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1 Toxicity assessment of microplastics from Loire River in the clam *Corbicula*
2 *fluminea* exposed to environmentally relevant exposure conditions

3
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10

11 **Abstract**

12 The diffuse pollution caused by microplastic (MPs) represents a current and global concern. While the
13 fragmentation of plastic debris into smaller particles occurs in rivers, few MPs research on freshwater
14 species is available compared to the marine environment. The Loire River, the longest French river, is
15 subjected to moderate to high anthropic pressure while it represents major societal and economic
16 issues. However, effects of environmental MPs (EMPs) on aquatic organisms have not been studied yet,
17 and no policies can take place to monitor the plastic pollution. In this study, freshwater bivalves,
18 *Corbicula fluminea*, were exposed for 21 days to environmentally relevant concentrations of a mixture
19 of < 200 µm MPs generated from plastic litter collected directly along the Loire river banks. This mixture
20 was composed of 40% polyethylene (PE), 40% polypropylene (PP), 10% polyethylene terephthalate
21 (PET) and 10% polyvinylchloride (PVC) (mass percentage). Ecotoxicological effects were assessed from
22 the individual to the molecular levels on several endpoints: condition index, filtration efficiency, enzyme
23 activities, lipid peroxidation, energy reserves and gene expression. The ingestion of EMPs caused
24 damages at the biochemical level. Indeed, we reported an increase in catalase activity in gills and
25 digestive mass, a decrease in TBARs in gills, a decrease in acetylcholinesterase activity in the digestive
26 mass, a decrease of glycogen and lipid contents in the whole organisms and a significant induction of
27 the expression of *gst*, *cat*, *mp*, *acp* genes. The current results suggest therefore that long-term exposure
28 to realistic doses of EMPs causes toxicity towards freshwater benthic biota. Biomarkers at the
29 biochemical and molecular levels are complementary tools allowing to prevent an early plastic
30 contamination for aquatic organisms.

31 **Keywords:** Microplastics; Bivalves; Biomarkers; Gene expression; Energy reserves

32

33 **Introduction**

34 Since the last decade, plastic pollution has become a major environmental concern due to its
35 increasing production, estimated at 390.7 million metric tons in 2021 (Statista Research Department,
36 2023). Among these plastics, many are single-used and immediately discarded, in most cases

37 improperly. Then, they reach the terrestrial environment and ultimately end up in aquatic environments
38 (Oliveira et al., 2019; Ferreira et al., 2019). At the global level, it is estimated that between 70 and 80%
39 of plastics in oceans come from land-based sources (Duis and Coors, 2016). While rivers are the main
40 entryway for plastic litter into the marine compartment, these streams are still poorly studied (Li et al.
41 2018). The Loire River, the longest French river with a length of 1 006 km, presents some major societal
42 and economic issues for the country (Managing Rivers Wisely, World Wildlife Fund (WWF)). In some
43 cities, such as Angers, the Loire water is used as a resource of drinking water and fishing is a widespread
44 activity in this area. Since anthropic activities of towns may affect the Loire River it seems helpful to
45 characterise the plastic contamination around metropolises in order to implement adequate policies.
46 Moreover, once plastics reach the aquatic compartment, they can be fragmented into smaller size
47 polymers by photodegradation (UV radiation), weathering and physical abrasion (e.g., wind and wave
48 action), and biotic factors (e.g., microorganisms' action). Nowadays, small plastic particles (≤ 5 mm) so-
49 called microplastics (MPs) represent a current and a global environmental concern. Indeed, numerous
50 scientific studies reported that MPs are increasingly being observed in all components of most aquatic
51 ecosystems worldwide, as suspended particulates in the water column, in sediments and in wastewater
52 (Burns and Boxall 2018; Frias et al. 2020; Habib et al., 2020; Bayo et al. 2020; Dusaucy et al. 2021; Uddin
53 et al. 2021; Vidal et al. 2023). In addition to being found in abiotic compartments, it has been
54 demonstrated that MPs are ingested by a wide variety of aquatic biota, at different levels of food webs,
55 from microalgae to mammals (Ferreira et al. 2019, Wang et al. 2019; Bringer et al. 2020; Qu et al. 2020;
56 Latchere et al. 2021; Pires et al. 2023).

57 Among invertebrates, bivalves are considered as good bioindicators to assess plastic pollution levels
58 (Boening, 1999). They are often used in laboratory to study the ecotoxicological effects of xenobiotics.
59 Indeed, bivalves are filter-feeders and have the ability to bioaccumulate pollutants from their
60 surrounding environment. Since years, Asian clam (*Corbicula fluminea*) is considered as a relevant
61 bioindicator to assess the anthropic pollution in aquatic compartments, and recently for plastic pollution
62 (Su et al. 2018). The presence of MPs is global in the aquatic compartments, especially in the marine
63 environment, on which many studies have focused. Despite a major research effort conducted to study
64 the effects of MPs on freshwater streams in general and on freshwater bivalves in particular, literature
65 on MPs contamination of freshwater bivalves remains still sparse compared to the publication on
66 marine bivalves. Indeed, Sendra et al. (2021) reported 56 studies dealing with marine bivalves *versus* 13
67 studies dealing with freshwater bivalves, of which 6 of them used *Corbicula fluminea* as their model
68 species, the rest (*Unio pictorum*, *Perna canaliculus*, *Dreissena burgensis*, *Elliptio complanate* and
69 *Dreissena polymorpha*) counted for either 1 or 2 works. In bivalves, it has been demonstrated that MPs
70 can be retained in the gut over timescales beyond those expected for other ingested matter (Browne
71 et al. 2008), and that MPs can be ingested by bivalves under environmental conditions (Ward et al.

72 2019). Once MPs are ingested, they may be retained in the digestive tract, egested through defecation,
73 or translocated from the gut to the circulatory system (Browne et al. 2008). In addition, diverse
74 chemicals named 'additives' are integrated to plastic polymers during their manufacture to improve
75 their properties and extend their lifespan (Hahladakis et al. 2018). Studies focused more and more on
76 the effects of these additives on aquatic organisms (Ateia et al. 2020; Hermabessiere et al. 2017). The
77 few studies addressing this topic reported worrying effects on bivalves, but toxicological studies using
78 EMPs (involving additives and aging of plastics) are still scarce. There is therefore a lack of knowledge
79 about their effects on aquatic organisms. This was demonstrated in the review of Latchere et al. (2021)
80 who reported that scientists often used unrealistically high concentrations of MPs in laboratory
81 exposures. This may lead to misleading conclusions about the impacts of plastic particles on aquatic
82 ecosystems. Also, it appears necessary to test MPs coming from macro-sized plastics collected in the
83 environment to be as closed as possible to the environmental conditions. In this context, this study aims
84 to evaluate the toxicity of environmental MPs (EMPs) on the freshwater bivalve *Corbicula fluminea*
85 under environmental realistic conditions such as a long-time exposure (21 days), environmental
86 exposure concentrations and use of plastics coming directly from the field. Ecotoxicological effects of
87 EMPs on clams were assessed at different biological organisation levels, from molecular to individual
88 levels on both relevant tissues (gill and digestive gland) or on the whole organism. We wondered in this
89 study if the ecotoxicological effects are dependent on the exposure concentrations and/or the tested
90 tissues. Also, we wondered if there is a link between effects at the different biological organisation
91 levels. Results will help establishing first insights about the links between molecular effects and
92 individual ones in an Adverse Outcome Pathway concept.

93 **Materials and methods**

94 **Collection, preparation and characterization of EMPs**

95 Plastic wastes were collected by hand along the Loire River banks in March 2022, near Angers,
96 a medium city in West of France (47°42'38.34"N, -0°60'30.10"W). There is an accumulation of plastic
97 specially at this site due to the river currents. In the laboratory, plastic wastes have been dried at room
98 temperature for few days and cleaned with demineralised water to take off duff, sand and other dirt to
99 avoid this matter to be transformed into microparticles. Plastic litter (bottles, rope, food packaging,
100 diverse coloured pieces) was pre-cut and analyzed by Attenuated Total Reflection Fourier Transform
101 Infrared spectrometer (ATR-FTIR Vertex70v, Bruker) equipped with a diamond crystal ATR accessory and
102 a denudated triglycine sulfate (DTGS) detector. A few grams of each plastic: polyethylene (PE),
103 polypropylene (PP), polyethylene terephthalate (PET) and polyvinylchloride (PVC) were put into a
104 sample container with a magnetic bar, and both sides of the container were closed with two stainless
105 steel caps. The container was inserted into a cryogenic mechanical miller (SPEX, 6770 Freezer-mill) filled

106 with liquid nitrogen. During the milling process, an oscillating magnetic field was generated, which drove
107 the magnetic bar forward and backward inside the container and make the impacts with the caps.
108 Milling consisted in 2 min of pre-cooling followed by 2 cycles of milling. Each cycle lasted 4 min with a
109 1-min pause between the 2 cycles. The obtained powders of plastic particles were then sieved during
110 60 min on a stain sieve of 200 μm , to select particles size reflecting that of food particles ingested in the
111 natural environment by Asian clams (Parra et al. 2021; Li et al. 2019). Fractions of PE, PP, PET and PVC
112 under 200 μm were then stored in a glass bottle at ambient temperature in the dark until preparation
113 of the stock solution.

114 Based on the composition of polymers found in French Atlantic coastal sediments by Phuong et
115 al. (2018), the following mixture of plastic particles has been realized (mass percentage): 40% PE, 40%
116 PP, 10% PET and 10% PVC. This is also consistent with a recent study (Wu et al. 2022), which reported
117 that the mostly common polymers analyzed in Asian clams sampled in the field (China) were PE (40%),
118 PET (23%) and PP (18%). A first stock suspension at 1 $\text{mg}\cdot\text{mL}^{-1}$ was prepared in milliQ water at pH 7 and
119 sonicated for 3 minutes. Then, solutions at 0.1 $\text{mg}\cdot\text{mL}^{-1}$ and 80 $\mu\text{g}\cdot\text{L}^{-1}$ were obtained by successive
120 dilutions of the stock suspension in milliQ water at pH 7. After sonication (3 minutes), a specific volume
121 of the stock suspension (1 $\text{mg}\cdot\text{mL}^{-1}$) or the working solutions (0.1 $\text{mg}\cdot\text{mL}^{-1}$ and 80 $\mu\text{g}\cdot\text{L}^{-1}$) was distributed
122 every 3 days after a complete water renewal in the aquariums to obtain the following final
123 concentrations: 0.008, 10, 100 and 500 $\mu\text{g}\cdot\text{L}^{-1}$. These concentrations were based on Revel et al. (2019):
124 a low concentration of 0.008 $\mu\text{g}\cdot\text{L}^{-1}$ for the coast regions, a medium concentration of 10 $\mu\text{g}\cdot\text{L}^{-1}$ for coastal
125 region/gyres, a higher concentration of 100 $\mu\text{g}\cdot\text{L}^{-1}$ for gyres, and the highest concentration of 500 $\mu\text{g}\cdot\text{L}^{-1}$
126 for freshwater streams. In addition, a Dynamic Light Scattering (DLS) analysis was performed on the
127 stock solution and the working solutions (Zetasizer Nano-ZS90) to detect the presence of particles
128 smaller than 1 μm . It allowed to measure the hydrodynamic diameter and the aggregation state of
129 particles in solution. In parallel, a spectroscopic MPs quantification by μ -FTIR imaging (micro-Fourier
130 Transform InfraRed; Thermo Nicolet iZ10) was performed on the stock solution to characterize the
131 number, the size and the composition of particles in the used plastic mixture. A pixel with a resolution
132 of 25 x 25 μm was selected. μ -FTIR imaging was performed for the entire sample *i.e.*, the entire filtration
133 zone of the sample. Acquisition parameters were described in a recent study (Treilles et al. 2021). The
134 μ -FTIR maps were treated with siMPle software (v.1.1. β) and the library
135 MP_Library_extended_grouped_1_5.txt. The default matching weight of 0.5 for the first derivative of
136 the spectra and 0.5 for the second derivative of the spectra was used, and the AAU pipeline was chosen
137 for data processing. The minimal particle size provided by siMPle is 25 μm . The major dimension of
138 plastic fragments provided in the output of the siMPle software were considered.

139 **C. *fluminea* sampling and laboratory experimental set up**

140 *C. fluminea* were collected in May 2022 in the Maine Lake located in Angers (France)
141 (47°27'47.9"N 0°35'21.0"W). Organisms were transported to the laboratory in containers filled with
142 sediment and water from the sampling site. They were then immediately transferred into aquaria (54
143 L) containing lake water (0.45 L per individual), in a temperature-controlled room at 13°C, with a
144 photoperiod maintained at 12:12. A total of 265 clams (mean length = 25.4 ± 0.83 cm and mean weight
145 = 8.09 ± 0.66 g) were divided among five glass aquaria equipped with bubblers and covered with a glass
146 plate to avoid plastic contamination by air. During the 10-days acclimation period, the water was
147 replaced by tap water to progressively decrease the volume of the lake water: 100/0, 75/25, 50/50,
148 25/75 until 0/100. At the end of the acclimation period, 15 individuals were sampled to provide
149 biometrical data (mass and length). Organisms were then exposed to MP for 21 days via water spiked
150 by the four concentrations mentioned above (0.008 µg.L⁻¹, 10 µg.L⁻¹, 100 µg.L⁻¹ and 500 µg.L⁻¹). A control
151 tank without added plastic was also put in place (Control). During both acclimation and exposure steps,
152 the organisms were fed every 3 days with a suspension of unicellular algae *Scenedesmus subspicatus*,
153 provided by Greensea. Before each feeding, algal concentration was counted on a Malassez cell to
154 obtain 70,000 cells per individual per liter of water. Individuals were fed one hour before each water
155 renewal. At the end of the exposure period, individuals were sampled in each aquarium: n=3 for
156 filtration efficiency; n=10 for condition index and also used for the verification of MP ingestion; n=10 for
157 biomarker analysis; and n=10 for energy reserves quantification. Whole soft bodies were sampled for
158 the quantification of energy reserves and immediately frozen at -80°C, while gills and digestive glands
159 were dissected and immediately frozen at -20°C for further biomarkers analysis.

160 **Analysis of MPs in the organisms**

161 A verification of the presence of MP in clams was performed to ensure their ingestion. Since the
162 microscopy method does not allow to know the plastic composition and the µ-FTIR method does not
163 give the colour nor the shape of MPs, it seems relevant to use both techniques for a complete
164 characterization of MPs ingested by the organisms. Therefore, a first analysis was performed by
165 microscopy to determine the size, the colour and the shape of MPs ingested by individuals for each
166 condition (n=3 organisms), including the control. Thus, each clam was entirely digested in ethanol
167 precleaned beakers containing 10% potassium hydroxide (KOH). Beakers, covered with aluminium, were
168 placed on a heating plate at 60°C with stirring for 24 h to eliminate organic matter. The digesta was then
169 filtered under a Büchner system using a 1.6 µm filter (GF/A 1820-037). The complementary analysis by
170 µ-FTIR was conducted to determine the composition and the size of MPs ingested by organisms (pool
171 of 3 individuals). The digestion protocol was the same as the microscopy analysis and the filtration was
172 performed on Anodisc (Whatman) membrane filters (0.2 µm, 25 mm).

173 **Ecophysiology measurement**

174 ***Condition index***

175 The total weight and the soft tissue weight were determined for each clam and all conditions at
176 the beginning (T0) and at the end of the 21 days exposure period (T21). The condition index (CI) was
177 determined as follows (AFNOR, 1985):

178
$$CI = \frac{\text{wet weight of soft tissues}}{\text{total wet weight}} \times 100$$

179 ***Filtration efficiency***

180 The filtration efficiency indicates the ability of bivalves to intake microalgae and other food
181 items from the water column and is a key function for evaluation of the individual fitness. The filtration
182 efficiency was defined as the volume of water cleared of suspended particles per unit time, and was
183 estimated for *C. fluminea* previously exposed for 21 days to clean water (Control) or MPs (0.008, 10,
184 100 and 500 $\mu\text{g.L}^{-1}$). Clams were individually placed in a 1 L beaker filled with 400 mL of water from the
185 aquaria one hour before measurements started. Mixing was promoted by fine bubble aeration to
186 minimize the re-suspension of faeces and individuals were placed on metal grids to let the faeces fall
187 into the bottom of the beaker. An initial concentration of 300 000 cells.mL⁻¹ of *S. subspicatus* was added
188 in each beaker. The filtration efficiency was then measured by an indirect method assessing the
189 concentration of the algal solution every 15 min for 45 min according to classical algal cell counting
190 technique (Utermöhl, 1958). Briefly, algal cells were fixed by adding a drop of Lugol Iodine solution
191 (Iodine I₂, 5%; Potassium Iodide KI 10%) in 10 mL of sample. Cells were counted under a light microscope
192 using a Malassez counting chamber. The filtration efficiency was expressed in cells.mL⁻¹ of *S. subspicatus*.

193 **Biochemical analyses**

194 ***Energy reserves***

195 Protein, lipid and glycogen contents were quantified in the total homogenate using a thawed
196 subsample for each energy storage fraction. For each condition, 10 clams were individually ground in
197 liquid nitrogen, then the powder was homogenized with a motorized grinder in a Tris magnesium buffer
198 (0.1 M Tris-HCl pH 8.5, 0.4 M MgSO₄, 15% polyvinylpyrrolidone (antioxidant) and 0.2% (w/v) Triton X-
199 100 (cell membrane permeabilizer)). Lipid and protein fractions were directly stored at -80°C while the
200 glycogen fraction was boiled during 4 minutes before storage at -80°C. Total lipids were determined by
201 the sulfo-phospho-vanillin reaction and measured spectrophotometrically at 540nm (TECAN® Sunrise)
202 (Frings et al. 1972). Glycogen content was estimated using enzymatic digestion by amyloglucosidase and
203 measured spectrophotometrically (TECAN® Sunrise) at 420 nm (Carr and Neff 1984). For protein
204 quantification, the homogenates were centrifuged at 3000 g, 10 min at 4°C. Protein concentration was

205 quantified using Bradford method (Bradford 1976). Lipid, glycogen and protein concentrations were
206 expressed as mg.g⁻¹ of wet weight (ww).

207 ***Oxidative stress, neurotoxicity and lipid peroxidation***

208 Enzyme activities: catalase (CAT), glutathione S-transferases (GST) and acetylcholinesterase
209 (AChE), and lipid peroxidation were quantified in the gills and digestive gland tissues of clams. Firstly,
210 the tissues were thawed on ice and homogenized in Tris buffer at pH 7.2 (Tris 0.8 M (Trizma-base Sigma®
211 T6791); NaCl 2 M (Sigma® S9888); DTT 100 mM (Sigma® 8349-12-3); CIP (Protease Inhibitor Cocktail,
212 Sigma® P8340)). The homogenates were then centrifuged at 9000 g at 4°C for 30 min. The amount of
213 total proteins was quantified using the Bradford method (Bradford, 1976). CAT activity was measured
214 spectrophotometrically at 240 nm ($\epsilon = 0.04 \text{ mM}^{-1}.\text{cm}^{-1}$; UviLine® 9600), following the dismutation of
215 hydrogen peroxide (H₂O₂) (Claiborne, 1985). GST activity was measured spectrophotometrically at
216 340nm ($\epsilon = 9.6 \text{ mM}^{-1}.\text{cm}^{-1}$ TECAN® Sunrise) at 25°C by observing the conjugation of the 1-chloro-2,4-
217 dinitrobenzene (CDNB) and the L-glutathione reduced (GSH) (Habig et al. 1974). AChE activity was
218 determined spectrophotometrically at 412nm (TECAN® Sunrise) at 25°C, monitoring the conversion of
219 acetylthiocholine into thiocholine in the presence of 5,5-dithiobis (2-nitrobenzoic acid) called DTNB
220 (Ellman et al. 1961; Galgani and Bocquene 1991). To determine lipid damage, the formation of
221 thiobarbituric acid reactive substances (TBARs), including malondialdehyde (MDA) was measured
222 according to the method of Sunderman et al. (1985) with a colorimetric assay (540 nm) that quantifies
223 lipid degradation products as an index of lipid peroxidation. All specific activities were expressed either
224 in $\mu\text{moles}.\text{min}^{-1}.\text{mg protein}^{-1}$ (CAT) or in $\text{nmoles}.\text{min}^{-1}.\text{mg protein}^{-1}$ (GST, AChE and MDA).

225 **Analysis of gene expression by quantitative PCR**

226 *Corbicula fluminea* were pooled by three for each condition and were analyzed by quantitative
227 RT-PCR for each condition. Total RNA was extracted using TRIzol reagent® (Life Technologies) from the
228 gills and the digestive mass, according to the manufacturer's recommendations and precipitated with
229 propan-2-ol. RNA concentration ($\mu\text{g}.\mu\text{L}^{-1}$) was quantified using a NanoDrop 2000 spectrophotometer
230 (ThermoScientific®). First-strand cDNA was synthesized from 5 μg of total RNA using the Invitrogen™
231 SuperScript™ III kit (ThermoScientific®; T100™ Thermal Cycler, BIORAD®) according to the
232 manufacturer's recommendations. The expression levels of seven genes involved in oxidative stress,
233 detoxication, respiratory chain, immunity, neurotoxicity and apoptosis were analyzed using a set of
234 forward and reverse primers by quantitative RT-PCR. The ribosomal protein 7 gene (rpl7), previously
235 tested and compared with other genes in our laboratory, was used as housekeeping gene. Previously,
236 the quality of each pair of primers was checked: cDNA tests were amplified by PCR (T100™ Thermal
237 Cycler, BIORAD®. 30 cycles: 30 s à 95°C, 30 s 181 à 60°C, 30 s à 72°C), then the amplification products
238 were separated on 1.5% agarose electrophoresis gel. After staining with ethidium bromide, the
239 presence and size of each amplicon were verified. Quantitative PCR (qPCR) amplifications were carried

240 out in triplicate in 96-well microplates (CFX Connect™ Real-Time System, BIORAD®) using SYBR™ Master
241 Mix PCR Power SYBR™ Green (Invitrogen) containing the SYBR Green dye, DNA Taq Polymerase and
242 dNTPs. For each reaction, 1 μL of each primer (50 pmoles.μL⁻¹), 6.25 μL of SYBR Green mix, 3.75 μL of
243 water treated with DEPC (DNase-free water) and 0.5μL of cDNA were added in each well. The qPCR
244 reactions consisted of a first step of 10 min at 95°C (enzyme activation) followed by 45 cycles (95°C for
245 30 s, 60°C for 30 s and 72°C for 30 s) and 5 min at 72°C. For each primer pair, amplification specificity
246 was verified revealing the presence of a single product on a melting curve and a no-template control
247 assay was realised. Expression levels were estimated by evaluating the fluorescence signal emitted by
248 SYBR-Green®. This fluorescent marker binds to double-stranded DNA (dsDNA) and the fluorescence
249 emitted is proportional to the dsDNA present in the reaction mix. Calculations are based on cycle
250 threshold (Ct) values. The relative gene expression ratio of each target gene was calculated following
251 the delta-delta method normalized with reference genes (Livak and Schmittgen, 2001), which is defined
252 as:

$$\text{ratio} = \frac{2 - \Delta\Delta\text{Ct}(\text{exposed})}{2 - \Delta\Delta\text{Ct}(\text{control})}$$

254 **Quality control**

255 To prevent a cross-contamination, all equipment used in the laboratory was previously rinsed
256 with milliQ water and ethanol, dried at room temperature in a hood and kept under aluminum foil to
257 avoid contact with the ambient air. During the experiments, cotton laboratory coats and nitrile gloves
258 were worn. For both methods (microscopy and μ-FTIR analysis), during the digestion, beakers were
259 covered with aluminum foil. After filtration, the filters were stored in closed glass Petri dishes until
260 microscopy and μ-FTIR analysis. Due to the remaining risk of cross-contamination of the samples, blanks
261 were performed in KOH 10%, following the same protocol. For each series of digestion, a blank sample
262 (without organisms nor water) was performed in parallel. Blanks were also performed for the filtration
263 step, and for the microscopy and μ-FTIR analysis. Finally, the use of plastic material was avoided at best
264 throughout this experimental study.

265 **Statistical analysis**

266 Statistical analysis was carried out using XLSTAT 2019 (version 21.4.63762) and R software
267 (version 4.0.3). Normality of data distribution and homogeneity of variance were tested using the
268 Shapiro-Wilk and Bartlett tests, respectively. If data did not meet the conditions for parametric tests,
269 Kruskal-Wallis test was therefore used to test for differences between the treatments. If the overall test
270 was significant, a Dunn procedure was performed to determine which means were significantly
271 different. If the data followed the conditions for application of parametric tests, an ANOVA was used to
272 test for differences between the treatments. If the overall test was significant, a Tukey post hoc test was
273 performed to determine which means were significantly different. In all cases, p values ≤ 0.05 were

274 considered statistically significant. Also, Principal Component Analysis (PCA) was realised using all data
275 of biomarkers obtained for gills and digestive gland (quantitative variables) to evaluate the influence of
276 MPs concentrations, on control and exposed *C. fluminea*, along the period of 21 days exposure. PCA
277 was performed with the ade4 package (R software) and all missing values were removed from database
278 to ensure the PCA realisation. Finally, overlaps were performed with qualitative variables (not used to
279 build the PCA).

280 Results

281 Analysis of MPs in the exposure solutions and clams

282 The DLS analysis revealed the presence of particles under 25 µm in the stock solution and in the
283 two working solutions (Figure S1). This is due to the non-selection of the grinded microplastics (<200
284 µm). In this study, all MPs ≤ 200 µm were considered but only particles ranging between 25 and 200 µm
285 (analyzed by both visual and spectrometry methods) were described below. The visual observation of
286 the stock solution (3 replicates of 100 µL) revealed the presence of 92% of fragments and 8% fibers.
287 Among the fragments, 73% ranged from 25 to 100 µm and 27% from 100 to 200 µm, and 55% were red,
288 30% were black, 10% were grey and 5% were transparent. 42% of the fibers measured between 275
289 and 1 000 µm, and 58% between 1 000 and 1 250 µm, with 50% being grey, 39% black and 11% blue.
290 The µ-FTIR analysis confirmed the presence of PE, PP, PVC and PET in the stock solution. Indeed, among
291 the 851 MPs identified in 100 µL of the stock solution, PE particles counted for 71.1%, PP for 19.9%, PET
292 for 7.9% and PVC for 0.1%. The 1.1%-part remaining was constituted of PA (0.4%) and PS (0.7%). Also, 3
293 particles of acrylic and 3 of polyamide (PA) were analyzed, probably due to a cross-contamination during
294 a step of the experiment. The sizes of the MPs analyzed by µ-FTIR were identified as follows: between
295 31.8 and 1790.9 µm for PE; between 31.8 and 632.9 µm for PP; between 38.2 and 220.3 µm for PVC;
296 and between 25 and 183.1 µm for PET. The mean size ± SD associated to this µ-FTIR analysis were: 144.8
297 ± 121.7 µm for PE; 105.4 ± 72.1 µm for PP; 80.2 ± 34.0 µm for PET and 103.6 ± 56.0 µm for PVC.
298 Therefore, based on the mean values, the exposure was realised using plastic particles with a size less
299 than 200 µm.

300 The verification performed to ensure that this specific size range of EMPs was ingested by the
301 exposed clams confirmed this hypothesis (Figure S2). Indeed, both the visual count under the binocular
302 microscope and the µ-FTIR analysis revealed an increasing presence of MPs in the clams, related to the
303 increase in MPs concentration used in the different treatments (Figure S2). Analysis performed on the
304 control group also revealed the presence of MPs despite subtracting the MPs found on the control filter,
305 although fewer than those found in the contaminated conditions. MPs found on the blank filters
306 (between 2 and 7 particles per filter) were mainly fibers (78%) measuring from 150 to 1 500 µm, either
307 transparent, blue, grey or pink. The two fragments were blue and pink, and measured 25 and 250 µm

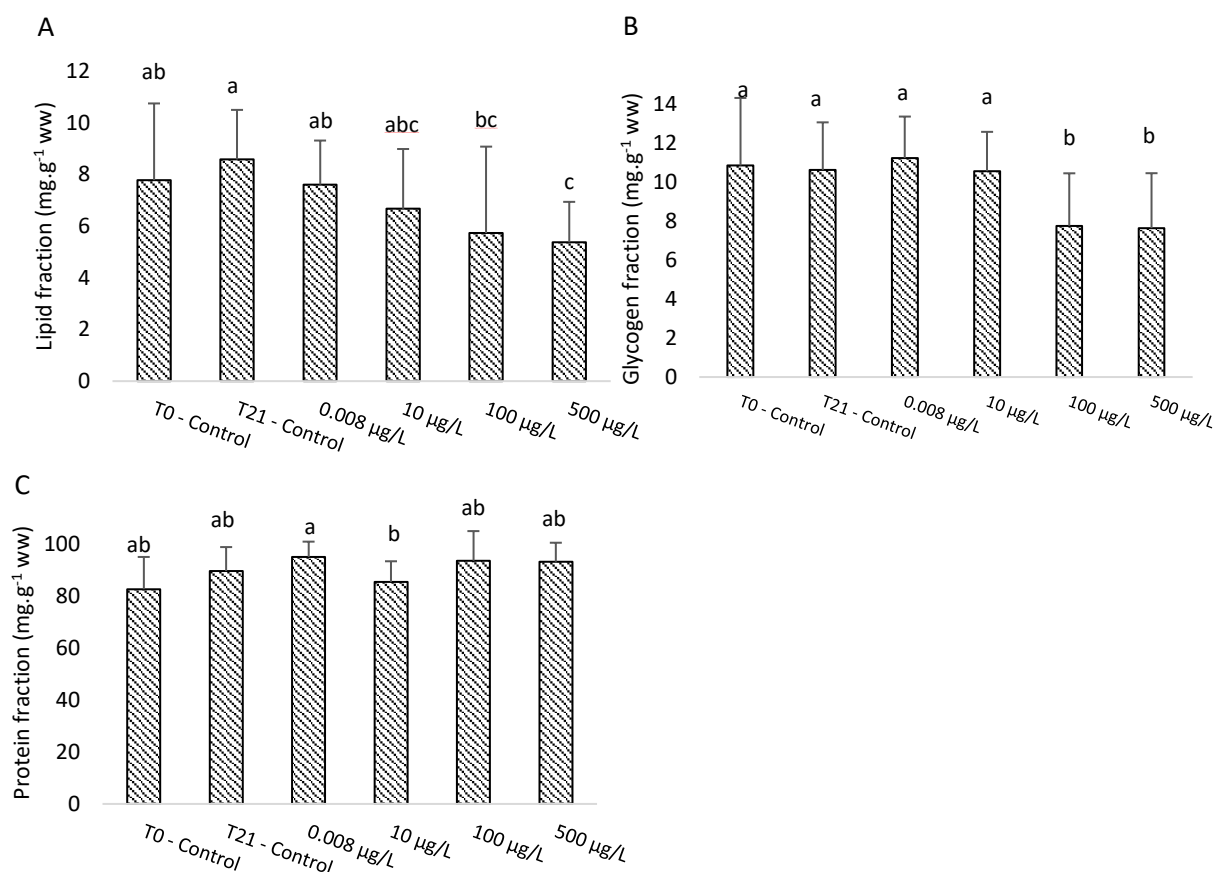
308 (major dimension), respectively. A light cross-contamination by acrylic and PA was also observed,
309 probably coming from textiles worn during the experiment period, despite the precautions. Fragments
310 and fibers with different colours (transparent, whitish/opaque, blue, grey and red) have been found in
311 all organisms (Figure S2). PE was the dominant plastic polymer in the clams followed by PP, reflecting
312 the MP profile of the stock solution. Neither PVC nor PET particles were found in the clams. The size of
313 the analyzed MPs in clams measured by μ -FTIR confirmed that determined by microscopy (Figure S2)
314 and was of the same range as the sizes of MPs determined in the stock solution.

315 **Condition index and filtration efficiency**

316 No mortality was recorded during the exposition period of 21 days. No significant CI difference
317 (Figure S3) was observed between the clams sampled at the beginning of the experiment (T0, $19.30 \pm$
318 1.52) and between the clams sampled from the control treatment and from the two lowest
319 concentrations: $0.008 \mu\text{g.L}^{-1}$ (19.08 ± 2.28) and $10 \mu\text{g.L}^{-1}$ (17.73 ± 2.68) after 21 days of exposure (T21,
320 17.90 ± 2.42). However, a significant decrease of the CI was observed between the clams sampled at T0
321 and those sampled at T21 at the highest concentrations: $100 \mu\text{g L}^{-1}$ (16.20 ± 2.21) and $500 \mu\text{g L}^{-1}$ (16.82
322 ± 3.02). In addition, after 21 days exposure, the MPs contamination had no significant impact on the
323 filtration efficiency of *C. fluminea*, independently of MPs concentration and measurement time (Figure
324 S4).

325 **Energy reserves**

326 The exposure to EMPs during 21 days caused a decrease in glycogen and lipid contents in *C.*
327 *fluminea*. Results indicated a significant post-exposure decrease in lipid content for individuals exposed
328 to $500 \mu\text{g MP.L}^{-1}$, and in glycogen content for individuals exposed to 100 and $500 \mu\text{g MP.L}^{-1}$ (Figure 1, A
329 and B). No significant change in protein contents was observed between T0 organisms and control
330 organisms at 21 days, and the treated organisms.



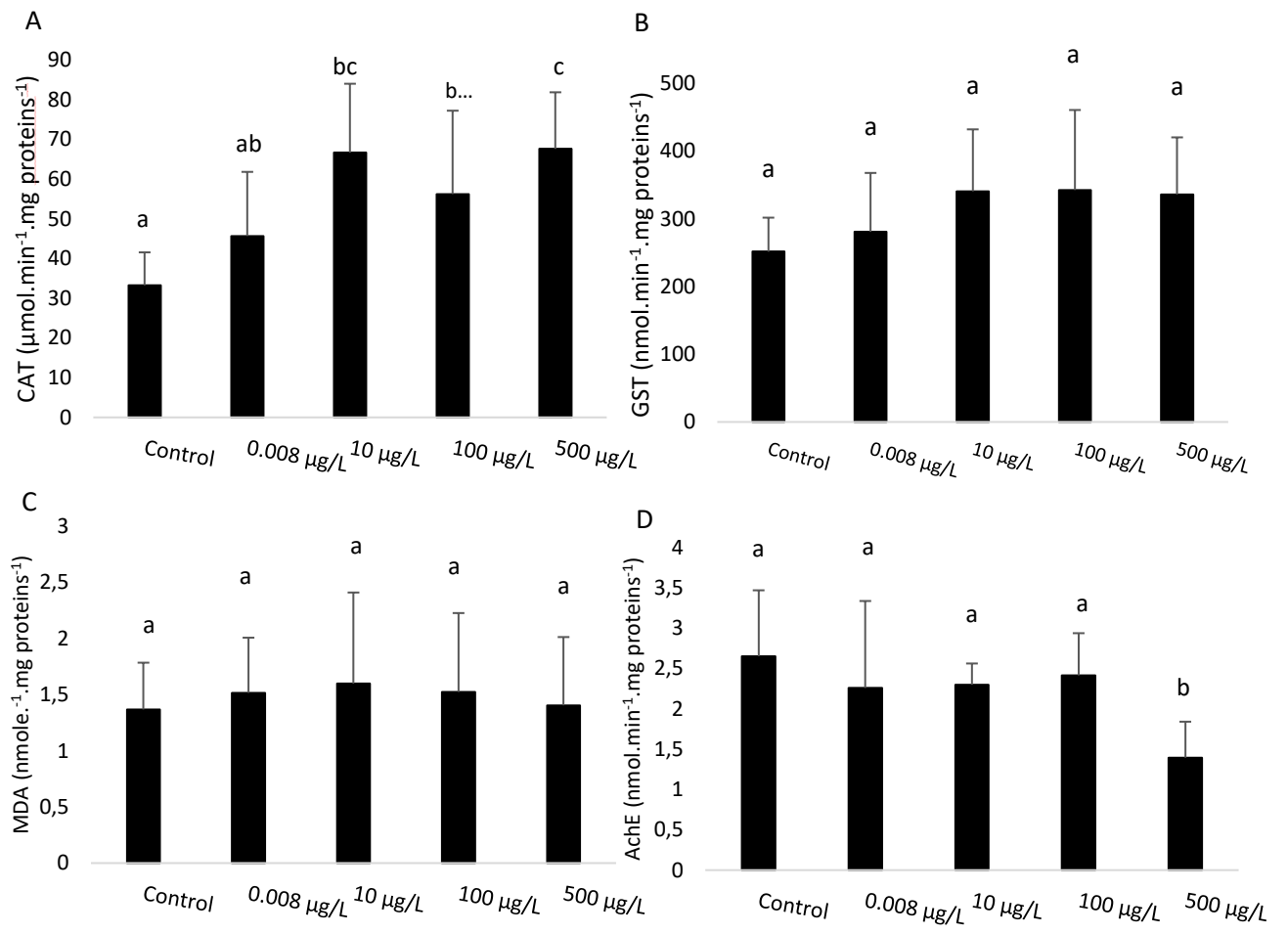
331
 332 **Figure 1.** Energy reserve contents (mg/g wet weight ± standard deviation) in *C. fluminea* (control and exposed to
 333 EMP). A) Lipid fraction; B) Glycogen fraction; C) Protein fraction. Different lower-case letters indicate significant
 334 differences between all groups ($p < 0.05$).

335 Biomarkers: oxidative stress, lipid peroxidation and neurotoxicity

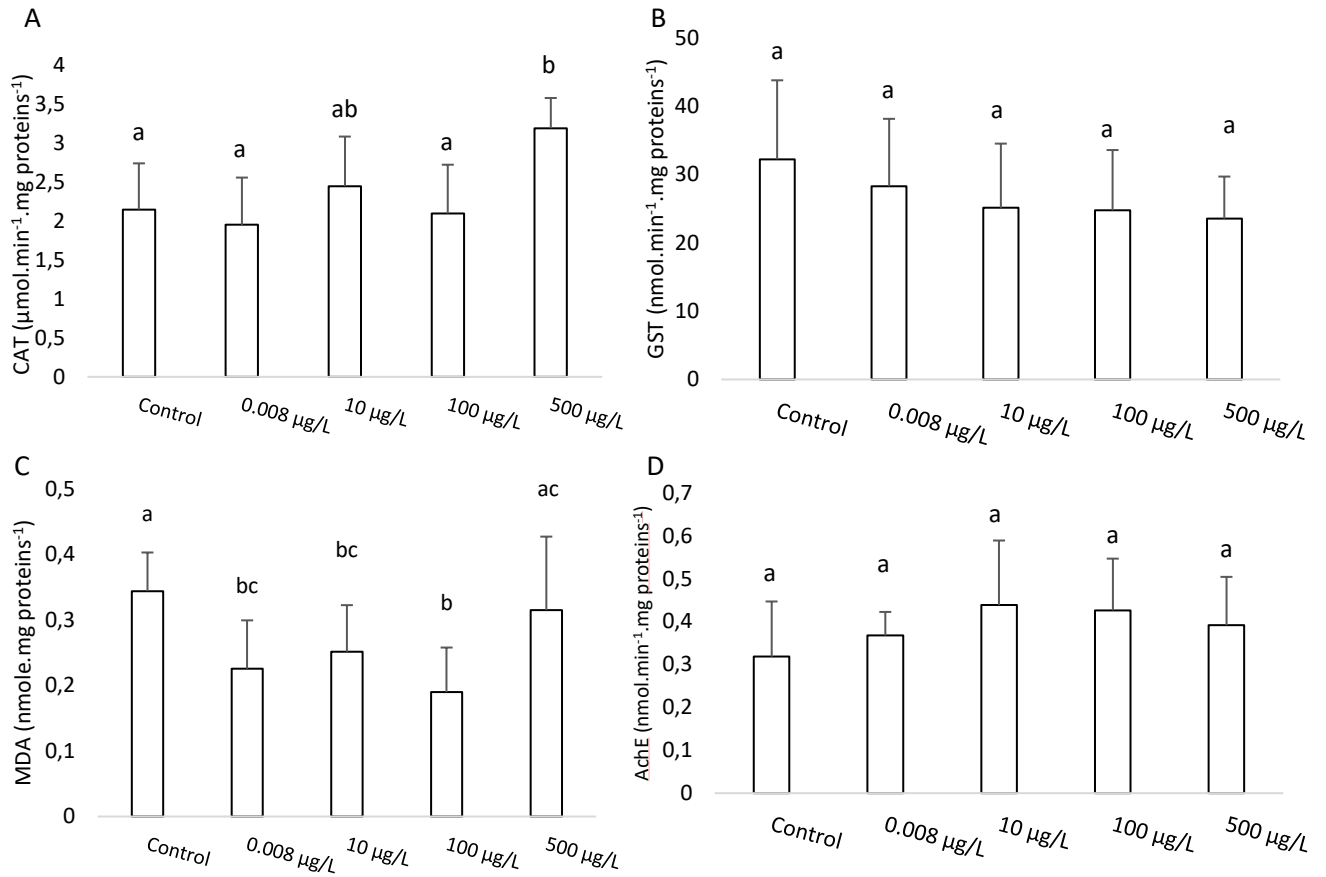
336 For all biomarkers measured after 21 days of exposure, the levels of enzymatic activities were
 337 higher in the digestive mass than in gills (Figure 2). The biomarker responses to the MP contamination
 338 differed according to the tissues (Figure 3). CAT activity was significantly increased in digestive mass of
 339 organisms exposed to 10, 100 and 500 µg.L⁻¹ compared to the control organisms, while in gills, CAT
 340 activity was increased significantly only at the highest concentration (500 µg.L⁻¹) compared to control.
 341 In both digestive mass and gills, no significant difference in GST activity was observed for the treated
 342 organisms compared to the control ones. Concerning MDA levels, significant differences were measured
 343 only in the gills. Indeed, MDA levels were significantly lower in the clams exposed to 0.008, 10 and 100
 344 µg.L⁻¹, whereas no significant difference was measured at the highest concentration (500 µg.L⁻¹)
 345 compared to the control. On the contrary, significant differences in AChE activity were depicted only in
 346 the digestive mass. Clams exposed to 500 µg MP.L⁻¹ showed a significant decrease in AChE activity
 347 relative to the controls.

348 The first PCA (Figure S5) was based on all biochemical data: CAT, GST and AChE activities, and
 349 MDA content measured on both organs (gills and digestive mass) of *C. fluminea* exposed to the 5

350 treatments (Control, 0.008, 10, 100 and 500 $\mu\text{g.L}^{-1}$). This PCA showed that the studied variables related
351 to biochemical biomarkers were organ-dependant. Therefore, we chose to represent two distinct
352 components for a better visual interpretation knowing that PCA representations on a single axis are
353 rarely presented. Two PCA were then performed separately, one using data obtained on gills and the
354 other on digestive mass. These two PCA showed the relationships between biochemical biomarkers and
355 exposure of clams to EMPs (Figure 3). Regarding the PCA performed on digestive mass (Figure 3A), the
356 two principal components represented 73.76% of total variance, the first axe PC1 and the second axe
357 PC2 representing 46.30% and 27.46% respectively. Exposure conditions (control, 0.008, 10, 100 and 500
358 $\mu\text{g.L}^{-1}$) were separated and distributed proportionally according to both axes. PC1 is mostly explained
359 by CAT and GST variables (34.79 and 40.31 % respectively). The exposure condition at the highest
360 concentration (500 $\mu\text{g.L}^{-1}$) was also separated from the others. Relative to PC1, a positive correlation
361 was established between CAT and GST; likewise, a positive correlation was depicted between MDA and
362 AChE related to PC2. In the gills (Figure 3B), the two principal components represent 59.73% of total
363 variance, with PC1 representing 32.26% and PC2 27.46%. PC1 was mostly explained by CAT and AChE
364 activities (35.35 and 38.25 % respectively), while PC2 was mostly explained by GST activity and MDA
365 (50.45 and 37.48 % respectively). No evident separation between the conditions was established except
366 for the highest concentration (500 $\mu\text{g.L}^{-1}$) separated from the control. PCA showed that exposure to the
367 highest concentration of EMPs (500 $\mu\text{g.L}^{-1}$) was positively correlated to an increase in the activity of
368 oxidative stress enzymes CAT and GST in the digestive mass, and only to CAT in gills. In digestive mass,
369 exposure to high concentration of EMPs was negatively correlated to the activity of AChE (inhibition).

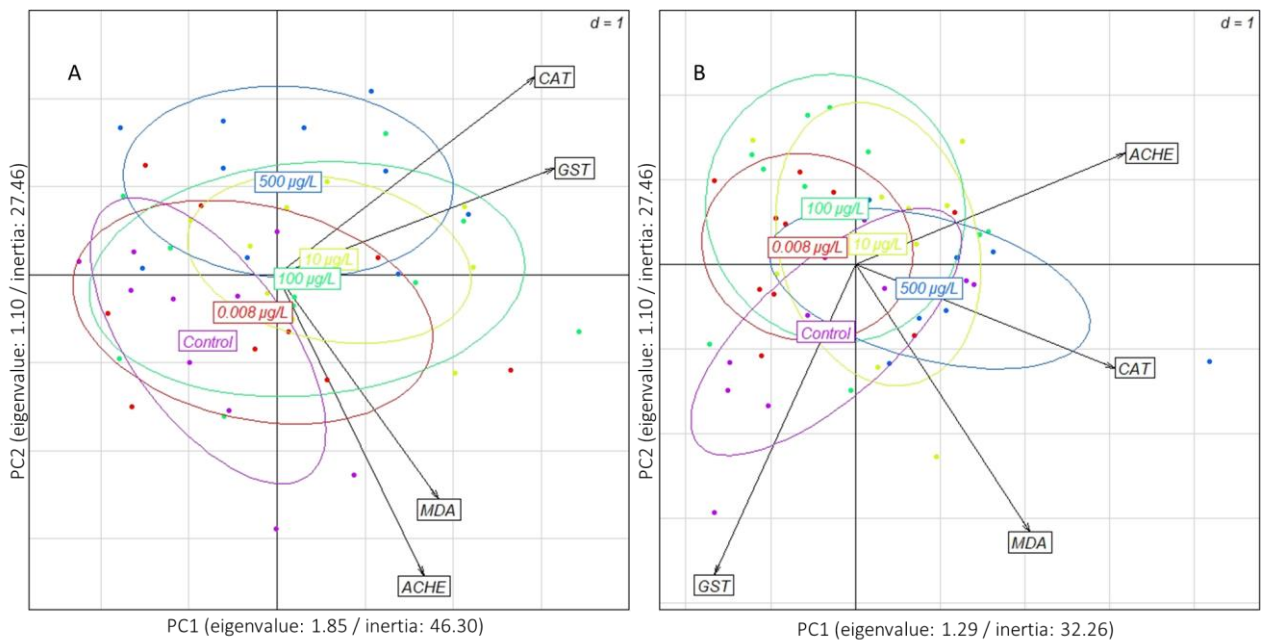


370



371

372 **Figure 2.** Activities of CAT (A), GST (B), AChE (C) and MDA (D) levels in digestive mass (black) and gills (white) of *C.*
 373 *fluminea* after 21 days of EMPs exposure at 0 (Control), 0.008, 10, 100 and 500 µg MP.L⁻¹ (mean + sd, N = 10).
 374 Different lower-case letters indicate significant differences between control and MPs exposed treatments (p <
 375 0.05).



376

377

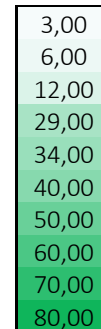
Variables	A: Digestive mass				B: Gills			
	CAT	GST	MDA	AChE	CAT	GST	MDA	AChE
Contribution to PC1 (%)	34.79	40.31	13.63	11.27	35.35	10.40	16.00	38.25
Contribution to PC2 (%)	20.41	5.90	26.38	47.32	5.63	50.45	37.48	6.43

378 **Figure 3.** Principal component analysis (PCA) based on biochemical variables analyzed in *C. fluminea* exposed to
379 EMPs at 0 (Control), 0.008, 10, 100 and 500 $\mu\text{g MP.L}^{-1}$ in different organs, (A) digestive mass and (B) gills. The table
380 showed the contributions of the variables to the axis (in %).

381 Gene expression

382 As shown in Table 1, at 21 days post-exposure, analyzed genes of *C. fluminea* were only up-
383 regulated in both gills and digestive mass. These genes are involved in detoxication (*gst* and *mt*),
384 oxidative stress (*cat*), respiratory chain (*cox1*), immunity (*acp*) and neurotoxicity (*ache*). In gills, the
385 induction of *gst*, *mt*, *cat* and *acp* was detected only for the lowest EMPs concentration (0.008 $\mu\text{g.L}^{-1}$),
386 except *acp* gene induced also at 10 $\mu\text{g.L}^{-1}$. In the digestive mass, *cat*, *acp* and *ache* genes were induced
387 only for the second lowest EMPs concentration (10 $\mu\text{g.L}^{-1}$), except *cat* gene induced from 0.008 $\mu\text{g.L}^{-1}$.
388 Therefore, significant gene inductions appeared only for the two lowest concentrations (0.008 and 10
389 $\mu\text{g.L}^{-1}$), while there was no significant up or downregulation of genes for *C. fluminea* exposed at 100 and
390 500 $\mu\text{g.L}^{-1}$. Overall, there was no evidence of dose-dependent effect for EMPs treatments.

Gills					
		0.008 $\mu\text{g.L}^{-1}$	10 $\mu\text{g.L}^{-1}$	100 $\mu\text{g.L}^{-1}$	500 $\mu\text{g.L}^{-1}$
Detoxication	<i>gst</i>	6.08			
	<i>mt</i>	11.8			
Oxidative stress	<i>cat</i>	5.7			
Respiratory chain	<i>cox1</i>				
Immunity	<i>acp</i>	62.86	38.35		
Neurotoxicity	<i>ache</i>				
Digestive mass					
		0.008 $\mu\text{g.L}^{-1}$	10 $\mu\text{g.L}^{-1}$	100 $\mu\text{g.L}^{-1}$	500 $\mu\text{g.L}^{-1}$
Detoxication	<i>gst</i>				
	<i>mt</i>				
Oxidative stress	<i>cat</i>	16.22	76.2		
Respiratory chain	<i>cox1</i>				
Immunity	<i>acp</i>		32.99		
Neurotoxicity	<i>ache</i>		7.21		



Fold change
scale bar

391 **Table 1.** Differential gene expression observed in the gills and the visceral mass of *Corbicula fluminea* after 21 days of exposure
392 to the different treatments. Results are presented as fold-change factors between gene expression in controls and gene
393 expression in exposed organisms (> 1: induction, in green). Only significant differences are represented (p values ≤ 0.05) and
394 only factors <0.5 and >1.5 are considered as significant.

396 Discussion

397 The Loire River, the longest French river, represents major societal and economic issues for the
398 country. Indeed, this stream crosses a multitude of cities of varying sizes, some of them using the Loire
399 as a resource for drinking water. In the region, scientist federations have even been created to inform
400 local inhabitants and industrialists of the potential risks of the Loire pollution, and the potential sanitary

401 and environmental impacts. Indeed, since the anthropic activities of cities, such as Angers, may affect
402 the Loire River, it would be helpful to characterise the plastic contamination in order to implement
403 adequate policies. In this context, the study of the effects of MPs derived from plastic litter coming
404 directly from the Loire banks, on the ubiquitous freshwater organism (*C. fluminea*), appeared crucial to
405 collect data that will help to implement public policies for environmental risk management and
406 contamination monitoring in the Loire. This is the first study reporting the toxicity of EMPs from the
407 Loire on *C. fluminea* using a multi-biomarker approach at different levels of biological organization. This
408 approach allowed 1) to characterize the effects of these plastics by quantifying the biological responses,
409 aiming to link molecular to individual effects and 2) to propose tools for monitoring plastic
410 contamination of the Loire by measuring early biological responses in order to predict potential
411 consequences at higher levels.

412 **Environmental realism of the experiment**

413 One of the current questions that arise from MPs exposures conducted in laboratory is whether the
414 conditions of experiments are realistic from an environmental point of view. In nature, MPs is an
415 umbrella term that covers particles with various physical and chemical properties, and organisms are
416 not exposed to uniform sizes and shapes of MPs (Rochman et al. 2017). Laboratory studies often
417 conduct bivalve exposure at higher concentrations of MPs than the ones found in the environment.
418 Also, they generally use MPs with a consistent shape and with a calibrated size, particles composed of
419 only one polymer type (no plastic mixture) (Li et al. 2019; Latchere et al. 2021), and without additives,
420 which may be responsible for the observed effects on aquatic organisms (Ateia et al. 2020;
421 Hermabessiere et al. 2017). More precisely, one type of spherical plastic microbeads, calibrated at 1-
422 100 µm in size, was often used for MPs exposure to clams (Li et al. 2019; Ribeiro et al. 2017; Oliveira et
423 al. 2018; Moreschi et al. 2020; Fu et al. 2022). However, it has been demonstrated that uptake of MPs
424 by *C. fluminea* varies based upon physicochemical properties, such as polymer composition and particle
425 size (Li et al. 2019). Accounting for this, in this study, *C. fluminea* were exposed to a mixture of different
426 types of EMPs (PP, PE, PVC and PET) generated from separated plastic waste collected directly from the
427 field. Since these plastics have been more or less aged, they still contain additives integrated during
428 their manufacture or adsorbed during their release in the environment. Lower exposure concentrations
429 of EMPs than those used in most bivalve laboratory experiments have been selected for this study. Each
430 grinded powder presented similar size range of MPs, but not perfectly uniform due to differences in
431 how the polymers micronized. These size ranges fall within the range of particles ingested by *C. fluminea*
432 sampled in the field (Wu et al. 2022; Su et al. 2016). Also, laboratory experiments have been conducted
433 to confirm preferential MPs sizes ingested by Asian clams. For example, Li et al. (2019) reported that
434 uptake of PET fibers in exposed clams was greater in the size range 100-250 µm than in other size classes
435 (5-100 µm, 100-250 µm, 250-500 µm, 500-1000 µm and 1000-5000 µm). In our study, size particle

436 ranges of MPs determined by μ -FTIR in organisms were similar for all plastic types and are also similar
437 to the ones measured in the stock solution. The μ -FTIR analysis of the clams after the 21-days exposure
438 revealed the presence of PE and PP but neither PVC nor PET particles were found. This is probably due
439 to the small amount detected in the stock solution or these particles may aggregate in the solution and
440 being less bioavailable for organisms (Revel et al. 2019).

441 **Effects of EMPs at individual level**

442 After confirming the ingestion of MPs by exposed organisms, biological changes at individual level
443 were analyzed. Indeed, statistically significant alteration in the CI was observed between the clams at
444 the beginning of the exposure (T0) and the ones after 21-days of exposure, exposed to the two highest
445 MP concentrations. The organisms exposed to these two concentrations did not grow as much as the
446 control clams and the ones exposed to lower MP concentrations. Naidu et al. (2022) calculated the CI
447 of yellow clams sampled at three different locations of the northwest coast of India. The lesser CI could
448 be due to the occurrence of a high number of < 100 μ m size MPs, as smaller size MPs can result in
449 oxidative damage, altered feeding, depressed reproductive output in bivalves (Naidu et al. 2022). This
450 reduced general condition may therefore be explained by disorders at lower biological levels such an
451 allocation of energy reserves to face a stress rather than for growth, and might cause negative impacts
452 on a long-term period. Therefore, the effects of EMPs on the whole organisms or on particular tissues
453 (gills and digestive mass) of the bivalves were evaluated by assessing their ecotoxicological impacts using
454 a battery of biomarkers and by comparing the expression of some genes involved in the stress
455 responses.

456 **Effects of EMPs at molecular and biochemical levels: relationship between gene expression 457 and enzymatic activities**

458 The comparison of gene expression levels between exposed and control organisms is useful for
459 identifying the mechanisms involved in the toxicant-specific responses. These molecular markers can
460 also provide early-detection of a stress. The biochemical markers were chosen to evaluate the possible
461 effects related to damage caused by MPs, including the oxidative stress, lipid oxidative damage and
462 neurotoxicity (AChE activity). Relationship between both, molecular and biochemical markers, is useful
463 to know if gene expression induces the activity of the associated protein. The results first highlighted
464 that an exposure to EMPs led to changes in gene expression patterns and biochemical activities at
465 environmentally relevant concentrations, in the two tissues, with a differential response of biomarkers
466 between gills and digestive mass (Figure 1 and Figure 2). For all biomarkers, the specific activity levels
467 were higher in digestive mass compared to gills (Figure 1), and ecotoxicological effects were sometimes
468 significant in gills and not in digestive mass, or inversely, reflecting the distinct physiological and
469 metabolic pathways present in the two tissues.

470 Firstly, *cat* and *gst* gene expressions were quantified, both involved in the detoxication pathways:
471 *cat* gene encodes catalase, an enzyme that acts as a defence mechanism against reactive oxygen
472 species, allowing the disproportionation of hydrogen peroxide into water and dioxygen; *gst* gene
473 encodes GST enzymes protecting cells against toxicants by conjugating the glutathione as substrate to
474 xenobiotics. The induction of both *cat* and *gst* genes observed in the gills and in the digestive mass of *C.*
475 *fluminea* is a sign of response to an oxidative stress. However, the GST activities was not significantly
476 modified in both organs of clams in this study and the CAT activity was significantly higher only at the
477 highest concentrations. The non-significant change of GST activity might be due to a compensatory and
478 earlier response of the antioxidant enzyme, such as CAT or SOD (not analyzed in this study). However,
479 some studies have reported a response of the GST in bivalves exposed to MPs. For example, Parra et al.
480 (2021) found a significant increase of GST activity in gill, digestive gland and gonad of Asian clams
481 exposed to the combination MPs with Cd, but not with MPs alone (PS commercial particles). The
482 individuals were exposed to higher concentration (2 mg MP.L⁻¹) than those used for our study. Métais
483 et al. (2023) reported a significant decrease of GST activity, after 21-days exposure, in gills and digestive
484 glands of *S. plana* exposed to low concentration of MPs (0.008 µg.L⁻¹) compared to the control but not
485 for the highest concentrations (10 and 100 µg.L⁻¹). Therefore, GST activity is probably not induced at
486 low environmental concentrations, and expression of *gst* appeared only at low MP exposure
487 concentrations, while the *cat* gene takes over for high MP concentrations.

488 The PCA realised on the data obtained from the digestive mass analysis, showing that exposure to
489 EMPs was positively correlated to an increase in the activity of CAT and GST, and negatively correlated
490 to the activity of AChE, suggested the clam response to an oxidative stress, the establishment of a
491 detoxication process and a neurotoxic effect. This response was not observed in gills except for the CAT
492 activity. This has been previously reported by Revel et al. (2019), who showed an increase in CAT activity
493 in gills of mussel (*Mytilus spp.*) at the highest tested concentration of MPs (100 µg.L⁻¹), while there was
494 a significant increase in digestive gland at the lowest and medium concentrations of MPs (0.008 and 10
495 µg.L⁻¹). A similar result was reported by Métais et al. (2023): after 21 days of exposure, significant higher
496 levels of CAT activity were observed in the gills of *S. plana* exposed to 100 µg.L⁻¹ of MPs compared to
497 the control ones, but there was no significant change for CAT activity in the digestive glands. This
498 difference may be explained by the differential exposure level between tissues due to particle
499 aggregation in stock suspensions (Revel et al. 2019). Another reason of the difference of the antioxidant
500 enzyme response in gills and digestive glands would be the natural differences in physiology of the
501 different organs. Gill is the first organ exposed to environmental contaminants, while the digestive gland
502 is the major organ for the accumulation of metals and other contaminants (Wirnkor et al. 2019). In our
503 study, CAT activity significantly increased already at lower concentration in digestive mass (at 100 µg.L⁻¹)
504 compared to the gills (at 500 µg.L⁻¹). Parra et al. (2021) also reported a higher induction of CAT activity

505 in the digestive gland of *C. fluminea* compared to gills, after an exposure to the mixture MPs and Cd,
506 while there is no difference between these two organs when organisms were exposed only to MPs (PS
507 commercial particles). In the green-lipped mussel *Perna canaliculus*, Chandurvelan et al. (2013)
508 observed an increase of CAT activity in digestive gland but not in gills from organisms exposed to
509 cadmium ($200 \mu\text{g.L}^{-1}$) for 21 and 28 days. Borković-Mitić et al. (2013) also observed higher CAT activity
510 in the digestive gland compared to the gills, after exposure of the freshwater bivalve *Unio tumidus* to
511 metals. Metals and some other contaminants are considered as additives integrated or adsorbed to the
512 plastics found in the environment. More and more studies focus on the toxicity of additives on bivalves,
513 and some of them highlighted an oxidative stress (Wu et al. 2018; Capolupo et al. 2021). Since the
514 plastics used in this study were directly collected from the field to evaluate the toxicity of
515 environmentally relevant plastic particles, the oxidative stress could therefore be induced by additives
516 like metals. In this context, it would be interesting to characterize additives associated to the EMPs used
517 in this exposure experiment, while the number of scientific studies on the effects of additives on aquatic
518 organisms is still too scarce.

519 The induction of the *MT* gene expression in gills at the lowest concentration suggests a detoxication
520 mechanism involvement. This can be induced by the presence of metal or other organic compounds
521 probably integrated or adsorbed as additives on plastics. In both tissues, the overexpression of *acp* gene
522 indicated an immune system response at low EMP concentration exposures. A similar result in the
523 immune response has been reported for bivalves exposed to MPs (Mkuye et al. 2022). Métais et al.
524 (2023) reported a significant higher level of AcP activity from the seventh day of exposure in *S. plana*
525 after exposure to the lowest concentration of MPs ($0.008 \mu\text{g.L}^{-1}$) compared to $10 \mu\text{g.L}^{-1}$, corroborating
526 the overexpression of *acp* found in this study.

527 Moreover, AChE, that splits acetylcholine into choline and acetate, is essential in transmitting nerve
528 impulses among synapses (Modesto and Martinez, 2010). In gills, neither a change of AChE activity nor
529 of the *ache* gene expression was depicted. However, in digestive mass, a significant inhibition of the
530 AChE activity of clams exposed at $500 \mu\text{g.L}^{-1}$ and a significant increase of the *ache* gene expression at
531 $10 \mu\text{g.L}^{-1}$ have been reported. This may be related to the inflammation of the visceral mass since it has
532 been reported that inflammatory conditions can trigger the up-regulation of *ache* gene expression
533 (Oliveira et al., 2012). Consistent with some previous studies (Guilhermino et al. 2018; Parra et al. 2021;
534 Fu et al. 2022), AChE activity inhibition induced by MPs was observed in the current study. As explained
535 by Fu et al. (2022), a possible reason contributing to the inhibition of AChE is that the enzyme structure
536 was changed through adsorption (Tan et al. 2020). Thus, the MPs could act as AChE inhibitors, that may
537 affect bivalve behaviour, including the feeding ability and the ability to recognize and avoid predators
538 (Pan and Dutta, 1998). Fu et al. (2022) highlighted that disturbance to AChE activity may lead to
539 weakened ability to uptake food from water. These changes were confirmed by a decrease in the

540 filtration rate in the exposed *Corbicula fluminea*. However, in our study, the filtration efficiency was not
541 disturbed when comparing control and exposed clams. This is in accordance with the results of Revel et
542 al. (2019). The authors did not report a significant difference in the clearance rate between control and
543 exposed mussels. Moreover, the expression of the *cox* gene did not change, neither in gills nor in
544 digestive gland in the exposed clams. This suggests that the respiratory chain was not impacted by an
545 exposure to EMPs at the chosen concentrations.

546 In addition, MDA levels are an important biomarker of oxidative damage in aquatic organisms, which
547 contain high amounts of lipids with polyunsaturated fatty acid, a substrate for oxidation (Lushchak,
548 2011). In our study, no significant increase of MDA level was reported in the digestive mass, which is
549 consistent with the studies of Parra et al. (2021) and Guilhermino et al. (2018). Indeed, Parra et al. (2021)
550 did not report significant lipid peroxidation in the digestive gland of Asian clams exposed to 2 mg MP.L⁻¹
551 and Guilhermino et al. (2018) did not report significant lipid peroxidation in the digestive gland of Asian
552 clams exposed at both 0.2 and 0.7 mg MP.L⁻¹. These results indicate that lipid peroxidation does not
553 occur in this organ, even at high concentrations of MPs, because it might be compensated by the
554 antioxidant enzymatic activities. However, in our study, a significant decrease of MDA content was
555 reported in gills of exposed clams to 0.008, 10, 100 µg MP.L⁻¹ compared to the control group but not
556 for the highest concentration (500 µg MP.L⁻¹). Ribeiro et al. (2017) also reported a decrease of oxidative
557 damage to lipids in the gills in PS MPs exposed clams (*Scrobicularia plana*) compared to the control
558 group. The authors explained that this could be linked to antioxidant defences, which consequently limit
559 the attack of reactive oxygen species to membrane lipids (Ribeiro et al. 2017). Parra et al. (2021) showed
560 that exposure to Cd alone, MPs alone and the mixture of both induced significant increase lipid
561 peroxidation levels in gills. Oliveira et al. (2018) also observed a significant oxidative lipid damage in *C.*
562 *fluminea* exposed to a mixture of MPs (0.13 mg.L⁻¹ of fluorescent red polymer microspheres of unknown
563 composition between 1 and 5 µm diameter) with mercury, but not in clams exposed to MPs only. This
564 suggests that mercury is responsible for this damage.

565 Finally, results of the current study showed the interest of using different complementary
566 ecotoxicological tools. Indeed, biomarker activities and energy reserves level changes were significant
567 for the highest concentrations of the EMPs exposure, while changes of gene expressions occurred at
568 the lowest EMP concentrations. This means that the transcriptomic tools targeting expression of genes
569 involved in the main pathways of stress responses would help to predict microplastic effects in bivalves
570 at high biological levels (biochemical and individual) but also at high concentrations of MP exposure.

571 **Effects of EMPs at biochemical level: energy reserves**

572 Exposure to EMPs caused a decrease of the energy reserves in *C. fluminea*. To our knowledge,
573 this is the first study reporting data on energy reserves on *C. fluminea* exposed to MPs. Although no
574 difference was observed in protein levels, a decrease in lipid and glycogen contents was depicted in

575 clams exposed to the two higher concentrations of MPs (100 and 500 $\mu\text{g}\cdot\text{L}^{-1}$), after 21 days of exposure.
576 A protein breakdown is considered as a last resort, only occurring during prolonged starvation or periods
577 of extreme energy demands (Haider et al. 2018). A decrease in the protein content in MPs-exposed
578 bivalves indicate therefore severe energy deficiency, which is not the case in the current study. A change
579 in energy reserves in bivalves exposed to MPs was supported by other studies. For example, Bour et al.
580 (2018) concluded that larger PE particles (125–500 μm) had more severe effects on energy reserves of
581 *Ennucula tenuis* and *Abra nitida* than smaller ones (4–6 μm and 20–25 μm), possibly due to a longer
582 residence time in the gut that can cause physical damage. Also, Shang et al. (2021) reported a decrease
583 of energy reserves of mussels *Mytilus coruscus* exposed to 2 μm polystyrene microspheres. MPs have
584 been demonstrated to significantly suppress the energy assimilation efficiency of mussels (Gardon et al.
585 2018; Pedersen et al. 2020). In our study, both glycogen and lipid reserves undergo depletion at 21 days
586 of exposure at 100 and 500 $\mu\text{g}\cdot\text{L}^{-1}$ indicating an energy cost for *C. fluminea* at high MP concentrations.
587 However, it is possible that these different energy reserves have already decreased before 21 days of
588 exposure. In their study, Shang et al. (2021) demonstrated an early depletion of glycogen (day-7) but
589 also a rapid resynthesis after the end of MP exposure of the mussels reaching the baseline levels after
590 7 days of recovery. In contrast, depletion of lipids and proteins occurs later during the MP exposure and
591 takes longer to recover after the exposure end. Lipid metabolism is the primary pathway through which
592 energy is supplied in organisms, playing an important role in the response to environmental stress in
593 aquatic invertebrates (Lee et al. 2018). However, to deep knowledge of the biological mechanisms,
594 energy reserves might be analyzed in separate tissues.

595 **Conclusion**

596 This study aimed to evaluate the toxicity of environmental MPs on the freshwater bivalve
597 *Corbicula fluminea* under environmental realistic conditions. Therefore, for this laboratory experiment
598 exposure microplastics were made up from plastic litter collected *in situ* and the exposure
599 concentrations were considered as environmental. This paper is the first that reports ecotoxicological
600 effects on aquatic organisms exposed to EMPs from the Loire, the longest French river. We
601 demonstrated that the effects were dependant on the tested concentrations: biochemical responses
602 have been generally reported at the two highest exposure concentrations while molecular responses
603 were reported at the two lowest concentrations in both tissues. Also, we showed that effects were
604 dependant on the tissues. Finally, from these results, it appears that a transcriptomic approach targeting
605 genes involved in stress response helps evaluating the slight differences in toxicity of the different tested
606 plastic conditions on the Asian clam. Also, it suggests that this approach could be relevant in order to
607 predict effects at higher levels (biochemical and individual) of the biological organization in the concept

608 of Adverse Outcome Pathway (AOP). This is particularly important with regard to plastic pollution whose
609 effects on humans are difficult to investigate.

610 **Declarations**

611 **Ethical Approval**

612 Not applicable. This article does not contain any studies with human participants. The animal used in
613 this article is an aquatic invertebrate.

614 **Competing interests**

615 The authors declare no competing interests.

616 **Authors' contributions**

617 Alice Vidal, Oihana Latchere, Isabelle Métais and Amélie Châtel conceived and designed the
618 experimental set up. Alice Vidal performed the field campaign and the laboratory analyses, assisted by
619 Isabelle Métais and Amélie Châtel. Alice Vidal and Johnny Gasperi realised the spectroscopy analyses.
620 Antoine Le Guernic did all the statistics on the results. Alice Vidal collected all the data, interpreted it
621 and wrote the first draft of the manuscript. All authors commented on previous versions of the
622 manuscript. All authors read and approved the final manuscript.

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626 **Availability of data and materials**

627 The authors confirm that the data supporting the findings of this study are available within the article
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629 **References**

- 630
631 AFNOR (1985). Norme Française. Huitres creuses. Dénominations et Classification, NF V 45-056.
632 Publication de l'association française de normalisation (AFNOR).
633 Ateia M., Zheng T., Calace S., Tharayil N., Pilla S., Karanfil T. (2020) Sorption behavior of real microplastics
634 (MPs): insights for organic micropollutants adsorption on a large set of well characterized MPs. *Sci*
635 *Total Environ* 720:137634. <https://doi.org/10.1016/j.scitotenv.2020.137634>
636 Bayo, J., Olmos, S., López-Castellanos, J. (2020) Microplastics in an urban wastewater treatment plant:
637 The influence of physicochemical parameters and environmental factors. *Chemosphere* 238,
638 124593. DOI: 10.1016/j.chemosphere.2019.124593
639 Boening, D.W. (1999) An Evaluation of Bivalves as Biomonitors of Heavy Metals Pollution in Marine
640 Waters. *Environmental Monitoring and Assessment*, 55, 459-470.
641 <http://dx.doi.org/10.1023/A:1005995217901>
642 Borković-Mitić, S., Pavlović, S., Perendija, B., Despotović, S., Gavrić, J., Gačić, Z., Saičić, Z. 2013. Influence
643 of some metal concentrations on the activity of antioxidant enzymes and concentrations of vitamin
644 E and SH-groups in the digestive gland and gills of the freshwater bivalve *Unio tumidus* from the
645 Serbian part of Sava River, *Ecological Indicators*, 32, 212-221.
646 <https://doi.org/10.1016/j.ecolind.2013.03.024>.

647 Bour, A., Avio, C.G., Gorbi, S., Regoli, F., Hylland, K (2018) Presence of microplastics in benthic and
648 epibenthic organisms: influence of habitat, feeding mode and trophic level. *Environ. Pollut.* 243 :
649 1217–1225. <https://doi.org/10.1016/j.envpol.2018.09.115>.

650 Bradford M (1976) A rapid and sensitive method for the quantitation of microgram quantities of protein
651 utilizing the principle of protein-dye binding. *Anal Biochem* 72:248–254.
652 [https://doi.org/10.1016/0003-2697\(76\)90527-3](https://doi.org/10.1016/0003-2697(76)90527-3)

653 Bringer, A., Cachot, J., Prunier, G., Dubillot, E., Clerandeanu, C., Helene, T (2020) Experimental ingestion
654 of fluorescent microplastics by pacific oysters, *Crassostrea gigas*, and their effects on the behaviour
655 and development at early stages. *Chemosphere* 254: 126793.
656 <https://doi.org/10.1016/j.chemosphere.2020.126793>.

657 Browne MA, Dissanayake A, Galloway TS, Lowe DM, Thompson RC. Ingested microscopic plastic
658 translocates to the circulatory system of the mussel, *Mytilus edulis* (L). *Environ Sci Technol.* 2008 Jul
659 1;42(13):5026-31. doi: 10.1021/es800249a.

660 Burns, E.E.; Boxall, A.B.A (2018) Microplastics in the aquatic environment: Evidence for or against
661 adverse impacts and major knowledge gaps. *Environ. Toxicol. Chem.* 37: 2776–2796.
662 <https://doi.org/10.1002/etc.4268>

663 Capolupo, M., Gunaalan, K., Booth, A., Sørensen, L., Valbonesi, P., Fabbri, E. 2021. The sub-lethal impact
664 of plastic and tire rubber leachates on the Mediterranean mussel *Mytilus galloprovincialis*,
665 *Environmental Pollution*. 283, ISSN 0269-7491. <https://doi.org/10.1016/j.envpol.2021.117081>.

666 Carr RS, Neff JM, 1984. Quantitative semi-automated enzymatic assay for tissue glycogen. *Comparative*
667 *Biochemistry and Physiology Part B: Comparative Biochemistry* 77B,3:447-449.
668 [https://doi.org/10.1016/0305-0491\(84\)90258-X](https://doi.org/10.1016/0305-0491(84)90258-X)

669 Chandurvelan, R., Marsden, I. D., Gaw, S., and Glover, C. N. (2013). Biochemical biomarker responses of
670 green-lipped mussel, *Perna canaliculus*, to acute and subchronic waterborne cadmium toxicity.
671 *Aquat. Toxicol.* 140–141, 303–313. doi: 10.1016/j.aquatox.2013.06.015

672 Claiborne A (1985) Catalase activity. In: Greenwald RA (ed) *Handbook of methods for oxygen radical*
673 *research*. CRC Press, Boca Raton, pp 283–284

674 Duis, K., Coors, A (2016) Microplastics in the aquatic and terrestrial environment: sources (with a specific
675 focus on personal care products), fate and effects. *Environ Sci Eur* 28, 2.
676 <https://doi.org/10.1186/s12302-015-0069-y>

677 Dusaucy, J.; Gateuille, D.; Perrette, Y.; Naffrechoux, E (2021) Microplastic pollution of worldwide lakes.
678 *Environ. Pollut.* 284: 117075. <https://doi.org/10.1016/j.envpol.2021.117075>

679 Ellman GL, Courtney KD, Andres V, Featherstone RM, 1961. A new and rapid colorimetric determination
680 of acetylcholinesterase activity. *Biochemical Pharmacology* 7(2):88-95.
681 [https://doi.org/10.1016/0006-2952\(61\)90145-9](https://doi.org/10.1016/0006-2952(61)90145-9)

682 Ferreira, I., Venâncio, C., Lopes, I., Oliveira, M (2019) Nanoplastics and marine organisms: what has been
683 studied? *Environ. Toxicol. Pharmacol.* 67: 1–7. <https://doi.org/10.1016/j.etap.2019.01.006>

684 Frias JPGL, Lyashevskaya O, Joyce H, Pagter E, Nash R. (2020) Floating microplastics in a coastal
685 embayment: A multifaceted issue. *Mar. Pollut. Bull.* 158, 111361. doi:
686 10.1016/j.marpolbul.2020.111361.

687 Frings CS, Dunn RT, Queen CA, 1972. Improved determination of total serum lipids by the sulfo-
688 phospho-vanillin reaction. *Clinical Chemistry* 18(7):673-674.
689 <https://doi.org/10.1093/clinchem/18.7.673>

690 Fu, L., Xi, M., Nicholas, R., Wang, Z., Wang, X., Kong, F. & Yu, Z. 2022 Behaviors and biochemical
691 responses of macroinvertebrate *Corbicula fluminea* to polystyrene microplastics. *Sci. Total Environ.*
692 813, 152617. <https://doi.org/10.1016/j.scitotenv.2021.152617>

693 Galgani F, Bocquene G, 1991. Semi-automated colorimetric and enzymatic assays for aquatic organisms
694 using microplate readers. *Water Research*, 25(2) :147-150. [https://doi.org/10.1016/0043-1354\(91\)90023-J](https://doi.org/10.1016/0043-1354(91)90023-J)

696 Gardon, T., Reisser, C., Soyeux, C., Quillien, V., and Le Moullac, G. (2018). Microplastics affect energy
697 balance and gametogenesis in the pearl oyster *Pinctada margaritifera*. *Environ. Sci. Technol.* 52,
698 277–286. doi: 10.1021/acs.est.8b00168

699 Guilhermeino, L., Vieira, L.R., Ribeiro, D., Tavares, A.S., Cardoso, V., Alves, A., Almeida, J.M., 2018. Uptake
700 and effects of the antimicrobial florfenicol, microplastics and their mixtures on freshwater exotic
701 invasive bivalve *Corbicula fluminea*. *Sci. Total Environ.* 622–623, 1131–1142.
702 <https://doi.org/https://doi.org/10.1016/j.scitotenv.2017.12.020>

703 Habib, R.; Thiemann, T.; Kendi, R (2020) Microplastics and Wastewater Treatment Plants—A Review. *J.*
704 *Water Resour. Prot.* 12: 1–35. DOI: 10.4236/jwarp.2020.121001

705 Habig WH, Pabst MJ, Jakoby WB (1974) Glutathione S-transferases. The first enzymatic step in
706 mercapturic acid formation. *J Biol Chem* 249(22):7130–7139.

707 Hahladakis, J. N., Velis, C. A., Weber, R., Iacovidou, E., & Purnell, P. (2018). An overview of chemical
708 additives present in plastics: Migration, release, fate and environmental impact during their use,
709 disposal and recycling. *Journal of hazardous materials*, 344, 179-199.

710 Haider, F., Sokolov, E. P., and Sokolova, I. M. (2018). Effects of mechanical disturbance and salinity stress
711 on bioenergetics and burrowing behavior of the soft-shell clam *Mya arenaria*. *J. Exp. Biol.* 221(Pt
712 4):jeb172643. <https://doi.org/10.1242/jeb.172643>

713 Hermabessiere L, Dehaut A, Paul-Pont I, Lacroix C, Jezequel R, Soudant P, Duflos G. Occurrence and
714 effects of plastic additives on marine environments and organisms: A review. *Chemosphere.* 2017.
715 182:781-793. doi: 10.1016/j.chemosphere.2017.05.096.

716 Latchere O, Audroin T, Hétier J, Métails I, Châtel A. 2021. The need to investigate continuums of plastic
717 particle diversity, brackish environments and trophic transfer to assess the risk of micro and
718 nanoplastics on aquatic organisms, *Environmental Pollution*, Volume 273, 116449, ISSN 0269-7491,
719 <https://doi.org/10.1016/j.envpol.2021.116449>.

720 Lee, M.C., Park, J.C., Lee, J.S., 2018. Effects of environmental stressors on lipid metabolism in aquatic
721 invertebrates. *Aquat. Toxicol.* 200, 83-92. <https://doi.org/10.1016/j.aquatox.2018.04.016>

722 Li, J., Liu, H., Paul Chen, J. 2018. Microplastics in freshwater systems: a review on occurrence,
723 environmental effects, and methods for microplastics detection. *Water Res.* 137, 362-374.
724 <https://doi.org/10.1016/j.watres.2017.12.056>

725 Li, L.; Su, L.; Cai, H.; Rochman, C M.; Li, Q.; Kolandhasamy, P.; Peng, J.; Shi, H. (2019). The uptake of
726 microfibers by freshwater Asian clams (*Corbicula fluminea*) varies based upon physicochemical
727 properties. *Chemosphere*, 221, 107–114. doi:10.1016/j.chemosphere.2019.01.024

728 Livak, K.J., Schmittgen, T.D., 2001. Analysis of relative gene expression data using real-time quantitative
729 PCR and the 2- $\Delta\Delta$ CT method. *Methods* 25, 402–408.2 <https://doi.org/10.1006/meth.2001.1262>.

730 Lushchak, V.I. Environmentally induced oxidative stress in aquatic animals. *Aquat. Toxicol.* 2011, 101,
731 13–30. DOI: 10.1016/j.aquatox.2010.10.006

732 Managing Rivers Wisely, WWF: <https://wwfint.awsassets.panda.org/downloads/mrwloirecasestudy.pdf>

733 Métails I, Latchere O, Roman C, Perrein-Ettajani H, Mouloud M, Georges D, Audroin T, Catrouillet C,
734 Gigault J, Agnès-Feurtet-Mazel, Baudrimont M, Châtel A. Continuum from microplastics to
735 nanoplastics: effects of size and source on the estuarine bivalve *Scrobicularia plana*. *Environ Sci Pollut*
736 *Res Int.* 2023 Apr;30(16):45725-45739. doi: 10.1007/s11356-023-25588-3

737 Mkuye, R., Gong, S., Zhao, L., Masanja, F., Ndandala, C., Bubelwa, E., Yang, C., Deng, Y., 2022. Effects of
738 microplastics on physiological performance of marine bivalves, potential impacts, and enlightening
739 the future based on a comparative study. *Sci. Total Environ.* 838, 155933.
740 <https://doi.org/10.1016/j.scitotenv.2022.155933>

741 Modesto, K.A., Martinez, C.B.R., 2010. Roundup® causes oxidative stress in liver and inhibits
742 acetylcholinesterase in muscle and brain of the fish *Prochilodus lineatus*. *Chemosphere* 78 (3), 294–
743 299. <https://doi.org/10.1016/j.chemosphere.2009.10.047>.

744 Moreschi, A.C., Callil, C.T., Christo, S.W., Ferreira Junior, A.L., Nardes, C., de Faria, É., Girard, P. 2020.
745 Filtration, assimilation and elimination of microplastics by freshwater bivalves, *Case Stud. Chem.*
746 *Environ. Eng.* 2. 100053. <https://doi.org/10.1016/j.cscee.2020.100053>.

747 Naidu, B.C., Xavier, K.M., Shukla, S.P., Jaiswar, A.K., Nayak, B.B., 2022. Comparative study on the
748 microplastics abundance, characteristics, and possible sources in yellow clams of different
749 demographic regions of the northwest coast of India. *J. Hazard. Mater. Lett.* 3, 100051.

750 Oliveira, M., Almeida, M., Miguel, I (2019) A micro(nano)plastic boomerang tale: a never ending story?
751 TrAC – Trends Anal. Chem. 112: 196–200. <https://doi.org/10.1016/j.trac.2019.01.005>

752 Oliveira, P. De, Gomes, A.Q., Pacheco, T.R., Almeida, V.V. De, Saldanha, C., 2012. Cell-specific regulation
753 of acetylcholinesterase expression under inflammatory conditions 51, 129–137.
754 <https://doi.org/10.3233/CH-2011-1520>

755 Oliveira, P., Barboza, L.G.A., Branco, V., Figueiredo, N., Carvalho, C., Guilhermino, L., 2018. Effects of
756 microplastics and mercury in the freshwater bivalve *Corbicula fluminea* (Müller, 1774): Filtration
757 rate, biochemical biomarkers and mercury bioconcentration. *Ecotoxicol. Environ. Saf.* 164, 155–163.
758 <https://doi.org/10.1016/j.ecoenv.2018.07.062>

759 Pan, G., Dutta, H.M., 1998. The inhibition of brain acetylcholinesterase activity of juvenile largemouth
760 bass *micropterus salmoides* by sublethal concentrations of diazinon. *Environ. Res.* 79 (2), 133–137.
761 <https://doi.org/10.1006/enrs.1998.3868>.

762 Parra, S., Varandas, S., Santos, D., Fernandes, L., Cabecinha, E., Monteiro, S.M., 2021. Multi-Biomarker
763 Responses of Asian Clam *Corbicula fluminea*. *Water*.

764 Pedersen, A. F., Gopalakrishnan, K., Boegehold, A. G., Peraino, N. J., Westrick, J. A., and Kashian, D. R.
765 (2020). Microplastic ingestion by quagga mussels, *Dreissena bugensis*, and its effects on physiological
766 processes. *Environ. Pollut.* 260:113964. doi: 10.1016/j.envpol.2020.113964

767 Phuong, N.N., Poirier, L., Lagarde, F., Kamari, A., Zalouk-Vergnoux, A., 2018. Microplastic abundance and
768 characteristics in French Atlantic coastal sediments using a new extraction method. *Environmental
769 Pollution* 243, 228–237. <https://doi.org/10.1016/j.envpol.2018.08.032>

770 Pires A, Cuccaro A, Sole M, Freitas R (2022) Micro(nano)plastics and plastic additives effects in marine
771 annelids: A literature review. *Environ Res.* 214, 113642. doi: 10.1016/j.envres.2022.113642.

772 Qu H, Ma R, Barrett H, Wang B, Han J, Wang F, Chen P, Wang W, Peng G, Yu G. 2020. How microplastics
773 affect chiral illicit drug methamphetamine in aquatic food chain? From green alga (*Chlorella
774 pyrenoidosa*) to freshwater snail (*Cipangopaludian cathayensis*). *Environ Int.* 136:105480. doi:
775 10.1016/j.envint.2020.105480.

776 Revel, M., Lagarde, F., Perrein-Ettajani, H., Bruneau, M., Akcha, F., Sussarellu, R., Rouxel, J., Costil, K.,
777 Decottignies, P., Cognie, B., Châtel, A., Mouneyrac, C., 2019. Tissue-specific biomarker responses in
778 the blue mussel *Mytilus* spp. exposed to a mixture of microplastics at environmentally relevant
779 concentrations. *Front. Environ. Sci.* 7, 1–14. <https://doi.org/10.3389/fenvs.2019.00033>

780 Ribeiro, F., Garcia, A.R., Pereira, B.P., Fonseca, M., Mestre, N.C., Fonseca, T.G., et al. (2017).
781 Microplastics effects in *Scrobicularia plana*. *Mar. Pollut. Bull.* 122, 379–391. doi:
782 10.1016/j.marpolbul.2017.06.078

783 Rochman CM, Parnis JM, Browne MA, Serrato S, Reiner EJ, Robson M, Young T, Diamond ML, Teh SJ.
784 Direct and indirect effects of different types of microplastics on freshwater prey (*Corbicula fluminea*)
785 and their predator (*Acipenser transmontanus*). *PLoS One.* 2017 Nov 6;12(11):e0187664. doi:
786 10.1371/journal.pone.0187664.

787 Sendra M, Sparaventi E, Novoa B, Figueras A (2021) An overview of the internalization and effects of
788 microplastics and nanoplastics as pollutants of emerging concern in bivalves. *Sci Total Environ.*
789 753:142024. doi: 10.1016/j.scitotenv.2020.142024.

790 Shang, Y., Wang, X., Chang, X., Sokolova, I. M., Wei, S., Liu, W., ... & Wang, Y. (2021). The effect of
791 microplastics on the bioenergetics of the mussel *Mytilus coruscus* assessed by cellular energy
792 allocation approach. *Frontiers in Marine Science*, 8, 754789.

793 Statista Research Department, Jun 12, 2023. [https://www.statista.com/statistics/282732/global-
794 production-of-plastics-since-1950/](https://www.statista.com/statistics/282732/global-production-of-plastics-since-1950/)

795 Su, L., Cai, H., Kolandhasamy, P., Wu, C., Rochman, C. M., & Shi, H. (2018). Using the Asian clam as an
796 indicator of microplastic pollution in freshwater ecosystems. *Environmental pollution*, 234, 347-355.
797 <https://doi.org/10.1016/j.envpol.2017.11.075>

798 Su L, Xue Y, Li L, Yang D, Kolandhasamy P, Li D, et al. Microplastics in Taihu Lake, China. *Environ Pollut*
799 2016;216:711–9. <https://doi.org/10.1016/j.envpol.2017.11.075>

800 Sunderman F.W., Marzouk A., Hopfer S.M., Zaharia O., Reid M.C. Increased lipid peroxidation in tissues
801 of nickel chloride-treated rats *Ann. Clin. Lab. Sci.*, 15 (1985), pp. 229-236

802 Tan, H., Yue, T., Xu, Y., Zhao, J., Xing, B., 2020. Microplastics reduce lipid digestion in simulated human
803 gastrointestinal system. *Environ. Sci. Technol.* 54 (19), 12285–12294.
804 <https://doi.org/10.1021/acs.est.0c02608>.

805 Treilles, R., Gasperi, J., Gallard, A., Saad, M., Dris, R., Partibane, C., ... & Tassin, B. (2021). Microplastics
806 and microfibers in urban runoff from a suburban catchment of Greater Paris. *Environmental Pollution*,
807 287, 117352. <https://doi.org/10.1016/j.envpol.2021.117352>

808 Uddin, S.; Fowler, S.W.; Uddin, M.F.; Behbehani, M.; Naji, A (2021) A review of microplastic distribution
809 in sediment profiles. *Mar. Pollut. Bull.* 163, 111973.
810 <https://doi.org/10.1016/j.marpolbul.2021.111973>

811 Utermöhl, H. (1958). Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. *Mitteilungen-*
812 *Int. Ver. Für Limnol.* 9, 1–38.

813 Vidal A., Phuong N.N., Métails I., Gasperi J., Châtel A. 2023. Assessment of microplastic contamination in
814 the Loire River (France) throughout analysis of different biotic and abiotic freshwater matrices,
815 *Environmental Pollution*. 122167, ISSN 0269-7491, <https://doi.org/10.1016/j.envpol.2023.122167>.

816 Wang J, Liu X, Li Y, Powell T, Wang X, Wang G, Zhang P (2019) Microplastics as contaminants in the soil
817 environment: A mini-review. *Sci Total Environ.* 691: 848-857. doi: 10.1016/j.scitotenv.2019.07.209.

818 Ward JE, Rosa M, Shumway SE (2019) Capture, ingestion, and egestion of microplastics by suspension-
819 feeding bivalves: a 40-year history. *Anthropocene coasts* 2(1):39–49. [https://doi.org/10.1139/anc-](https://doi.org/10.1139/anc-2018-0027)
820 [2018-0027](https://doi.org/10.1139/anc-2018-0027)

821 Wirnkor, V.A.; Ngozi, V.E.; Ebere, E.C. Microplastics, an emerging concern: A review of analytical
822 techniques for detecting and quantifying microplastics. *Anal. Methods Environ. Chem. J.* 2019, 2, 15–
823 32. <https://doi.org/10.24200/amecj.v2.i2.57>

824 Wu, H., Zhong, M., Lu, Z., Shan, X., Li, F., Ji, C., Cong, M. 2018. Biological effects of tris (1-chloro-2-propyl)
825 phosphate (TCPP) on immunity in mussel *Mytilus galloprovincialis*. *Environmental Toxicology and*
826 *Pharmacology*. 61, 102-106. <https://doi.org/10.1016/j.etap.2018.05.022>.

827 Wu, Y., Yang, J., Li, Z., He, H., Wang, Y., Wu, H., Xie, L., Chen, D. Wang, L. 2022. How does bivalve size
828 influence microplastics accumulation? *Environmental Research*, 214, Part 1, 113847.
829 <https://doi.org/10.1016/j.envres.2022.113847>.