Abundance of plastic microbeads in Hong Kong coastal water

Wing Kwan So[a,b], Kayi Chan[a,b], Christelle Not[a,b*]

[a]Department of Earth Sciences, University of Hong Kong, Pokfulam, Hong Kong

[b]Swire Institute of Marine Science, University of Hong Kong, Cap d'Aguilar, Hong Kong

*corresponding author: Christelle Not cnot@hku.hk

Keywords:

Microplastic, Microbeads, Hong Kong, coastal water, pollution, environment

ABSTRACT

To address the rising concern over the use of plastic microbeads in personal care and cosmetic products, countries worldwide have started taking legislative actions to ban microbeads. Yet, the degree of contamination of coastal waters by plastic microbeads is rarely reported. Surface manta trawls were conducted to investigate the presence of microbeads in the southern coastal waters of Hong Kong. Considering only the size fraction of 0.3 to 1 mm, 60% of samples were found to contain microbeads. Microbeads accounted for 3.6% of the total microplastics collected and microbead abundance ranged from 0 to 380,129 pcs/km^2. The shapes, sizes, colours, and composition of microbeads found in our samples were similar to those from tested facial scrubs, suggesting that pelagic microbeads collected in this study very likely originated from the cosmetic products available locally. Microbeads represent a non-negligible part of the microplastics found in surface coastal waters.

1. Introduction

The increasing presence of plastic litter in the ocean is a growing concern of the public. Approximately 5.25 trillion plastic particles weighing 268,940 tonnes are believed to have reached the ocean between 2007 to 2013 (Eriksen et al., 2014). Plastic pieces bigger than 5 mm are identified as macroplastic and pieces equal or smaller than 5 mm are identified as microplastic by the scientific community (Arthur et al., 2009). Microplastic particles have been found in several marine environments such as gyres, coastal surface waters, the deep sea, and beaches, and their size restricts their recovery during cleanup efforts (Lee et al., 2015; Lusher et al., 2014; Moore et al., 2001; Van Cauwenberghe et al., 2013). In the marine environment, microplastic originates from either the degradation of macroplastic pieces, which then are referred to as secondary microplastic, or direct release in their original size, which then are referred to as primary microplastic (Barnes et al., 2009).

Primary microplastics are manufactured by the industry in differing sizes depending on their usage. For example, microbeads are primary microplastics smaller than 1 mm, mainly used in personal care and cosmetic products (Schneiderman, 2014). Several studies have investigated the variety of physical characteristics (i.e. shape, colour) and composition of microbeads from personal care or cosmetics products (Cheung and Fok, 2017; Fendall and Sewell, 2009; Napper et al., 2015; Schneiderman, 2015). They observed that microbeads are principally blue, white or transparent with spherical shape and mainly polyethylene (PE) but some have polypropylene (PP) or polystyrene (PS) compositions (Zitko and Hanlon, 1991; Fendall and Sewell, 2009; Napper *et al.*, 2015). Microbeads from personal care products sold in Hong Kong are principally a combination of colourless particles in granular shape and blue or colourless spherical pieces, mostly composed of low density polyethylene (LDPE; Cheung and Fok, 2017). Effluent outfall is considered the major source contributing to microbead pollution (Leslie, 2015; Magnusson and Norén, 2014;

Murphy et al., 2016). As the substitutes of natural scrubbers in personal care and cosmetics products, plastic microbeads are designed to be ditched down the drain (Rochman et al., 2015; Schneiderman, 2014). Murphy et al. (2016) estimated that microbeads are effectively but not totally captured in the grease of the wastewater treatment, leading to a non-negligible release into the aquatic environment. Between 80 to 98% of microbeads retention by wastewater treatment have been suggested, depending on the technology of the treatment plant (Duis and Coors, 2016; Murphy et al., 2016) still it was estimated that over 94,000 microbeads could wash down the drain in a single use and in each cubic meter of treated sewage discharge up to 7,000 microbeads could pass through wastewater treatment screens and directly enter the sea (Napper et al., 2015; Rochman et al., 2015). Only a few studies have investigated the presence of microbeads in the ocean as both their size and their identification are difficult in the natural environment (Cheung and Fok, 2016; Isobe, 2016). Microbeads, which have a diameter around 100-200 μm (Fendall and Sewell, 2009; Napper et al., 2015), are not collected by regular surface trawling as the mesh net is typically 335 μm. In addition, the differentiation between non-spherical microbeads and secondary microplastics in samples is problematic. With these limitations, the identification of microbeads in previous studies are clearly underestimations; yet they are the best estimates available to evaluate the role of cosmetics and personal care products on microplastic pollution in coastal waters (Eriksen et al., 2013; Isobe, 2016).

Even though recent experiments have demonstrated the role of microbeads as a transport vector for chemicals (Napper et al., 2015), their threat to the marine environment is still debated (Koelmans et al., 2017; Rist et al., 2018). However, the source of microbeads, in comparison to secondary microplastics, is unique and therefore, any changes in personal care and cosmetics regulations will directly impact the presence of microbeads in the marine environment. Over the last five years, several countries such as the UK, Canada, and New Zealand, several states in the

USA, and private sector industries have started to ban or stop the usage of microbeads in personal care products, limiting their input into the ocean (Rochman et al., 2015). Bans can be perceived as extreme but several natural replacement solutions for microbeads are available to cosmetics companies, allowing them to easily stop the contribution of microbeads to the plastic pollution observed in the ocean.

Here, we aim to quantify the microbeads present in Hong Kong coastal waters and evaluate their relative importance within the microplastic fraction. Using 147 samples from the southern region of Hong Kong, we investigated the spatial and temporal variability of the presence of microbeads.

2. Materials & Methods

2.1. Sample collection

Hong Kong is surrounded by the South China Sea and situated at the estuary of the Pearl River. Seven locations around the Hong Kong territories were chosen as sampling stations, namely Aberdeen (n = 37), Chi Ma Wan (n = 27), Discovery Bay (n = 1), Pak Kok (n = 60), Po Toi (n = 13), South Lamma (n = 1), and Tung O (n = 8) (Fig. 1). From February 2016 to April 2017, opportunistic surface water trawls were conducted at each site and a total of 147 samples were gathered throughout the year. A 15-minute surface manta trawl, using a sampling net with opening of 0.5 x 1m and a mesh size of 335μm, was performed at a constant speed of 2 knots to collect samples. Samples were first stored in a glass bottle, transported to the laboratory at the University of Hong Kong, and kept inside the refrigerator until they were processed. Abundance of microbeads is presented by surface area estimated as follow:

Area (km^2) = speed of vessel (ms^{-1}) x duration of the trawl (s) x width of the net (m) / 1000000

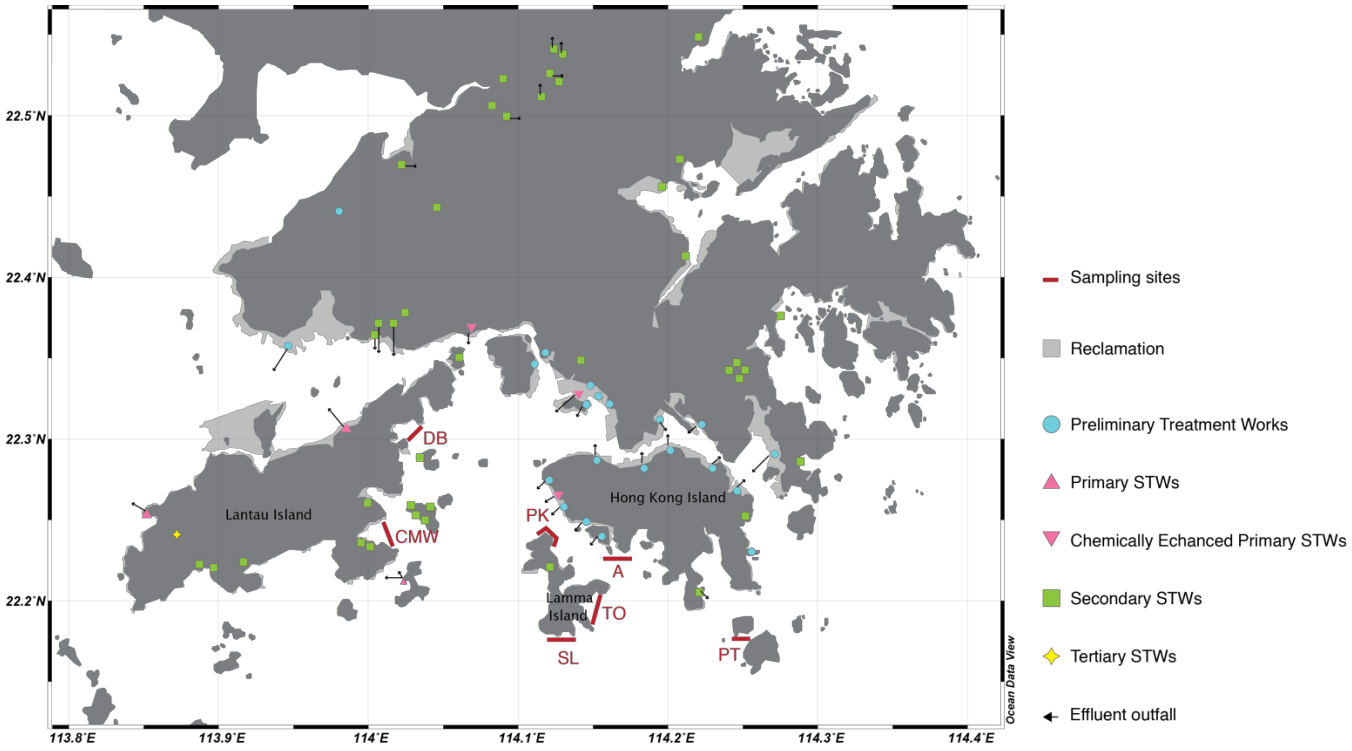


Figure 1. Map of Hong Kong with the seven sampling locations: Aberdeen (A), Chi Ma Wan (CMW), Discovery Bay (DB), Pak Kok (PK), Po Toi (PT), South Lamma (SL), and Tung O (TO). Sewage treatment works are located with information about their effluent outfall direction and the type of treatment (Drainage Servicess Department, 2015). Dark grey represents the natural coastline whereas the light grey represents coastline after land reclamation.

2.2. Sample processing

Each sample was separated into two size fractions by sieving through a 5 mm and 0.3 mm stainless-steel mesh sieve. All materials remaining on the sieves were transferred to clean beakers and dried completely in an oven at 50°C for 24 hours. The smaller size fraction samples (0.3-5 mm) underwent wet peroxidation using 30% hydrogen peroxide and 0.05M aqueous iron (II) sulphate solution at 70°C to oxidize organic matters (Masura et al., 2015). If organic matter was still visible, samples were further subjected to alkaline digestion with 10% potassium hydroxide solution (Foekema et al., 2013). Sodium chloride was later added to the samples to increase the aqueous density and samples were settled in filter funnels overnight for density separation. Only the floating plastic debris were collected and screened through a 0.3 mm mesh sieve (Masura et al., 2015).

112 Samples were again thoroughly dried in an oven at 50ºC for 24 hours and separated into two size
113 fractions of 1 to 5 mm and 0.3 to 1 mm by sieving through stainless-steel sieves with mesh sizes
114 of 1 mm and 0.3 mm. Plastic samples smaller than 1 mm were preserved in glass petri dishes for
115 visual sorting of microbeads under a stereomicroscope (40x; stemi 305; Carl Zeiss Microscopy
116 GmbH). Effort to limit sample contamination included ensuring that all materials used in sample
117 processing were thoroughly cleaned before usage and covered after each step. Microplastic
118 particles with spherical shapes within the 0.3-1 mm size fraction were considered microbeads and
119 as such were sorted and counted. Due to the impossibility to distinguish granular microbeads with
120 secondary microplastic produced by the degradation of bigger plastic pieces combined with the
121 limitation of the mesh size of the net used to collect the sample, microbeads abundance presented
122 below is underestimated as in the only two previous studies, yet it represents a primary estimate.
123
124 *2.3. Fourier Transform Infrared (FTIR) Spectroscopy Verification*
125 Suspected microbeads (hereafter regarded as "microbeads") were selected for FTIR (Spectrum
126 Two, Perkin Elmer) verification. A random selection of 95 microbeads, with at least one from each
127 sample where microbeads were present, were scanned with a spectral resolution of 4 cm^{-3} at a
128 range of 4000 cm^{-1} to 400 cm^{-1}. Samples were identified with the aid of Perkin Elmer standard
129 polymer spectrums and compared to the composition of microbeads from the locally sold facial
130 scrubs tested in Cheung and Fok (2017).
131
132 *2.4. Data analysis*
133 All statistics were performed in R Studio using R version 3.4.0 (R Core Team, 2017). In order to
134 test the potential spatial and temporal variability of microbead abundance in coastal surface waters,
135 a one-way ANOVA comparing the difference in microbead abundance between different sites,

months and seasons was performed. In addition, to evaluate the impact of weather parameters such as rainfall and wind magnitude, and sewage volume outflow and total suspended solids on the abundance of microbeads, we performed correlation analyses using the 'ggpubr' package (Kassambara, 2017). Spearman correlations were chosen due to a lack of normality in the data. Rain and wind data were obtained from the Hong Kong Observatory and sewage data was obtained from the Drainage Services Department of Hong Kong.

3. Results

More than half of the samples (60%) collected from seven sampling sites contained microbeads. A total of 1698 spherical microbeads, accounting for 3.59% of all microplastics, were found in the 147 samples examined (Table 1). The abundance of microbeads found per sample ranged from 0 to 380,129 pcs/km^2 with an average of 12,429 ± 3,929 pcs/km^2.

Table 1: Mean abundance and percentage of microbeads in Hong Kong.

Site		Size (mm)	Mean abundance (pcs/km^2)	Standard deviation (pcs/km^2)	Range (pcs/km^2)	no. of MBs	no. of MPs	%
Southern Hong Kong	**(n = 147)**	**0.3-1**	**12429**	**47634**	**0 - 380129**	**1698**	**47241**	**3.59**
Aberdeen	(n = 37)	0.3-1	3495	6310	0 - 29158	127	3518	3.61
Chi Ma Wan	(n = 27)	0.3-1	8759	19666	0 - 87473	219	3768	5.81
Discovery Bay	(n = 1)	0.3-1	15119	n.a.	15119	14	83	16.87
Pak Kok	(n = 60)	0.3-1	17298	56660	0 - 380129	960	30132	3.19
Po Toi	(n = 13)	0.3-1	2907	6821	0 - 24838	35	972	3.60
South Lamma	(n = 1)	0.3-1	0	n.a.	0	0	27	0.00
Tung O	(n = 8)	0.3-1	46301	127925	0 - 362851	343	8741	3.92

Microbeads found in the samples were mainly transparent (62%), black (22%), white (10%), blue (3%), and green (2%) (Figure 3). Most of them were spherical in shape (97%) and some were grape-like (3%). Out of the 95 microbeads analysed by FT-IR, 55% were identified as polyethylene (PE), 27% as polypropylene (PP), 7% as polystyrene (PS) and 11% as mixed plastic composition (Figure 2 and Figure 3), indicating a 100% rate of identification of plastic microbeads.

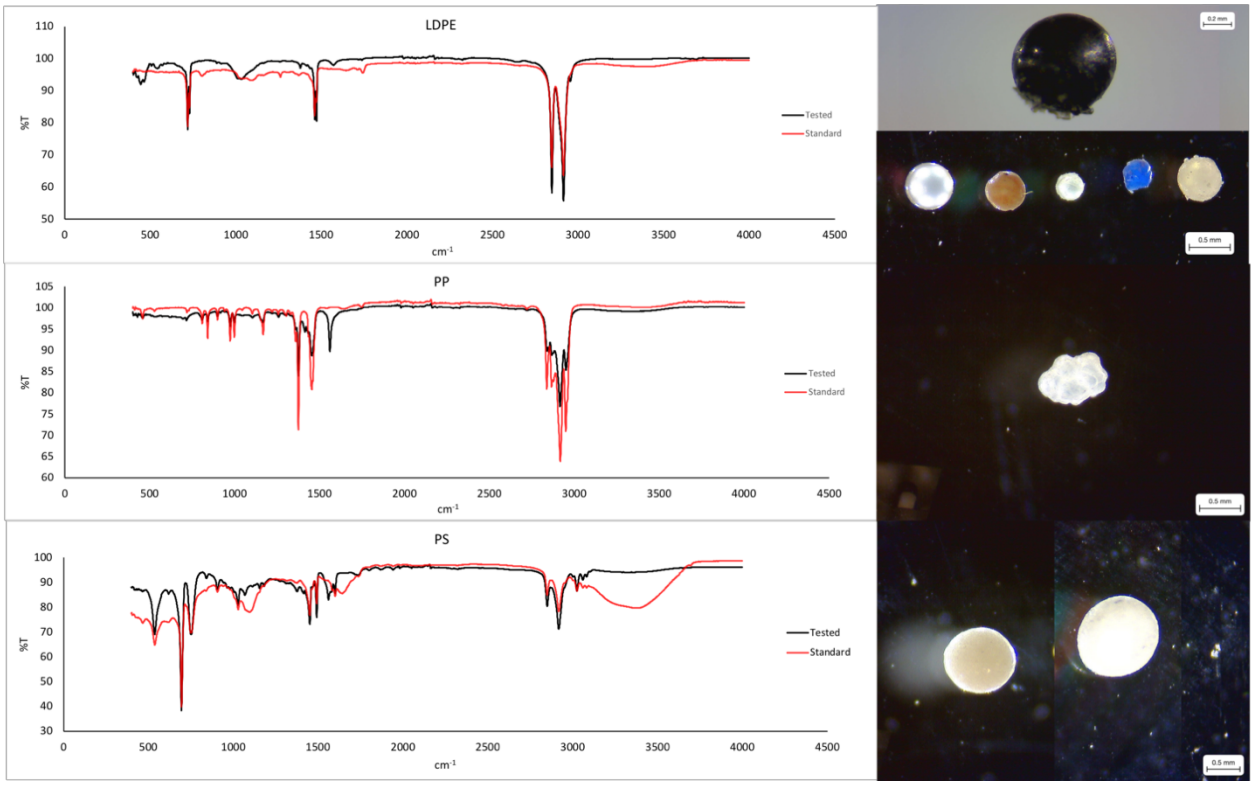


Fig. 2. Photo of microbeads present in Hong Kong water with their composition spectrum obtained by FT-IR.

Samples from Tung O showed the highest mean concentration (46301± 45228, n = 8), followed by Pak Kok (17298 ± 7315, n = 60), Discovery Bay (15119, n = 1), Chi Ma Wan (8759 ± 3785, n = 27), Aberdeen (3495 ± 1037, n = 37), Po Toi (2907 ±1892, n = 13), and finally South Lamma which was free of microbeads (n=1; Table 1). However no statistically significant difference was observed between the locations (p = 0.16). Interestingly, the percentage of microbeads per site had a different spatial variability. Samples from Discovery Bay showed the highest percentage (16.9%), followed Chi Ma Wan (5.8%), Tung O (3.9%), Aberdeen (3.6%), Po Toi (3.6%), Pak Kok (3.2%), and lastly South Lamma (0%; Figure 3) but this observation is driven by Discovery Bay where only one sample was collected and without this sample, the proportion of microbeads is quite similar between the sites.

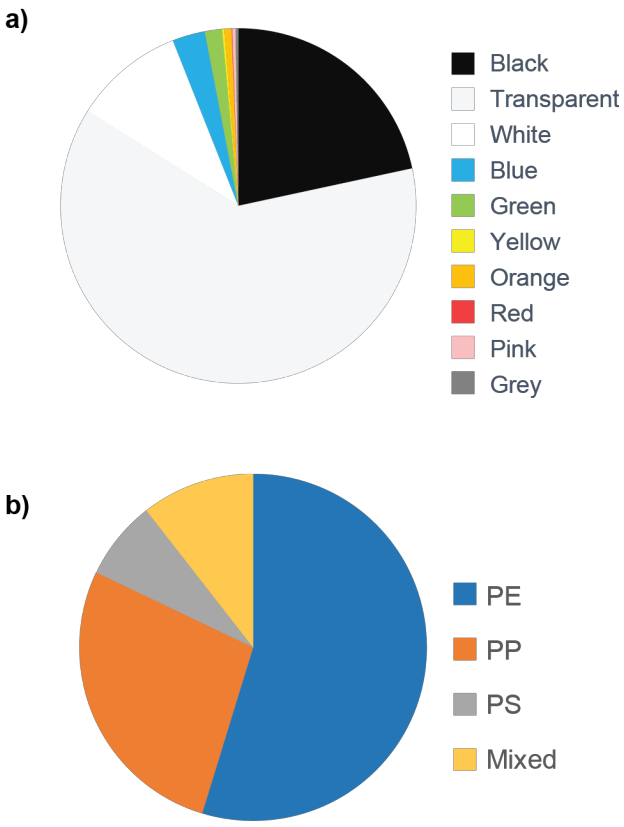
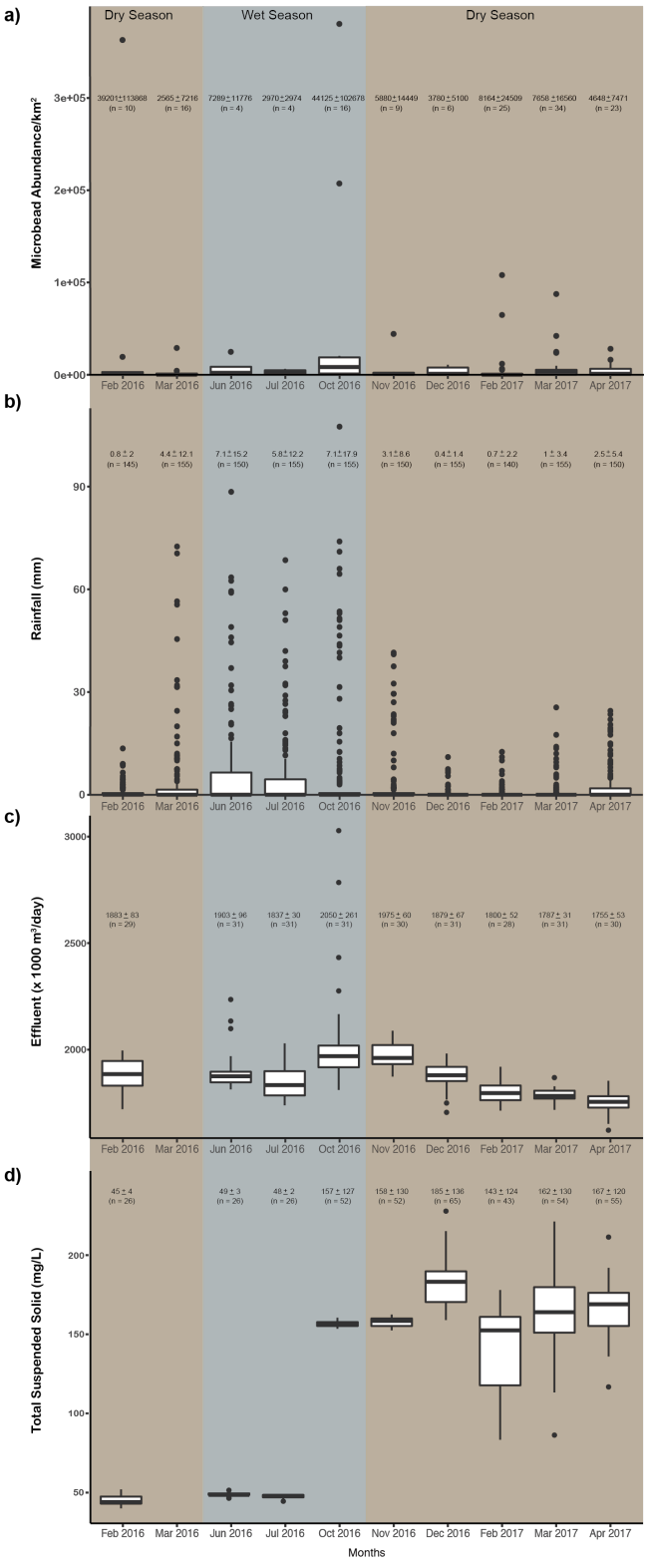


Fig. 3. Proportion colour composition from the 1698 microbeads collected within Hong Kong water (a) Proportion of the chemical composition of the 95 microbeads measured by FT-IR (b).

The average concentration of microbeads found in the sampling sites remained around 12429 ± 47634 pieces/km^2 throughout the year and no significant difference was observed between months. However, three samples with very high abundance were considered as outliers. One sample was taken on February 4th, 2016 and two were taken on October 20th, 2016. Both in absence and presence of the outliers, a significant difference is observed between seasons (wet: November to April vs dry: May to October), with the wet season having a higher abundance of microbeads.



182

183 Fig. 4. Temporal variability in the abundance of microbeads in surface water from Hong Kong (a),
184 in the rainfall (b), volume outflow of waste water treatment plant (c) and quantity of total
185 suspended solid (d) between February 2016 to April 2017.

4. Discussion

Hong Kong, with its high population density, is believed to be at high risk of pollution by plastic microbeads. We found a total of 1698 microbeads within the 75 days of sampling. Colours and composition of the microbeads found in our samples matched with the study of microbeads from local personal care and cosmetic products done by Cheung and Fok (2017) where transparent microbeads and LDPE/PE composition were dominant (Figure 3). Therefore, we are confident that all microbeads gathered originated from the personal care and cosmetic products sold in Hong Kong or in the Pearl River area. Considering various factors such as the percentage of the population using facial scrubs, density of microbeads in facial scrubs, escape rate of microbeads, etc., it was estimated that 342.2 billion microbeads were emitted into Hong Kong waters (Cheung and Fok, 2016). Here we collected a total of 1698 microbeads, suggesting that only a small quantity of microbeads is found in coastal waters. However, direct comparison between microbeads discharge and abundance in surface water have to be done carefully since several factors will affect the quantification of microbeads in the water. As mentioned in the introduction, microbeads correspond to synthetic exfoliators which are usually spherical, but irregularly shaped microbeads are also present with general size ranges between 0.1 mm and 1 mm (Gouin et al., 2015; Napper et al., 2015; Rochman et al., 2015). The size of microbeads present in locally sold cosmetic products was typically between 0.024 mm and 1 mm (Cheung and Fok, 2017). Therefore, counting only spherical microplastic from the 0.3 to 1 mm size fraction underestimates the microbeads present in the surface water; thus, our estimate should be taken as the minimum abundance of microbeads in marine waters. More precise estimations are technically limited by the complexity involved in differentiating between irregularly shaped microbeads and secondary microplastics,

and the common usage of 335μm mesh net. Consequently, 60% of the surface water samples, with between 1 and 380129 pcs/km^2 microbeads, constitute a conservative evaluation of the microbead contamination of Hong Kong waters. In addition, sewage treatment works have been shown to effectively but not totally captured microbeads in the grease of the wastewater treatment, leading to non-negligible release in the aquatic environment (Murphy et al., 2016). In Hong Kong, 69 sewage treatment works with different levels of technological advances are operated to handle the domestic sewage from different districts (Figure 1; Drainage Services Department, 2017) but for all of them the grease removal is undertaken, which results in a significant quantity of microbeads retained. Yet, about 7% of the population (510,000 persons) in Hong Kong reside in rural areas that are not connected to the public sewage facilities but rely on some rather simple wastewater treatment systems such as septic-tank-and-soakaway and dry-weather-flow interceptors to handle the untreated sewage (Aduit Commissiom, 2016; Cheung and Fok, 2016). Here, despite the fact that most of the waste waters treated will limit the input of microbeads in coastal water, the technical limitation of identification of irregular microbeads, and the absence of collection of the microbeads smaller than 0.3 mm, we show that microbeads form a non-negligible proportion of the microplastics present in Hong Kong surface coastal waters. The close match between both the colour and the composition of the microbeads found in coastal waters with those from cosmetics and personal care products sold in Hong Kong confirm the sources of the microbeads present in the surface coastal waters.

No significant spatial difference is observed in the abundance of microbeads in surface waters or the number of samples containing microbeads between the different locations sampled, suggesting that the distance or the type of the wastewater treatment plant (primary vs secondary) does not influence the quantity of microbeads present. For example microbead abundance at Discovery Bay or Chi Ma Wan, which are close to several secondary sewage treatment plants, is not lower than

234 in Aberdeen or Pak Kok, which are close to primary sewage treatment plants. Overall, our data
235 suggests the absence of a link between the proximity or type of sewage treatment plant and the
236 abundance of microbeads in the surrounding waters. Thus, it supports the idea that grease removal,
237 which is performed at all sewage water treatment plants in Hong Kong, is efficient at retaining
238 microbeads (Murphy et al., 2016).
239 No significant difference in the microbead abundance is observed between the different months
240 (Figure 4). We also observed no correlation between the abundance of microbeads and rainfall
241 ($p=1.00$), wind intensity ($p=0.18$), or quantity of total suspended solids ($p=0.23$), suggesting that
242 none of these parameters influence the quantity of microbeads present in surface coastal water.
243 However, a significant difference is observed between seasons, where the wet season (June to
244 October) has a significant higher abundance of microbeads when rainfalls are higher, more
245 variable and driven by extreme events. Also, a positive correlation between rainfall and sewage
246 outflow is observed, validating the link between sewage effluent and microbeads release in coastal
247 water. In addition, we observed three outliers with abundance at least an order of magnitude higher
248 than the average abundance observed during the sampling period. One of these sample was collect
249 on February 4th, 2016 and two were collected on October 20th, 2016. These two days are
250 characterized by rainfall and sewage outflow higher than the average for the sampling period and
251 higher than the average of their respective month, suggesting that these three parameters influence
252 the abundance of microbeads, especially during "extreme" events. For example, on October 20th,
253 the average rainfall of the previous two days was 71 mm and the average sewage outflow was
254 2905 (x1000 m^3/day) in comparison to the October 2016 month average of 7 mm of rainfall and
255 2050 (x 1000 m^3/day). Previous studies have also highlighted the role of major rain events and
256 storm water increase the abundance of microplastic collected (Lattin et al., 2004; Moore et al.,
257 2002; Yonkos et al., 2014). A positive correlation is observed between the abundance of

258 microbeads and the volume of sewage outflow when outliers are included, validating the impact
259 of extreme event in the abundance of microbeads in surface waters. In comparison, wind has been
260 recognized to enhance microplastic transport to the deeper part of the ocean and therefore decrease
261 the abundance of microplastic in surface waters (Kukulka et al., 2012; Reisser et al., 2015). Here,
262 wind intensity was not different between these specific days and the average of the sampling period
263 or the average of the month. In addition, we observed a much higher variability in the data from
264 the dry months (November 2016 to April 2017), when sampling efforts were concentrated. This
265 suggests that the opportunistic sampling conducted had an influence and did not allow us to cover
266 the full range of weather conditions, especially rainfall. When looking at the rainfall data over the
267 sampling period we noticed that several days have more than 100 mm of rain, but none of these
268 days were sampled. Our data suggests a link between rainfall, and therefore sewage outflow, with
269 microbeads abundance in surface water. More samples are needed to confirm the direct link or the
270 existence of a threshold.
271

272 **5. Conclusion**

273 Microbeads, spherical plastic pieces from 0.3 to 1 mm in size, has been identified in 88 surface
274 water samples from Hong Kong coastal waters. Based on comparable shapes, colours and
275 compositions, the microbeads collected most likely originated from personal care and cosmetics
276 products sold in Hong Kong. The average abundance of microbeads in the surface water was
277 estimated at 12429 ± 47634 pieces/km^2. However, this estimate represents a minimum value of
278 the microbeads present since it does not take into account microbeads smaller than 0.3 mm and
279 microbeads with irregular shape. Irrespective of this limitation, our study confirms that microbeads
280 represent a non-negligible part of the microplastic pollution of Hong Kong coastal water.
281

Acknowledgement

We would like to express our gratitude to the non-governmental organization, Plastic Free Seas, which collected all the samples as part of their outreach program. We thank the students of the environmental oceanography course at the University of Hong Kong years 2016 and 2017 for the preliminary sample processing as well as the Gallant Ho Experiential Learning Fund, HKU granted to C. Not. We also want to thank Poon Sing Chau and Tang Tsz Wing Lilia from The University of Hong Kong for their assistance with sample processing and analysis. Finally, part of the research was supported by the start-up fund of the University of Hong Kong awarded to C. Not.

References

Aduit Commissiom, 2016. SEWERAGE SYSTEMS IN RURAL AREAS Executive Summary.

Arthur, C., Baker, J., Bamford, H., 2009. Proceedings of the International Research Workshop on the Occurrence , Effects , and Fate of Microplastic Marine Debris. Group 530.

Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. B Biol. Sci. 364, 1985–1998. doi:10.1098/rstb.2008.0205

Cheung, P.K., Fok, L., 2017. Characterisation of plastic microbeads in facial scrubs and their estimated emissions in Mainland China. Water Res. 122, 53–61. doi:10.1016/j.watres.2017.05.053

Cheung, P.K., Fok, L., 2016. Evidence of microbeads from personal care product contaminating the sea. Mar. Pollut. Bull. 1–4. doi:10.1016/j.marpolbul.2016.05.046

Drainage Service Department, 2017. List of Sewage Treatment Facilities in Hong Kong [WWW Document].

Drainage Service Department, 2015. Sustainability Report 2014-2015.

306 Duis, K., Coors, A., 2016. Microplastics in the aquatic and terrestrial environment: sources (with
307 a specific focus on personal care products), fate and effects. Environ. Sci. Eur. 28, 2.
308 doi:10.1186/s12302-015-0069-y
309 England, N., Estuary, S., Channel, B., Atlantic, N., Estuary, S., Zealand, N., 1991. Another
310 Source of Pollution by Plastics : Skin Cleaners with Plastic Scrubbers Pesticide Residues in
311 Sediments from the West Coast of India 22, 41–42.
312 Eriksen, M., Lebreton, L.C.M., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F.,
313 Ryan, P.G., Reisser, J., 2014. Plastic Pollution in the World's Oceans: More than 5 Trillion
314 Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. PLoS One 9, 1–15.
315 doi:10.1371/journal.pone.0111913
316 Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S.,
317 2013. Microplastic pollution in the surface waters of the Laurentian Great Lakes. Mar.
318 Pollut. Bull. 77, 177–182. doi:10.1016/j.marpolbul.2013.10.007
319 Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face:
320 Microplastics in facial cleansers. Mar. Pollut. Bull. 58, 1225–1228.
321 doi:10.1016/j.marpolbul.2009.04.025
322 Foekema, E.M., De Gruijter, C., Mergia, M.T., Van Franeker, J.A., Murk, A.J., Koelmans, A.A.,
323 2013. Plastic in north sea fish. Environ. Sci. Technol. 47, 8818–8824.
324 doi:10.1021/es400931b
325 Gouin, T., Avalos, J., Brunning, I., Brzuska, K., Graaf, J. de, Kaumanns, J., Koning, T.,
326 Meyberg, M., Rettinger, K., Schlatter, H., Thomas, J., Welie, R. van, Wolf, T., 2015. Use of
327 Micro-Plastic Beads in Comestic Products in Europe and Their Estimated Emissions to the
328 North Sea Environment. SOFW-Joural 141.
329 Isobe, A., 2016. Percentage of microbeads in pelagic microplastics within Japanese coastal

waters. Mar. Pollut. Bull. 110, 432–437. doi:10.1016/j.marpolbul.2016.06.030

Kassambara, A., 2017. ggpubr: 'ggplot2' Based Publication Ready Plots.

Koelmans, A.A., Besseling, E., Foekema, E., Kooi, M., Mintenig, S., Ossendorp, B.C., Redondo-Hasselerharm, P.E., Verschoor, A., Van Wezel, A.P., Scheffer, M., 2017. Risks of Plastic Debris: Unravelling Fact, Opinion, Perception, and Belief. Environ. Sci. Technol. 51, 11513–11519. doi:10.1021/acs.est.7b02219

Kukulka, T., Proskurowski, G., Morét-Ferguson, S., Meyer, D.W., Law, K.L., 2012. The effect of wind mixing on the vertical distribution of buoyant plastic debris. Geophys. Res. Lett. 39, 1–6. doi:10.1029/2012GL051116

Lattin, G.L., Moore, C.J., Zellers, A.F., Moore, S.L., Weisberg, S.B., 2004. A comparison of neustonic plastic and zooplankton at different depths near the southern California shore. Mar. Pollut. Bull. 49, 291–294. doi:10.1016/j.marpolbul.2004.01.020

Lee, J., Lee, J.S., Jang, Y.C., Hong, S.Y., Shim, W.J., Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Kang, D., Hong, S., 2015. Distribution and Size Relationships of Plastic Marine Debris on Beaches in South Korea. Arch. Environ. Contam. Toxicol. 69, 288–298. doi:10.1007/s00244-015-0208-x

Leslie, H.A., 2015. Plastic in Cosmetics: Are we polluting the environment through our personal care?

Lusher, A.L., Burke, A., O'Connor, I., Officer, R., 2014. Microplastic pollution in the Northeast Atlantic Ocean: Validated and opportunistic sampling. Mar. Pollut. Bull. 88, 325–333. doi:10.1016/j.marpolbul.2014.08.023

Magnusson, K., Norén, F., 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant, IVL Swedish Environmental Research Institute. doi:naturvardsverket-2226

354 Masura, Julie, Baker, Joel, Foster, Gregory, Arthur, C., 2015. Laboratory Methods for the
355 Analysis of Microplastics in the Marine Environment.
356 Moore, C.J., Moore, S.L., Leecaster, M.K., Weisberg, S.B., 2001. A comparison of plastic and
357 plankton in the North Pacific Central Gyre. Mar. Pollut. Bull. 42, 1297–1300.
358 doi:10.1016/S0025-326X(01)00114-X
359 Moore, C.J., Moore, S.L., Weisberg, S.B., Lattin, G.L., Zellers, A.F., 2002. A comparison of
360 neustonic plastic and zooplankton abundance in southern California's coastal waters. Mar.
361 Pollut. Bull. 44, 1035–1038. doi:10.1016/S0025-326X(02)00150-9
362 Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater Treatment Works (WwTW)
363 as a Source of Microplastics in the Aquatic Environment. Environ. Sci. Technol. 50, 5800–
364 5808. doi:10.1021/acs.est.5b05416
365 Napper, I.E., Bakir, A., Rowland, S.J., Thompson, R.C., 2015. Characterisation, quantity and
366 sorptive properties of microplastics extracted from cosmetics. Mar. Pollut. Bull. 99, 178–
367 185. doi:10.1016/j.marpolbul.2015.07.029
368 R Core Team, 2017. R Core Team (2017). R: A language and environment for statistical
369 computing. R Found. Stat. Comput. Vienna, Austria. URL http//www.R-project.org/. R
370 Foundation for Statistical Computing.
371 Reisser, J., Slat, B., Noble, K., Du Plessis, K., Epp, M., Proietti, M., De Sonneville, J., Becker,
372 T., Pattiaratchi, C., 2015. The vertical distribution of buoyant plastics at sea: An
373 observational study in the North Atlantic Gyre. Biogeosciences 12, 1249–1256.
374 doi:10.5194/bg-12-1249-2015
375 Rist, S., Carney Almroth, B., Hartmann, N.B., Karlsson, T.M., 2018. A critical perspective on
376 early communications concerning human health aspects of microplastics. Sci. Total
377 Environ. 626, 720–726. doi:10.1016/j.scitotenv.2018.01.092

Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Scientific Evidence Supports a Ban on Microbeads. Environ. Sci. Technol. 49, 10759–10761. doi:10.1021/acs.est.5b03909

Schneiderman, E.T., 2015. Discharging Microbeads to Our Waters: An Examination of Wastewater Treatment Plants in New York.

Schneiderman, E.T., 2014. Unseen Threat: How Microbeads Harm New York Waters, Wildlife, Health and Environment.

Van Cauwenberghe, L., Vanreusel, A., Mees, J., Janssen, C.R., 2013. Microplastic pollution in deep-sea sediments. Environ. Pollut. 182, 495–499. doi:10.1016/j.envpol.2013.08.013

Yonkos, L.T., Friedel, E.A., Perez-Reyes, A.C., Ghosal, S., Arthur, C.D., 2014. Microplastics in Four Estuarine Rivers in the. doi:10.1021/es5036317