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Spatio-temporal comparison of neustonic microplastic density in Hong Kong waters under the influence of the Pearl River Estuary

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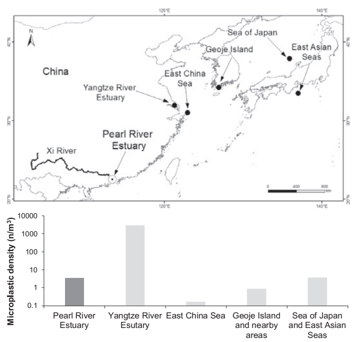
HIGHLIGHTS

- Sea surface plastics were collected in Hong Kong waters in two seasons.
- Mean microplastic (MP) density on west coast was 3.627 n/m^3 or 304,816 n/km^2.
- Plastic debris densities were significantly higher in rainy season than dry season.
- No spatial difference in MP density.
- Major types of polymers were PP/EPR and PE (~30 % each).

GRAPHICAL ABSTRACT



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ABSTRACT

Rivers are recognised as an important source of plastic debris in the open sea. The Pearl River in China is estimated to transport 0.1 million tonnes of plastic waste to the open sea annually. However, no empirical study has been conducted to assess the plastic contamination levels in the Pearl River Estuary. Hong Kong is situated in the east of the Pearl River Estuary; its western waters are strongly influenced by river discharge, whereas the eastern waters are unaffected by the freshwater plume. In this study, we quantified the neustonic plastic debris density in the western and eastern waters of Hong Kong. The mean microplastic (0.355–4.749 mm) and large plastic debris (≥4.75 mm) densities in the western side were 3.627 and 0.758 n/m^3, respectively. Seasonal comparisons indicated that both size classes of plastic debris were significantly more abundant by number in the rainy season than the dry season ($p < 0.001$). However, the influence of rivers on plastic density at the sea surface may be highly restricted to the estuarine delta, as no significant spatial difference was found between the western and eastern waters.

1. Introduction

Plastic refers to a type of synthetic polymer that possesses distinctive properties. Plastic is light, inexpensive, flexible, mouldable and resistant to biodegradation. These properties provide a high cost-performance advantage, promoting the wide use of this material in industrial production of consumer goods. Despite the fact that plastic materials have only

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been mass-produced for approximately 70 years, the annual global production reached 322 million tonnes in 2015, and approximately 40% of that plastic was used in single-use applications, such as packaging (PEMRG, 2016). Plastic surpassed steel in terms of production volume and became the most-used material in the 1980s. Without question, humans are currently living in the age of plastics (Zalasiewicz et al., 2016). At the same time, the mass production of this convenient material has allowed a throwaway culture to flourish. The development of effective waste management solutions to address post-consumption plastic products has not been able to catch up. The net result is a large amount of plastic waste that is disposed of every year, and some of it eventually ends up in the final sink of the planet: our oceans. It has been estimated that 4.8 to 12.7 million tonnes of plastic waste entered the ocean from land in 2010 (Jambeck et al., 2015). Plastic debris in the ocean was recognised by the United Nations Environment Programme (UNEP) as an emerging global environmental issue (Kershaw et al., 2011) and represents a threat to the sustainable development of a region.

Recently, the concerns over microplastics in the environment have increased dramatically in the scientific community (Halden, 2015). The Group of Experts on Scientific Aspects of Marine Environmental Protection (GESAMP) defined microplastics as plastic debris with particle sizes <5 mm in diameter (Kershaw, 2015). Microplastics are potentially harmful, both physically and chemically, to a wide range of biota, including zooplankton (Cole et al., 2013), bivalves (Li et al., 2016a; Van Cauwenberghe and Janssen, 2014) and fishes (Güven et al., 2017; McGoran et al., 2017; Steer et al., 2017). Microplastics can cause physical damage to an organism through ingestion (Wright et al., 2013). Moreover, although a modelling study suggested that the transfer of chemical pollutants from microplastics to animals upon ingestion is unlikely to be an important route of contamination (Koelmans et al., 2016), microplastics themselves constitute a serious threat to organisms and should be classified as persistent organic pollutants (Lohmann, 2017) and hazardous materials (Rochman et al., 2013). The origins and potential impacts of plastic debris in general and microplastics in particular have been reviewed in several articles (Kershaw, 2015; Li et al., 2016b; Peng et al., 2017). The impact of microplastic pollution may extend to human health as some bivalves and small fishes are consumed without removal of the gastrointestinal tract. It was estimated that the consumption of 225 g mussels would introduce 900 microplastics (7 μg) to the human body in the worst case scenario (Li et al., 2016a; Lusher et al., 2017). However, the entailed exposure to persistent, bioaccumulative and toxic compounds (PBTs) only contributes to <0.1% of the total dietary exposure (Lusher et al., 2017). In addition, it is difficult to assess human health impact of microplastics, given our simultaneous exposure to a variety of substances (Sedlak, 2017).

The Pearl River Delta featured the largest urban area in the world in 2010 (Deuskar et al., 2015). The amount of plastics produced and discarded in this region is enormous. In 2015, 78 and 5.8 million tonnes of raw plastic materials were produced in China and Guangdong Province, respectively (National Bureau of Statistics of China, 2017). In addition, waste collection in the rural areas of China is rudimentary at best and is often non-existent (Wang et al., 2017a). As it rains, mismanaged plastic waste is transported into the oceans due to the generally low relative densities of plastic materials. A recent study estimated that the Pearl River, as the third largest contributor of river plastics to the ocean, emits 0.1 million tonnes of plastics to the South China Sea every year (Lebreton et al., 2017). Large amounts of marine litter have been observed in both the littoral and marine environments in the South China Sea (Zhou et al., 2011). The microplastics in the littoral zones of the region have also been recorded as some of the most abundant in the world (Fok et al., 2017; Qiu et al., 2015) and have been measured at >5000 items per m^2 on sandy beaches (Fok and Cheung, 2015). In addition, the spatial distribution of marine debris has suggested that the Pearl River Estuary is a significant source of marine plastic pollution in the region (Cheung et al., 2016), while the numerical abundance of microplastics at the surface of semi-enclosed seas of Hong Kong reportedly range from 0.5–279 particles per m^3 (Tsang et al., 2017).

Although a modelling study suggested that the Pearl River is the third most polluting river in the world (Lebreton et al., 2017), no empirical study has been conducted to quantify the plastic pollution level in this region. This study aims to assess the spatio-temporal distribution of large plastic debris and microplastics in Hong Kong waters.

2. Materials and methods

2.1. Study area

The Pearl River is a macroscale catchment with an area of 453,700 km^2 (Pearl River Water Resources Commission, 2017). The population living inside the Pearl River catchment in China was over 200 million in 2015 (Cheung and Fok, 2017). The lower reaches and the delta of the river lie in Guangdong Province, which has a humid subtropical climate under the influence of the East Asian monsoon (Köppen-Geiger Climate Classification: Cwa/Cfa). In 2015, the average annual rainfall in Guangdong Province was 1841.7 mm (source: Guangdong Meteorological Service, http://www.grmc.gov.cn/qxgk/tjsj/index.html), of which >70% fell during the summer (April–September) (Fig. S1).

Fifteen sites were sampled in Hong Kong waters, covering both the western waters (influenced by the Pearl River) and the eastern waters (uninfluenced by the Estuary) of Hong Kong (Fig. 1; Table S1). Sites W1 to W9 were chosen to represent the Pearl River Estuary (PRE) because previous studies indicated that the freshwater plume of the river can cover these sites during the wet season (Lee et al., 2006) and significantly augments the abundance of plastic debris at the estuarine beaches (Fok and Cheung, 2015). Sites E1 to E6 were largely unaffected by the Pearl River and could reflected local influence. Sampling further towards the centre and west side of the PRE will better reveal the plastic pollution level in the estuary. However, we were only able to sample within the boundary of Hong Kong SAR.

2.2. Sampling

Sea surface samples were collected at the fifteen sites once during the dry season (February 2015) and once during the rainy season (July 2015). The samples were collected by a Manta net system (Ocean Instruments Inc., San Diego, USA) with a net aperture of 87 × 16 cm and a mesh of 333 μm attached to a cod-end. Attached to the front centre of the net frame, a mechanical flowmeter (General Oceanics Inc., model: 2030R) were used to measure the volume of filtered sea water. This system has been adopted by several pelagic microplastic studies (e.g. Kang et al., 2015; Moore et al., 2002). At each sampling location, two nets were towed, one from the port and one from the starboard, at an average speed of 2 knots for 20 min. The average density of the two duplicates were used in subsequent statistical analysis. The trawls were carried out at a skewed horizontal angle away from the wakes of the vessel. Vertical mixing in the wake zone is known to reduce the amount of floatable debris on the surface, including microplastics (Kooi et al., 2016). After each trawl, the nets were rinsed with sea water from the outside until all debris was washed into the cod-ends. The volume-reduced samples were then transferred from the cod-ends to sealable plastic bags and delivered to the laboratory on the same day.

2.3. Digestion and size separation

In the laboratory, the working benches were cleaned, and the tools were rinsed using filtered and deionised water before use. Cotton lab coats were worn to prevent synthetic fibres from falling off clothes. All