



HAL
open science

Extraction of microplastics from sediment matrices: Experimental comparative analysis

Mel Constant, Gabriel Billon, Noémie Breton, Claire Alary

► **To cite this version:**

Mel Constant, Gabriel Billon, Noémie Breton, Claire Alary. Extraction of microplastics from sediment matrices: Experimental comparative analysis. *Journal of Hazardous Materials*, 2021, 420, pp.126571. 10.1016/j.jhazmat.2021.126571 . hal-03338741

HAL Id: hal-03338741

<https://hal.science/hal-03338741>

Submitted on 2 Aug 2023

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution - NonCommercial 4.0 International License

1 **Extraction of microplastics from sediment matrices:** 2 **experimental comparative analysis**

3 Mel Constant^{1*}, Gabriel Billon², Noémie Breton² and Claire Alary¹

4 ¹Univ. Lille, Institut Mines-Télécom, Univ. Artois, Junia, ULR 4515 - LGCgE, Laboratoire de
5 Génie Civil et géo-Environnement, F-59000 Lille, France

6 ²Univ. Lille, CNRS, UMR 8516 - LASIRE, Laboratoire Avancé de Spectroscopie pour les
7 Interactions, la Réactivité et l'Environnement, F-59000 Lille, France

8 *corresponding author: mel.constant@lilo.org

9 **Keywords:** Microplastics, soil, sediments, extraction methods, comparative analysis.
10

11 **1. Introduction**

12 Anthropogenic litter is a widespread, ubiquitous, and global threat for marine and terrestrial
13 ecosystems. Plastics are the most abundant litter by number in most environments, e.g., >80% at the
14 surface of European rivers (González Fernández *et al.*, 2018). Plastics have low degradability,
15 especially under low UV-radiation, oxygen concentration, and temperature, such as on the sea floor
16 (Andrady, 2017). The degradation of large plastic pieces creates small fragments called
17 microplastics (MPs; <5 mm; GESAMP, 2015), which are more numerous than macroplastics (>5
18 mm). MPs can also be manufactured directly at micrometric size by plastic industries (GESAMP,
19 2015). MPs have been found in most biotopes worldwide, even within remote and/or pristine areas
20 (Allen *et al.*, 2019; Bergmann *et al.*, 2017). Due to their small size, they can be ingested by a wide
21 range of organisms, which can be associated with potential harmful toxicology (Wright *et al.*, 2013).
22 In addition, MPs can adsorb a broad range of pollutants from the surrounding environment
23 (Rochman, 2015). They can also desorb pollutants which were used during their fabrications, such
24 as bisphenol A and phthalates (Hirai *et al.*, 2011; Teuten *et al.*, 2009).

25 The quantification of MPs can be divided into four main steps: sampling, sample treatment,
26 counting, and analysis. The ideal protocol should enable the quantification of all MPs, regardless of
27 length or density. It should also be convenient and cheap to enable large-scale investigations. This
28 challenge is particularly important for studies on sediment matrices (e.g., beach, riverbed, and deep
29 sea) where MPs must be extracted from sediments before the identification step. Most extraction
30 protocols are based on the difference of volumetric mass density between sediments and plastics
31 (He *et al.*, 2021). Recently, oil has been proposed as an interesting alternative for this separation
32 (Mani *et al.*, 2019). Plastics are lipophilic, and therefore, attracted by oil. In addition, plastic and oil
33 are hydrophobic, so they tend to join each other in a water environment. Electrostatic separation
34 (Felsing *et al.*, 2018), magnetic separation (Grbic *et al.*, 2019), solvent extraction (Corti *et al.*, 2020;

35 Fuller and Gautam, 2016), and depolymerization with phase transfer catalysis (Castelvetto *et al.*,
36 2020) have also been investigated for this purpose. However, they require expensive equipment,
37 which restrict their uses, despite their remarkable results (Bellasi *et al.*, 2021).

38 In density separation methods, the use of an extracting solution (ES) with an appropriate volumetric
39 mass density (i.e., less dense than the sediment and denser than the plastic) enables the recovery of
40 MPs at the solution surface. Zinc chloride ($ZnCl_2$) and sodium iodide (NaI) are the most common
41 high-density solutions used for this purpose (>1.6 g/mL), and they can be applied to approximately
42 93% of the plastics produced in Europe (Kedzierski *et al.*, 2017b). However, they are both toxic to
43 the environment and humans, and expensive. Therefore, a hypersaline solution (NaCl) is often
44 preferred because it is non-toxic and cheap. Nevertheless, NaCl only allows the extraction of low-
45 density polymers (<1.2), which represents only half of the European plastic production (Kedzierski
46 *et al.*, 2017b).

47 This study aims to compare several protocols for the extraction of MPs from sediments. Several ES
48 (oil, water, oil-in-water, NaCl, oil-in-NaCl, and NaI) were tested in combination with three isolation
49 methods (IM) (hand stirring, centrifugation, and aeration). The results are expected to help future
50 studies to solve the challenge of extracting MPs from sediments and soils. Our findings do not
51 emphasize one unique protocol. Instead, they provide information for the selection of the most
52 suitable process, by providing the basis to avoid common mistakes and enabling a more accurate
53 comparison between the methods.

54 **2. Methods**

55 **2.1 Environmental sample collection**

56 Sediments were sampled in May 2019 within a dredging disposal area belonging to *Voies*
57 *Navigables de France* (VNF, Navigable Waterways of France), along the Aa River (St-Omer,

58 France; 50.773479N, 2.265114E). Two sub-sites with contrasting granulometry were selected. In
59 the first one, the sediments were coarse and heterogeneous, whereas in the second one, they were
60 fine and homogeneous (Table S1). At each sub-site, three samples of approximately 2 kg of wet
61 sediment were collected using a metal trowel just below the surface and between 10 and 20 cm
62 depth. The 2-kg samples were transferred into glass bottles and stored at 4 °C before the tests.
63 Environmental MPs trapped within those sediments have been quantified and characterized in a
64 concomitant study and cannot be confused with added pristine MP used in this work (See details in
65 Fig S1; Constant et al., 2021).

66 **2.2 Recovery experiments**

67 **2.2.1 Spiked pristine microplastics**

68 For each experiment, approximately 50 g of wet sediments were transferred to a 100 mL glass
69 container, to which 50 mL of Milli-Q water was added to liquefy the sediment. After one week, the
70 water was removed and discarded using a glass pipette. Subsequently, pristine MPs were added to
71 the container (Table S2). A mixture of low and high-density polymers were used: 10 pellets (2 ± 0.2
72 mm long) of polyethylene (PE; 0.9 g/cm^3) and polyethylene terephthalate (PET; 1.4 g/cm^3), 10
73 fragments (1 ± 0.3 mm long) of PE and PET, 10 foams (1 ± 0.2 mm long) comprising polyethylene
74 and acrylonitrile butadiene styrene (PE-ABS; 1.0 g/cm^3), and 10 fibers (4 ± 0.5 mm long) of
75 polyamide (PA; density: 1.2 g/cm^3) and polyester (PES; density: 1.4 g/cm^3). In addition, 10 pieces
76 of cotton fiber were added to represent fibers of heavy density (1.5 g/cm^3). Pellets were purchased
77 from a plastic manufacturer (Acordis©). Fragments were obtained by cryo-crushing the pellets, and
78 fibers were obtained by cutting bobbin threads. The added MPs were easily recognizable from the
79 other particles already present in the samples.

80 **2.2.2 Separation methods**

81 He *et al.* (2021) reviewed 144 publications extracting MPs within sediment, soil, sand, sludge, and
82 other complex environments. They noticed that MPs are mostly separated from sediment matrices
83 by gravity. As most synthetic polymers are lighter than sediments, MPs can be extracted with a
84 solution of appropriate density. In density-based and lipophilicity-based separation methods, an ES
85 is added to the sediment, mixed, left to stand for a separation period, and the supernatant is
86 collected from the top or the sediment is removed from the bottom. A wide range of IMs and ESs
87 have already been proposed. In the present study, three different IMs were tested:

- 88 • Hand stirring: manual agitation for 2 min followed by 5 h of decantation, adapted from
89 Thompson *et al.* (2004)
- 90 • Centrifugation: manual agitation for 30 s followed by 5-min centrifugation at 500 rpm, as
91 performed by Phuong *et al.* (2018)
- 92 • Aeration: airflow input for 2 min at 100 mL/s followed by 5 h of decantation, adapted from
93 Nuelle *et al.* (2014)

94 For each IM, six ESs were tested:

- 95 • Water (1 g/mL; Phuong *et al.*, 2018)
- 96 • Canola oil (0.9 g/mL; Rodrigues *et al.*, 2018)
- 97 • Water and a thin layer of canola oil (Crichton *et al.*, 2017)
- 98 • Concentrated NaCl solution (1.2 g/mL; Thompson *et al.*, 2004)
- 99 • Concentrated NaCl solution with a thin layer of canola oil (Karlsson *et al.*, 2017)
- 100 • Concentrated NaI solution (1.6 g/mL; Claessens *et al.*, 2013)

101 For all tests, 100 ± 1 mL of solution was added to 50 ± 5 g of sediment in a glass container. After each
102 density separation process, the supernatant was filtered by GF/F Whatman filters (47 mm diameter;
103 $0.7\ \mu\text{m}$ porosity). This extraction procedure was repeated 3 times. Filters were dried at $50\ ^\circ\text{C}$
104 overnight and then examined under a Leica MZ12 dissecting stereo-microscope ($\times 10$

105 magnification). First, 36 tests were performed (3 IMs × 6 ESs × 2 sediments × 1 replicate).
106 Subsequently, the three ES with the best extraction yield (i.e., oil-in-water, concentrated NaCl and
107 NaI solutions) were tested two additional times (3 IMs × 3 ESs × 2 sediments × 2 replicates),
108 resulting in additional set of 36 tests, for a total of 72 tests for the study (Table 1).

109 **2.3 Data analysis**

110 The extraction rate (%) was calculated for each pristine MP based on the number of MP recovered
111 compared to the number of spiked MP. Statistical analyses were performed using R software (R
112 Core Team, 2018), including the caption package “captioner” (Alathea, 2015), manuscript package
113 “rmarkdown” (Allaire *et al.*, 2018), data manipulation package “dplyr” (Wickham *et al.*, 2017), and
114 graphical package “ggpubr” (Kassambara, 2017). As normality of distribution was not observed
115 (Shapiro-Wilk test), four non-parametric tests were used: the Wilcoxon-Man-Whitney test to
116 compare two groups, Kruskal-Wallis test to compare more than two groups, Scheirer-Ray-Hare test
117 to investigate the influence of two different factors and the interaction among factors, and post hoc
118 Dunn's test to compare differences of all possible pairs and pinpoint specific medians that are
119 significantly different from the others.

120 **3. Results**

121 The recovery rates for 3 IMs and 6 ESs were determined by spiking pristine MPs in fine and coarse-
122 grained sediments. All results (i.e., for every polymer and shape in both sediment types) are shown
123 in Fig. S2. Some trends were observed after the first set of 36 tests (i.e., no replicate). Overall, the
124 recovery rates for low-density (PE and PE-ABS pooled) and high-density polymers (PA, PES, and
125 PET pooled) were significantly different for the various ES (Scheirer-Ray-Hare (SHR) test, both $p <$
126 0.01 ; Table S3) but not for different IM (SHR test, $p = 0.11$ and 0.16 , respectively; Table S3).
127 Interactions between ES and IM also created significant differences (SHR test, $p = 0.02 < 0.01$;

128 Table S3). Extraction efficiencies were heterogeneous for every ES, ranging from 0% to 100% (Fig.
129 1). The recovery rate upon use of oil alone was $37\pm 32\%$ (0–83%; $n = 6$), and it was significantly
130 lower for low-density polymers (Kruskal-Wallis test, $p < 0.01$; Table S3). Excluding oil, $84\pm 17\%$
131 (7–100%; $n = 66$) of low-density MPs were recovered, and more than two thirds (68%) of the tests
132 presented an extraction rate greater than 80%. Excluding the lowest extraction rate of oil-in-water,
133 the results for ESs (except oil) varied within the range of 50%–100% with the relative standard
134 deviation (ratio of standard deviation to the mean) varying between 0.11 and 0.21.

135 For high-density polymers (PA, PES, PET), extraction efficiencies were also heterogeneous.
136 Excluding the highest extraction rate (1/6 value), results for water and NaCl were all lower than 3%.
137 The results of other ES varied substantially, and their relative standard deviation ranged between
138 0.23 and 0.73. Recovery efficiencies for water and oil were $6\pm 11\%$ (0–27%; $n = 6$) and $6\pm 10\%$ (0–
139 28%; $n = 6$) respectively, which are significantly lower than those for the NaI solution and the oil-
140 in-NaCl solution (Kruskal-Wallis test, $p < 0.01$; Table S3), at $71\pm 17\%$ (42–90%; $n = 18$) and
141 $46\pm 26\%$ (0–82%; $n = 18$), respectively. The oil-in-water and NaCl solutions had significantly lower
142 efficiencies than the NaI solution (Kruskal-Wallis test, $p < 0.01$; Table S3), at $26\pm 20\%$ (10–65%, n
143 = 18) and $29\pm 22\%$ (2–60%, $n = 6$), respectively, but these results were not significantly different
144 from those of other ESs. In addition, the NaI solution presented a significantly higher efficiency
145 than other ESs.

146 The replicate experiments performed with oil (in water and NaCl solution) and NaI solution
147 confirmed the previously observed high heterogeneity (Fig. 2). For PA, PET, and PES MPs, the
148 extraction rates were widely dispersed between 0% and 100% for each of the three selected ES,
149 with mean values ranging between 13% and 84%, and relative standard deviation ranging between
150 0.22 and 1.36. NaI was significantly more efficient than oil for PA fibers, PET pellets, and PET
151 fragments, but not for PES fibers (Kruskal-Wallis tests; Fig. 2). For this last ES, no significant
152 differences were observed between different IM (i.e., hand stirring, centrifugation, and fluidization;

153 Kruskal-Wallis tests; Fig. 3) for PA, PET, and PES MPs. For fine sediments, the recovery rate was
154 slightly better with a centrifugation step than other IM for PET, but not for fibers. Extraction rates
155 were slightly better for coarse than fine sediments, but the differences were not significant (SHR
156 test, $p = 0.09$; Table S3).

157 For each test, the extraction process was repeated three times. The first extraction recovered
158 $67 \pm 35\%$ (0–100%; $n = 72$) of the total recovered MP (i.e., after three extractions). The second and
159 third extractions represented $19 \pm 27\%$ (0%–100%) and $14 \pm 23\%$ (0%–100%) of the total recovered
160 MP, respectively. The extraction efficiency of NaI after only one extraction was $45 \pm 28\%$ (17%–
161 100%; $n = 18$) for low-density polymers and $47 \pm 23\%$ (5%–85%) for high-density polymers. After
162 three extractions, this rate doubled ($89 \pm 11\%$; 67%–100%) and increased one and a half times
163 ($71 \pm 17\%$; 42%–90%) for low- and high-density polymers, respectively.

164 **4. Discussion**

165 The extraction rates varied highly within and among the 18 separation protocols tested (Fig. S1).
166 Regardless of MP and sediment types, extraction efficiencies changed remarkably between the
167 replicates. Such variability implies a low repeatability, which was under the minimum requirements
168 for a reliable MP analysis (Cadiou *et al.*, 2020). Nevertheless, the NaI-centrifugation protocol
169 presented a lower variability (<10%), which was more acceptable, particularly for coarse sediments.
170 The extraction rates within a protocol did not change significantly for different IM (hand stirring,
171 centrifugation, and aeration) or sediment feature (coarse or fine). Therefore, the most convenient IM
172 can be chosen without influencing the recovery efficiency. Hand stirring was the simplest, easiest,
173 and fastest method. Centrifugation resulted in a solid sediment pellet at the bottom of the container,
174 which limited the resuspension of fine sediments, but it did not require a settling step (5 h). Aeration
175 was the smoothest method, as it limited the fragmentation of weak MP, and the bubbles promoted
176 the separation of MPs adhered to soil or sediments (Nuelle *et al.*, 2014). The different size fractions

177 of sediment grains may lead to different physical properties (Konechnaya *et al.*, 2020). For instance,
178 fine particles can adhere to the surface of MPs, increasing their densities and decreasing the
179 efficiency of density separation methods (Radford *et al.*, 2021). To address this issue, some authors
180 have proposed the use of ultrasonic waves to separate particles (Wang *et al.*, 2018; Zhang *et al.*,
181 2018). The absence of significant differences between fine and coarse sediments in this study may
182 have occurred due to the use of pristine MPs, whereas the surfaces of weathered MPs may react
183 differently (Andrady, 2017).

184 Fragments, pellets, and foams of low-density polymers (PE and PE-ABS) were well extracted
185 (>80%) by all ESs tested, except oil alone. Contrary to Rodrigues *et al.* (2018), which obtained a
186 mean weight recovery of 65% for both low and high-density polymers (PE, PET, PP, PS, and PVC;
187 results not provided for each individual polymer), less than half of MPs spiked were recovered
188 using only oil. Rodrigues *et al.* (2018) mentioned that the recoveries may be overestimated because
189 oil can stay attached to particles even after filtration and drying, thereby increasing the final weight
190 values.

191 To a large extent, extraction rates of high-density polymers (PET, PA, and PES) were lower than
192 those obtained in previous studies (Fig. 4 and Table S4). Part of this relatively lower efficiency
193 might be attributed to the difficulty in collecting MPs from the supernatant. We observed MPs
194 adhered to the glass walls of the container used for extraction, which likely occurred because the
195 MPs used were pristine, and therefore, more prone to electrostatic interactions. Repeated extractions
196 increase the probability to catch those particles. Increasing the number of separation steps may also
197 improve the separation of MPs and sediment matrices. In the literature, several interesting
198 alternative methods have been investigated. Nuelle *et al.* (2014) reported an overflow recovery
199 process that prevents the adhesion and/or removes the MPs adhered to glass. Nakajima *et al.* (2019)
200 proposed a glass device for an easier supernatant separation. Mani *et al.*, (2019) proposed the

201 removal of sediments from the bottom, instead of collecting the supernatant at the top, with the risk
202 of resuspension of fine particles.

203 Denser polymers were poorly extracted when NaCl and oil-in-water were used. NaCl has a low
204 density and has been reported by many studies to be less efficient than other substances, as shown
205 in Figure 4 (Han *et al.*, 2019; Li *et al.*, 2019; Zhang *et al.*, 2020). Oil-in-water has been
206 recommended as a good ES as it is neither harmful nor expensive. The separation of sediments and
207 MPs is not based on densities, but on the hydrophobicity of plastics and oil. Nevertheless, lipophilic
208 properties of plastics may be altered by contaminants on MP surfaces (He *et al.*, 2021). Several
209 authors obtained recovery rates >90% for dense polymers in sediments and wastewater effluents
210 (Crichton *et al.*, 2017; Mani *et al.*, 2019; Scopetani *et al.*, 2020). As previously mentioned, oil is
211 difficult to remove and interferes with the Fourier transform infrared spectroscopy and Raman
212 spectra, but the use of aggressive detergents can be efficient for the elimination of oil (Mani *et al.*,
213 2019). In this study, the addition of few milliliters of oil (oil-in-water and in saline solutions) did
214 not significantly change the recovery rates of water and saline solutions alone. Finally, NaI
215 presented the best extraction efficiency for high-density fragments, pellets, and fibers. Most studies
216 using ES with a density higher than 1.5 achieved quantitative extraction rates (>90%; Table S4).
217 This indicates that the ES (i.e., chemicals) probably exerts less influence than density. NaI has been
218 commonly used because it can reach a density of 1.8, and the solution can be reused several times
219 (Kedzierski *et al.*, 2017b). In addition, some methods can significantly reduce the amount of
220 sediment required for the process. For example, the elutriation method developed by Kedzierski *et*
221 *al.* (2016) is based on particle-size subdivision and can extract >90% of dense plastics, with a sand
222 recovery <1%, which considerably reduces the amount of NaI required. However, this method was
223 developed and improved only for sand sediments (Kedzierski *et al.*, 2018, 2017a). Its efficiency
224 would likely be lower for fine and/or organic matter-rich sediments, which can aggregate and/or

225 react with plastics (Zhang and Liu, 2018). Organic matter can be removed using an oxidant such as
226 hydrogen peroxide, and aggregation can be prevented using a dispersal agent.

227 Currently, the methodologies are relatively similar to extract MPs from the different complex solid
228 matrices (He et al., 2021; Junhao et al., 2021). However, matrix features (organic matter, particle-
229 size, etc.) may severely influence the extracting rate, promoting one approach rather than the others.
230 Some methods, recently developed, are less reliant on matrix characteristics. Solvent extractions
231 ensure high recoveries, but dissolve the MP items (La Nasa et al., 2021; Stile et al., 2021; Wen et al.,
232 2021). Tötzke et al., (2021) proposed to use complementary neutron and X-ray tomography to
233 identify MPs in sediment, without extracting them. Finally, current methods cannot accurately extract
234 all types of plastics (size, shape, and density). Therefore, monitoring studies should choose and
235 optimize their protocols according to their scientific requirements, and associate different methods
236 to widen the extractable range of plastics.

237 **5. Conclusions**

238 This study investigated a major challenge for the quantification of MPs within sediment matrices:
239 the separation between sediments and MPs. Eighteen combinations of IM and ES were tested for
240 light and dense pristine polymers on fine and coarse sediments. The main conclusions based on 72
241 tests are as follows:

242 (1) The IMs (hand stirring, centrifugation, and aeration) and sediment features (coarse or fine) did
243 not affect extraction efficiencies;

244 (2) Fragments, pellets, and foams of low-density polymers (PE and PE-ABS) were well extracted
245 (>80%) for all tested ESs, excepted pure oil;

246 (3) Overall, NaI presented the best extraction efficiency for the extraction of fragments, pellets, and
247 fibers of high-density polymers;

248 (4) The addition of few milliliters of oil to the aqueous solutions did not improve the recovery of
249 pristine MPs. Nevertheless, the oil-in-saline solution presented a slight better extraction than water
250 and oil alone;

251 (5) Repeated extractions improved recovery efficiency.

252 This work should be continued using MPs artificially altered in laboratory and MPs directly
253 recovered from the environment so that possible variations in electrostatic effects and other
254 adsorption mechanisms can be evaluated. The interactions between MPs and sedimentary particles
255 (such as clays and organic matter), which can modify densities and probably their extractability
256 rates, should also be investigated.

257

258 References

- 259 Alatheia, L., 2015. Captioner: numbers figures and creates simple captions. [https://CRAN.R-](https://CRAN.R-project.org/package=captioner)
260 [project.org/package=captioner](https://CRAN.R-project.org/package=captioner)
- 261 Allaire, J., Xie, Y., McPherson, J., Luraschi, J., Ushey, K., Atkins, A., Wickham, H., Cheng, J.,
262 Chang, W., 2018. Rmarkdown: dynamic documents for R. [https://CRAN.R-](https://CRAN.R-project.org/package=rmarkdown)
263 [project.org/package=rmarkdown](https://CRAN.R-project.org/package=rmarkdown)
- 264 Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S.,
265 Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain
266 catchment. *Nat. Geosci.* 12, 339–344. <https://doi.org/10.1038/s41561-019-0335-5>
- 267 Andrady, A.L., 2017. The plastic in microplastics: A review. *Mar. Pollut. Bull.* 119, 12–22.
268 <https://doi.org/10.1016/j.marpolbul.2017.01.082>
- 269 Bellasi, A., Binda, G., Pozzi, A., Boldrocchi, G., Bettinetti, R., 2021. The extraction of microplastics
270 from sediments: An overview of existing methods and the proposal of a new and green
271 alternative. *Chemosphere* 278, 130357. <https://doi.org/10.1016/j.chemosphere.2021.130357>
- 272 Bergmann, M., Wirzberger, V., Krumpfen, T., Lorenz, C., Primpke, S., Tekman, M.B., Gerdt, G.,
273 2017. High Quantities of Microplastic in Arctic Deep-Sea Sediments from the
274 HAUSGARTEN Observatory. *Environ. Sci. Technol.* 51, 11000–11010.
275 <https://doi.org/10.1021/acs.est.7b03331>
- 276 Cadiou, J.-F., Gerigny, O., Koren, Š., Zeri, C., Kaberi, H., Alomar, C., Panti, C., Fossi, M.C.,
277 Adamopoulou, A., Digka, N., Deudero, S., Concato, M., Carbonell, A., Bains, M., Galli, M.,
278 Galgani, F., 2020. Lessons learned from an intercalibration exercise on the quantification and
279 characterisation of microplastic particles in sediment and water samples. *Mar. Pollut. Bull.*
280 154, 111097. <https://doi.org/10.1016/j.marpolbul.2020.111097>
- 281 Castelvetro, V., Corti, A., Bianchi, S., Ceccarini, A., Manariti, A., Vinciguerra, V., 2020.
282 Quantification of poly(ethylene terephthalate) micro- and nanoparticle contaminants in marine
283 sediments and other environmental matrices. *J. Hazard. Mater.* 385, 121517.
284 <https://doi.org/10.1016/j.jhazmat.2019.121517>
- 285 Claessens, M., Van Cauwenberghe, L., Vandegehuchte, M.B., Janssen, C.R., 2013. New techniques
286 for the detection of microplastics in sediments and field collected organisms. *Mar. Pollut. Bull.*
287 70, 227–233. <https://doi.org/10.1016/j.marpolbul.2013.03.009>
- 288 Constant, M., Alary, C., De Waele, I., Dumoulin, D., Breton, N., Billon, G., 2021. To What Extent
289 Can Micro- and Macroplastics Be Trapped in Sedimentary Particles? A Case Study
290 Investigating Dredged Sediments. *Environ. Sci. Technol.* 55, 5898–5905.
291 <https://doi.org/10.1021/acs.est.0c08386>
- 292 Corti, A., Vinciguerra, V., Iannilli, V., Pietrelli, L., Manariti, A., Bianchi, S., Petri, A., Cifelli, M.,
293 Domenici, V., Castelvetro, V., 2020. Thorough Multianalytical Characterization and
294 Quantification of Micro- and Nanoplastics from Bracciano Lake’s Sediments. *Sustainability*
295 12, 878. <https://doi.org/10.3390/su12030878>
- 296 Crichton, E.M., Noël, M., Gies, E.A., Ross, P.S., 2017. A novel, density-independent and FTIR-
297 compatible approach for the rapid extraction of microplastics from aquatic sediments. *Anal.*
298 *Methods* 9, 1419–1428. <https://doi.org/10.1039/C6AY02733D>
- 299 Felsing, S., Kochleus, C., Buchinger, S., Brennholt, N., Stock, F., Reifferscheid, G., 2018. A new
300 approach in separating microplastics from environmental samples based on their electrostatic
301 behavior. *Environ. Pollut.* 234, 20–28. <https://doi.org/10.1016/j.envpol.2017.11.013>
- 302 Fuller, S., Gautam, A., 2016. A Procedure for Measuring Microplastics using Pressurized Fluid
303 Extraction. *Environ. Sci. Technol.* 50, 5774–5780. <https://doi.org/10.1021/acs.est.6b00816>

304 GESAMP, 2015. Sources, fate and effects of microplastics in the marine environment: a global
305 assessment, IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group
306 of Experts on the Scientific Aspects of Marine Environmental Protection). ed, Rep. Stud.
307 GESAMP. <https://doi.org/10.13140/RG.2.1.3803.7925>

308 González Fernández, D., Hanke, G., Kideys, A., Alicia, N., Sanchez-Vidal, A., Bruge, A., Bayram,
309 öztürk, Palma, C., Santinelli, C., Daan, D., Damia, B., Elias, D., Rojo-Nieto, E., Fernando, F.,
310 Bessa, F., Suaria, G., Siedlewicz, G., Castro-Jiménez, J., Joana, G., Yuri, G., 2018. Floating
311 Macro Litter in European Rivers-Top Items. <https://doi.org/10.2760/316058>

312 Grbic, J., Nguyen, B., Guo, E., You, J.B., Sinton, D., Rochman, C.M., 2019. Magnetic Extraction of
313 Microplastics from Environmental Samples. *Environ. Sci. Technol. Lett.* 6, 68–72.
314 <https://doi.org/10.1021/acs.estlett.8b00671>

315 Han, X., Lu, X., Vogt, R.D., 2019. An optimized density-based approach for extracting
316 microplastics from soil and sediment samples. *Environ. Pollut.* 254, 113009.
317 <https://doi.org/10.1016/j.envpol.2019.113009>

318 He, D., Zhang, X., Hu, J., 2021. Methods for separating microplastics from complex solid matrices:
319 Comparative analysis. *J. Hazard. Mater.* 409, 124640.
320 <https://doi.org/10.1016/j.jhazmat.2020.124640>

321 Hirai, H., Takada, H., Ogata, Y., Yamashita, R., Mizukawa, K., Saha, M., Kwan, C., Moore, C.,
322 Gray, H., Laursen, D., Zettler, E.R., Farrington, J.W., Reddy, C.M., Peacock, E.E., Ward,
323 M.W., 2011. Organic micropollutants in marine plastics debris from the open ocean and
324 remote and urban beaches. *Mar. Pollut. Bull.* 62, 1683–1692.
325 <https://doi.org/10.1016/j.marpolbul.2011.06.004>

326 Junhao, C., Xining, Z., Xiaodong, G., Li, Z., Qi, H., Siddique, K.H.M., 2021. Extraction and
327 identification methods of microplastics and nanoplastics in agricultural soil: A review. *J.*
328 *Environ. Manage.* 294, 112997. <https://doi.org/10.1016/j.jenvman.2021.112997>

329 Karlsson, T.M., Vethaak, A.D., Almroth, B.C., Ariese, F., van Velzen, M., Hassellöv, M., Leslie,
330 H.A., 2017. Screening for microplastics in sediment, water, marine invertebrates and fish:
331 Method development and microplastic accumulation. *Mar. Pollut. Bull.* 122, 403–408.
332 <https://doi.org/10.1016/j.marpolbul.2017.06.081>

333 Kassambara, A., 2017. Ggpubr: ‘Ggplot2’ based publication ready plots. [https://doi.org/
334 https://CRAN.R-project.org/package=ggpubr](https://doi.org/https://CRAN.R-project.org/package=ggpubr)

335 Kedzierski, M., Le Tilly, V., Bourseau, P., Bellegou, H., César, G., Sire, O., Bruzaud, S., 2017a.
336 Microplastics elutriation system. Part A: Numerical modeling. *Mar. Pollut. Bull.* 119, 151–
337 161. <https://doi.org/10.1016/j.marpolbul.2017.04.060>

338 Kedzierski, M., Le Tilly, V., Bourseau, P., Bellegou, H., César, G., Sire, O., Bruzaud, S., 2016.
339 Microplastics elutriation from sandy sediments: A granulometric approach. *Mar. Pollut. Bull.*
340 107, 315–323. <https://doi.org/10.1016/j.marpolbul.2016.03.041>

341 Kedzierski, M., Le Tilly, V., Bourseau, P., César, G., Sire, O., Bruzaud, S., 2018. Microplastics
342 elutriation system: Part B: Insight of the next generation. *Mar. Pollut. Bull.* 133, 9–17.
343 <https://doi.org/10.1016/j.marpolbul.2018.05.011>

344 Kedzierski, M., Le Tilly, V., César, G., Sire, O., Bruzaud, S., 2017b. Efficient microplastics
345 extraction from sand. A cost effective methodology based on sodium iodide recycling. *Mar.*
346 *Pollut. Bull.* 115, 120–129. <https://doi.org/10.1016/j.marpolbul.2016.12.002>

347 Konechnaya, O., Luchtrath, S., Dsikowitzky, L., Schwarzbauer, J., 2020. Optimized microplastic
348 analysis based on size fractionation, density separation and μ -FTIR. *Water Sci. Technol.* 81,
349 834–844. <https://doi.org/10.2166/wst.2020.173>

350 La Nasa, J., Biale, G., Mattonai, M., Modugno, F., 2021. Microwave-assisted solvent extraction and
351 double-shot analytical pyrolysis for the quali-quantitation of plasticizers and microplastics in
352 beach sand samples. *J. Hazard. Mater.* 401, 123287.
353 <https://doi.org/10.1016/j.jhazmat.2020.123287>

354 Li, Q., Wu, J., Zhao, X., Gu, X., Ji, R., 2019. Separation and identification of microplastics from
355 soil and sewage sludge. *Environ. Pollut.* 254, 113076.
356 <https://doi.org/10.1016/j.envpol.2019.113076>

357 Mani, T., Frehland, S., Kalberer, A., Burkhardt-Holm, P., 2019. Using castor oil to separate
358 microplastics from four different environmental matrices. *Anal. Methods* 11, 1788–1794.
359 <https://doi.org/10.1039/C8AY02559B>

360 Nakajima, R., Tsuchiya, M., Lindsay, D.J., Kitahashi, T., Fujikura, K., Fukushima, T., 2019. A new
361 small device made of glass for separating microplastics from marine and freshwater sediments.
362 *PeerJ* 7, e7915. <https://doi.org/10.7717/peerj.7915>

363 Nuelle, M.-T., Dekiff, J.H., Remy, D., Fries, E., 2014. A new analytical approach for monitoring
364 microplastics in marine sediments. *Environ. Pollut.* 184, 161–169.
365 <https://doi.org/10.1016/j.envpol.2013.07.027>

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

369 R Core Team, 2018. R: A Language and Environment for Statistical Computing. R Foundation for
370 Statistical Computing, Vienna, Austria. <https://www.R-project.org/>

371 Radford, F., Zapata-Restrepo, L.M., Horton, A.A., Hudson, M.D., Shaw, P.J., Williams, I.D., 2021.
372 Developing a systematic method for extraction of microplastics in soils. *Anal. Methods*.
373 <https://doi.org/10.1039/D0AY02086A>

374 Rochman, C.M., 2015. The Complex Mixture, Fate and Toxicity of Chemicals Associated with
375 Plastic Debris in the Marine Environment, in: Bergmann, M., Gutow, L., Klages, M. (Eds.),
376 *Marine Anthropogenic Litter*. Springer International Publishing, Cham, pp. 117–140.
377 https://doi.org/10.1007/978-3-319-16510-3_5

378 Rodrigues, M.O., Gonçalves, A.M.M., Gonçalves, F.J.M., Nogueira, H., Marques, J.C., Abrantes, N.,
379 2018. Effectiveness of a methodology of microplastics isolation for environmental monitoring
380 in freshwater systems. *Ecol. Indic.* 89, 488–495. <https://doi.org/10.1016/j.ecolind.2018.02.038>

381 Scopetani, C., Chelazzi, D., Mikola, J., Leiniö, V., Heikkinen, R., Cincinelli, A., Pellinen, J., 2020.
382 Olive oil-based method for the extraction, quantification and identification of microplastics in
383 soil and compost samples. *Sci. Total Environ.* 733, 139338.
384 <https://doi.org/10.1016/j.scitotenv.2020.139338>

385 Stile, N., Raguso, C., Pedruzzi, A., Cetojevic, E., Lasagni, M., Sanchez-Vidal, A., Saliu, F., 2021.
386 Extraction of microplastic from marine sediments: A comparison between pressurized solvent
387 extraction and density separation. *Mar. Pollut. Bull.* 168, 112436.
388 <https://doi.org/10.1016/j.marpolbul.2021.112436>

389 Teuten, E.L., Saquing, J.M., Knappe, D.R.U., Barlaz, M.A., Jonsson, S., Björn, A., Rowland, S.J.,
390 Thompson, R.C., Galloway, T.S., Yamashita, R., Ochi, D., Watanuki, Y., Moore, C., Viet, P.H.,
391 Tana, T.S., Prudente, M., Boonyatumanond, R., Zakaria, M.P., Akkhavong, K., Ogata, Y.,
392 Hirai, H., Iwasa, S., Mizukawa, K., Hagino, Y., Imamura, A., Saha, M., Takada, H., 2009.
393 Transport and release of chemicals from plastics to the environment and to wildlife. *Philos.*
394 *Trans. R. Soc. B Biol. Sci.* 364, 2027–2045. <https://doi.org/10.1098/rstb.2008.0284>

395 Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D.,
396 Russell, A.E., 2004. Lost at Sea: Where Is All the Plastic? *Science* 304, 838–838.
397 <https://doi.org/10.1126/science.1094559>

398 Tötze, C., Oswald, S.E., Hilger, A., Kardjilov, N., 2021. Non-invasive detection and localization of
399 microplastic particles in a sandy sediment by complementary neutron and X-ray tomography.
400 *J. Soils Sediments*. <https://doi.org/10.1007/s11368-021-02882-6>

401 Wang, Z., Taylor, S.E., Sharma, P., Flury, M., 2018. Poor extraction efficiencies of polystyrene
402 nano- and microplastics from biosolids and soil. *Plos One* 13, e0208009.
403 <https://doi.org/10.1371/journal.pone.0208009>

404 Wen, D., Chen, Y., Tong, Y., Wang, H., Zhang, H., Luo, Y., 2021. Quantification of Microplastics in
405 Soils Using Accelerated Solvent Extraction: Comparison with a Visual Sorting Method. *Bull.*
406 *Environ. Contam. Toxicol.* <https://doi.org/10.1007/s00128-021-03200-z>
407 Wickham, H., François, R., Henry, L., Müller, K., 2017. dplyr: a grammar of data ma-nipulation.
408 <https://CRAN.R-project.org/package=dplyr>
409 Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on
410 marine organisms: A review. *Environ. Pollut.* 178, 483–492.
411 <https://doi.org/10.1016/j.envpol.2013.02.031>
412 Zhang, G.S., Liu, Y.F., 2018. The distribution of microplastics in soil aggregate fractions in
413 southwestern China. *Sci. Total Environ.* 642, 12–20.
414 <https://doi.org/10.1016/j.scitotenv.2018.06.004>
415 Zhang, S., Yang, X., Gertsen, H., Peters, P., Salanki, T., Geissen, V., 2018. A simple method for the
416 extraction and identification of light density microplastics from soil. *Sci. Total Environ.* 616,
417 1056–1065. <https://doi.org/10.1016/j.scitotenv.2017.10.213>
418 Zhang, X., Yu, K., Zhang, H., Liu, Y., He, J., Liu, X., Jiang, J., 2020. A novel heating-assisted
419 density separation method for extracting microplastics from sediments. *Chemosphere* 256,
420 127039. <https://doi.org/10.1016/j.chemosphere.2020.127039>
421

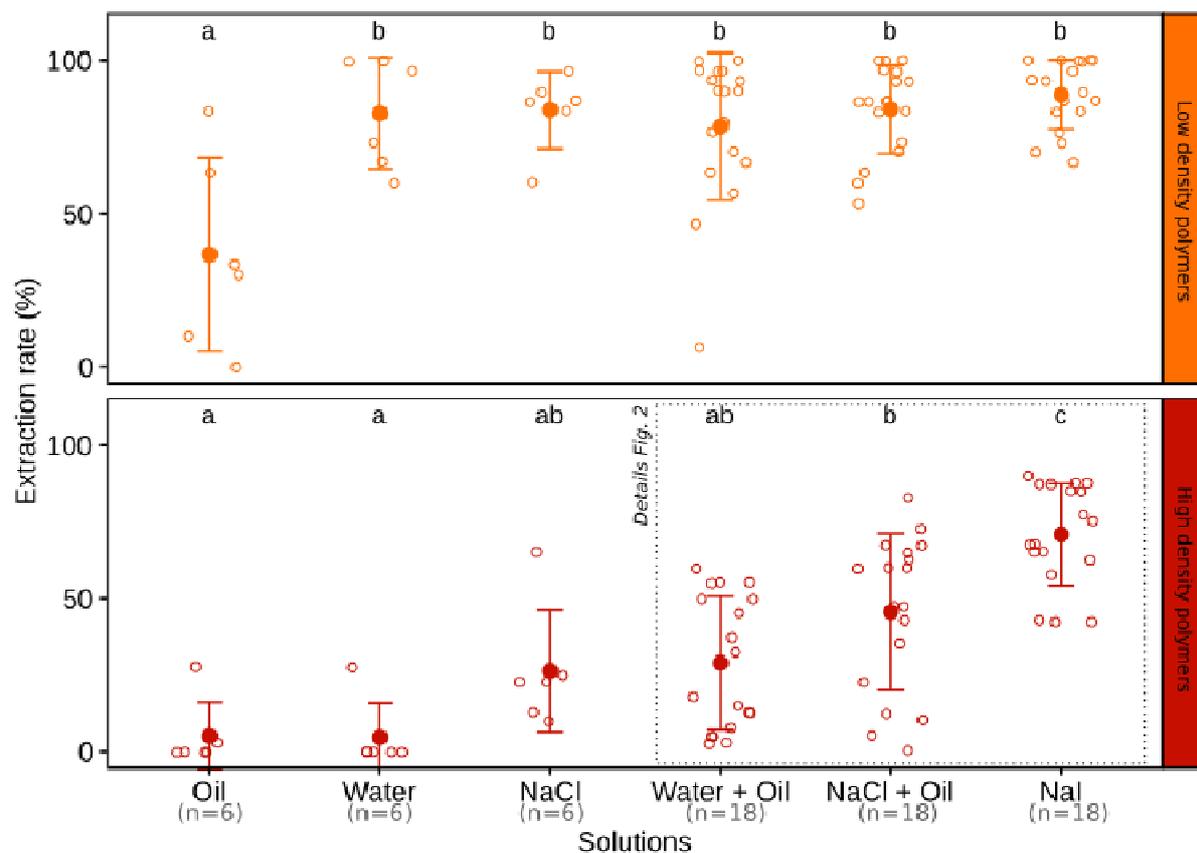


Fig. 1. Extraction rates (%) of low (orange) and high (red) density polymers for each extracting solutions. Small empty circles show results of each tests (i.e. 3 agitation methods \times 2 sediment type \times 1 or 3 replicates). Large solid circles and bars represent means and standard deviations, respectively. Compact letters (a, b and c) indicate groups with significant differences. Extraction rate for each 7 types of spiked MPs were pooled by polymer density. Low density: PE and PE-ABS. High density: PET, PA and PES. See details about statistical tests in section 2.3.

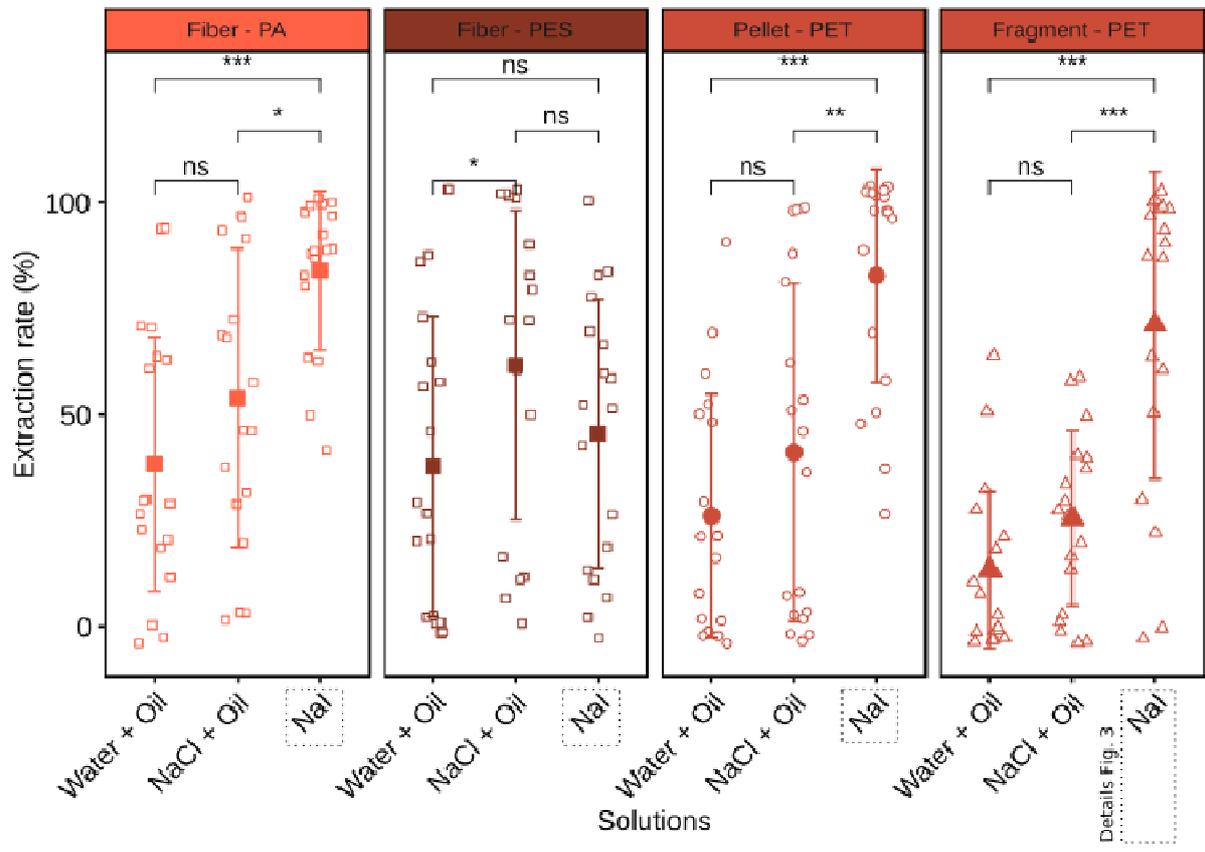


Fig. 2. Extraction rates (%) of each high density polymers for three extracting solutions. Small empty circles, triangles and squares show results for 10 pellets, fragments and fibers, respectively. Large solid circles and bars represent means and standard deviations, respectively. ns: Non-significant (p-value > 0.05). *: Significant (p-value < 0.05). **: Highly significant (p-value < 0.01). ***: Very highly significant (p-value < 0.001). See details about statistical tests in section 2.3.

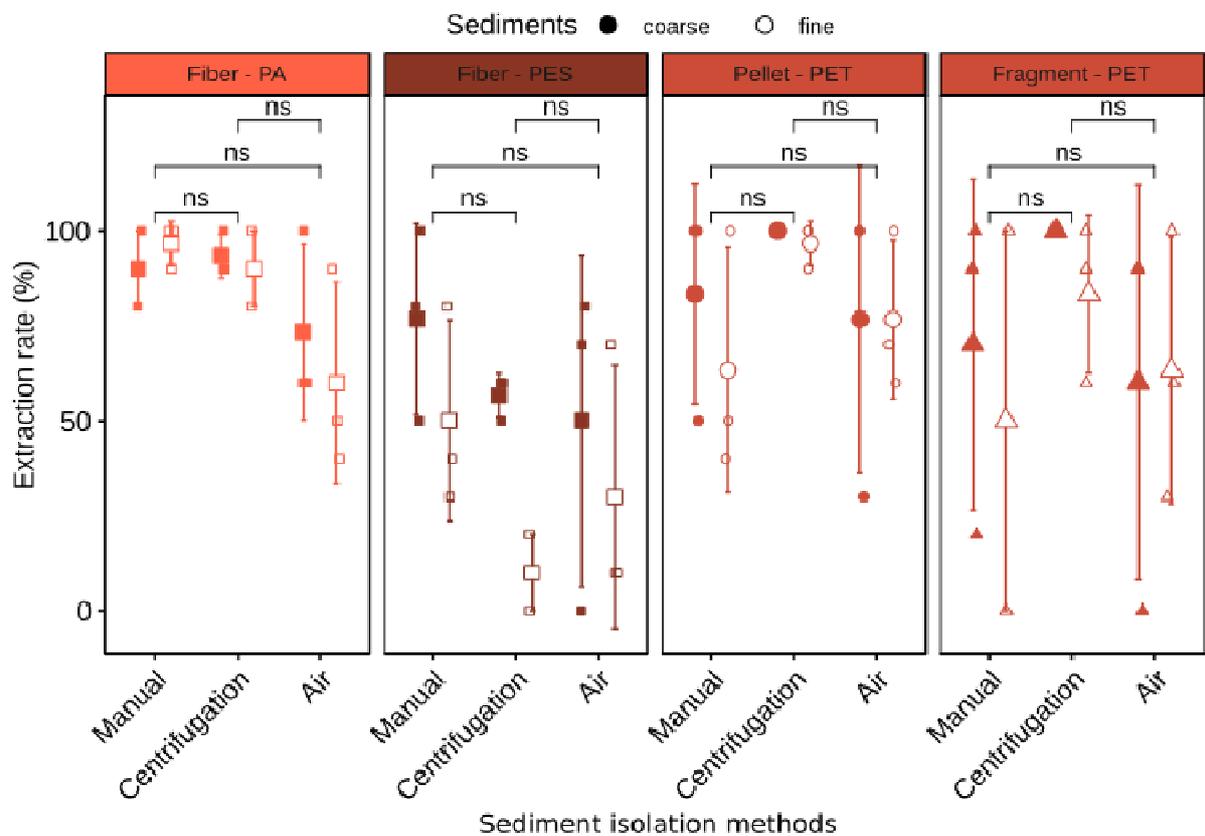


Fig. 3. Extraction rates (%) of each high density polymers obtained with the NaI solution for each methods and both sediment types. Small circles, triangles and squares show results for 10 pellets, fragments and fibers, respectively. Large icons and bars represent means and standard deviations, respectively. Coarse and fine sediments were pooled for statistical tests (*i.e.* 6 replicates per group). ns: Non-significant (p-value > 0.05). *: Significant (p-value < 0.05). **: Highly significant (p-value < 0.01). ***: Very highly significant (p-value < 0.001). See details about statistical tests in section 2.3

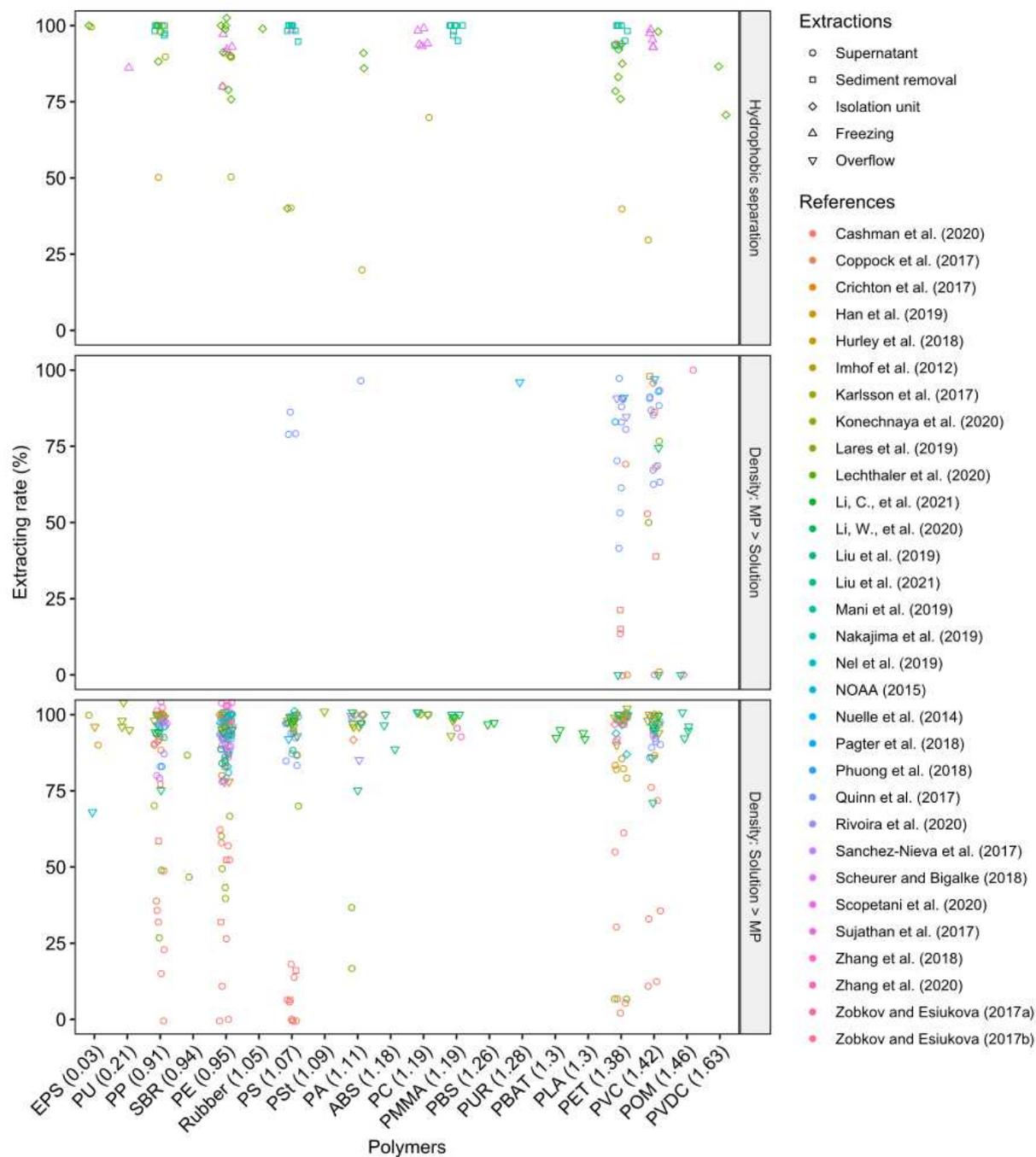


Fig. 4. Extraction rates (%) retrieved from 30 studies. Extractions: methods to collect MPs after being isolated from sediment.

Table 1. Procedures performed for extracting spiked microplastics in fine and coarse sediments. 18 combinations with 3 stirring methods and 6 solutions were tested with 7 types of pristine microplastics. Each combination was tested for coarse and fine sediments (2), tests for 3 solutions (Oil in water and in NaCl, and NaI) were repeated 3 times ($\times 3$).

Solution	Manual	Centrifugation	Air	Total
Oil	2	2	2	6
Water	2	2	2	6
Water + Oil	2×3	2×3	2×3	18
NaCl	2	2	2	6
NaCl + Oil	2×3	2×3	2×3	18
NaI	2×3	2×3	2×3	18
Total	24	24	24	72

Materials



Sediments

+



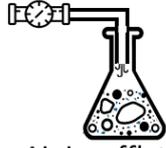
Microplastics

+

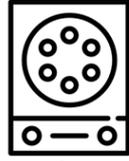


Extracting solutions

Separation methods



Air insufflation



Centrifugation



Hand-stirring

Extraction rate (%)

