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1 **Nanoplastic occurrence in a soil amended with plastic debris**

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4

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9

10 **ABSTRACT:** While several studies have investigated the potential impact of nanoplastics,
11 proof of their occurrence in our global environment has not yet been demonstrated. In the
12 present work, by developing an innovative analytical strategy, the presence of nanoplastics in
13 soil was identified for the first time. Our results demonstrate the presence of nanoplastics with
14 a size ranging from 20 to 150 nm and covering three of the most common plastic families:
15 polyethylene, polystyrene and polyvinyl chloride. Given the amount of organic matter in the
16 soil matrix, the discrimination and identification of large nanoplastic aggregates are
17 challenging. However, we provided an innovative methodology to circumvent the organic
18 matter impact on nanoplastic detection by coupling size fractionation to molecular analysis of
19 plastics. While photodegradation has been considered the principal formation pathway of
20 nanoplastics in the environment, this study provides evidence, for the first time, that plastic
21 degradation and nanoplastic production can, however, occur in the soil matrix. Moreover, by
22 providing an innovative and simple extraction/analysis method, this study paves the way to
23 further studies, notably regarding nanoplastic environmental fate and impacts.

24 **Key Words:** Nanoplastics, soil, municipal wastes, Py-GCMS, AF4

25

26 **1. Introduction**

27 Despite a consensus that plastic debris will accumulate in the environment, its
28 presence has drastically increased, reaching almost 360 million tons in 2018 ([PlasticsEurope](#)
29 [2019](#)). Plastic pollution is now considered one of the main environmental challenges and
30 represents an emerging threat for all living species, especially due to the chemical species
31 (additives and other adsorbed elements) that are associated with plastics ([Velzeboer, Kwadijk,](#)
32 [et Koelmans 2014](#); [Alimi et al. 2018](#); [Davranche et al. 2019](#); [Shen et al. 2019](#)). Leading
33 governments have even classified plastic pollution as a critical problem comparable to climate
34 change ([Programme 2016](#)).

35 In marine systems, despite knowledge gaps regarding the occurrence and fate of
36 microplastics, the recent discovery of nanoplastics ([Ter Halle et al. 2017](#)) has opened the door
37 to new considerations in terms of their properties and environmental fate and behavior.
38 Nanoplastics are defined as plastic particles with sizes covering the colloidal range (1 nm to 1
39 μm) and with a Brownian motion in aqueous system ([Gigault et al. 2018](#)). By contrast,
40 compared to marine systems, terrestrial systems are largely ignored, as noted by [Rillig \(2012\)](#).

41 However, plastic-based materials such as plastic mulches and polytunnels are widely
42 used to increase agricultural production efficiency or life duration ([Steinmetz et al. 2016](#); [Liu](#)
43 [et al. 2018](#); [Gao et al. 2019](#)). Sewage sludge products also contribute to the incorporation of
44 microplastics into soils ([Habib, Locke, et Cannone 1998](#); [Corradini et al. 2019](#)). Both inputs,
45 together with many others ([Hurley et Nizzetto 2018](#)), such as landfills ([He et al. 2019](#)), lead to
46 the accumulation of plastics in soils, which could represent an even larger pollution pool than
47 that in the oceans ([Horton et al. 2017](#)). Once plastic debris is present in soil, it mixes and

48 reacts with soil organic matter (OM) and minerals and may persist for up to a few hundred
49 years (Bläsing et Amelung 2018). The prolonged residence time of plastics in soil could lead
50 to the formation of micro- and nanodebris. The formation of such debris represents a severe
51 challenge for environmental protection, especially for long-term farming based on plastics
52 utilization (Steinmetz et al. 2016). Some recent studies have started investigating nanoplastic
53 impacts on soil biota (Zhu et al. 2018; Awet et al. 2018), their interactions with contaminants
54 (Velzeboer, Kwadijk, et Koelmans 2014; Davranche et al. 2019) and their transport through
55 model soils (Hu et al. 2020; Pradel et al. 2020). However, to the best of our knowledge,
56 evidence of nanoplastics in a natural soil environment has never been demonstrated. This lack
57 is clearly due to the need for appropriate analytical methodologies for the identification of
58 nanoplastics in the complex and heterogeneous soil matrix (Pinto da Costa et al. 2019). The
59 size, shape, concentration (at the trace level) and carbon composition of nanoplastics make
60 their identification and quantification highly challenging, especially due to the amount of
61 natural OM, which is also composed of carbon and covers the entire colloidal size range.

62 Due to their small size and high surface reactivity, nanoplastics could exert a direct
63 and indirect effect on soil ecosystems, including groundwaters. They could indirectly
64 influence the soil physicochemical properties, whereas direct effects were already reported on
65 soil microorganisms and fauna communities (Zhu et al. 2018; Awet et al. 2018). Since soils
66 and groundwaters are major human resources, evaluating the occurrence, fate and impact of
67 nanoplastics in soils is crucial and is a main political and governmental priority, leading to
68 legislation such as the European Commission directive on plastics on soil regulation.

69 Regarding the difficulties to extract and detect nanoplastics in a soil matrix, our
70 objective is not to realize an exhaustive study of nanoplastic presence in various kind of soils,
71 but to demonstrate that nanoplastics are present and can be produced directly in the soil
72 matrix. The challenge was here to find a soil sufficiently contaminated by plastics and for

73 enough years to allow the potential production of nanoplastics. For this purpose, we chosen to
74 work on a soil contaminated by plastic debris through several municipal waste amendments
75 performed around fifteen to thirty years ago. Nanoplastics were extracted using a water
76 extraction procedure and subsequently identified by an innovative high-resolution analytical
77 method combining size fractionation and molecular analysis.

78 **2. Methods**

79 *2.1. Soil sampling and geochemical analysis.*

80 The soil samples are agricultural soils collected in central France in February 2018.
81 They correspond to the upper most horizon (0 to 10 cm, organo-mineral (Ah) horizon) of a
82 well-drained cambisol (VWR classification), enriched in pebbles and developed from alluvial
83 deposits. The contaminated soil received two household wastes amendments mostly
84 composed of plastics around thirty and fifteen years ago. The wastes were crushed and mixed
85 with organic compost that was spread on the soil. The field was regularly tilled and used as a
86 meadow for the last ten years. Plastic debris is thus clearly visible on and in the soil horizons
87 from the surface to 40 cm depth. Plastic crushing has probably promoted the plastics
88 degradation in the soil (Ng et al. 2018). Regarding the contamination origin, treatment and
89 dates, this soil is therefore the ideal target for studying the potential nanoplastic production in
90 a soil matrix.

91 A soil control was sampled from a non-amended plot close to the contaminated plot. The
92 same extraction protocol and characterization methods were used for the amended and control
93 soil samples. Soil samples were dried in ambient air, sieved at 2 mm and stored in ambient air
94 in the dark. To ensure no plastic pollution, both soils were similarly stored and treated. The
95 geochemical composition of both soil samples is summarized in the Supplementary Materials,
96 [Table S1](#). Geochemical analyses were performed at the “Service d’Analyse des Roches et des
97 Minéraux” (SARM). The major element concentrations were determined by inductively

98 coupled plasma-atomic emission spectrometry (ICP-AES, Thermo ICap 6500). Trace element
99 concentrations were determined by ICP-mass spectrometry (Thermo Elemental X7). The
100 samples were digested by alkaline fusion using lithium metaborate (LiBO_2) as a fusion flux.

101

102 *2.2. Nanoplastic water extraction experiments.*

103 To extract nanoparticles from the soil samples, soil-water extraction was performed.
104 Ultrapure water was added to soil at a soil/water ratio of 1:4 (200 g/800 g) and stirred at 300 r
105 min^{-1} for 72 h without any pH adjustment. Soil solutions were sampled over time. After pH
106 and conductivity measurements, each sample was filtered to 0.8 μm (Sartorius filters). Total
107 organic carbon (TOC) was determined using a TOC analyzer (Shimadzu TOC-V CSH). The
108 accuracy of the TOC measurements was estimated at $\pm 5\%$ for all samples using a standard
109 solution of potassium hydrogen phthalate. The extraction experiments were performed in
110 duplicate.

111

112 *2.3. Nanoplastics identification and characterization.*

113 Water-extract filtrates were fractionated by asymmetric flow-field flow fractionation
114 coupled to UV spectroscopy and static light scattering (AF4-UV-SLS, Wyatt Technology,
115 Germany, and Agilent Technologies, France). The AF4 channel thickness was fixed by a 250-
116 μm Mylar film. The channel had a length of 26.5 cm and a width that narrowed from 2.1 to
117 0.6 cm. The accumulation wall was defined by a 10-kDa PES membrane purchased from
118 Wyatt Technology. The method used was described by [Gigault et al. \(2017\)](#). The detailed
119 parameters are presented in the Supplementary Materials. Three hundred microliters of each
120 sample were injected into the AF4 instrument. The sizes (gyration radius, R_g) obtained from
121 SLS were determined by the first-order Berry model using ASTRA-6 software. The AF4

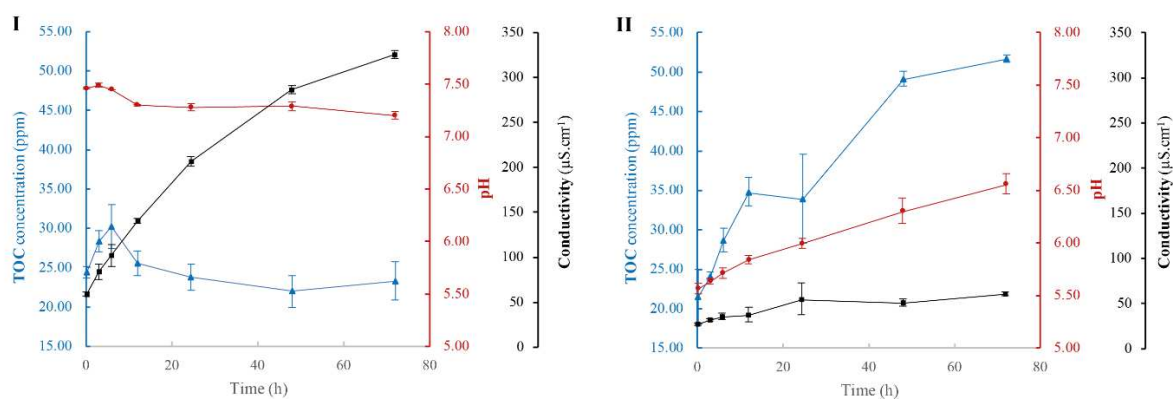
122 instrument was calibrated (Supplementary Materials, [Fig. S1](#)) using polystyrene spherical
123 models (PSL, NIST traceable standard) to convert the elution time for a sphere with the
124 equivalent hydrodynamic diameter (d_{zH}). Both the AF4 extracts and $< 0.8 \mu\text{m}$ water extract
125 fractions were analyzed by pyrolysis (PY-3030D - Frontier Lab, Japan) coupled to gas
126 chromatography and mass spectrometry (Py-GCMS). For this experiment, $40 \mu\text{L}$ of sample
127 was introduced into an analysis cup and placed on a heating table at 40°C until complete
128 evaporation of the solvent. This step was repeated one time to concentrate the samples before
129 injection. The method followed for Py-GC/MS was described by [Dehaut et al. \(2016\)](#), and the
130 corresponding settings are detailed in the Supplementary Material. A blank control was
131 performed to ensure that none of the instruments or materials contaminated the samples.

132 **3. Results and Discussion**

133 *3.1. Extraction and aggregation of the nanoparticle fraction.*

134 During soil-water extraction, pH, conductivity and TOC were monitored in the $<0.8 \mu\text{m}$
135 fraction of the soil water-extracts ([Fig. 1](#)). For the contaminated soil ([Fig. 1.I](#)), in the first 6 h,
136 TOC increased from 24 to 30 mg L^{-1} and then decreased to reach a pseudo equilibrium from
137 24 h with an average concentration of $23.7 \pm 1.8 \text{ mg L}^{-1}$. The pH remained stable (7.35 ± 0.2),
138 while the conductivity continuously increased along the experiment, from 59.0 to $325.7 \mu\text{S}$
139 cm^{-1} . The TOC variations could be explained by an aggregation. Aggregates were formed
140 with the running time experiments and were removed by the filtration at $0.8 \mu\text{m}$, leading to
141 the TOC decrease. This hypothesis is also supported by the conductivity increase, which
142 could be responsible of the colloids material aggregation. By contrast, the TOC, pH and
143 conductivity of the control soil all increased along the experiment ([Figure 1.II](#)) but with
144 values much lower than for the contaminated soil. The control soil did not reach equilibrium
145 during the experiment. Moreover, the control soil has lower concentrations of CaO and CO_2
146 total than the contaminated soil ([Table S1](#), Supplementary Materials). The contaminated soil

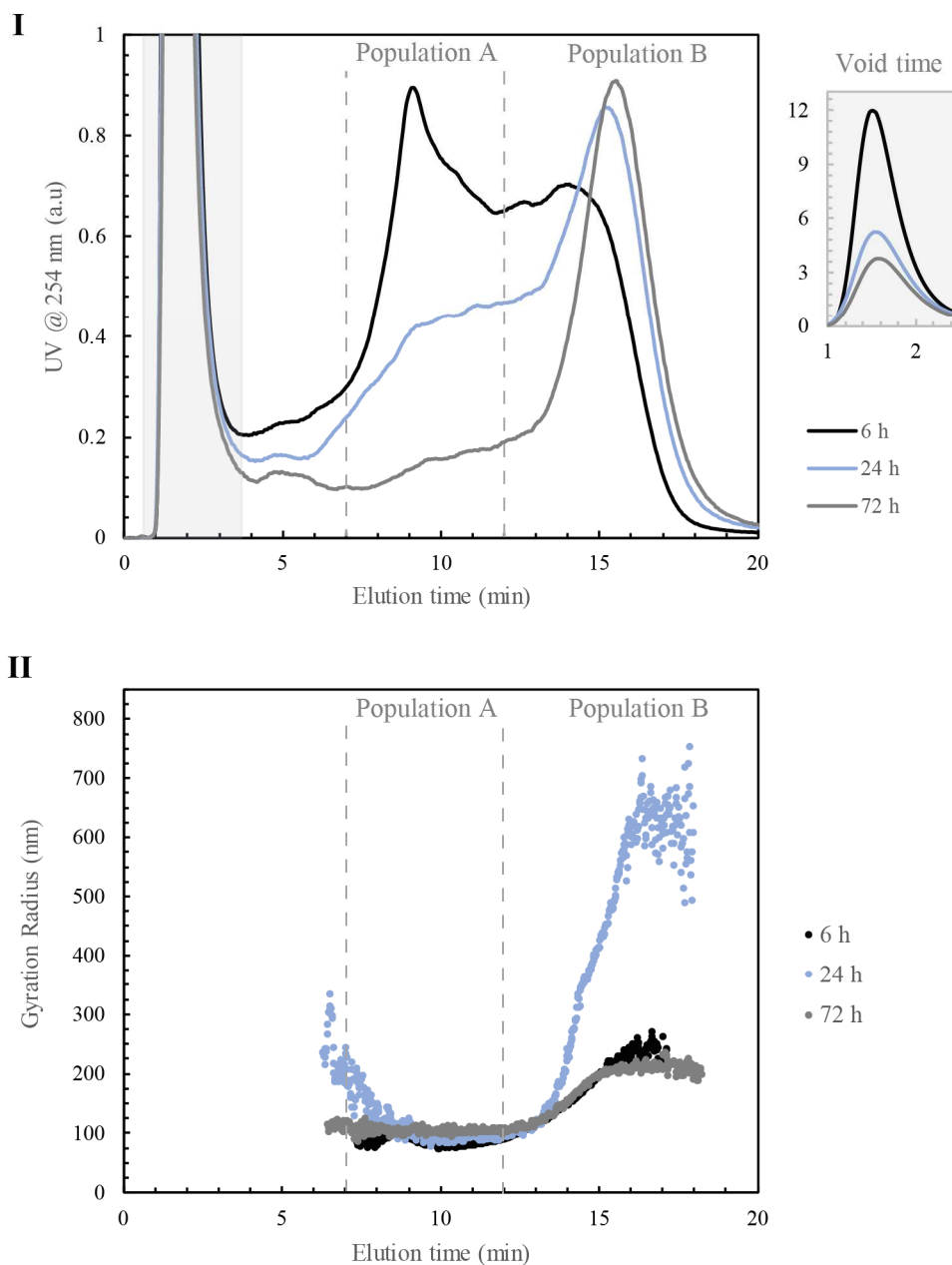
147 is thus enriched in carbonates as compared to the control sample. Such carbonates are
 148 dissolved during the water-extraction which buffer the pH and involve the conductivity
 149 increase. This carbonates enrichment could be the result of agricultural practices such as
 150 liming but also of the presence of weathered plastics debris. [Hahladakis et al. \(2018\)](#) showed
 151 that plastics can have carbonate-based fillers representing until 50% of the total plastic
 152 weight.



153
 154 **Fig. 1:** Evolution of TOC, pH, and conductivity with extraction time. **I:** contaminated soil, **II:** soil control. The
 155 error bars correspond to the standard deviation calculated from duplicates.

156 To verify such hypothesis and better understand the behaviors in presence, the soil
 157 water extracts were characterized by AF4-UV-SLS. Regarding the TOC evolution, only 3
 158 samples were collected and analyzed at 3 extraction times (6 h, 24 h and 72 h). [Fig. 2](#) presents
 159 the AF4-UV-SLS analysis results at different sampling times for the <0.8 µm solutions.
 160 According to the UV results, 3 populations (void time, A and B) were identified for all
 161 extraction times ([Fig. 2.I](#)). The void time population from 0 to 2.5 min corresponds to small
 162 particles (0 to 5 nm) that are generally not colloidal ([Chevalier et al. 2018](#)) and correspond to
 163 low-molecular-weight OM. They were not considered in the present study. Two colloidal
 164 populations were identified: **A** from 7 to 12 min and **B** from 12 to 19 min, which correspond
 165 to hydrodynamic sizes (d_{ZH}) of 20 to 150 nm and 150 nm to 500 nm, respectively (*eq. (1)*,
 166 [Supplementary Materials](#)). The variation in R_g (gyration radius), obtained by SLS, provides

167 information on the particle size. From 6 h to 24 h, while R_g did not vary for **A**, R_g increased
168 from 200 to 600 nm for **B** and finally decreased at 72 h to reach 200 nm, the initial value (Fig.
169 2.II). Such R_g variations indicate particle aggregation (Frimmel, Kammer, et Flemming 2007).
170 To characterize the nature of the particle aggregation, R_g was combined with the
171 hydrodynamic radius (r_H – obtained from AF4 elution time, *eq. (1)*) to determine the shape
172 factor S (i.e., R_g/r_H). While $S = 0.778$ corresponds to hard spherical particles, a deviation to 1
173 indicates the presence of inhomogeneities on the sphere structure (Brewer et Striegel 2011).
174 Here, $S \gg 1$ indicates the formation of polymorphic aggregates. Our results therefore show
175 the formation of large and polymorphic aggregates. This aggregation process can be
176 confirmed by the UV signal evolution with time. From 6 to 72 h, the UV signal of **A**
177 decreased, while it increased for **B**, indicating the aggregation and transfer of the **A**
178 population to **B**. Moreover, from 6 to 72 h, the maximum of the peaks in the **B** population is
179 shifted to the right, which means that the particles size increased. In addition, from 6 to 72h,
180 the total UV signal decreased for the same elution time interval (7 to 20 min). The total area
181 of the peaks was 367 at 6 h, 314 at 24 h and 213 at 72 h which correspond to the TOC
182 decrease and particles loss. Thus, after 24 h, large aggregates with sizes $> 0.8 \mu\text{m}$ were
183 formed, and the remaining particles in the $<0.8 \mu\text{m}$ fraction were smaller, as confirmed by the
184 R_g decrease.



185
 186 **Fig. 2: I:** Fractograms (UV trace at 254 nm) of the soil water extracts at 6 h, 24 h and 72 h from the
 187 contaminated soil with respect to elution time. **II:** Evolution of the gyration radius (R_g) with the elution time for
 188 the 3 soil water extracts.

189

190 *3.2. Are extracted nanoparticles nanoplastics?*

191 To discriminate the presence of anthropogenic nanoparticles, populations **A** and **B**
 192 identified by AF4 were collected and further analyzed by Py-GCMS. Analysis by Py-GCMS

193 of different plastic materials provides specific molecular markers, which are reported in
194 commercial databases (Tsuge, Ohtani, et Watanabe 2011; Supplementary Materials, table S2).
195 Fig. 3 presents the pyrograms obtained at different extraction times for A only. The most
196 abundant signal was obtained at 24 h (no signal at 72 h). At 6 h and 24 h, the global
197 pyrograms present a series of multiple peaks at constant time intervals that are characteristic
198 of polyethylene (PE) (Tsuge, Ohtani, et Watanabe 2011). By extracting the appropriate ion
199 masses (m/z , Supplementary Materials, table S2), specific plastic molecular markers were
200 identified at particular retention times (t_R), as follows:

- 201 - Naphthalene (m/z 128) at $t_R=11.3$ min and naphthalene-1-methyl (m/z 142) at $t_R=12.3$
202 min are characteristic of polyvinyl chloride (PVC; Fig. 3.II)
- 203 - Styrene monomer (m/z 104) at $t_R=8.5$ min and toluene ($t_R=7$ min) are markers of
204 polystyrene (PS; Fig. 3.III).
- 205 - Finally, as previously described, the pyrogram (Fig. 3.IV) shows the presence of PE
206 with a series of alkenes: 1-decene (C10), 1-undecene (C11), 1-dodecene (C12), 1-
207 tridecene (C13), 1-tetradecene (C14) and 1-pentadecene (C15).

208 Moreover, as demonstrated by Ter Halle et al. (2017), the presence of the triplet n-
209 alkadiene, n-alkene and n-alkane with a bimodal distribution indicates a typical molecular
210 formation pathway of PE during pyrolysis, rather than OM (Fig. 3.IV). Considering the Py-
211 GCMS results and by comparison with the control soil, nanoplastics are effectively present in
212 the amended soil. Indeed, for the control soil, 24 h of extraction time also allowed the most
213 important signals to be obtained, but these signals had typical markers that are characteristic
214 of natural organic matter (Supplementary Materials, Fig. S2). No specific markers of plastics
215 were determined in the control soil water extracts at any time.

216 For the contaminated soil, a first approximation gives a size ranging from 20 to 150
217 nm (A population) with heterogeneous shapes. Surprisingly, no signal was obtained for B

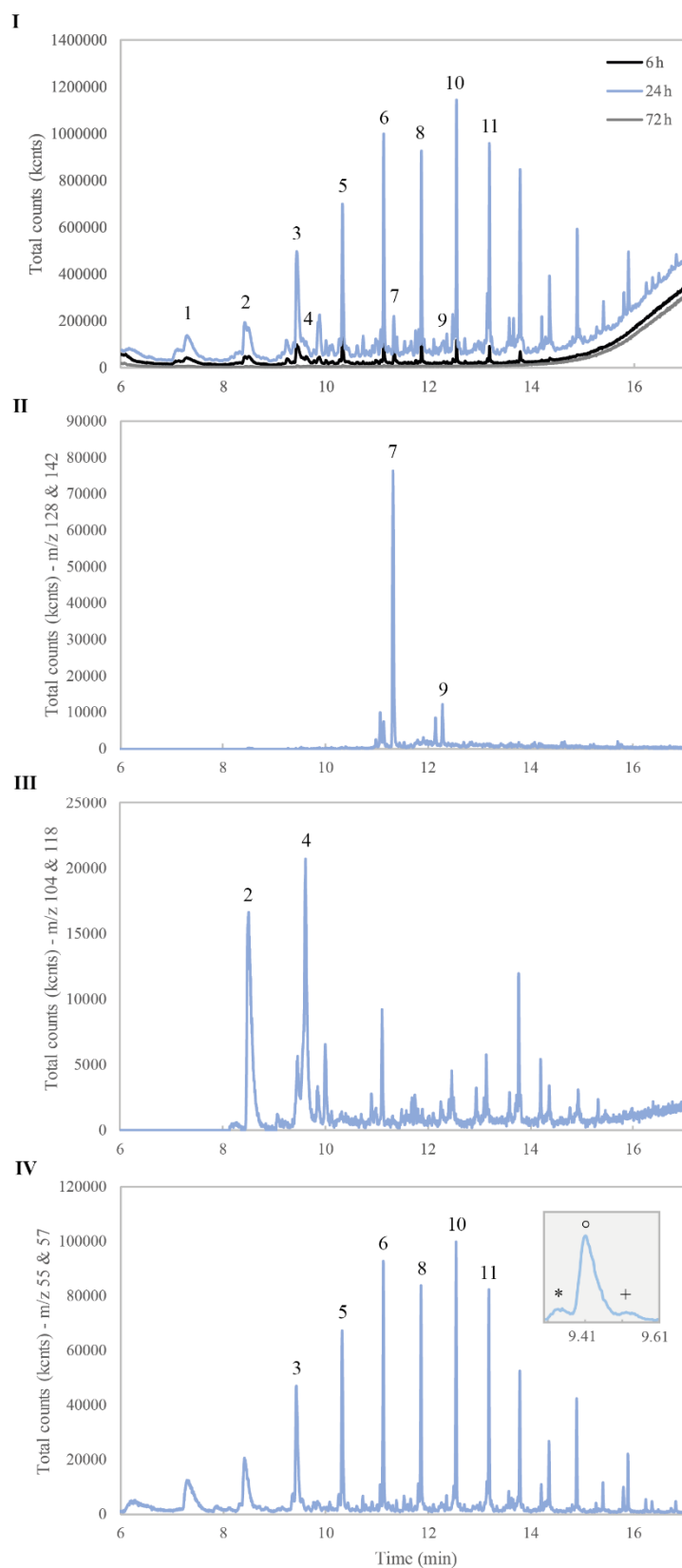
218 despite the aggregation suggested by the AF4 results and TOC evolution. Different
219 hypotheses could be made: (i) the nanoparticles identified in **A** aggregated to form
220 micrometric particles that were transferred to **B** while the nanoplastics remained stable; or (ii)
221 OM became predominant in **B** and interfered with the pyrolysis signal. Nanoplastics could be
222 stabilized by their bindings with organic macromolecules or present a different aggregation
223 behavior. However, the comparison of the water extracts with and without AF4 analysis and
224 fraction collection (Fig. 4) showed that the pyrolysis signals were enhanced after size
225 fractionation despite sample dilution. Therefore, AF4 can be used as a purification technique
226 to remove organic and dissolved species that may interfere with detection. The amplification
227 of the Py-GCMS signal of plastics after AF4 purification demonstrates that OM mitigates the
228 detection of plastics. A higher proportion of OM than plastics could hide the plastic signature
229 in the Py-GCMS results for the **B** population. This assumption explains the higher pyrolysis
230 signal at 24 h than at 6 h and the plastic disappearance at 72 h. To summarize, in the first 6 h,
231 OM was quickly released and aggregated after 24 h. In both cases, OM was present in a
232 higher proportion than plastics and interfered with the plastic pyrolysis signal until plastic
233 detection became impossible at 72 h. By contrast, at 24 h, OM had just begun to aggregate,
234 and its proportion was less than that of nanoplastics, enhancing the plastic PY-GCMS signal.

235 Finally, despite the increase in detection sensitivity achieved using A4F or other
236 colloidal purification techniques (ultracentrifugation and ultrafiltration), it is still challenging
237 to attribute specific markers to nanoplastics or OM. An interesting way to discriminate the
238 OM contribution from that of plastics is the toluene-to-styrene ratio (Tol/Sty). Styrene and
239 toluene are also produced during the pyrolysis of OM (Fabbri, Trombini, et Vassura 1998).
240 Tol/Sty ranges from 4 to 5 for substrates of OM origin, in contrast to values of 0.001 to 1 for
241 polystyrene plastics (Fabbri, Trombini, et Vassura 1998; Watteau et al. 2018). This difference
242 in ratios was confirmed by analyzing polystyrene nanoplastics and humic acid standards

243 (Dignac et al. 2005). Tol/Sty did not exceed 0.03 for the polystyrene standards, while 6.3 was
244 obtained for the humic acids. In the soil-water extracts, for A, Tol/Sty was 2.8. Recently,
245 Watteau et al. (2018) compared the Tol/Sty values of leachates from a soil amended with
246 municipal solid wastes. For the 0-2 μm fraction, potentially containing nanoplastics, Tol/Sty
247 was approximately 3 to 5 for their nonamended soil sample (Watteau et al. 2018). Despite this
248 difference in range from typical values, the authors claimed no plastic occurrence since
249 Tol/Sty was over 1. However, the confidence of the nanoplastics detection in our samples is
250 based on (i) the coherent polystyrene detection associated with PE and PVC nanoplastics
251 identification and (ii) the contribution of OM to plastics markers at the nanoscale (i.e., high
252 specific surface area). Note that in the literature, Tol/Sty ratios were determined from
253 microplastic debris with sizes generally $>500 \mu\text{m}$. Nevertheless, plastic debris (micro- and
254 millimetric) is generally coated by biofilms and OM with a thickness less than $0.4 \mu\text{m}$
255 (Besseling et al. 2017). Therefore, the mass proportion of OM on large debris is insignificant.
256 Nanoplastics (like other nanoparticles) are also associated with OM (surface sorption,
257 heteroaggregate formation, surfactant-type associations, etc.) (Hotze, Phenrat, et Lowry
258 2010). However, regarding the high specific surface area and similar size of OM and
259 nanoplastics, the OM mass contribution to core materials (i.e., plastics) is substantially larger
260 (Delay et al. 2011). Indeed, by decreasing the size from $200 \mu\text{m}$ to 200nm , the OM volume
261 on plastic increases by more than 4 000 times (Supplementary Materials, Fig. S3). This OM
262 distribution on nanoplastics leads to different Tol/Sty values from those of large
263 microplastics.

264 To summarize, in our experiment, nanoplastics are released into the soil solution
265 concomitantly with OM, which controls the stability and aggregation pathways of
266 nanoplastics (Hotze, Phenrat, et Lowry 2010). OM-nanoplastics associations are thus a key
267 factor in their detection in complex environmental matrices. This study demonstrated the

268 mandatory use of AF4 or other size-separation techniques for proper detection. The
269 development of an analytical strategy based on Py-GCMS is therefore needed (pyrolysis
270 temperature optimization, ramp program, etc.). Preliminary sample treatment (UV or H₂O₂) to
271 separate nanoplastics from OM is also of major concern in such analytical strategies. Finally,
272 this study also rises the question of plastics impacts on soil chemical properties. The
273 occurrence of plastic debris seems indeed to involve different geochemical behaviors of both
274 studied soils.



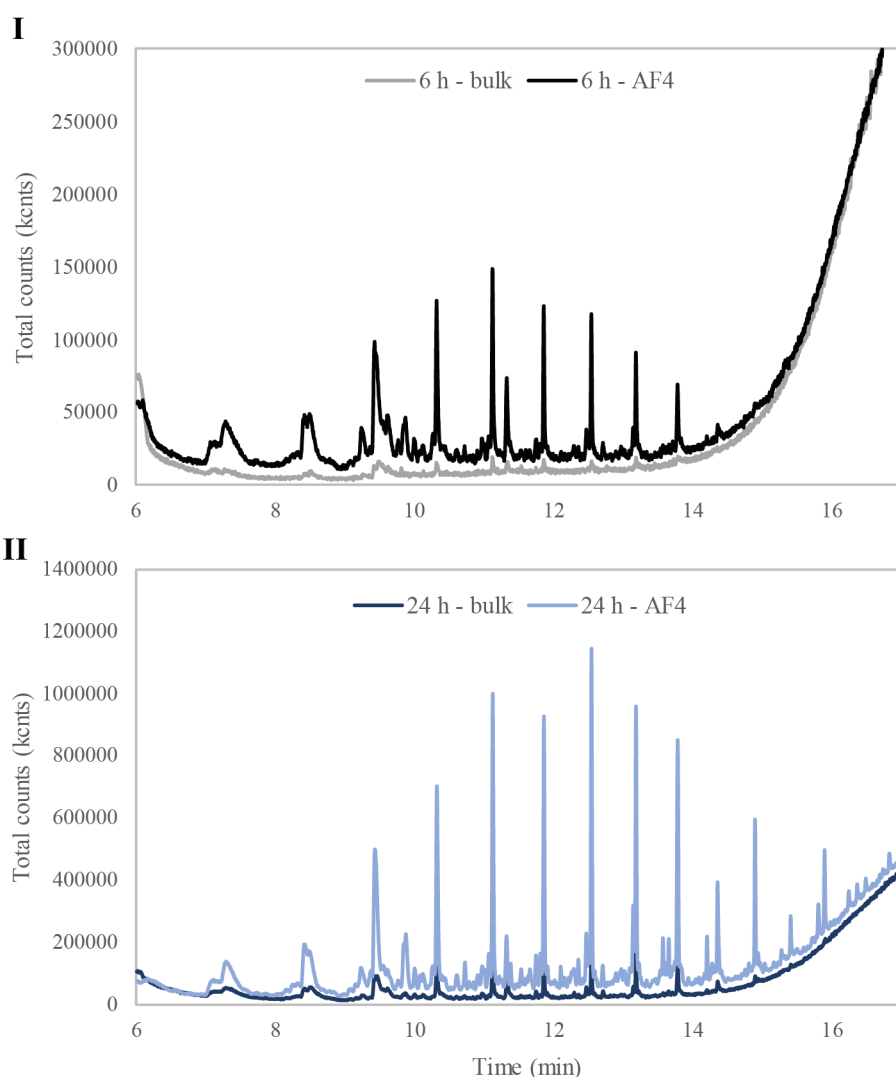
275

276 **Fig. 3: I.** Pyrograms of the 3 soil water extracts at 6, 24 and 72 h. Numbers correspond to specific markers of

277 plastic: 1: toluene, 2: styrene, 3: 1-decene, 4: a-methyl styrene, 5: 1-undecene, 6: 1-dodecene, 7: naphthalene, 8:

278 1-tridecene, 9: naphthalene 1-methyl, 10: 1-tetradecene, 11: 1-pentadecene. **II, III** and **IV**: Ion chromatograms at

279 24 h for $m/z=128$ and 142 , $m/z=104$ and 118 , and $m/z=55$ and 57 , respectively. Finally, the box in the top right
280 corner in **IV** details the triplet n-alkadiene *, n-alkene ° and n-alkane †.



281
282 **Fig. 4:** Comparison between bulk solutions and solutions fractionated by AF4. **I:** 6 h, **II:** 24 h.

283 3.3. Environmental implications.

284 In addition to the first proof of nanoplastic presence in a soil contaminated by plastic
285 debris, this study demonstrates that such debris is formed from larger pieces of plastic directly
286 in the soil matrix. While the main degradation pathways of plastics in ocean and surface
287 waters are mechanical abrasion and photo-thermo-oxidative degradation (Andrady 2015),
288 these mechanisms only occur in the first centimeters of the soil. Unfortunately, experimental
289 data on plastic degradation in soils are limited and rather scattered, as is knowledge of the

290 parameters that control these degradation processes (Scalenghe 2018). To elucidate this
291 uncertainty concerning nanoplastic formation pathways, the degradation of carrier plastic bags
292 in soil, open-air and marine environments was recently compared over a 3-year period
293 (Napper et Thompson 2019). After 3 years, plastic bags were still functional after exposure to
294 soil and marine environments but had degraded into microplastics under open-air conditions.
295 Therefore, degradation is slower in soil and marine environments than under open-air
296 conditions. Our results suggest that in a 30-year time frame, plastic degradation in soil occurs
297 to produce nanoplastics, which is similar to the degradation time observed in oceanic systems.
298 This degradation pathway in soil thus suggests the implication of mechanisms other than
299 photo-thermo-oxidative processes. Scalenghe (2018) reported that plastics degradation in soil
300 could be enhanced by microorganisms.

301 Nanoplastics production in soil also raises the question of their environmental fate and
302 their final impact on living organisms. The nanoscale of nanoplastics allows them to pass
303 through the cell membranes (Bouwmeester, Hollman, et Peters 2015). As soils support our
304 food sources, there is an urgent need to investigate the fate of nanoplastics: Are they taken up
305 by plants? Do they accumulate in the food chain? Are they transferred to surface and
306 groundwater? Their ability to cotransport contaminants (Velzeboer, Kwadijk, et Koelmans
307 2014; Davranche et al. 2019) as well as to release additives in the environment (Shen et al.
308 2019) could also impact water quality. These issues represent considerable societal and
309 economic impacts that need to be urgently characterized to better anticipate them.

310 To conclude, it is important to note that the aim of the present study was not to provide
311 an exhaustive study of the nanoplastic occurrence in various soil families and pedo-climatic
312 conditions. Regarding that no study has already provided evidence of the potential nanoplastic
313 production and presence in natural soil under environmental conditions, we chose to study a
314 soil sample contaminated in plastics during a sufficient timeframe to be able to prove that

315 nanoplastics can be produced directly in a soil matrix. For this, we developed an innovative
316 and single extraction/analytical procedure allowing to detect nanoplastics even with a high
317 amount of OM. Moreover, the present results suggest that nanoplastic production in soil
318 seems to be rather slow, 30 years maximum, since photo-degradation is only limited to the
319 first cm of the uppermost soil horizon.

320 **4. Supplementary Materials**

321 Table of the elementary composition of the top soil of the contaminated and the
322 uncontaminated soil (Table S1), detailed methods, A4F calibration (Fig. S1), plastic markers
323 for the Py-GCMS detection (Table S2), pyrograms of the soil water extracts from the
324 uncontaminated soil (Fig. S2), comparison between OM volume and plastic debris surface
325 (Fig. S3).

326 **5. Acknowledgments**

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329 coordinated by Julien Gigault.

330

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