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Microplastic analysis in the South Funen Archipelago, Baltic Sea, implementing manta trawling and bulk sampling

Matthias Tamminga, Elena Hengstmann, Elke Kerstin Fischer*

Center for Earth System Research and Sustainability (CEN), University of Hamburg, Bundesstraße 55, 20146 Hamburg, Germany

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ABSTRACT

Microplastic contamination in surface waters of the South Funen Archipelago in Denmark was assessed. Therefore, ten manta trawls were conducted in June 2015. Moreover, 31 low-volume bulk samples were taken to evaluate, whether consistent results in comparison to the net-based approach can be obtained. Microplastic contamination in the South Funen Archipelago (0.07 ± 0.02 particles/m^3) is slightly below values reported before. The sheltered position of the study area, low population pressure on adjacent islands and the absence of any major potential point sources were identified as major factors explaining the low concentration of microplastics. Within the Archipelago, harbors or marinas and the associated vessel traffic are the most probable sources of microplastics. The concentration of microplastics in low-volume bulk samples is not comparable to manta trawl results. This is mainly due to insufficient representativeness of the bulk sample volumes.

1. Introduction

The contamination of natural habitats with plastic litter has become an emerging topic for environmental scientists in recent decades (Galgani, 2015; Galgani et al., 2013; Wright et al., 2013; Thompson, 2004). The long-term durability of most plastic polymers, increasing production rates on a global scale, the unsustainable usage of plastics and inadequate waste management have led to the accumulation of plastics in ecosystems worldwide (PlasticsEurope, 2016; Barnes et al., 2009).

Microplastics in particular can pose various threats to ecosystems. Plastic particles may act as a transporting vector for toxic contaminants, when compounds such as Persistent Organic Pollutants (POPs) or heavy metals adsorb to the surface (GESAMP, 2015). Moreover, it was shown that various species can misinterpret microplastics for their actual food (Lusher et al., 2016, Battaglia et al., 2016). Besides potential negative impact on the respective metabolism (Lu et al., 2016; Watts et al., 2016), an accumulation of adsorbed contaminants into the food chain has to be considered, which includes potential threats to humans, as well (Galgani et al., 2015; Van Cauwenberghe et al., 2015; Cole et al., 2013). However, the impacts of plastic debris exceed solely ecological consequences. Economical (e.g. disadvantages for fishery and tourism) as well as social effects (e.g. reduced recreational value of natural landscapes) have to be taken into account and emphasize the need for further research (Newman et al., 2015). In general, the harmonization of sampling and analysis methods and the extension of the data base on microplastic abundances have largely been identified as important research topic (GESAMP, 2015; Löder and Gerdts, 2015).

Microplastics are commonly defined as plastic particles being smaller than 5 mm in their longitudinal orientation (GESAMP, 2015; Arthur et al., 2009). To the present, the lower size limit of about 10 µm is determined by analytical limitations (Enders et al., 2015; Lenz et al., 2015), but theoretically covers particles down to 1 µm in diameter (Magnusson et al., 2016).

A further distinction is made between primary and secondary microplastics (Cole et al., 2011; Fendall and Sewell, 2009). While the first enter the ecosystem directly, i.e. in the form of raw pellets or abrasive scrubs as an ingredient of cosmetics (Fendall and Sewell, 2009), the latter originate from the fragmentation of larger particles. UV(B)-oxidation and mechanical disintegration due to abrasion (e.g. in sand matrix), wave-action and turbulence are dominant processes in this regard (Cole et al., 2011; Barnes et al., 2009).

Microplastics are ubiquitous in the marine environment. To the present day microplastic contamination has been reported in surface waters (e.g. Setälä et al., 2016; Lusher et al., 2014; Song et al., 2014a), the water column (e.g. Enders et al., 2015; Reisser et al., 2015), embedded into sea ice (Obbard et al., 2014), seabed sediments (e.g. Zobkov and Esiukova, 2017; Woodall et al., 2014), coastal sediments (e.g. Graca et al., 2017; Stolte et al., 2015) and organisms such as fishes (e.g. Bellas et al., 2016) or annelids (e.g. Gusmão et al., 2016).

In situ littering from fishing or shipping (commercial and recreational) directly adds to microplastic pollution in the marine ecosystem.

* Corresponding author.
E-mail address: elke.fischer@uni-hamburg.de (E.K. Fischer).

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For terrestrial sources river based input, including waste water (WWTP effluents), stormwaters, lateral input via beaches and shore lines and atmospheric deposition have to be considered (Magnusson et al., 2016).

Major driving factors for the distribution of microplastics in surface waters and the water column are sea currents, waves as well as predominant wind patterns (Liubartseva et al., 2016; Gago et al., 2015; Reisser et al., 2015). The first is demonstrated by the concentration of microplastics in the large ocean gyres (Gago et al., 2015; Eriksen et al., 2013; Law et al., 2010). The latter can lead to a reduction of microplastic concentrations in the surface layer due to wind driven vertical mixing (Reisser et al., 2015; Kukulka et al., 2012).

The assessment of microplastic contamination in marine surface waters mostly relies on net-based (volume-reduced) sampling approaches (Eriksen et al., 2013; Hidalgo-Ruz et al., 2012). Manta trawl is a commonly used device, which was originally designed for the collection of plankton (Moore et al., 2002). Occasionally, neuston nets are used, which sample the air-water interface for microplastics (Morét-Ferguson et al., 2010). The lower detection size of both systems is defined by the specific net that usually (for sampling of aquatic environments) has a mesh size of 300 µm or 333 µm (Setälä et al., 2016; Mani et al., 2015; Eriksen et al., 2013). For sampling within the water column, bongo (Doyle et al., 2011) and other plankton nets have been applied (Hidalgo-Ruz et al., 2012). It has become evident that a majority of particles found in environmental samples is sized smaller than 300 µm (e.g. Enders et al., 2016). Since the application of finer nets could be hampered due to clogging effects, the need to implement bulk sampling techniques was identified (Setälä et al., 2016). Though, for this method sample volumes differ on a great scale. While Dubaish and Liebezeit (2013) took two replicates of 100 ml, Lusher et al. (2015) sampled 2000 l investigating sub-surface water. Several studies relied on sample volumes well below 100 l (Bagaev et al., 2017; Zhao et al., 2014; Dubaish and Liebezeit, 2013; Ng and Obbard, 2006).

To quantify microplastic in water samples it has to be distinguished from biogenic matter that can interfere the identification process (Song et al., 2015; Hidalgo-Ruz et al., 2012). To reduce the organic content of microplastic samples acidic (De Witte et al., 2014; Claessens et al., 2013), alkaline (Cole et al., 2014; Foekema et al., 2013), oxidizing (Avio et al., 2015; Collard et al., 2015; Tagg et al., 2015; Nuelle et al., 2014) or enzymatic digestion protocols have been applied (Löder and Gerdts, 2015; Cole et al., 2014).

Micro-Fourier transform infrared (FTIR) spectroscopy (Song et al., 2015; Tagg et al., 2015; Cole et al., 2014), micro-Raman-spectroscopy (Cole et al., 2013; Van Cauwenberghe et al., 2013) and Pyrolysis-gas-chromatography with mass spectrometry (Dekiff et al., 2014; Nuelle et al., 2014; Fries et al., 2013) are frequently applied to assess polymer abundance and composition (Ivleva et al., 2017; Löder and Gerdts, 2015).

Moreover, differential staining based on the fluorescent lipophilic dye Nile Red (9-diethylamino-5H-benzo[α]phenoxazine-5-one) has been implemented for microplastic quantification (Erni-Cassola et al., 2017; Maes et al., 2017; Fischer et al., 2016; Shim et al., 2016; Desforges et al., 2014; Shim et al., 2014; Song et al., 2014b; Andrady, 2010). The spectral characteristic of the fluorescence emission depends on the respective solvent as well as the polarity of the stained polymer (Maes et al., 2017; Shim et al., 2016). This approach allows a quick and inexpensive estimation of the microplastic load in a sample without giving detailed information on the chemical composition of the particles (Tamminga et al., 2017; Song et al., 2014b).

2. Material & methods

2.1. Study area

The South Funen Archipelago is situated in the transition zone between the Baltic Sea and the Kattegat in Denmark (Fig. 1). In total, it consists of 55 islands and is delimited by the islands of Lyø in the west, Ærø in the south, Langeland and Tåsinge in the east and Funen in the north. Svendborg (population 2016: 27,074) and Faaborg (population 2016: 7178), located at the southern coast of Funen are the largest cities in the region (Statistics Denmark, 2017).

Due to its remoteness and natural landscape, the region is a favorable destination for tourists. Especially water sports (e.g. sailing, angling or sea kayaking) are popular among both, the local population and visitors from abroad. Several small coastal villages such as Ærøskøbing or Søby on the island of Ærø are harboring marinas, which are frequented by smaller and larger vessels. Additionally, regular ferry connections between all major islands exist.

Complex hydrographic processes can be identified within the Skagerrak-Kattegat region. In general, a surface layer of low salinity (up to 1.5%) flows northwards through the Little and Great Belt straits (Omstedt et al., 2014). This is compensated by a denser, high-saline (up to 33%) and near-bottom stream of Skagerrak water entering the Baltic Sea. In the South Funen Archipelago the water column is mixed almost constantly without developing any considerable strata (Rask et al., 1999).

Mean wind conditions at Tåsinge Island (closest climate station with data available) are displayed in Fig. 2. Wind patterns are dominated by westerlies, in general. East to west wind flows develop as well, especially in winter. At this time increased frequencies of heavy winds lead to intensified mixing within in the water column (Rask et al., 1999).

2.2. Strategy for microplastic sampling in surface and subsurface water

All sampling was conducted between June 16th and 19th, 2015 aboard the sailing ship *Lovis* in the South Funen Archipelago, Denmark. A total of ten manta trawls were carried out to investigate microplastic contamination within the uppermost water layer. Additionally, 27 bulk-samples in open waters in three depths (0.5 m, 2.0 m and 5.0 m) and four bulk-samples in harbors (0.5 m depth in Faaborg, Ærøskøbing, Søby and Lyø) were taken throughout the area to test for the comparability of low-volume bulk-samples towards manta trawls. Table 1 provides an overview of manta sampling related data.

The manta trawl was positioned at the windward side of the ship's hull to exclude any vessel-based contamination of the sample. The manta was recovered after 20 min of trawling. Special care was given to avoid any backflow during the recovery, which could have possibly flushed out parts of the sample at the cod-end. In previous sampling protocols, trawling times between one and three hours were suggested. However, Fischer et al. (2016) recommended trawling durations of < 60 min to prevent potential clogging effects of the net in case of high organic contents, which might lead to minor results. In order to avert these issues, trawling times were kept at 20 min in this study. Mean trawling speed was 4.2 km/h (maximum 5.95 km/h) and thereby well within the recommended range (5-Gyres Institute, 2014). The covered distance was tracked via D-GPS (Trimble Geo 7×) and ranged from 1146 m to 1746 m depending on the trawling speed. To calculate the sampled area and volume the recorded distance was multiplied with the width or the area of the manta opening, respectively.

Aboard the *Lovis,* the cod-end was detached and the sample volume was transferred into a stainless-steel bowl. Afterwards, the cod-end was inverted and thoroughly rinsed with purified water, until all sample material was recovered. It was then transferred into brown glass jars and treated with 10 ml of hydrochloric acid (HCl, 37%, Merck Emsure®) per jar to stop biological processes.

The bulk sampling was done by means of an Integrated Water Sampler (IWS, HYDRO-BIOS GmbH). After the entire sample volume (5 l) was automatically pumped into the plexiglass hull via a valve at the bottom of the IWS, the device was recovered on deck. Here, the sample was poured through a sieving cascade with mesh sizes of 5.0 mm, 1.0 mm and 0.3 mm. Subsequently, the sieve contents (5 mm– > 1 mm, 1 mm– > 0.3 mm) were transferred into brown glass vials via rinsing with little purified water. The sample