

1 **Microplastics in sub-surface waters of the Arctic Central Basin**

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11

12 **Abstract**

13 Polar oceans, though remote in location, are not immune to the accumulation of plastic debris. The
14 present study, investigated for the first time, the abundance, distribution and composition of
15 microplastics in sub-surface waters of the Arctic Central Basin. Microplastic sampling was carried
16 out using the bow water system of icebreaker Oden (single depth: 8.5 m) and CTD rosette sampler
17 (multiple depths: 8 – 4369 m). Potential microplastics were isolated and analyzed using Fourier
18 Transform Infrared Spectroscopy (FT-IR). Bow water sampling revealed that the median
19 microplastic abundance in near surface waters of the Polar Mixed Layer (PML) was 0.7 particles
20 m⁻³. Regarding the vertical distribution of microplastics in the ACB, microplastic abundance
21 (particles m⁻³) in the different water masses was as follows: Polar Mixed Layer (0 - 375) > Deep
22 and bottom waters (0 – 104) > Atlantic water (0 – 95) > Halocline i.e. Atlantic or Pacific (0 – 83).

23

24 **Keywords**

25 Microplastic, Marine debris, Arctic Ocean, Sub-surface waters, Pollution, Water column

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29

30 **Introduction**

31 The Arctic Ocean, though the smallest in the world, is unique due to its distinct abiotic features
32 and the highly specialised ecosystem it supports. Key anthropogenic drivers which may put
33 pressure on this ecosystem include (i) climate change, (ii) harvest and fisheries, (iii) persistent,
34 bio-accumulative and toxic contaminants, (iv) industrial development, (v) shipping, and (vi)
35 invasive alien species (CAFF 2017). Plastic contaminants in the world's oceans have emerged as
36 an issue of global importance due to their ubiquitous distribution, long-range transport potential,
37 persistence and perhaps most importantly the potential threat they pose to marine organisms
38 (UNEP 2011). Remote polar oceans such as the Arctic Ocean have not been immune to the entry
39 of plastics as a combination of long-range transport processes and local anthropogenic activities
40 have contributed to the plastic debris in these areas.

41

42 Characteristic abiotic features which set the Arctic Ocean apart from other oceanic basins include
43 (i) a central area of perennial pack ice, (ii) seasonal extremes in solar irradiance, ice and snow
44 cover, temperature and riverine inflow, and (iii) an upper layer of lower salinity water due to
45 freshwater input from rivers and seasonal sea-ice melt (CAFF 2013). This unique ecosystem is a
46 habitat for a vast array of marine organisms, some of which are (i) endemic to the region, (ii)

47 commercially important, (iii) apex predators, (iv) central to the functioning of the ecosystem, and
48 (v) threatened as evidenced by their inclusion in the IUCN Red List of Threatened Species (CAFF
49 2013, CAFF 2017).

50

51 Despite its remote location away from major population centres and the low coastal population in
52 its surrounding shelf areas, both macro and microplastics were detected in the various
53 environmental compartments of the Arctic Ocean. Between 2002 and 2014, macroplastics were
54 detected on the seafloor (2500 m depth) of the eastern Fram Strait at the HAUSGARTEN
55 observatory (Bergmann and Klages 2012; Tekman et al. 2017). Sightings of buoyant macroplastics
56 were also made during ship and helicopter observation surveys in the Barents Sea and Fram Strait
57 (Bergmann et al. 2016). A citizen-science study also recently reported the presence of
58 macroplastics on six beaches of the Svalbard Archipelago (Bergmann et al. 2017a). Arctic sea ice
59 was reported by Obbard et al. (2014) as having microplastic concentrations (38 – 234 particles m³
60 of ice) several orders of magnitude greater than highly contaminated oceanic waters. Lusher et al.
61 (2015) first reported on microplastic abundances in surface and sub-surface waters south and
62 southwest of Svalbard. Amélineau et al. (2016) later reported on microplastic abundance in surface
63 waters east of Greenland. Regarding Arctic species, microplastics have been detected in the gular
64 pouches of Little Aulks (*Alle Alle*), (Amélineau et al. 2016), as well as in the stomachs of juvenile
65 polar cod (*Boreogadus saida*), (Kuhn et al. 2018). Microplastics were also detected in sediments
66 (collection depths 2340 – 5570 m) from the Fram Strait (Bergmann et al. 2017b). Recently, results
67 from a circumpolar expedition of the Arctic indicated that concentrations of floating plastic ranged
68 between 0 – 320 000 items km⁻² in the Greenland and Barents Sea and 0 – 27 000 items km⁻² in
69 the rest of the Arctic Ocean (Cózar et al. 2017).

70
71 Plastic contaminants are introduced to the Arctic Ocean due to a combination of (i) long-range
72 transport processes, e.g. via oceanic currents, biotransport and riverine input, and (ii) local
73 anthropogenic activities, e.g. shipping. The three oceanic currents which supply the greatest water
74 volumes to the Arctic Ocean are the (i) West Spitsbergen Current i.e. the polar limb of the North
75 Atlantic circulation which carries warm water from the North Atlantic Current (9.5 Sverdrup, Sv
76 = $10^6\text{m}^3\text{s}^{-1}$), (ii) a cold ocean current that enters from the Pacific Ocean via the Bering Strait (1.5
77 Sv) and, (iii) a branch of the North Atlantic Current, which flows along the Siberian coastline (1.0
78 Sv), (Zarfl and Matthies 2010). These oceanic currents may also transport plastics to the Arctic
79 Ocean with the estimated plastic flux to this region ranging between 62 000 to 105 000 tons per
80 year (Zarfl and Matthies 2010). Models based on a particle-trajectory approach for studying the
81 fate of marine debris in the open ocean highlighted the northward transport of marine debris to
82 polar regions and the formation of a sixth so-called garbage patch in the Barents Sea (van Sebille
83 et al. 2012). Bio-transport is another long-range transport process via which plastics may enter
84 polar regions. Plastic ingestion was reported in Northern Fulmars (*Fulmaris glacialis*) and Thick-
85 billed Murres (*Uria lomvia*) in the Arctic (Mallory 2008; Provencher et al. 2012; Trevail et al.
86 2015). Some studies suggested that the seabirds had ingested plastics during their wintering in the
87 North Atlantic Ocean and had then transported the contaminants to the Arctic upon migration
88 (Mallory 2008; Provencher et al. 2012). Riverine discharge from Siberian (Ob, Yenisei and Lena)
89 and Canadian (Mackenzie) rivers are other potential sources of plastics to the Arctic. Obbard et al.
90 (2014), however, point out that the contribution of riverine discharge to plastic input in the Arctic
91 is projected to be low due to the fact that these rivers flow through sparsely populated watersheds.
92 Local anthropogenic activities are another source of plastics to the Arctic. Increased ship traffic

93 due to shipping and tourism was found to be positively correlated with increased litter densities in
94 the Fram Strait (Bergmann and Klages 2012; Tekman et al. 2017).

95

96 The intense focus by scientists on the near-surface layer of the ocean for microplastics has been
97 due in part to the presumption that the majority of particles would be found in this region of the
98 water column given the inherent densities of individual synthetic polymers. Such a theorization
99 led to traditional techniques that involved nets, manta trawls as well as the seawater intake of
100 vessels that sampled only the upper few metres of the water column for microplastics. Yet, several
101 studies indicated that a mismatch existed between observed and expected plastic concentrations in
102 surface oceanic waters when estimated plastic production and projected inputs to the oceans were
103 considered (Cózar et al. 2014; Eriksen et al. 2014). It was therefore proposed that several
104 mechanisms potentially influenced the vertical distribution of microplastics within the water
105 column and led to their transport out of surface waters. Some of these mechanisms included (i)
106 incorporation into marine aggregates (Long et al. 2015), (ii) biofouling (Fazey and Ryan 2016),
107 (iii) incorporation into faecal matter (Cole et al. 2016) and, (iv) hydrodynamic factors such as wind
108 (Kukulka et al. 2012). Despite the theorization that surface waters are not the ultimate repository
109 for plastic debris in the marine environment (Cózar et al. 2014), few studies ventured beyond
110 traditional near-surface microplastic monitoring to investigate their vertical distribution in the
111 water column.

112

113 Microplastic pollution in the Arctic Ocean is an issue that warrants attention due to the potential
114 threats that these contaminants may pose to the inhabitants of this unique ecosystem. A practical
115 step towards addressing this issue and evaluating the extent of the problem involves assessing the

116 abundance, distribution and composition of microplastics in Arctic waters. Whilst microplastic
117 monitoring in the marine environment has traditionally focused on surface waters, the reality is
118 that the vast majority of marine organisms inhabit sub-surface waters. Monitoring microplastics in
119 sub-surface waters is particularly relevant as it can also provide some insight into the whereabouts
120 of the ‘missing plastic’ from surface waters. To our knowledge, the present study sought for the
121 first time (i) to provide a spatial overview of microplastic abundance, distribution and composition
122 in the Polar Mixed Layer (PML) of the Arctic Central Basin (ACB) and, (ii) to determine whether
123 microplastics in the ACB were being transported out of surface waters by assessing their vertical
124 distribution in the water column.

125

126 **2.0 Materials and Method**

127 **2.1 Study Area**

128 The Arctic Ocean is comprised of a deep central basin surrounded by extensive continental shelves
129 (CAFF 2013). The bathymetry of the Arctic Ocean is such that the Lomonosov Ridge separates
130 the central basin into the Canadian (Amerasian) and Eurasian basins with the basins being further
131 sub-divided by the (i) Gakkel Ridge, into the Amudsen and Nansen basins and, (ii) Alpha Ridge,
132 into the Makarov and Canada basins (Jakobssen et al. 2004; Rudels 2015, Figure 1). A major
133 structuring element of the Arctic marine ecosystem is sea ice which floats on the surface layer
134 impeding surface mixing and influencing freshwater and heat fluxes (CAFF 2013). In the Arctic
135 Ocean, there is a distinct vertical stratification of the water column giving rise to three major water
136 layers (i) Polar Surface Water (PSW) which includes the Polar Mixed Layer (PML) and the
137 halocline, (ii) Atlantic Water and, (iii) deep and bottom waters (Rudels 2015, Figure 1). The PML
138 (approximately 50 m deep) is the uppermost surface layer of low salinity water formed as a result

139 of sea ice melt and the influx of freshwater from riverine sources (Rudels et al. 1991; CAFF 2013;
140 Jakobsson et al. 2014). Beneath the PML is a halocline (50 – 250 m), characterised by a strong
141 salinity increase with depth and comprised of either Pacific waters or Atlantic waters with the
142 Pacific halocline being deeper than the Atlantic halocline (Rudels et al. 1991; Jakobsson et al.
143 2004). Below the halocline lies an intermediate water layer comprising of dense saline Atlantic
144 water. The deep and bottom waters also referred to as Arctic deep water ranges from a depth of
145 approximately 900 m and extend to the seafloor (Rudels et al. 1991; CAFF 2013).

146

147 **2.2 Sample Collection**

148 *Underway samples*

149 This study was conducted onboard the Swedish icebreaker Oden during the Arctic Ocean 2016
150 expedition. The vessel departed Longyearbyen, Svalbard on August 8th 2016 and traversed
151 approximately 4943 nautical miles in the Arctic Ocean until its return on September 19th 2016
152 (Figure 2). Sub-surface oceanic water pumped onboard the vessel via the bow water system was
153 sampled for microplastics according to Lusher et al. (2014). Sampling was conducted for a period
154 of approximately 6 weeks (9 August to 16 September 2016). Since each sample constituted the
155 filtration of approximately 2000 L of water, the total survey effort for this study was approximately
156 116 000 L of water (58 samples).

157

158 Seawater from a continuous intake located at the keel of the ship (depth 8.5 m) was pumped
159 onboard the vessel using a rotary positive displacement pump (Universal II Series Pump,
160 Waukesha Cherry-Burrell) at a flow rate of 85 L/min (at optimal capacity) and transported to the
161 laboratory via stainless steel pipes. Prior to reaching the laboratory, the seawater passed through a

162 stainless steel primary filter (pore size 2.5 mm) which was beyond the control of the investigator.
163 The discharge of grey water in relation to the seawater intake was not an issue since the seawater
164 intake was located towards the front of the vessel whilst grey water was discharged mid-vessel. In
165 the laboratory, seawater from the vessel's bow water system was allowed to flow through a covered
166 stainless steel sieve (250 μm) by means of a connection hose fitted into the wooden sieve cover.
167 For the duration of the sampling, the stainless steel sieve was supported in a wooden stand. Based
168 on Lusher et al. (2014), approximately 2000 L of water was filtered for each sample. The length
169 of time taken for the filtration of the specified volume of water was determined by calculation of
170 the flow rate of the seawater. A flow meter, attached at a point prior to the entry of the water into
171 the sieve, was also used to verify the volume of water filtered. Once the specified volume of water
172 was filtered, the sieve was removed and Milli-Q water was used to wash retained material from
173 the sieve into a clean container. The collected material was then filtered under vacuum onto glass
174 microfiber paper (GF/C); Whatman: 47 mm, pore size: 1.2 μm , using a Buchner funnel and a
175 vacuum flask (Lusher et al. 2014). Each filter paper was then placed into a clean plastic petri dish,
176 covered and stored in a freezer (-20 °C) until returned to the laboratory. At the start and at the end
177 of each sample, positioning data were collected. Data for various environmental variables were
178 obtained from the vessel's (i) thermosalinometer (water temperature, salinity) and (ii) weather
179 station (wind speed and direction).

180

181 *CTD samples*

182 A rosette water sampler containing 24 Niskin bottles coupled to a Sea-Bird SBE 911 conductivity-
183 temperature-depth (CTD) sensor suite (hereafter referred to as CTD) was used to collect sub-
184 surface water samples and hydrographic data at 9 sampling locations in the Arctic Ocean. Upon

185 deployment from the vessel, the CTD entered the water and was allowed to descend to the bottom
186 layer. During the descent, Niskin bottles were open with water flowing through them. It was during
187 the up-cast that multiple Niskin bottles were closed at specific depths in order to facilitate the
188 collection of a specified volume of water. A total of 48 water samples were retrieved during the 9
189 CTD casts to sample for microplastics. At 7 of the CTD casts, 6 water depths were sampled with
190 48 L of water collected at each depth i.e. 4 Niskin bottles (12 L) per depth. At 2 CTD casts, 3 water
191 depths were sampled with 21 L of water collected at each depth i.e. 3 Niskin bottles (7 L) per
192 depth. At a particular sampling location, the overall goal was to collect samples in the near-surface,
193 mid-water and bottom layers within the water column thereby reflecting the main water masses.
194 As such, exact sampling depths were determined by the information provided by the salinity and
195 temperature sensors on each downcast. The deepest samples at each CTD cast were collected at
196 least 10 m above the seafloor at a given location. Following each CTD cast, Niskin bottle taps
197 were rinsed with Milli-Q water and a clean hose was attached. Water from bottles closed at the
198 same depth was passed through the same stainless steel sieve (250 μm) held in a covered wooden
199 stand. Once water from all bottles at a specific depth had been filtered, the sieve was removed and
200 Milli-Q water was used to wash retained material from the sieve into a clean container. The
201 collected material was filtered under vacuum onto glass microfiber paper (GF/C); Whatman: 47
202 mm, pore size: 1.2 μm , using a Buchner funnel and a vacuum flask. Each filter paper was then
203 folded and placed into an aluminium foil packet and stored in a freezer (-20 °C) until returned to
204 the laboratory.

205

206 **2.3 Method Validation and Contamination Prevention**

207 For the underway samples, potential contamination during sample processing was evaluated by (i)
208 leaving clean plastic petri dishes with filter paper exposed to the air during vacuum filtration, and
209 (ii) passing an aliquot (250 mL) of Milli-Q water through clean GF/C filter paper under vacuum.
210 For the CTD samples, potential contamination was assessed by filling a clean Niskin bottle with
211 Milli-Q water and subjecting it to the exact process a sample underwent. Measures taken to prevent
212 contamination in the laboratory included (i) wearing lab coats (cotton/polyester blend), cotton
213 clothing and gloves (nitrile) during sample processing, (ii) placing a wooden cover over the
214 stainless steel sieve during filtration to prevent airborne contamination, and (iii) washing all
215 containers used during sample processing with Milli-Q water before reuse.

216

217 **2.4 Laboratory analyses**

218 Filter papers were removed from the freezer, left to dry and then visually examined under a
219 dissecting microscope (Olympus SZX10) equipped with a polariser and camera (Q Imaging Retiga
220 2000R). Potential microplastics were isolated and processed (photographed and length
221 measurements taken) prior to transferring to a clean filter paper in a labelled petri dish (Kanhai et
222 al. 2017). All potential microplastics were analysed by Fourier transform infrared (FT-IR)
223 spectroscopy on a Thermo Scientific Nicolet iN10 FT-IR spectrometer. The instrument was
224 equipped with a potassium bromide (KBr) beamsplitter and an internal mercury cadmium telluride
225 (MCT) detector which was cooled with liquid nitrogen. Microscope-reflectance sampling was
226 performed and spectra were recorded as the average of 256 scans in the spectral wave number
227 range of 4000 - 675 cm^{-1} at a resolution of 4 cm^{-1} . Thermo Scientific's OMNIC Picta Version 9
228 spectroscopy software was used for processing and evaluating all spectra. Prior to analysing each
229 sample, background scans were performed and sample spectra were automatically corrected. Each

230 sample spectrum was compared with those of known standard polymers in the (i) Hummel
231 Polymer Sample library, (ii) Polymer Laminate Films library, and (iii) Wizard library. Values of
232 between 0 and 100 % were produced for each match between sample and reference spectra with
233 the highest percentage representing the closest match. Particles for which there was uncertainty
234 regarding the identity of the polymer (specifically fragments and some fibres) were subjected to
235 further FT-IR spectroscopy on a Bruker Vertex 70 Infrared Spectrometer coupled to a Hyperion
236 1000 microscope (Kanhai et al. 2017). Samples which produced spectra with a match < 60 % were
237 automatically rejected while those with a match of > 70 % were automatically accepted. All spectra
238 with matches > 60 % were individually examined to ensure that there was clear evidence of peaks
239 from the sample corresponding to known peaks of standard polymers and that instances of the
240 misidentification of natural and semi-synthetic polymers was reduced (Comneau-Stancu et al.
241 2017).

242

243 **2.5. Statistical analyses**

244 All statistical analyses were performed using R version 3.2.3 (R Core Team 2015). Descriptive
245 statistics, histograms and box plots were generated and tests of normality (Supplementary Tables
246 1 and 2) were conducted on all data to determine whether parametric or non-parametric statistical
247 analyses were appropriate. Correlation analyses were performed between individual
248 environmental variables and microplastic abundance for both underway and CTD samples. A
249 generalized additive model (GAM) was developed using the underway data and a generalized
250 linear mixed model (GLMM) was developed using the CTD data to determine which
251 environmental variables had an effect on microplastic abundance.

252

253 **3. Results**

254 **3.1. Quality Control**

255 In conjunction with the collection of samples via the underway system of the vessel, a total of 24
256 blanks (air contamination-12, method-12) were run (Supplementary Table 3). No synthetic
257 polymers were found in the method blanks. However, a single synthetic fibre (blue, polyethylene
258 terephthalate, 0.438 mm) was found in the last air contamination blank. For 6 of the 9 CTD casts,
259 at least one method blank was run (Supplementary Table 4). Between 0 and 3 synthetic fibres were
260 found in each of the method blanks. The synthetic fibres that were found included polyethylene
261 terephthalate (n = 8), polyacrylonitrile (n = 1) and polyvinyl chloride (n = 1).

262

263 **3.2. Overview of findings**

264 *Underway samples*

265 A total of 303 particles were isolated from the underway samples and analysed by FT-IR
266 spectroscopy. Of these, 46 particles were excluded because of uncertainty regarding their identity
267 (< 60 % match to reference spectra) and in the minority of cases (n = 6) due to their length (< 250
268 μm). Of the remaining particles (n = 257), 14 were macro-particles (i.e. > 5 mm in length) and 243
269 were micro-particles (< 5 mm in length). Of the macro-particles, 11 were confirmed as
270 macroplastics having the following polymer types: polyethylene terephthalate (4), polyamide
271 blend (4) and polyacrylonitrile (3). Of the 243 micro-particles, 110 were natural (cellulosic), 16
272 were semi-synthetic (cellulose-based e.g. rayon) and 117 were synthetic. All further analyses and
273 discussions focus on the 117 confirmed microplastics.

274

275 The majority (94 %) of microplastics were fibres and 6 % were fragments. In terms of colour, the
276 most prevalent were blue (49 %) and transparent (25 %) (Figure 3a). Approximately 62 % of the
277 microplastics occurred in the larger size classes of 1.0 – 2.0 mm and 2.0 – 5.0 mm (Figure 4a).
278 Microplastic polymer types included polyester (n = 88), blends (n = 11), polyacrylonitrile (n = 8),
279 polyamide (n = 5) and polyvinyl chloride (n = 5), (Figure 5a). The overall category of ‘polyester’
280 included both polyethylene terephthalate (PET) and other polyesters while blends included either
281 polyamide blends or polyester blends.

282

283 *CTD samples*

284 A total of 157 particles were isolated from the CTD samples and analysed by FT-IR spectroscopy.
285 Of these, 14 were excluded for the reasons mentioned above for the underway samples. Of the
286 remaining particles, 2 were categorised as macroplastics (> 5 mm) and included PET and
287 polyacrylonitrile. Of the 141 micro-particles (< 5 mm), 39 were natural (cellulosic), 8 were semi-
288 synthetic (cellulose-based e.g. rayon) and 94 were synthetic. All further analyses and discussions
289 focus on the 94 confirmed microplastics.

290

291 Overall, the characteristics of the microplastics isolated from CTD samples were similar to those
292 from the underway samples in that (i) the majority (96 %) of microplastics were fibres and 4 %
293 were fragments, (ii) the most prevalent colours were blue (46 %) and transparent (22 %) (Figure
294 3b), (iii) the majority (64 %) of microplastics were in the larger size classes of 1.0 – 2.0 mm and
295 2.0 – 5.0 mm (Figure 4b), and (iv) microplastic polymer types included polyester (n = 74), blends
296 (n = 12), polyacrylonitrile (n = 6), polyamide (n = 1) and polyvinyl chloride (n=1), (Figure 5b).

297 The overall category of polyester included both polyethylene terephthalate (PET) and polyester
298 while blends included only polyamide blends.

299

300 **3.2 Microplastic abundance and distribution in the Arctic Ocean**

301 Based on the underway samples (collection depth 8.5 m), microplastic abundance in sub-surface
302 waters in the Arctic Ocean ranged between 0 – 7.5 particles m^{-3} with a median of 0.7 particles m^{-3}
303 (interquartile range 0.4 – 1.0), (Figure 2, Supplementary Table 5). For the majority of the
304 sampling sites, microplastic abundance ranged between 0 – 1.0 particles m^{-3} . However, at a few
305 sites, microplastic abundances were between 2 – 2.5 particles m^{-3} and at two sites it was at 5 and
306 7.5 particles m^{-3} respectively. Based on the CTD samples (collection depths between 8 – 4369 m),
307 microplastic abundance in sub-surface waters in the Arctic Ocean ranged between 0 – 375 particles
308 m^{-3} with a median of 20.8 particles m^{-3} (interquartile range 20.8 – 62.5) (Figures 2 and 5,
309 Supplementary Table 6). With the exception of CTD cast 4, the CTD casts (1 – 3) nearer the
310 periphery of the Arctic Central Basin (ACB), i.e. in the Nansen Basin (Yermak Plateau), reflected
311 a comparatively higher abundance of microplastics in the water column than other CTD casts
312 within the ACB (Figures 2, 6). It must be noted however that CTD casts 1 – 3 sampled the upper
313 850 m of the water column and as such would have sampled particles from the Polar Mixed Layer
314 (PML), Atlantic halocline and Atlantic water (Figures 1, 6). This is in contrast to the other CTD
315 casts which sampled a much more extensive vertical range throughout the water column by
316 including deep bottom water in excess of 1000 m depth. Overall, there was no statistically
317 significant correlation between microplastic abundance and depth (Spearman's rank correlation,
318 $\rho = 0.06$, $p\text{-value} = 0.7$). However, upon examination of individual CTD casts, it is apparent that
319 microplastic abundance was not uniform at various depths in the water column and that there were

320 certain depths that reflected higher microplastic abundances (Figure 6). Additionally, microplastic
321 abundance (particles m^{-3}) in the different water masses of the ACB was as follows: Polar Mixed
322 Layer (0 - 375) > Deep and bottom waters (0 – 104) > Atlantic water (0 – 95) > Halocline i.e.
323 Atlantic or Pacific (0 – 83), (Table 1).

324

325 **3.3. Association between environmental variables and microplastic abundance in samples**

326 Correlation analyses were conducted to determine whether there was any association between
327 environmental variables and microplastic abundance in the samples. For both the underway and
328 CTD samples, there was no statistically significant correlation between microplastic abundance
329 and any of the ancillary environmental variables of temperature, salinity, wind direction, wind
330 speed, depth and density (Supplementary Tables 1 and 2). Specifically, there was no statistically
331 significant correlation between microplastic abundance at depth (Supplementary Table 2).
332 However, for the underway samples, there was a statistically significant weak negative correlation
333 between microplastic abundance and latitude (Spearman's rank correlation, $\rho = -0.286$, p-value
334 = 0.03).

335

336 Using the underway data, a Generalized Additive Model (GAM) was developed to further
337 determine whether environmental variables influenced microplastic count in the underway
338 samples. In this model, the response variable was microplastic count (number of microplastics per
339 sample) and initial explanatory variables included location (latitude, longitude), physico-chemical
340 properties associated with sub-surface waters (temperature, salinity) and weather data (wind
341 direction, wind speed). In the model, the Poisson family distribution of error terms was specified
342 with a log link function since microplastic abundance data were count data. The output of the

343 initial model was examined and based on this non-parametric smoothers were accordingly applied
344 to the explanatory variables. A scale invariant tensor product smooth (te) was applied to latitude
345 and longitude while a cubic regression spline (bs = cr) was applied to all other variables except
346 wind direction to which no smoother was applied (based on initial model plots). Non-significant
347 explanatory variables (as evidenced by their p-values) were eliminated in a stepwise manner until
348 a GAM with the lowest Akaike Information Criterion (AIC) score and the fewest explanatory
349 variables was obtained. The final GAM (R-sq = 0.396) was as shown below:

350
351 Microplastic count ~ te(**latitude, longitude**) + s(**temperature**, br = "cr") + s(**wind speed**, bs =
352 "cr")

353
354 All of the explanatory variables that were present in the final model (shown in bold) were found
355 to have a significant influence on microplastic count in water samples from the Arctic Ocean (wind
356 speed, p-value = 0.0006, latitude, longitude, p-value = 0.0007, temperature, p-value = 0.0483).

357
358 A generalized linear mixed model (GLMM) was developed using the CTD data to determine the
359 influence of environmental variables on microplastic count in the CTD samples. In this model, the
360 response variable was microplastic count (number of microplastics per sample) and initial
361 explanatory variables included location (latitude, longitude), physicochemical properties
362 associated with sub-surface waters (temperature, salinity), depth at which water was sampled and
363 ctd number. All the explanatory variables were included in the model as fixed effects, with the
364 exception of ctd number which was included as a random effect. In the model, the Poisson family
365 distribution of error terms was specified with a log link function since microplastic abundance data

366 were count data. Based on the preliminary finding that there was a statistically significant
367 correlation between depth and salinity (Spearman's rank correlation, $\rho = 0.852$, $p\text{-value} =$
368 $8.156e-13$), temperature and salinity (Spearman's rank correlation, $\rho = 0.506$, $p\text{-value} = 0.00074$)
369 and depth and density (Spearman's rank correlation, $\rho = 0.973$, $p\text{-value} = < 2.2 e-16$),
370 interactions between these variables (denoted by ':') were included in the initial model. Non-
371 significant explanatory variables (as evidenced by their p -values) were eliminated in a stepwise
372 manner until a model with the lowest Akaike Information Criterion (AIC) score and the fewest
373 explanatory variables was obtained. The significance of the random effect (ctd number) in the final
374 model was verified by using analysis of variance (ANOVA) to compare the full final model to a
375 reduced model (random effect deleted). The model with the random effect was shown to be
376 significantly different (ANOVA, $p\text{-value} = 0.0008$, $\Delta\text{AIC} = 9.19$) from the model without the
377 random effect. The model with the lower AIC score (AIC = 190.59) was retained as the final mixed
378 effects model as shown below:

379

380 Microplastic count ~ **latitude** + **temperature: salinity** + (1|ctd)

381

382 Latitude ($p\text{-value} = 0.0198$) and the physicochemical parameters of temperature and salinity (p -
383 value = $7.46 e-05$), as shown in bold, were the explanatory variables that were found to have a
384 significant influence on microplastic count in the CTD samples.

385

386 **4.0 Discussion**

387 The discovery of microplastics in virtually every environmental phase (sea ice, water, sediments,
388 biota) of the Arctic and Southern Oceans has revealed that polar oceans, though remote, are not

389 immune to the entry of plastic contaminants to their ecosystems (Bergmann and Klages 2012;
390 Obbard et al. 2014; Lusher et al. 2015; Amélineau et al. 2016; Bergmann et al. 2016; Bergmann
391 et al. 2017a; Bergmann et al. 2017b; Cincinelli et al. 2017; Cózar et al. 2017; Isobe et al. 2017;
392 Tekman et al. 2017; Waller et al. 2017). The present study expands the knowledge base about
393 plastics in the Arctic by providing evidence for the existence of microplastics in the Polar Mixed
394 Layer (PML) as well as some insight into the vertical distribution of microplastics in the Arctic
395 Central Basin (ACB). This region of the Arctic, though of low productivity, has been recognised
396 as an Ecologically/Biologically Significant Marine Area (EBSA) due to its uniqueness/rarity,
397 provision of a critical habitat and ability to support specialised biota (CAFF 2017). There is cause
398 for concern about microplastics in Arctic waters since laboratory studies have shown that these
399 contaminants may (i) hinder algal photosynthesis/growth (Bhattacharya et al. 2010; Besseling et
400 al. 2014), (ii) reduce feeding and energy reserves of lugworms (Besseling et al. 2013; Wright et
401 al. 2013), (iii) reduce filtering activity and decrease lysosomal membrane stability in mussels (Von
402 Moos et al. 2012; Wegner et al. 2012), (iv) reduce feeding and reproductive output in copepods
403 (Cole et al. 2015) and, (v) cause liver stress, negatively impact upon cholinergic neurotransmission
404 and lead to endocrine disruption in fish (Oliveira et al. 2013; Rochman et al. 2013; Rochman et al.
405 2014). It must be pointed out, however, that some laboratory experiments which reported negative
406 effects of microplastics on marine organisms used microplastic concentrations of 42 to 10 000
407 particles/mL or 42 million to 10 billion particles m^{-3} (Phuong et al. 2016). In context, microplastic
408 abundance in the ACB as reported by the present study ranged from 0 – 7.5 particles m^{-3} (based
409 on underway sampling) and 0 – 375 particles m^{-3} (based on CTD sampling). Although the
410 ecological impact of microplastics upon the Arctic ecosystem presently remains unknown, it is
411 plausible that these contaminants could pose a threat to its inhabitants.

412

413 The present study showed for the first time the pervasiveness of microplastics throughout the water
414 column of the Arctic Central Basin. Between depths of 8 – 4400 m, microplastic abundance in the
415 ACB ranged between 0 – 375 particles m⁻³ (based on CTD sampling). Such findings provide
416 evidence that in natural conditions microplastics are being vertically transported out of surface
417 waters. These findings also give some indication as to the whereabouts of the ‘missing plastic’
418 from oceanic surface waters (Cózar et al. 2014; Eriksen et al. 2014). Recently, Courtene-Jones et
419 al. (2017) also reported on microplastic abundance (70.8 particles m⁻³) in deep oceanic waters
420 (2227 m at the Rockall Trough, North East Atlantic Ocean) and similarly suggested the possibility
421 of vertical re-distribution of microplastics within the water column. Although it remains unclear
422 as to which mechanisms are specifically operating in the ACB to influence the vertical transport
423 of particles, previous studies have provided several possibilities. Specifically, some laboratory
424 experiments showed that aggregates of algae species (*Chaetoceros neogracile*, *Rhodomonas*
425 *salina*) were capable of incorporating and concentrating microplastics and that the microplastics
426 impacted the sinking rates of the aggregates (Long et al. 2015). In the Arctic Ocean, it is certainly
427 plausible that marine aggregates may be playing a role in the vertical transport of microplastics
428 due to the existence of phytoplankton in the ACB (CAFF 2017) and the fact that transparent
429 exopolymer particles (TEPs); which are excreted by algae and are important components of marine
430 aggregates, have been reported in sub-surface waters as far north as the Fram Strait (Engel et al.
431 2017). Biofouling is another possibility as field studies have shown that plastic particles exposed
432 to natural conditions became sufficiently fouled, had their average material density affected
433 leading them to sink (Fazey and Ryan 2016). The incorporation of microplastics into faecal matter
434 is another means by which microplastics may be vertically transported out of surface waters given

435 that laboratory experiments have shown that zooplankton may egest microplastics within densely
436 packed faecal pellets which in natural conditions would sink or in some cases be eaten by other
437 biota (Cole et al. 2016).

438

439 Sea ice is an integral component of the Arctic Ocean's ecosystem and as such possibly exerts an
440 influence on microplastic abundance in sub-surface waters. Sea ice floating on the surface of the
441 water column in the Arctic Ocean can potentially act as (i) a source of microplastics upon melting,
442 (ii) a physical barrier to wind and as such reduce vertical mixing of surface waters and, (iii) a
443 physical barrier to influx of polluted surface waters. Based on the analysis of sub-sections of four
444 ice cores, sea ice in the Arctic Ocean was reported to contain orders of magnitude more
445 microplastic than contaminated oceanic waters suggesting that sea ice potentially acts as both a
446 sink and a source of microplastics (Obbard et al. 2014). Apart from Obbard et al. (2014) no data
447 exists in the published literature regarding either the spatial or vertical distribution of microplastics
448 in sea ice from the Arctic Ocean. In the upper water column, the absence of sea ice cover means
449 that wind stress can generate turbulence and lead to vertical mixing of buoyant plastic debris
450 (Kukulka et al. 2012). More recently, Cózar et al. (2017) suggested that sea ice can also act as a
451 physical barrier preventing the surface advance of polluted Atlantic water into the Arctic Ocean.
452 The present study highlighted that the Polar Mixed Layer (PML) of the ACB reflected the highest
453 overall microplastic abundance (particles m^{-3}): Polar Mixed Layer (0 - 375) > Deep and bottom
454 waters (0 - 104) > Atlantic water (0 - 95) > Halocline i.e. Atlantic or Pacific (0 - 83). As previously
455 mentioned, the PML is the uppermost surface layer of low salinity water (approximately 50 m
456 deep) formed as a result of sea ice melt and the influx of freshwater from riverine sources (Rudels
457 et al. 1991; CAFF 2013). It is possible that one of the reasons that the highest microplastic

458 abundances were recorded in this layer is due to its proximity to microplastic sources such as
459 melting sea ice as well as sea-going vessels (especially if they are releasing wastewater to the
460 environment). Furthermore, in the present study, the highest microplastic abundances were
461 reported nearer to the periphery of the Arctic Central Basin (ACB), i.e. in waters north of Svalbard.
462 It is possible that the lack of permanent sea ice cover in this region of the Arctic allows incoming
463 Atlantic water to have a greater influence on near surface waters thereby resulting in higher
464 microplastic abundances. A recent circumpolar expedition of the Arctic Ocean similarly reported
465 that the north eastern Atlantic sector of the Arctic was a hotspot of plastic debris due to the
466 influence of incoming Atlantic water (Cózar et al. 2017).

467

468 The present study showed that there was a predominance of fibrous microplastics (> 90%) in sub-
469 surface waters of the ACB. This dominance of fibres in sub-surface waters was similarly reported
470 in (i) the north east Pacific Ocean (75 %), (ii) the north east Atlantic Ocean (96 %), (iii)
471 south/southwest of Svalbard (95 %), and (iv) the Atlantic Ocean (96 %) (Desforges et al. 2014;
472 Lusher et al. 2014; Lusher et al. 2015; Kanhai et al. 2017). Fibrous microplastics in the marine
473 environment most likely originate from textile materials and fishing gear (Andrady 2017). Studies
474 have indicated that washing clothes may lead to the release of fibrous materials in the order of >
475 1900 fibres per wash or as much as 700 000 fibres per 6 kg load of acrylic fabric (Browne et al.
476 2011; Napper and Thompson 2016). A recent study in the Ross Sea revealed that the highest
477 concentration of fibrous microplastics (54 %) was found close to the effluent of a sewage treatment
478 plant at the scientific Mario Zucchelli Station, Antarctica (Cincinelli et al. 2017). Fibrous
479 microplastics may enter the Arctic Ocean through a combination of long range transport processes
480 (e.g. via oceanic currents, riverine input) or more in-situ activities such as the release of wastewater

481 from vessels operating in the region. Another mechanism which was recently suggested as being
482 responsible for plastic fibres in Arctic sea ice was atmospheric transport (Cózar et al. 2017). This
483 is certainly plausible as there have been reports of the atmospheric fallout of synthetic polymers
484 in both urban and sub-urban environments in France (Dris et al. 2016). It must be highlighted that
485 while fibrous microplastics seem to be dominant in certain sub-surface waters, other studies
486 conducted in surface Arctic waters reported a predominance of filaments (97 %) and fragments
487 (73 %), (Amélineau et al. 2016; Cózar et al. 2017).

488

489 Analytical techniques such as FT-IR and Raman spectroscopy are a central component of
490 microplastic studies which aim to provide unambiguous identification of synthetic polymers in
491 environmental samples. Omission of this critical step is likely to be a key contributor to an
492 overestimation of microplastic abundances due to the inclusion of non-synthetic polymers in
493 microplastic counts. In the present study, for example, only a percentage (underway - 48 %, CTD
494 - 67%) of the particles were confirmed as synthetic polymers with the remainder being a
495 combination of natural and semi-synthetic polymers. Of the synthetic polymers in the present
496 study, the most abundant (underway – 74 %, CTD – 78 %) was polyethylene terephthalate (PET).
497 A member of the polyester family, PET is one of the five major types of commodity plastics
498 commonly found in the marine environment (Andrady 2011; Andrady 2017). As a thermoplastic,
499 PET is often used in manufacturing beverage containers and packaging materials and its fibres are
500 used in clothing. Overall, this finding of a high prevalence of polyesters in sub-surface waters of
501 the Arctic Central Basin was also corroborated by those of other Arctic studies which reported that
502 polyester was the most prevalent synthetic polymer in waters south/southwest of Svalbard (15 %),
503 in waters of the east Greenland Sea (53 %), and in sea ice (21 %), (Obbard et al. 2014; Lusher et

504 al. 2015; Amélineau et al. 2016). Other polymers in sub-surface Arctic waters in this study
505 included polyacrylonitrile, polyamide and polyvinyl chloride. Of note is the fact that the majority
506 of synthetic polymers in the present study had densities greater than that of seawater (Andrady
507 2017). Even though investigators are able to generate information regarding the identity of
508 polymers in environmental samples, definitive statements cannot be made about the origin of the
509 plastics. Based on the identity and type of synthetic polymers found in sub-surface Arctic waters,
510 it is likely that they originated from textiles, fishing gear, beverage containers and packaging
511 materials (Andrady 2011; Andrady 2017).

512

513 Within the water column, the distribution of microplastics is in a state of flux due to the influence
514 of multiple factors. The development of models based on simultaneously acquired environmental
515 and microplastic data is immensely useful in this respect as they can provide some discernment
516 regarding the variables influencing measures of microplastic abundances in the marine
517 environment. In the present study, the utilisation of a generalized additive model (GAM) and a
518 generalized linear mixed effects model (GLMM) was particularly relevant due to the ability of
519 both models to handle non-normal data and in the case of the GLMM to differentiate between
520 fixed and random effects. Visual inspection of microplastic abundances in sub-surface waters
521 (Figure 2) revealed that the highest microplastic abundances were located to some extent towards
522 the periphery of the Arctic Central Basin (ACB). Bearing this in mind, it was presumed that the
523 ‘location’ of sampling could have influenced the number of microplastics that were found in the
524 samples. Cózar et al. (2017) also noted this spatial heterogeneity in the distribution of plastics in
525 the Arctic. For these reasons, latitude and longitude were included as a proxy of location in the
526 models. In the GAM, both variables were included using a smoother and therefore it was not

527 assumed that latitude and longitude had a linear effect on microplastic count. Furthermore, during
528 the period of sampling (August/September 2016), the Arctic Central Basin (ACB) was not
529 completely covered by sea ice. Since the vessel traversed areas of open water, wind was included
530 in the model as it could have influenced microplastic abundance at specific sampling sites. Both
531 models suggested that location, oceanographic (temperature, salinity) and atmospheric variables
532 (wind) had a significant influence on microplastic counts in samples of water from the Arctic
533 Ocean. The findings of the present study must be taken in the context of the number of samples
534 used in the generation of the GAM ($n = 58$). Models that are based on a low number of samples
535 are weak e.g. standard errors are inflated, etc. However, such models allow us to delve a bit deeper
536 into the factors influencing measurements of microplastic abundance in the Arctic Ocean. Findings
537 of the present study were also corroborated by previous studies which indicated that water
538 temperature, salinity and wind also had a significant effect on microplastic abundance (Lusher et
539 al. 2014; Lusher et al. 2015; Kanhai et al. 2017).

540

541 Comparative assessments between oceanic basins are critical in providing an indication of the
542 extent of microplastic pollution in the marine environment. A major challenge, which demands
543 caution when drawing conclusions from such comparisons, is the lack of standardization of
544 microplastic sampling methods (depth of collection, mesh size of net/sieve, etc). Microplastic
545 abundances in the present study were not normally distributed and therefore the median was
546 reported as it is the most relevant measure of central tendency for such data. However, in order to
547 enable comparability with other published studies, which generally did not report median
548 microplastic abundances nor made statements about the normality of their data, the mean was also
549 reported in Supplementary Table 7. In the present study, sub-surface waters (depth 8.5 m) in the

550 Arctic Central Basin (ACB), sampled via the bow water system, had a mean microplastic
551 abundance of 0.97 ± 1.20 particles m^{-3} . In comparison to other studies that employed similar
552 methods (i.e. the underway system of vessels) to sample sub-surface waters, microplastic
553 abundance in the ACB was lower than values reported (i) in the north eastern Pacific Ocean (279
554 ± 178 particles m^{-3}), (ii) in the North Atlantic Ocean ($13 - 501$ particles m^{-3}), (iii) off Svalbard
555 (2.68 ± 2.95 particles m^{-3}), (iv) in the north east Atlantic Ocean (2.46 ± 2.43 particles m^{-3}), and (v)
556 in the Atlantic Ocean (1.15 ± 1.45 particles m^{-3}), (Desforges et al. 2014; Lusher et al. 2014; Enders
557 et al. 2015; Lusher et al. 2015; Kanhai et al. 2017, Supplementary Table 7). The only oceanic basin
558 for which there were reports of lower microplastic abundances in sub-surface waters was the Ross
559 Sea (0.17 ± 0.34 particles m^{-3}), (Cincinelli et al. 2017; Supplementary Table 7). Although the
560 methods used to sample sub-surface waters for microplastics in the above mentioned studies were
561 similar, the fact remains that the variation of several factors e.g. mesh size of sieve ($1 - 300 \mu m$),
562 sampling depth ($3 - 11$ m), etc., amongst the studies could have impacted the reported microplastic
563 abundances. In terms of assessing the vertical distribution of microplastics in the marine
564 environment, Bagaev et al. (2017) was the only other published study which utilised a similar
565 sampling method (Niskin bottles) in the Baltic Sea. Being cognisant of the fact that no
566 confirmatory analytical techniques or blanks were used by Bagaev et al. (2017), microplastic
567 abundance at multiple depths in sub-surface waters of the Arctic Ocean (mean: 46 ± 62 particles
568 m^{-3} ; range: $0 - 375$ particles m^{-3} ; depths sampled: $8 - 4400$ m) was lower than reported for the
569 Baltic Sea (mean: 310 ± 520 particles m^{-3} ; $70 - 2600$ particles m^{-3} ; depths sampled: $1 - 218$ m).
570 Similar to the findings of the present study whereby the highest microplastic abundances were
571 found in the uppermost water layer i.e. the PML, Bagaev et al. (2017) reported that near- surface
572 and near-bottom water layers in the Baltic Sea had higher fibre concentrations than intermediate

573 layers and that this was possibly due to greater turbulence and density stratification in those layers.
574 Of note is the fact that microplastic abundance in deep waters of the ACB (0 – 104 particles m⁻³,
575 depths 1000 – 4400 m, sieve 250 µm) was similar to those reported for deep waters at the Rockall
576 Trough, North East Atlantic Ocean (70.8 particles m⁻³, depth 2227 m, sieve 80 µm), (Courteney-
577 Jones et al. 2017). Overall, it must be acknowledged that an underestimation of microplastic
578 abundance in the Arctic Central Basin (ACB) could have occurred in the present study as the mesh
579 size of the sieve was only 250 µm leading to an exclusion of smaller sized particles. Nevertheless,
580 the Arctic's remote geographic location away from major population centres, its low population
581 in its surrounding continental shelves and relatively low in-situ anthropogenic activities (e.g.
582 shipping) are all factors which may explain the lower microplastic abundances in sub-surface
583 waters within the Arctic Central Basin (ACB). From an oceanographic perspective, the reduced
584 contribution of Atlantic water in its upper water layers due to the dominance of the polar mixed
585 layer is another possible explanation for lower microplastic abundances in the Arctic in
586 comparison to other oceanic basins. Presumably more polluted, Atlantic water which originates
587 from the more densely populated southern latitudes has its surface advance into the Arctic Ocean
588 hindered due to freshwater released from melting ice and other physical barriers such as the sea
589 ice itself and the Novaya Zemlya islands (Cózar et al. 2017). However, a plausible future scenario
590 for the Arctic in the context of a changing climate is that microplastic abundance in near-surface
591 layers of the Arctic Ocean may increase upon melting of contaminated sea ice and opening up of
592 shipping lanes due to a decrease in sea ice extent (Obbard et al. 2014; Cózar et al. 2017).

593

594 Of interest is the fact that the present study managed to sample microplastics in sub-surface waters
595 at approximately 8.5 m depth by two independent methods i.e. by the bow water system of the

596 vessel (underway sampling) and the rosette water sampler (CTD sampling). CTD samples (n = 9)
597 retrieved from an average depth of 8.5 m indicated that microplastic abundance in the ACB ranged
598 between 0 – 148 particles m⁻³, with a median of 20.8 particles m⁻³. By comparison, samples
599 collected via the underway system (n = 58) at 8.5 m indicated that microplastic abundance in the
600 ACB ranged between 0 – 7.5 particles m⁻³, with a median of 0.7 particles m⁻³. Although both
601 methods sampled water at an average depth of 8.5 m, calculated microplastic abundances from
602 both methods are not directly comparable due to the differences associated with the methods.
603 Whereas underway sampling involved filtration of a greater volume of water (approximately 2000
604 L) over a longer distance and a longer sampling time (> 2 hours), the CTD sampling involved the
605 collection and subsequent filtration of a smaller volume of water (21 L or 48 L) at a single location
606 in a shorter period of time (minutes). The advantage of using the underway system is that
607 microplastic abundances over a larger spatial area can be quantified whilst the vessel is in transit.
608 By comparison, CTD sampling facilitates the quantification of microplastic abundance at specific
609 locations making it less likely to mask contamination hotspots. However, some of the major
610 limitations associated with CTD microplastic sampling are (i) the vessel must stop at sampling
611 stations to collect samples, (ii) deployment and retrieval of the rosette water sampler is time
612 consuming and, (iii) only small volumes of water can be collected in comparison to the underway
613 sampling. The limitation of filtering smaller volumes of water is twofold in that there can be (i)
614 false negatives whereby microplastics are not sampled despite being present in the environment
615 or, (ii) microplastics are found in the samples but scaling up to relevant units (particles m⁻³) has a
616 greater effect on microplastic abundances.

617

618 One of the major challenges that investigators face when quantifying marine microplastic
619 abundance is sample contamination. In addition to employing strict measures to control
620 contamination during sampling and processing, it is important that checks are carried out to
621 quantify potential contamination of samples. For underway samples in the present study, although
622 method blanks were free of contamination by synthetic particles, a single synthetic fibre was found
623 in one air contamination check. In context, there were between 0 – 15 synthetic particles in each
624 underway sample, with an average of 2 synthetic particles per sample. For CTD samples in the
625 present study, between 0 – 3 synthetic fibres were found in the method blanks. In context, between
626 0 – 18 synthetic particles were found per CTD sample (21 L or 48 L), with an average of 2 particles
627 per sample. In both cases, if contamination were an issue, its contribution to the reported
628 microplastic abundances in the present study would be substantial. However, the possibility of
629 airborne contamination in the actual underway samples is projected to be low since (i) 92 % of the
630 air contamination checks (11 of the 12 petri dishes) were free of synthetic particles and, (ii) air
631 contamination checks had maximum exposure to the atmosphere while actual samples had
632 minimal exposure. With respect to the CTD samples, it is proposed that synthetic fibres in the
633 method blanks may have been introduced into the Niskin bottle during the transfer of Milli-Q water
634 or could have been present from the previous CTD cast and remained in the bottle due to
635 insufficient rinsing with Milli-Q water prior to the blank. This should not have been an issue for
636 the actual samples since Niskin bottles were rinsed during the downcast and were closed within
637 the water column thus preventing the possibility of airborne contamination.

638

639 **5.0 Conclusion**

640 The present study demonstrated the pervasiveness of microplastics in sub-surface waters of the
641 Arctic Central Basin (ACB). Two independent sampling techniques led to the discovery of
642 microplastics in near surface waters of the Polar Mixed Layer (PML) i.e. at a single depth of 8.5
643 m as well as throughout the water column i.e. at multiple depths (8 – 4369 m) of the ACB. Such
644 findings confirm that microplastics are entering the central Arctic Ocean, that they are being
645 vertically transported out of surface waters and that the water column is one of the reservoirs of
646 microplastics in this region. Presently, however, uncertainty exists regarding the actual
647 mechanisms responsible for the vertical transport of microplastics in the Arctic Ocean. Although
648 there was a predominance of fibrous microplastics, the majority of which were polyester, the exact
649 sources of microplastics to the Arctic Ocean remain unknown as they could have been introduced
650 to the ecosystem via long range transport processes or originated from more local sources. The
651 fact that the highest microplastic abundances were recorded in the PML nearer to the periphery of
652 the ACB suggests the influence of location-specific factors e.g. absence of sea ice, proximity to
653 microplastic sources, wind, etc. Knowledge about microplastic abundance, distribution and
654 composition in the Arctic Ocean is vital as it provides (i) quantitative data on the concentrations
655 and types of microplastics that polar organisms are exposed to, (ii) a sound starting point for
656 investigating the potential threat that microplastics pose to the Arctic ecosystem and, (iii) insight
657 into the whereabouts of the ‘missing plastic’ from oceanic surface waters.

658

659 **Acknowledgements**

660 The authors acknowledge the invaluable support of the staff of the Swedish Polar Research
661 Secretariat (especially Jeanette Axelsson, Robert Holden, Lars Lehnert, Asa Lindgren, Axel
662 Meiton and Per Salo) and the crew of icebreaker Oden with the Arctic Ocean 2016 expedition. The

663 expert guidance of Mr. Andrew Tonkin (Plymouth University) during FT-IR analyses is also
664 acknowledged. The first author also acknowledges the support of fellow early career scientists
665 (especially Hans-Martin Heyn, Luz Maria Ramirez, Grace Shephard), other researchers (especially
666 Asa Johannisson, Sofi Jonsson, Michelle Nerentorp, Carina Johansson) and staff (especially Bengt
667 Berglund, Ted Juliusen) during the expedition.

668

669 **Funding**

670 Sampling in the Arctic Ocean was funded by the Swedish Polar Research Secretariat (SPRS) under
671 the Early Career Scientist (ECS) Programme in which the first author was a participant. This work
672 was also co-funded through a MARES Grant. MARES is a Joint Doctorate programme selected
673 under Erasmus Mundus and coordinated by Ghent University (FPA 2011-0016). The funders had
674 no role in study design, data collection, analysis and interpretation, decision to publish, or
675 preparation of the manuscript.

676

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