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Study of microplastic occurrence on the sandy beaches of Šventoji, Lithuania

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Abstract. In this study, a sandy beach in Šventoji, Lithuania, a popular albeit not highly urbanised touristic destination, was analysed for microplastic pollution in the summer of 2019. The presence and abundance of microplastics in different sites of the beach were evaluated. Šventoji Beach was found to be significantly polluted with microplastic, its concentration ranging from 85 to 325 MPs kg^{-1}dw. Such concentrations are relatively high if compared to other Baltic Sea coasts and worldwide. The physico-chemical characteristics of the plastics showed limited variability. Blue fibres were prevailing over other types of microplastics. Blue and red/orange fibres were identified as nylon containing copper–phthalocyanine dyes, while red/orange fibres as high-density polyethylene. This study provides new insights into the application and development of microplastic analysis methods for the coastal sands of the Baltic Sea.

Keywords: *Baltic Sea; non-tidal beach; microplastic identification; marine debris; coastal pollutants*

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INTRODUCTION

Due to the widespread use, constantly increasing production and slow biodegradability of synthetic polymers, plastic pollution has become one of the most serious concerns of our time. According to estimates, approximately, 5–13 million tonnes of plastic debris enter the marine environment each year, and their degradation products are considered to be the most abundant solid-waste pollutants (Chubarenko et al. 2020; Jambeck *et al*. 2015; Urban-Malinga *et al*. 2020,). Numerous studies have already addressed the issue of microplastic pollution in the marine sediment and sandy beaches around the world (Chubarenko et al. 2020; De-La-Torre *et al*. 2020; Urban-Malinga *et al*. 2020; Fischer *et al*. 2015; Lusher 2015; Lots *et al*. 2017; Martinelli Filho, Pereira Monteiro 2019). Even in relatively remote and undisturbed locations, microplastic pollution has been reported (Barnes 2005; Zarfl, Matthies 2010).

Extreme vulnerability of beaches to litter pollution is explained by the following reasons: firstly, they are exposed to plastic pollution from the sea, and, secondly, their recreational function entails increased risk of pollution. Plastic debris was found to be the prevalent type of beach litter (Balčiūnas, Blažauskas 2014; Fernandino *et al*. 2016), the degradation of which leads to the *in situ* formation of microplastics in the beach environment (Andrady 2011). It is also noteworthy that sandy beaches act as filtering systems (McLachlan, Brown 2006), and thus microplastic occurrence on sandy beaches might (to some extent) reflect the overall litter pollution in adjacent waters and coastal areas (Urban-Malinga *et al*. 2020).

Knowledge of the spatial and temporal microplastics (MPs) distribution, their concentrations in various environments, size and the prevailing polymer types is essential for both understanding the MPs fate in the environment and for developing efficient regulation strategies. However, microplastic analysis is rather challenging and is still evolving. Visual inspection under an optical microscope is used for preliminary identification and quantification of microplastic fragments (Lots *et al*. 2017; Bridson *et al*. 2020). This method lacks precision as it relies basically on the observer's experience and misinterpretation of the results is difficult to avoid. Spectroscopic methods, such as Raman spectroscopy, could be applied to reduce misidentification (Lenz *et al*. 2015; Gillibert *et al*. 2019; Bridson *et al*. 2020; Álvarez-Hernández *et al*. 2019). Raman spectra reveals vibrational information of the molecular structure, thus providing unique information on sample composition (Lenz *et et al*. 2015). Even though the Raman-based polymer identification is hindered by contamination and weathering of samples, it is a powerful tool for the identification of microplastics (Lenz *et al*. 2015).

Conditions for pollutants accumulation in the Baltic Sea are favourable because it is shallow (52 m mean depth) and semi-enclosed with a highly anthropogenized catchment area and a slow water exchange rate (Franck *et al*. 1987). According to Haseler *et al.* (2017), the major problem facing the Baltic beaches in Lithuania and Germany is meso- and micro-litter. The beaches of the Baltic Sea are reported to be among the most microplastics-polluted beaches in Europe (Lots *et al*. 2017). In addition, in the same study, the sandy beach of Klaipėda was rated as the second most-polluted site after Vik (Iceland). Sandy beaches of the Baltic Sea in the Kaliningrad region were found to be polluted with industrial pellets, foamed plastics, fibres, granules and films, the total MPs concentration ranging from 1.3 to 36.3 MP kg^{-1} dw (Esiukova 2017). Similar values (1.8–30.2 MP kg^{-1} dw) were recorded in the Kiel Fjord (Schröder *et al*. 2021). A study covering a 100-km-long marine coast of the Curonian Spit (between the cities of Klaipeda (Lithuania) and Zelenogradsk (Russia) revealed greater variations in microplastics content. The minimum value of 5 MP kg^{-1} dw was recorded in Lesnove (Russia), whereas the maximum value of 177 MP kg^{-1} dw in Klaipėda (Lithuania) (Esiukova *et al*. 2020). Particularly high MPs loadings, ranging from 53 to 572 MP kg^{-1} dw were detected in Kaliningrad by Chubarenko *et al*. (2018). The detected MPs levels were significantly higher than those previously observed in the region (Esiukova 2017), indicating that conditions of microplastics pollution in the Baltic are varying. The authors stated that variations in MPs loadings are tightly related to oceanographic and atmospheric processes (wave- induced sediment transport, storms) (Chubarenko *et al*. 2018).

In this study, we seek to gain a deeper knowledge about the microplastic pollution on Lithuanian beaches focusing on application and development of laboratory methods for the Baltic sandy sediments. To this end, a sandy beach in Šventoji, a popular albeit not highly urbanised touristic destination, was chosen. The aim of our study was to assess the presence and abundance of microplastics at different sites of the beach, as well as to apply Raman spectroscopy for the identification of sample composition.

MATERIALS AND METHODS

Study area and sampling

This study was performed on the Šventoji Beach, located in the Lithuanian part of the southern Baltic Sea coast. The southern Baltic beaches are dominated by sands (Łabuz *et al.* 2012; Jarmalavičius *et al*. 2020, 2012), and the Šventoji Beach is classified as a non-tidal beach with water level fluctuations controlled by wind-generated waves (Jarmalavičius *et al*. 2012).

Three areas of the Šventoji Beach were selected for sampling (Fig. 1). Samples were collected from different sites of the Šventoji Beach in August 2019. Sampling points were evenly distributed along the Šventoji coastal line. The sampling stations were selected so as to reflect different morpholithodynamic patterns in the coastal zone (Jarmalavičius, Žilinskas 2007; Kriaučiūnienė *et al*. 2013). Relatively wide and high beaches with the prevailing fine sand fraction are located southward from the Šventoji port (stations ST1 and ST2) (Jarmalavičius, Žilinskas 2007; Kriaučiūnienė *et al*. 2013). The foredune at ST2 reaches 6–8 m in height and exceeds that at ST1. Moreover, the beach around ST2 is particularly wide (up to 100 m). North of the Šventoji port (ST3), the beach width varies from 40 to 60 m, the foredune is slightly eroded by waves and is considerably degraded (Kriaučiūnienė *et al*. 2013). Hydrotechnical constructions and piers of the Šventoji port inhibit the longitudinal migration of the sediment northwards.

Thus, coarser sand (medium-grained) prevails in the Northern part of the beach (Jarmalavičius,

Fig. 1 Map of the study area. Note: ST1–ST3 – sampling stations

Žilinskas 1996; Karlonienė *et al*. 2021).

Various environmental factors (wind, waves, etc.) can affect the distribution of microplastics along the beach (Chubarenko *et al*. 2018). The middle section of Baltic Sea beaches has been reported to tend to accumulate high amounts of microplastics compared to the end of the beach and the part constantly washed by the waves (Graca *et al*. 2017). Thus, beach samples were collected from the middle section of the Šventoji Beach. Sampling was performed as suggested by McDermid, McMullen (2004) and Besley *et al*. (2017) with some minor modifications. The beach was first divided into three sections. A 50 × 50 cm quadrant was placed at the sampling location in the middle section of the beach, and the surface sand (5 cm deep) was scooped with a metal spoon. Five replicate samples were collected from each sampling location (from the middle and from each corner of the quadrant, each subsample weighing at least 200 g).

Extraction

MPs were extracted from the sediment using a standardised density separation method suggested by Besley *et al*. (2017). The saturated NaCl was obtained by adding 358.9 g of NaCl to 1 L of deionized water, stirring and heating it up to 60 °C. After cooling, the concentrated saline solution was filtered through a glass fibre filter (pore size 0.6 µm, Merck). Replicate samples were carefully mixed, a total of 500 g of the sediment was weighed, put into a glass dish and dried for 48 h at 60 °C. As there were no large plastic particles visible, the sediment was further sieved through a 2.5-mm metal sieve. An aliquot of 100 g was used for the MPs extraction in the saturated NaCl solution: the required amount of the dry sediment was added to the glass bottle filled with 400 ml of the concentrated NaCl solution, the mixture was stirred at 600 RPM for 2 min and was left to settle overnight. Subsequently, the supernatant was carefully poured into a vacuum filtration system and filtered through 47 mm Millipore 0.6 µm glass fibre filters (Merck). The filters were dried in covered petri dishes at room temperature. The extraction step was repeated three times for each sample (Besley *et al*. 2017).

Quality control

To avoid contamination, no plastic equipment/storage containers were used during sample collection, preparation and analysis procedures. White robe made of 100% cotton was worn throughout sample processing and analysis. Prior to using, all the glassware was rinsed with nitric acid and finally, with deionised water. During sample preparation, all the vessels were covered with parafilm when not in use. Procedural blanks were also prepared. They were found to contain occasional MPs (approx. 2–3 MPs per blank sample).

Analysis

In order to reliably distinguish potential plastic fragments from organic debris (animal parts, dried algae/seagrasses, etc.) and mineral particles, it is necessary to carry out careful visual evaluation. First, filter papers were examined under an optical microscope (Motic Classmag 41, Motic, Germany). To facilitate the count of MPs fragments, the filter paper was divided into four equal parts and the top of the filter paper was marked.

Visual identification of the samples was performed in accordance with the rules proposed by Hidalgo-Ruz *et al*. (2012). Based on their morphological properties, the objects spotted on the filter papers were either accepted as suspected MPs or rejected. To be identified as MP, a specimen under examination had to meet the following criteria: a) no visible cells/organic structures; b) fibres equally thick throughout their entire length; c) clear and homogenous colour. However, as it is known, degradation and bleaching may affect both the colour and thickness of microplastic fragments (Lots *et al*. 2017). Thus, the primary identification was made very carefully, suspicious fragments being also marked as potential microplastics. Each suspected MP was described, with its colour, shape and the approximate location on the filter indicated. Potential MPs were classified according to their colour and shape.

Plastic pollutants were further analysed using Renishaw in Via Raman spectrometer coupled with a thermoelectrically cooled (-70 °C) CCD camera and a microscope equipped with the OLYMPUS LC Plan N 50 × /0.65 NA objective lens. Raman spectra were measured using the excitation laser wavelength of 785 nm with the output power of 0.92 mW. Each spectrum was collected from 10 scans with the exposure time of 10 s, yielding the total exposure time of 100 s.

RESULTS AND DISCUSSION

Morphological properties

Fibrous fragments were found to dominate other shapes of suspected MPs (Fig. 2). They accounted for 98% of the total MPs detected. One particle was spherical and the other suspected MPs were angular and irregular in shape. The fibres were of unequal length, which varied from 0.2 to 2.62 mm. The average length of blue, black and red fibres was 1.82 ± 0.8, 0.7 ± 0.5, 0.9 ± 0.7 mm, respectively. Most of the MPs measured (54.8%) were shorter than 1 mm. The fibres were predominantly blue in colour, followed by black. Also, there were several orange and red fibres found among the samples, the rest of the suspected MPs were transparent or yellow/yellowish. These results are similar to those reported in other studies: blue/black and red fibres are usually reported as the prevailing types of the fibres found in the marine environment (Frère *et al*. 2017; Alomar *et al*. 2016; Strand, Tairova 2016; Lots *et al*. 2017). Various studies report the prevalence of fibrous materi-

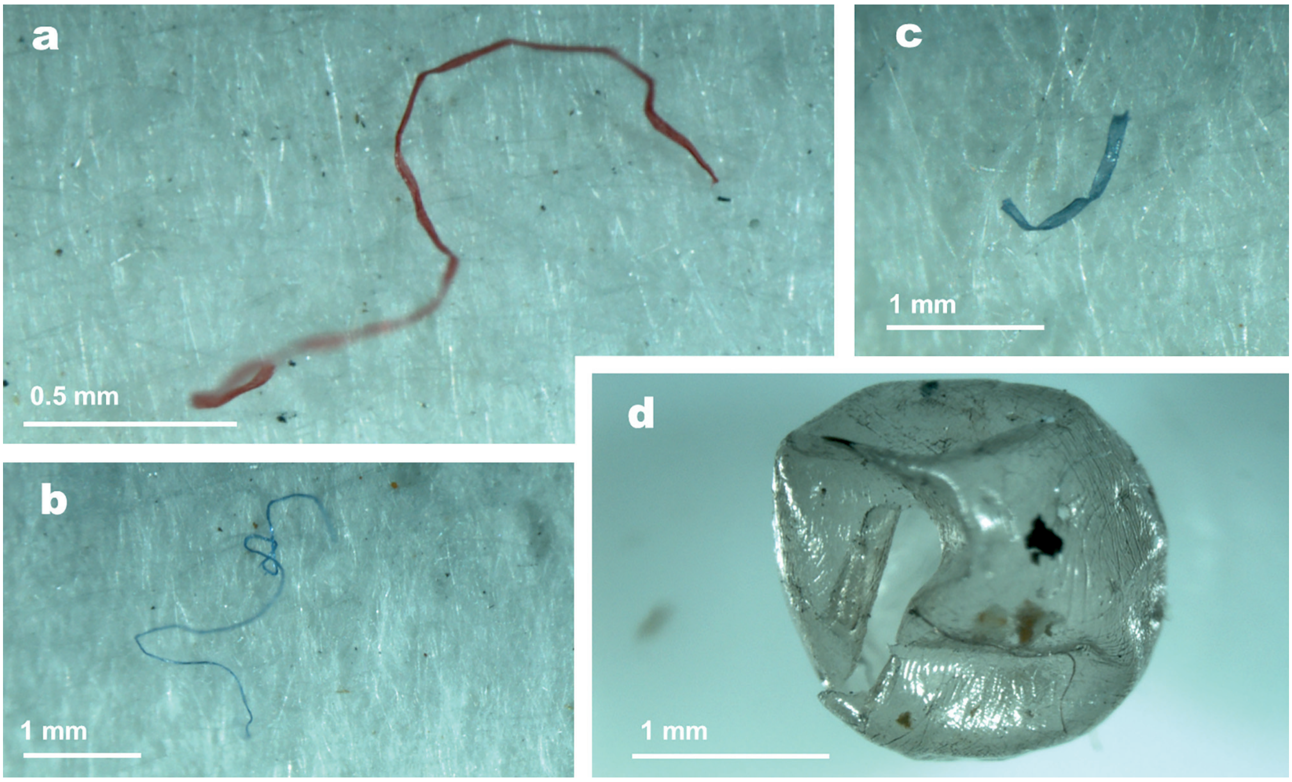

Fig. 2 Microplastics extracted from the sandy sediment of the Šventoji beach: a) Red fibre (ST1); b) Blue fibre (ST2); c) Blue MPs fragment (ST3), d) Spherical synthetic particle (ST3)

als, reaching up to 90 percent (Urban-Malinga *et al*. 2020; Zobkov, Esiukova 2017; Alomar *et al*. 2016; Graca *et al*. 2017; Blašković *et al*. 2017). The detected fibres could possibly originate from the fragmentation of fishing ropes and nets (Thompson 2004), some fragments of which were also found close to sampling locations. It is also possible that microfibres of machine-washed synthetic fabrics entered the marine environment with wastewater (Hernandez *et al*. 2017; Napper, Thompson 2016).

Characterisation of the visually detected microplastics

Some of the visually suspected MPs did not show discernible peaks in their Raman spectra. This was especially true for the irregularly shaped particles. Most of them were identified as mineral particles and were rejected from the further examination. When dealing with environmental samples, the rates of success achieved in matching a MP to a specific polymer type, are usually relatively low. The rate of success that Lots *et al*. (2017) achieved in polymer identification was 4.5%, whereas successful polymer identification made by Frère *et al*. (2017) was rated at 13%. Polymer identification success depends on a number of factors. Successful polymer identification may be hindered by the presence of biological material on the MP surface, which is difficult to avoid in environmental samples, as marine microplastic is a common substrate for various organisms whose residuals are likely to remain on the particles' surface and cover the polymer with an organic matrix. Even more important is the fact that commercial plastics contain not only the basic polymer matrix, but also various additives, e.g., dyes, plasticizers, fire retardants etc., which may also affect the Raman signal (Lenz *et al.* 2015; Urban-Malinga *et al*. 2020). One of the disadvantages of Raman spectroscopy is the luminescence effect produced by dyes and impurities present in plastic pollutants, which can be sometimes overcome by changing the excitation wavelength. In this study, to avoid luminescence, we chose the 785 nm laser excitation.

Under excitation of 785 nm, blue fibers exhibited well defined peaks in their Raman spectra (Fig. 4), while Raman spectra of red shades plastics were weak and evoked low signal/noise rate (Fig. 3). Identification was performed based on Raman spectra databases and results of other studies. However, precise identification of plastic fibers is far from being easy. The degradation processes of plastics lead to a large variety of degradation products with altered chemical and physical features that are also reflected in the Raman spectra. The Raman spectra of red shades fibers (Fig. 3) reflect plastic pollutants degraded to varying degrees. The different degree degradation of the same plastic is optically manifested in color fading,

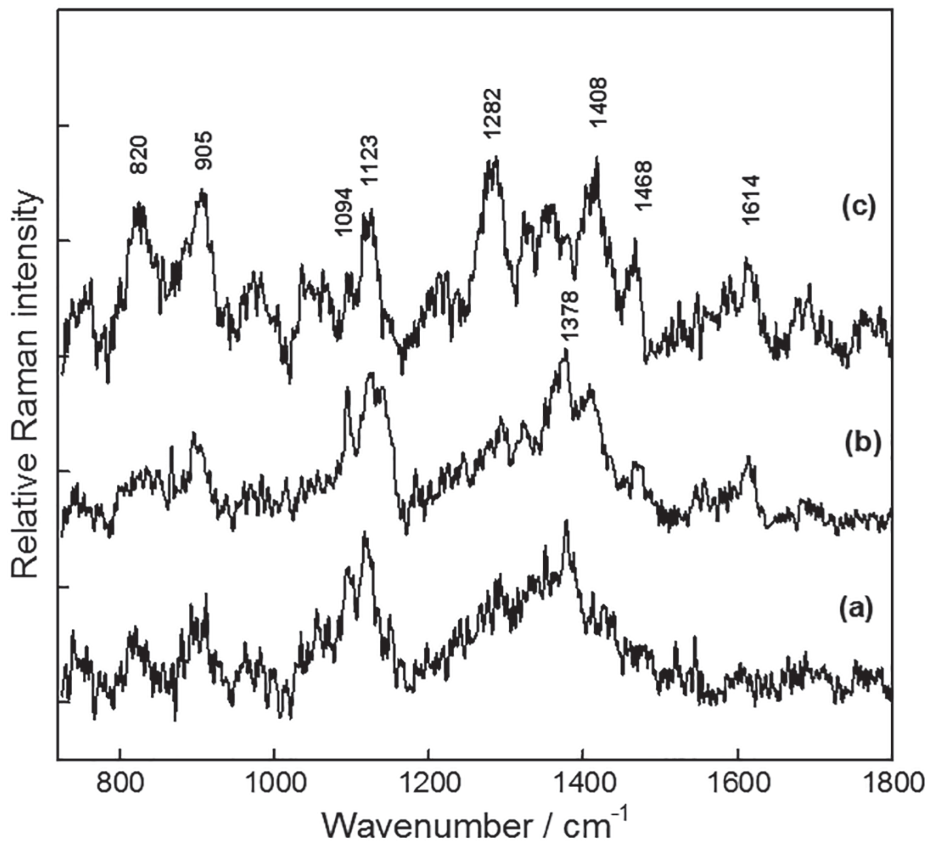

Fig. 3 Raman spectra of red shades fibers (a) colorless, (b) orange, (c) red

specifically, in a decreasing intensity of vibrational bands, as well as in their disappearance. According to the relevant literature, red shades plastics might be identified as high-density polyethylene with characteristic bands at 1463, 1436, 1294, 1123, 1064 cm^{-1}. Vibrational modes at 1463, 1436, 1294, cm^{-1} are related to CH_2 vibrations, while the bands at 1123 and 1064 cm^{-1} are assigned to CC asymmetric and symmetric stretching, respectively (Angelin *et al*. 2020; Gillibert *et al*. 2019). The sharp band at 1094 cm^{-1} and a broader band at 1130 cm^{-1} overlapping with a band at 1123 cm^{-1} indicate the presence of sulphates in a sample (Angelin *et al*. 2020).

Not all blue shades plastic pollutants are identical: varying shades were observed under an optical microscope, and the performed Raman analysis revealed differences in their composition. Four subgroups of blue fibers were distinguished: dark blue, light blue, grey blue and ink blue. The Raman spectra obtained for different fibers of the same subgroup were similar. The identification of blue shades plastics like that of red shades plastics was beset with similar problems: the plastics under study were affected by degradation and dye-bleaching processes. Another problem that we faced when identifying plastic fibers was their origin: we assume that plastic fibers might be copolymers. It should be pointed out that each additive contributes to the shift and peak intensity in the Raman spectra, making the identification complicated (Lenz *et al*. 2015).

Figure 4 shows the Raman spectra of the light blue plastic fiber next to the identical Raman spectrum of Heliogen Blue (copper–phthalocyanine blue) (Simon and Rohrs 2018). To observe and identify the plastic bands, we changed the scale of the graph (Fig. 4 B). Compared to Heliogen Blue, the plastic bands at 1484, 1428, 1373, 1160, 1038 and 850 cm^{-1} are weak and are, very likely, overshadowed by the dye bands. The ink blue plastic fiber was identified as nylon with its characteristic bands at 1630, 1461, 1364, 1311, 1248, 1226 cm^{-1}; depending on the density level of the polymer, there was a high variability

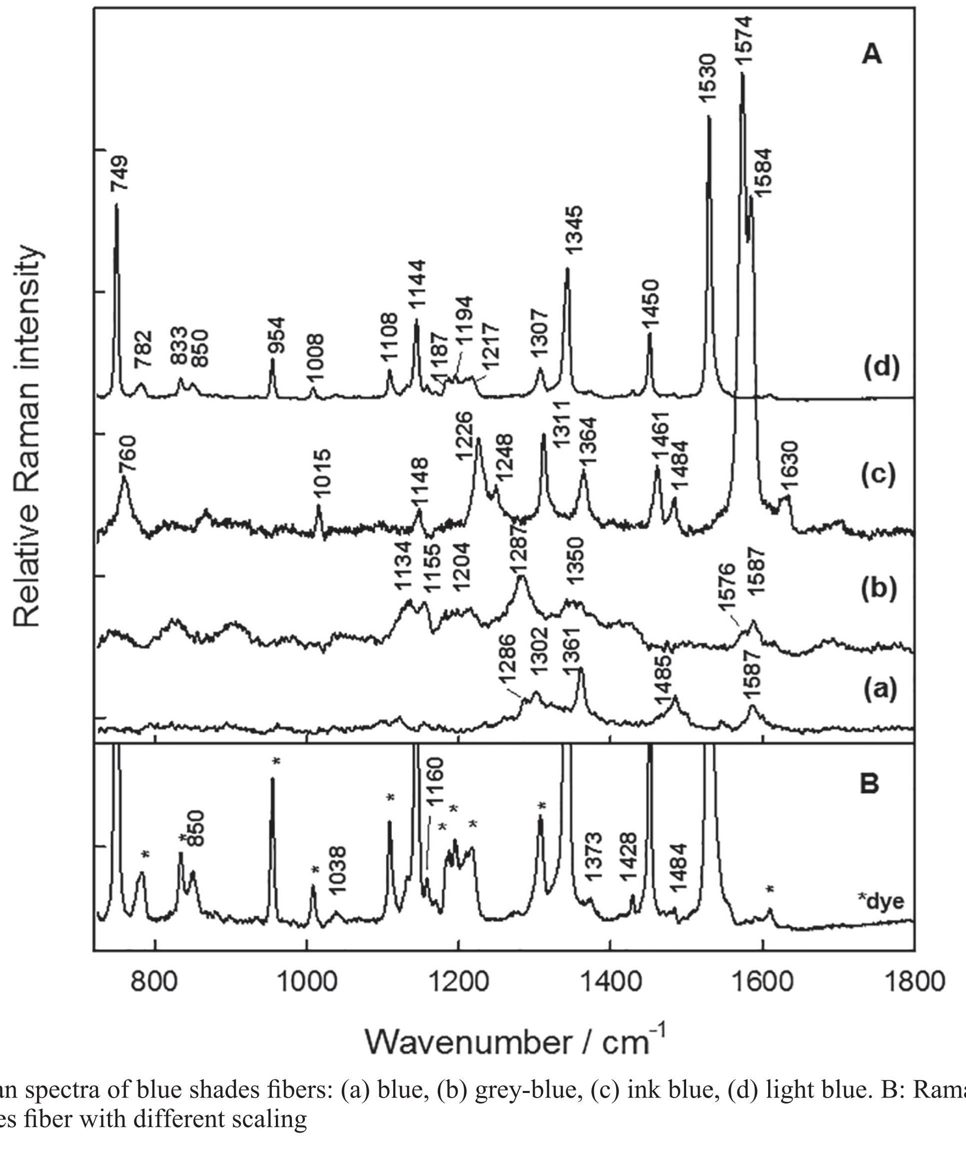

Fig. 4 A: Raman spectra of blue shades fibers: (a) blue, (b) grey-blue, (c) ink blue, (d) light blue. B: Raman spectrum of light blue shades fiber with different scaling

of spectral features recorded in the 1290–1000 cm^{-1} spectral region (Hendra *et al*. 1990). Therefore, we do not rule out the possibility that all blue and grey blue fibers are just highly degraded nylon. This polymer is commonly applied for the production of fishing ropes and nets, the degradation products of which account for approximately 20% of marine debris (Andrady 2011). Vibrational bands at 1584, 1574, 1484, 1144, 1016 cm^{-1} could be assigned to one of the copper–phthalocyanine dyes.Marine sediments and coastal sand are often polluted with fishing ropes/nets. Some larger, not yet degraded fragments of these artefacts were found in the vicinity of the sampling locations ST1 and ST2 (Fig. 1, Fig. 5). It is likely that the blue fibres detected in our study originated from similar items.

Abundance of microplastics

As mentioned above, some fragments were excluded from further examination and quantification. However, all fragments that were blue, red or orange in colour were accepted as plastic. Black fibres with the uniform morphology fitting the description of plastic fibres were also included. After this, a quantitative analysis was performed. Fig. 6 shows the average abundance of MPs per sampling location, which was in the 100 ± 15 - 300 ± 25 MPs kg^{-1} dw range. The site ST2 was characterised by the lowest pollution level, whereas sampling station ST1 by the highest MPs loadings.

Microplastics may enter the beaches from marine waters or originate from the on-site anthropogenic pollution (Chubarenko *et al*. 2018; Schröder *et al*. 2021; Stolte *et al*. 2015). Differences in sampling locations could be determined by a complex impact of several parameters, such as beach use and maintenance, location and exposition of the sampling site, wind patterns, hydrodynamics (Schröder *et al*. 2021; Stolte *et al*. 2015; Graca *et al*. 2017). ST1 is strongly affected by the open sea as it is located in the narrower section of the beach with a lower foredune than ST2 (Jarmalavičius, Žilinskas 2007; Kriaučiūnienė *et al*. 2013). Therefore, in this location, conditions for MP accumulation from the sea are favourable (ST1). The higher MP loadings recorded at ST1 can be also explained by the fact that this site is located farther from the main resort areas and is, therefore, less frequently cleaned. The lowest MP abundance was observed at ST2, which is located in the widest and the most frequently visited part of the Šventoji coastline. We assume that the relatively low MP concentrations recorded at this site could be related to the relatively frequent cleaning of the beach. ST3 was characterised by a higher MP content than that at ST2. ST3 is located in the narrow beach with an eroded and a highly degraded foredune (Kriaučiūnienė *et al*. 2013). This part of the coastline is exposed to stormy winds and erosion (Jarmalavičius *et al*. 2016), and contains coarser sand fraction than the remaining sampling stations (Kriaučiūnienė *et al*. 2013). According to Chubarenko *et al*. (2018), sites with stronger water dynamics tend to contain coarser sands within the beach, and the highest microplastics concentrations are considered to be related to stormy events. To sum up, variations in MPs abundance were most probably

Fig. 5 Images of the large fragments of fishing nets and ropes (potential MPs source) found on the sandy beaches of Šventoji

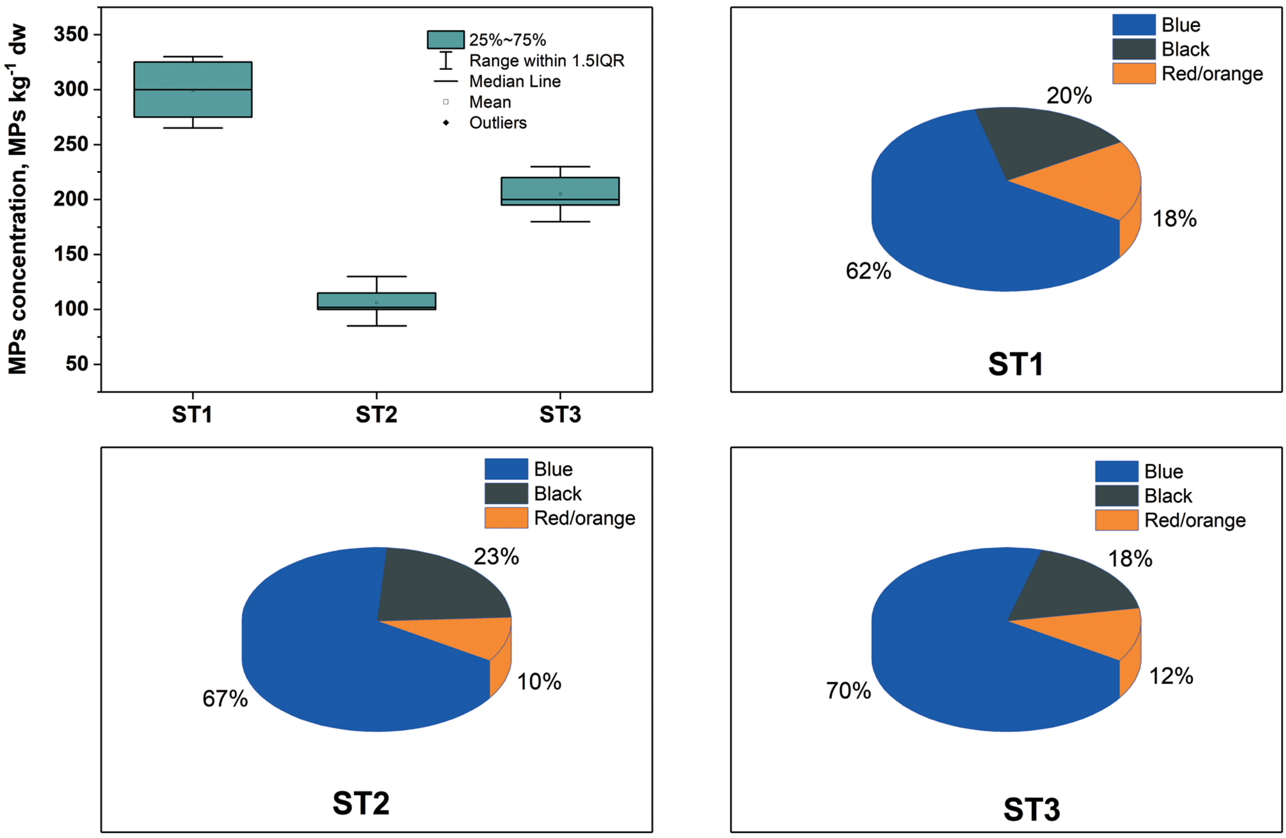

Fig. 6 MPs abundance on the Šventoji beach (ST1, ST2, ST3) and contribution of different MPs colours. Colours are expressed as a percentage of the total count

determined by the morpholithodynamic coastline differences (height of the foredune, width of the beach, grain size), hydrodynamic processes and differences in the maintenance and cleaning of the beaches.

Our results are in agreement with those obtained from a large-scale study performed by Lots *et al.* (2017), who found that the average abundance was in the range 72 ± 24 MPs kg^{-1} dw - 1512 ± 187 MPs kg^{-1} dw. This study also identified the Baltic zone as one of the most polluted sites in Europe with means of 270 MPs kg^{-1} dw. Moreover, MPs concentrations found in Klaipėda, Lithuania, reached 700 MPs kg^{-1} dw and were among the highest ones. MP concentrations recorded in our study are significantly lower than those found in Klaipėda. This is not surprising considering that the sampling in Klaipėda was performed at the outlet of the Curonian Lagoon, which is known to be significantly polluted (Lots *et al.* 2017; Christian *et al.* 2008). Similar results were obtained by Vianello *et al.* (2013), who investigated the Lagoon of Venice (Italy). However, significantly lower MPs concentrations were detected in the Vistula Lagoon (Poland) (Lots *et al.* 2017). Increased MPs concentrations detected in Klaipėda and Venice are most likely related to the fact that the samples were collected from the urban estuaries. Densely populated zones and the impact of river inflow create perfect conditions for the accumulation of microplastics. Several authors have highlighted the impact of river discharge on the MP pollution (Claessens *et al.* 2011; Faure *et al.* 2015). A study by Esiukova *et al.* (2020) identified Klaipėda (177 MP kg^{-1} dw) as the most MPs-polluted site on the 100-km-long marine coast of the Curonian Spit. Our results are also similar to those obtained in the study recently performed on the Polish coast (76 and 295 MPs kg^{-1} dw, Urban-Malinga *et al.* 2020). Comparable MP loadings (53 to 572 MP kg^{-1} dw) were also recorded in Kaliningrad (Chubarenko *et al.* 2018).

In this study, the samples were collected on the Šventoji Beach, which is located next to the Šventoji town and is a popular recreational site. The recorded MPs loadings were relatively high compared to different studies worldwide (Graca *et al.* 2017; Esiukova 2017; Esiukova *et al.* 2020; Hengstmann *et al.* 2018; Lots *et al.* 2017; Stolte *et al.* 2015). MPs concentrations found there were lower than those detected in Klaipėda in 2017 (Lots *et al.* 2017), but higher than those recorded in Klaipėda in 2020 (Esiukova *et al.* 2020). Such variations might be related to the changing dynamics of MP pollution in the coastal sand. Numerous studies have reported similar MPs abundance levels from various coasts of the south-western Baltic Sea (Graca *et al.* 2017; Esiukova 2017; Hengstmann *et al.* 2018; Stolte *et al.* 2015). However, some studies report high variations in MPs loadings even at small spatial scales (Schröder *et al.* 2021). Significant temporal variations in MPs abundance were also observed (Esiukova 2017; Esiukova *et al*. 2020; Chubarenko *et al*. 2018), indicating that microplastic pollution is a dynamic process, which depends not only on such anthropogenic factors as urbanization, touristic activity or the nearby industry (Schröder *et al*. 2021). Oceanographic and atmospheric processes may play even a more important role in MPs pollution (Chubarenko *et al*. 2018). According to Chubarenko *et al*. (2018), the main source of beach pollution with microplastics is marine waters. As reported by these authors, variations in MPs abundance are controlled by oceanographic and atmospheric processes and plastics beaching mainly occurs via wave-driven mechanism.

Wave- and wind-driven mechanisms can also affect MPs distribution on the Šventoji Beach, although no evidence for that has been obtained in the current study. It should be also noted that the present study covers a relatively short period of time and thus it does not reflect dynamics of microplastics abundance. Different results might be obtained in different seasons or years. Also, it is difficult to identify the potential sources of MPs pollution on the Šventoji Beach. Further wider studies are needed to fully answer these questions.

CONCLUSIONS

This study has provided a deeper insight into the development and application of methods for analysing microplastic pollution on Baltic Sea beaches. On the Šventoji Beach, fibrous microplastic fragments were found to be prevailing over other detected MPs, blue fibres being the most abundant microplastic found in the study area. Using the Raman spectroscopy, the blue shade fibres were identified as nylon of different degradation degree. In addition, the Raman spectra of the blue shades fibres indicated the presence of copper–phthalocyanine dyes. Red shades fibers were identified as high-density polyethylene. The observed differences in the Raman spectra of red/orange fibers were attributed to varying degradation degrees. However, it was impossible to identify some of the MPs fragments due to the high luminescence or weakness of the signal. Some particles that did not exhibit a well-defined Raman spectrum, but had characteristics of synthetic materials, were regarded as microplastic fragments. This study has revealed that concentrations of microplastics found on the Šventoji Beach, Lithuania are significant. The results are in agreement with those of previous studies and indicate that Lithuanian beaches are relatively heavily polluted with MPs. However, the present study covers a relatively short period of time. Considering the dynamics of microplastic abundance on Baltic Sea beaches, it is possible that such high concentrations are not typical of the Šventoji Beach in other seasons or years. Further more comprehensive studies are needed to identify the potential sources of MPs pollution on the Lithuanian beaches and to uncover the possible reasons for the high MPs concentrations observed in this study.

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