Bacchetta, R., & Tremolada, P. (2019).
558 Does mechanical stress cause microplastic release from plastic water bottles? *Water*
559 *Research*, 166, 115082. <https://doi.org/https://doi.org/10.1016/j.watres.2019.115082>

560 Yang, D., Shi, H., Li, L., Li, J., Jabeen, K., & Kolandhasamy, P. (2015). Microplastic
561 Pollution in Table Salts from China. *Environmental Science & Technology*, 49(22),
562 13622–13627. <https://doi.org/10.1021/acs.est.5b03163>

563 Zhao, S., Zhu, L., & Li, D. (2016). Microscopic anthropogenic litter in terrestrial birds from
564 Shanghai, China: Not only plastics but also natural fibers. *Science of the Total*
565 *Environment*, 550, 1110–1115. <https://doi.org/10.1016/j.scitotenv.2016.01.112>

566 Zimmermann, L., Dierkes, G., Ternes, T. A., Völker, C., & Wagner, M. (2019).
567 Benchmarking the in Vitro Toxicity and Chemical Composition of Plastic Consumer
568 Products. *Environmental Science & Technology*, 53(19), 11467–11477.
569 <https://doi.org/10.1021/acs.est.9b02293>
570