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Microplastics in the marine environment: a literature review and northeast England case study

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Abstract

The issue of microplastic pollution has seen an increasing focus over the last decade, both as a scientific research topic and due to intensive media interest. Despite this, there has been, until recently, a relative paucity of data relating to the volume and type of microplastics in the marine environment. The Cell 1 Regional Coastal Monitoring Programme, which covers approximately 300 km of the northeast England coastline, has recently collected subtidal sediment samples from the bed of the North Sea and analysed these for particle size distribution and microplastic content. This paper presents the results of this analysis along with a literature review of the sources, movements and concentration of microplastics in the marine environment.

Introduction

The term "microplastics" was established by Thompson *et al*. (2004) to describe microscopic pieces of plastics in European waters and sediments. The most common size description of microplastics is that of Arthur *et al*. (2009) defining microplastic particles as those less than 5 mm.

There are, however, a range of definitions used, with less than 1 mm being another favoured definition (Frias and Nash, 2019). While there is no definition of a minimum size of microplastics, a 0.33 mm mesh size is commonly used to collect microplastic samples (Masura *et al*., 2015). With a large amount of attention being given to microplastics in the environment, several researchers have begun to consider the fragmentation of plastics down to lower scales (i.e. the submicrometre scale). The term "nanoplastics" is still under debate for these fragments and different studies have set the upper size limit at either 1000 nm or 100 nm (Gigault *et al*., 2018). A review of the current opinion by Gigault *et al*. (2018) defines nanoplastics as particles unintentionally produced (i.e. from the degradation and the manufacturing of the plastic objects) and presenting a colloidal behaviour, within the size range from 1 to 1000 nm.

Microplastics are commonly split into three different categories (Barnes *et al*., 2009; Cole *et al*., 2011):

- *Microbeads* – Primary microplastics which have been manufactured to be microscopic in size, such as those in exfoliation cosmetics and plastic pellets used for the production of other plastic products.
- Microfragments – Secondary microplastics fragmented from larger plastics usually due to mechanical abrasion, photo-oxidative processes by ultra-violet (UV) radiation and thermo-oxidative or biological processes.
- Microfibres – Secondary microplastics in the form of synthetic fibres, such as those broken down from textiles.

Microplastic abundance within the marine environment is increasing. Claessens *et al*. (2011), whilst investigating sediment cores, found that microplastic concentrations in beach sediments in Belgium tripled between 1993 and 2008. Degradation of plastic objects can take several hundred years (Avio *et al*., 2016) so increasing the fragmentation of legacy or current plastic wastes is expected to continually increase microplastic concentrations.

Ubiquitous across the global marine environment (Bergman *et al*., 2015), microplastics have been found in a wide range of marine sediments from surface sediments on beaches (Lots *et al*., 2017) to the deep sea (Van Cauwenberghe *et al*., 2013). The impacts of this scale of microplastics on the marine environment are not yet fully understood; microplastics can be ingested by marine organisms and transferred up the food chain (Andrady, 2011) and harmful substances can leach from or adhere to microplastic debris (Cole *et al*., 2011).

Research topics have rapidly expanded to establish baseline surveys (Maes *et al*., 2017), develop sampling methodologies (Qiu *et al*., 2016) and to investigate sources and impacts of microplastic pollution (Auta *et al*., 2017). Marine impacts have remained the focus, but an increasing number

of studies are considering impacts on freshwater and terrestrial habitats (Horton *et al*., 2017b). This increasing interest in plastic pollution is reflected in the growing number of bans on certain types of microplastics such as microbeads in multiple countries, including the United Kingdom. Additionally, the European Union's (EU's) Marine Strategy Framework Directive (MSFD) requires its member states to provide specific information in relation to trends, amount, distribution and composition of microplastics.

Literature review: Summary of previous studies

Method

With ever-growing scientific research interest in the topical subject of microplastics in the marine environment, there is an increasing base of published literature becoming available. These literature sources largely cover individual extant studies and so a scoping review of these sources was deemed useful in collating and synthesising the overall cumulative endeavour in this area of intense emerging interest. This review was undertaken with the intent of examining the extent, range and nature of ongoing research activities and informing the design and undertaking of the northeast England case study. Particular aims of the literature review, which focused primarily (though not exclusively) on top-tier journals in the field of marine pollution, were to:

- Identify what has previously been researched and published on the subject.
- Determine the extent to which any interpretable trends or patterns are evident.
- Aggregate the findings from individual extant studies to provide a broad evidence-based overview of the subject.
- Understand the approaches (and their limitations) for sampling and laboratory testing for microplastic content.
- Provide background context for the northeast England case study.

The literature review identified that studies worldwide have increasingly been looking at quantifying the concentrations of microplastics. However, varying sampling techniques, metrics and even definitions of microplastics make direct comparisons between these studies difficult. A selection of microplastic concentration studies is provided in Table 1. These studies have been selected where the definition of microplastics of being 'less than 5 mm' in size is used and results are shown in the number of microplastic particles per kilogram of sediment (p/kg), rounded to the nearest decimal place.

Table 1 Mean and Max concentration (particles per kilogram of sediment (p/kg)) of microplastics in sediment of different areas. All studies identified microplastics as <5 mm

Study location	Mean (p/kg)	Max (p/kg)	Reference
Tunisia, Mediterranean	316	461	Abidli *et al*. (2018)
Balearic Islands, Mediterranean	–	900	Alomar *et al*. (2016)
Orkney, Scotland	730–2,300	–	Blumenröder *et al*. (2017)
Bohai Strait, China	102	256	Dai *et al*. (2018)
Canada	6656	25,368	Kazmiruk *et al*. (2018)
Europe (13 countries)	131–387	1,512	Lots *et al*. (2017)
Belgium	585	3,146	Maes *et al*. (2017)
France	481	1,509	Maes *et al*. (2017)
Netherlands	222	561	Maes *et al*. (2017)
English Channel, UK	306	643	Maes *et al*. (2017)
Halifax, Canada	–	8,000	Mathalon and Hill (2014)
Singapore	37	63	Nor and Obbard (2014)
Changjian Estuary, China	121	340	Peng *et al*. (2017)
Wanning, China	6,923	8,714	Qiu *et al*. (2015)
Baltic Sea	34	48	Zobkov and Esiukova (2017)
United States of America	1,636	6,110	Wilkens *et al*. (2019)

In the majority of studies in Table 1, microplastic pollution was found in all samples taken. Notably, a study taking five replicate samples from 23 beach locations across 13 different countries in Europe found microplastics in every sample, the lowest of which was 72 p/kg in Norway (Lots *et al*., 2017). However, this finding could be an artefact of studies specifically selecting sample locations with high anthropogenic inputs (see Qiu *et al*., 2015; Alomar *et al*., 2016; Peng *et al*., 2017; Abidli *et al*., 2018; Dai *et al*., 2018). Indeed, Maes *et al*. (2017) found 3 out of 27 sample sites containing no microplastic contamination when comparing sediments from four European countries. The samples containing no microplastics were found in subtidal sediments in the English Channel and the Netherlands. Conversely, a study specifically comparing sample locations in mangroves near and far from centres of human activity (Nor and Obbard, 2014) found microplastics in all samples.

Studies with sample locations in the United Kingdom are represented by two entries in Table 1 (Blumenröder *et al*., 2017 and Maes *et al*., 2017). Further studies include Thompson *et al*. (2004), which found microplastics in sediments for 23 of 30 estuarine, beach and subtidal samples in Plymouth, identifying them as fibres and fragments typically derived from clothing, packaging and rope. Browne *et al*. (2010) identified 952 microplastic items in 30 sediment samples throughout the Tamar estuary. However, in both cases, a size definition of less than 1 mm was used and

so these studies have not been included in Table 1. Additional UK studies cover a wider range of environments. For example, freshwater sediment studies have been completed in the River Thames, finding an average of 660 particles per kilogram of sediment using a definition of 1–4 mm for microplastics (Horton *et al*., 2017a), whilst 250–300 particles per kilogram of sediment were recorded in a shallow eutrophic lake in central Birmingham, using a size definition of 1.0–0.5 mm (Vaughan *et al*., 2017). A study has also been undertaken on the presence of microplastics within the water column in the Solent estuarine complex (Gallagher *et al*., 2016), finding an average of 172 particles within 16 ten minute trawls. The highest number of microplastics in a single trawl was 937 particles, found in the River Itchen.

Sources of microplastics

There are multiple sources of microplastics entering the marine environment. Drainage systems bring microplastics from cosmetics and clothing into the aquatic environment (Auta *et al*., 2017), either directly to sea or first into rivers and then into the sea. Recently, in one of the biggest Wastewater Treatment Plants in Northern Italy, despite 84% of microplastics being removed by the treatment system, it was estimated that 160 million microplastics particles were released daily into freshwater environments (Magni *et al*., 2019). Sewage sludge produced by Wastewater Treatment Plants is often used as fertilizers for agriculture and the same treatment plant produced 30 tons of sludge containing 3.4 billion microplastic particles each day.

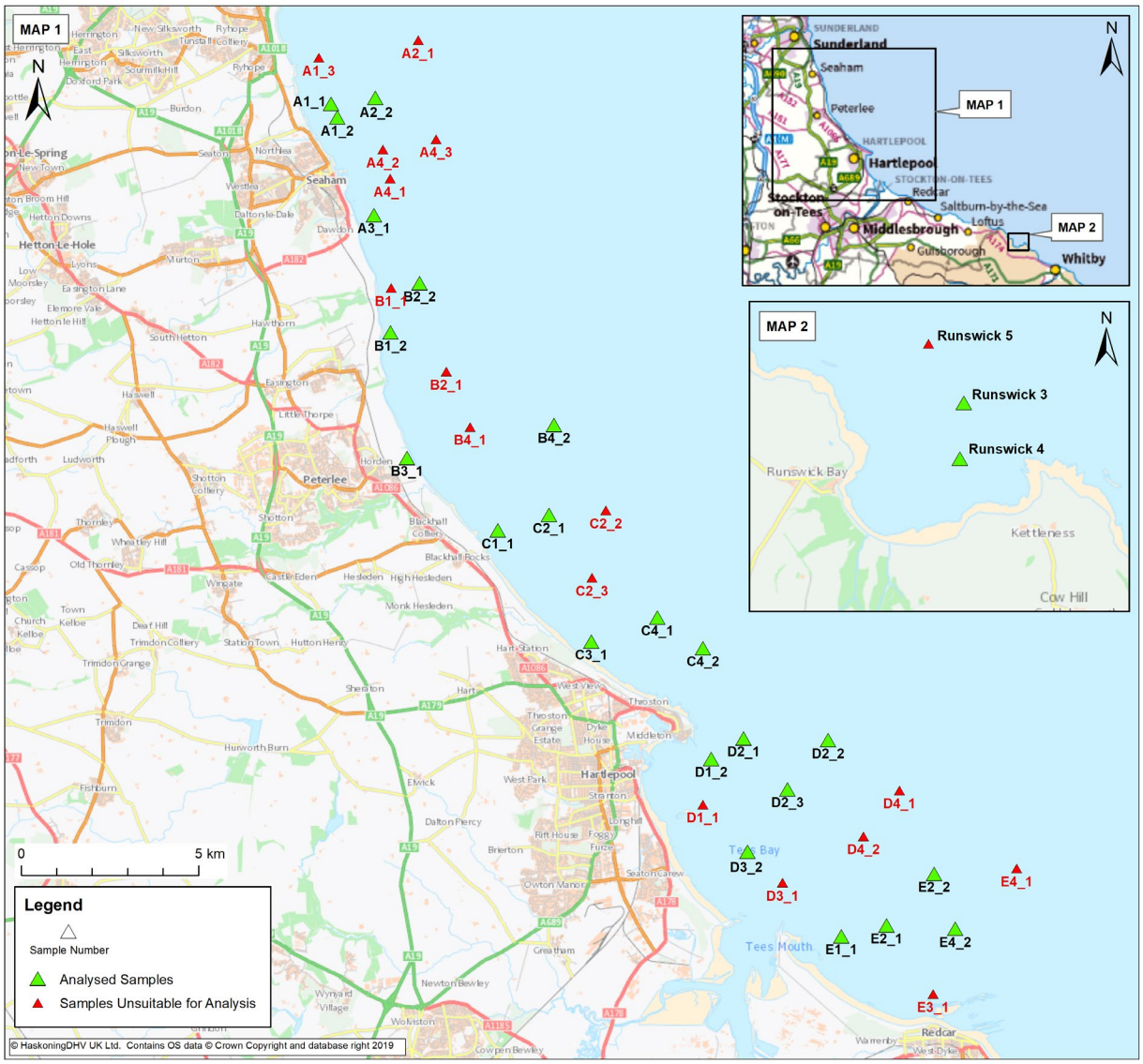

Fig. 1. Microplastics sample locations.

Plate 1. Drying of received samples.

Plate 2. Wet sieving of sample.

There are various mechanisms for the movement of microplastics from freshwater and terrestrial sources to the marine environment. Locally, run-off and wind are pathways for movement, as well as direct flows from storm drain systems (Auta *et al*., 2017) and even the atmospheric fallout of microplastic fibres can occur (Dris *et al*., 2016). Horton *et al*. (2017a) identified that microplastic fragments of paint from road markings were present in River Thames basin sediments, likely from run-off through storm drains.

Common anthropogenic sources are at centres of human activity, like cities, industrial sites and harbours. Within the marine environment activities of tourism, recreation, fishing, aquaculture and marine vessels can all contribute to the input of microplastics (Cole *et al*., 2011). Abidli *et al*. (2018)

Plate 3. Custom modified glass separating funnel.

investigated sites along the Tunisian coast finding microplastic concentrations highest near human and industrial activity. Peng *et al*. (2017) sampled the wider Changjian Estuary in China, finding the highest values in a geographical area with riverine inputs, a sewage treatment plant and near to Shanghai, the most populated city in China.

The highest maximum value for microplastic pollution in the studies identified in Table 1 was that found by Kazmiruk *et al*. (2018) in Canada. The high result was contributed to by both the geography of the area, a shallow coastal bay surrounded by tidal estuaries and inshore marshes, and the intense levels of shellfish aquaculture near the sampling site. The release of microfibres from fishing nets is a well-known source and Kazimuruk *et al*. (2018) hypothesised microbeads were likely to be sourced from the aquaculture industry as well. It has been suggested that fisheries-related activities are responsible for large amounts of the marine debris present in UK waters and beaches (Unger and Harrison, 2016).

Conversely, in a similar study sampling low energy (mangrove) areas near high levels of aquaculture, results found the second lowest concentrations in Table 1 (Nor and

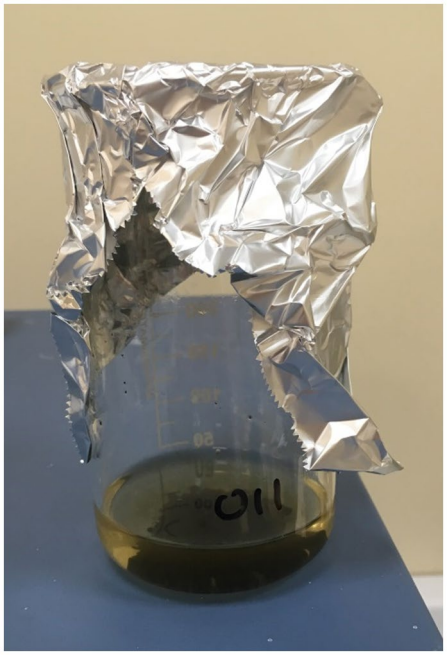

Plate 4. Sample undergoing WPO.

Plate 5. Sample undergoing Microscopic examination.

Obbard, 2014). In a study comparing results from sites within an urbanised and highly populated coastal bay against results from sites within two different Marine Protected Areas (MPAs) on the Balearic Islands in the west Mediterranean, it was found the highest microplastic concentration was in the MPA with the highest level of restrictions (Alomar *et al*., 2016). Additionally, Laglbauer *et al*. (2014) found no significant difference in microplastics between tourist and non-tourist beaches in Slovenia.

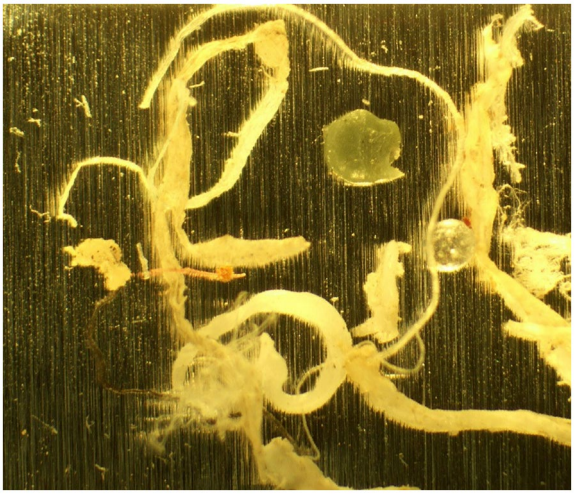

Plate 6. Microbeads, microfibres and microfragments under Microscopic examination.

From the studies discussed, it is clear that there is not always a clear correlation between microplastic concentration and proximity to anthropogenic sources. This variation in values can often be attributed to geographical conditions affecting the transport of microplastic particles.

Transportation of microplastics

Once within the marine environment, microplastics can be transported on a far-field scale. Due to their size and low density, currents can distribute microplastics across large distances leading to widespread transport (Eriksson *et al*., 2013). Winds, waves, tides and tsunamis can all contribute to transportation (Wang *et al*., 2016; Zhang, 2017).

Studies selecting sampling areas for their relative remoteness, such as Orkney in Scotland, found that microplastic contamination levels were comparable with mainland UK values in areas with much higher anthropogenic activity (Blumenröder *et al.,* 2017). It was found that microplastic concentration was significantly higher in remote Iceland when compared to wider European values (Lots *et al*., 2017), noting the potential travel of plastic debris on the North Atlantic current.

Dai *et al*. (2018) sampled the Bohai sea, a semi-enclosed inner sea surrounded by one of the most densely populated and industrialised zones in China with multiple riverine inputs including the Yellow River – the second largest river in the world for sediment load. However, as seen in Table 1, microplastic concentration was comparably low. This could relate to strong tidal and wind-driven currents within the sample area causing considerable dispersion (Dai *et al*., 2018).

In general, the low density of plastic means that a large proportion of plastic debris floats on the surface of the marine environment (Maes *et al*., 2017). However, where particles are denser than seawater some, such as polyvinyl chloride (PVC), may settle on the sea bed (Engler, 2012). Plastics that float can be entrained in marine sediments through biofouling, gaining weight and sinking to the sea bed (Ye and Andrady, 1991). Additionally, microplastics can become entrained in sediments through the ingestion and egestion of microplastics in faecal pellets by zooplankton (Auta *et al*., 2017). It has also been shown that the polychaete *Arenicola marina*, a marine worm species common in high numbers across coasts in Britain (Tyler-Walters, 2008), promotes the burial of microplastics in sediment at up to 20 cm depth due to ingestion and bioturbation (Gebhardt and Forster, 2018). Marine sediments have been found to act as sinks, areas of long-term burial, for microplastic pollution in areas like the deep sea and submarine canyons (Pham *et al*., 2014; Woodall *et al*., 2014).

There is a difference between the transport of microplastics to areas and the entrenchment of the microplastics into marine sediments. It was found in Dai *et al*. (2018) that the concentration of microplastic particles in the water column was not consistent with concentration within sediments, with areas with the highest water column microplastic concentrations having the lowest sediment concentration and *vice versa*. A study looking at the different beach environments in Halifax Harbour in Canada found no statistical difference in the average number of microplastic particles between different beaches. There was, however, a significant difference in where those particles were found, with more exposed beaches having higher values in the strandline in the upper shore and low energy beaches having increased concentrations on the lower shore (Mathalon and Hill, 2014).

It appears that the factors that determine the sedimentation of fine sediment particles may similarly govern the settlement of microplastics. Maes *et al*. (2017) found that in samples with a smaller median grain size a higher number of microplastic particles were found. Similar results had been found by Vianello *et al*. (2013), looking at microplastic presence in a lagoon in Italy.

However, both Peng *et al*. (2017) and Alomar *et al*. (2016) found no significant relationship between sediment grain size and microplastic concentrations, and similar results have been found in a number of studies (Thompson *et al*., 2004; Nor and Obbard, 2014). Browne *et al*. (2010) proposed that other processes may be playing a more important role, such as the aggregation of microplastic particles with organic material, consequently demonstrated by Strand *et al*. (2013).

Table 2 Microplastic particles per kg sediment, by microplastic type, sediment fractions and sorting coefficients

Reference	Microfibre	Microbead	Microfragment	Total Microplastics	%Gravel	%Sand	%Mud	Sorting Coefficient
A1_1	20	5	165	190	0.00	0.81	0.19	1.06
A1_2	3	0	31	34	0.00	0.87	0.13	1.07
A2_2	5	0	5	10	0.00	0.84	0.16	1.01
A3_1	9	0	9	18	0.00	0.93	0.07	0.85
B1_2	16	0	19	35	0.07	0.93	0.00	0.77
B2_2	40	0	22	62	0.00	0.71	0.29	1.48
B3_1	16	6	16	38	0.00	0.95	0.05	1.25
B4_2	45	3	27	75	0.00	0.17	0.83	2.31
C1_1	7	7	7	21	0.07	0.93	0.00	0.97
C2_1	14	0	23	37	0.00	0.88	0.12	1.20
C3_1	10	0	17	27	0.04	0.92	0.04	1.06
C4_1	191	10	57	258	0.00	0.69	0.31	2.11
C4_2	58	5	48	111	0.05	0.57	0.38	2.98
D1_2	10	0	13	23	0.00	0.95	0.05	0.76
D2_1	20	0	0	20	0.00	0.97	0.03	0.85
D2_2	97	17	46	160	0.00	0.37	0.63	2.72
D2_3	3	0	3	6	0.00	0.89	0.11	1.30
D3_2	13	0	31	44	0.00	0.94	0.06	0.75
E1_1	12	0	16	28	0.09	0.88	0.04	1.68
E2_1	21	4	14	39	0.00	0.93	0.07	0.63
E2_2	21	6	21	48	0.00	0.90	0.10	0.77
E4_2	20	0	4	24	0.00	0.96	0.04	0.47
Runswick 3	56	0	25	81	0	0.88	0.12	1.54
Runswick 4	329	31	172	532	0	0.20	0.80	2.12
Total	1,036	94	791	1,921				

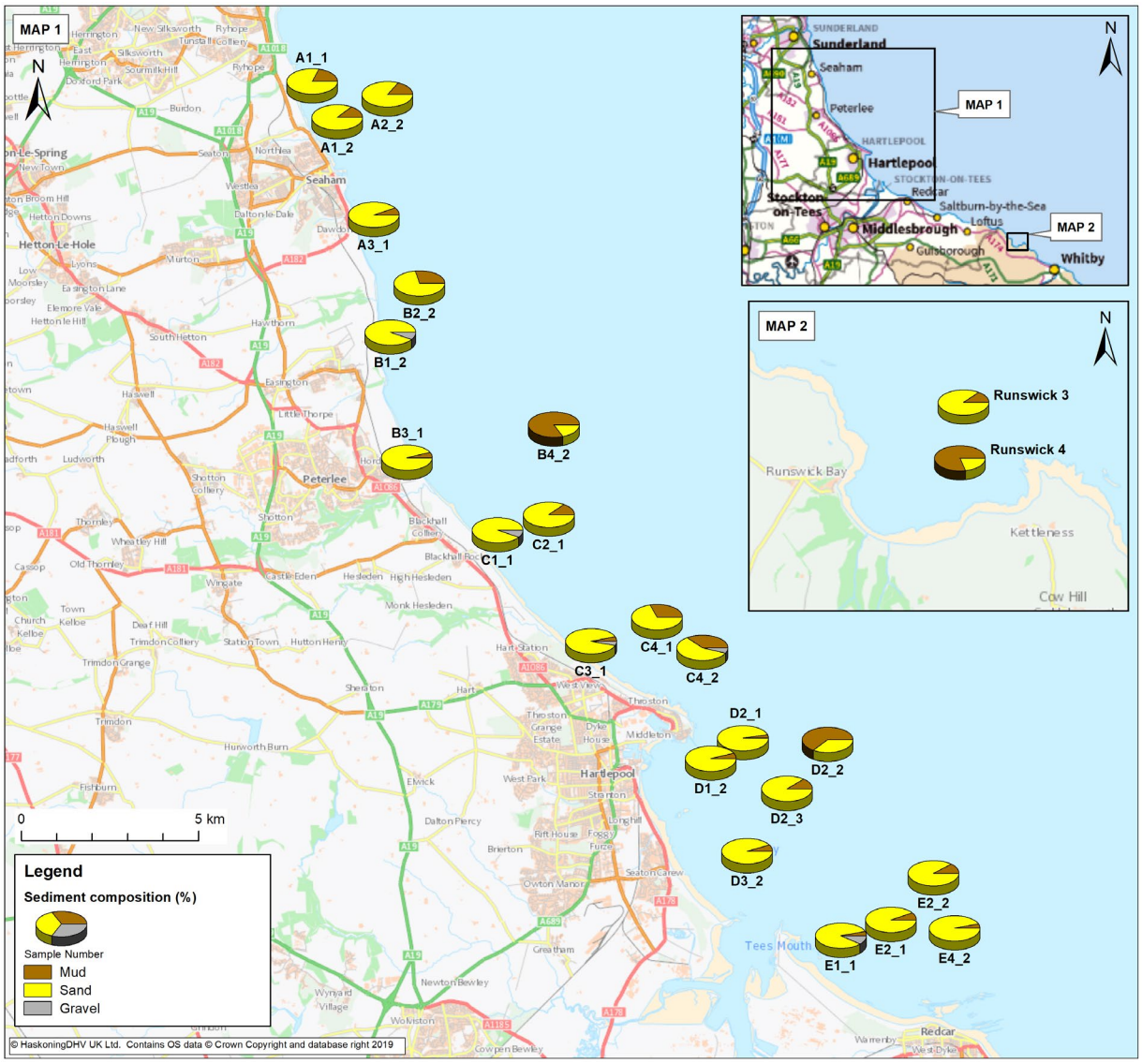

Fig. 2. Sediment composition.

Types of microplastics

More information on the complex relationships influencing microplastic concentration and spatial variation can be found when looking at the types of microplastics found in previous studies. Secondary microplastics, specifically fibres, are the predominant type of microplastics found with the least common being primary microbeads (Claessens *et al*., 2011; Laglbauer *et al*., 2014; Nor and Obbard, 2014; Peng *et al*., 2017; Abidli *et al*., 2018).

These differences could be attributed to the difference in how the types of microplastic can be transported. Alomar *et al*. (2016), comparing urban and marine protected area (MPA) microplastic concentrations, found fibres were higher in urban areas and MPAs had higher fragments. Indeed, fibres are the main output from sewage and wastewater treatment plants, with an estimated 1,900 fibres released into the environment from washing a single piece of clothing (Browne *et al*., 2011).

The relationships between microplastics and their spatial abundances are, therefore, complex and not yet fully understood. Multiple and varied sources as well as a number of varied factors govern movement, sedimentation and accumulation of microplastics. Factors range from geographical, biological to chemical and contradicting results from studies are common.

There is no single reason for such confusion, which could be increased due to several factors. Comparing results of microplastic concentrations in studies in Greece between 2013 and 2017 shows an increase from 1.5–15 p/kg (Kaberi

et al., 2013) to 232 p/kg (Lots *et al*., 2017). But when looking more closely the 2013 study only sampled microplastics below 1mm, while the majority of microplastics found in 2017 were above 1mm.

Varying sampling techniques can over- or under-estimate different types of microplastic (Song *et al*., 2015) and cross-contamination is still a significant issue (Prata *et al*., 2019). Additionally, there is a wide range of sediment sampling depths used throughout studies, such as 1 cm (Browne *et al*., 2011), 3 cm (Mathalon and Hill, 2014), 5 cm (Van Cauwenberghe *et al*., 2013) and 10 cm (Ng and Obbard, 2006). Differences in sampling depth may not produce an accurate estimation of microplastic concentrations as the top 1–5 cm commonly has higher concentrations than the top 10cm (Prata *et al*., 2019). There are also differences in the separation methodologies used depending on the nature of the samples (i.e. seawater, benthic sediment, aquatic).

Separation strategies for microplastics include floatation, evaporation, filtration, sieving and visual sorting (Herrera *et al*., 2018)). These techniques are useful for isolating microplastics from sediments, but isolating them from biological material requires a different treatment. While certain irregularities may be due to varying sampling techniques, undoubtedly the main constraint is the lack of available information. Further evidence from a wider number of studies is required to be able to show the wider state of microplastic pollution. As noted earlier within this section, many microplastic studies target areas known for plastics contamination sources or plastic debris accumulation,

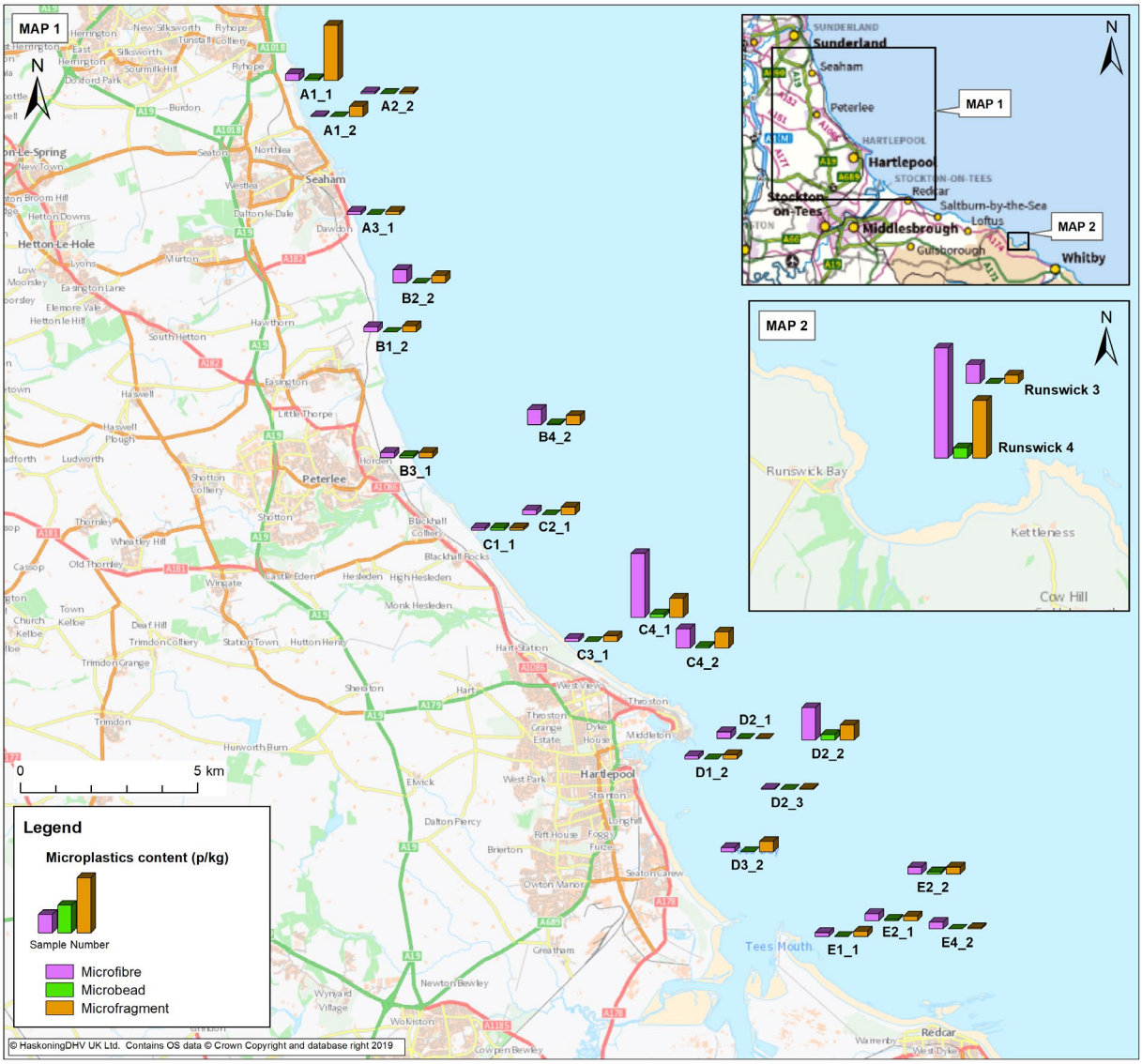

Fig. 3. Microplastics content.

further studies with wider scopes are, therefore, needed to provide a more holistic view.

Case study: North Sea, off northeast England

Methodology

Background

The literature review provided confidence that appropriate sampling and laboratory methods could successfully be employed to undertake a northeast England case study into microplastics in the North Sea. Whereas previously published research has been undertaken exclusively for scientific research purposes, the northeast England case study was intended to provide a baseline survey, for applied marine management purposes, that could be repeated in future years to determine trends over time. Furthermore, previously published research tended to focus on the areas of known, or expected, microplastics pollution (thereby skewing the findings towards generally high content values), whereas the northeast England case study was intended to cover a broad area of sea bed more generally to determine if there was a presence of microplastics and, if so, whether there were particular spatial variations.

Sample collection

Surface sediment samples (0.1 m depth) were taken using a Van Veen grab in subtidal sea bed sediments as part of the Cell 1 Regional Coastal Monitoring Programme covering northeast England (Cooper *et al*., 2009). This is a conventional and widely used sampling approach for sea bed samples. A total of 41 sites were sampled in December 2017 and January 2018 from Runswick Bay (three samples) and off the coast between south Sunderland and Redcar, comprising County Durham, Hartlepool and Tees Bay (38 samples). The location of all sample locations is displayed in Fig. 1.

Where sampling was not possible or where returned samples were not suitable for particle size analysis, these sites were not included in further analysis. Samples from the remaining 24 sites were sent to the laboratory for particle size analysis and then analysed for microplastics.

Microplastics analysis approach

Samples were subjected to a method designed for the extraction of potential microplastics from marine sediment samples. Potential microplastics were defined as solids that have a size between 0.3 and 5 mm, are resistant to a Wet Peroxide Oxidation (WPO) extraction, float in a density flotation liquid (density 1.6 g/ml) and pass a visual inspection under a microscope at 40x magnification.

Sample drying

Up to 400 g of each sample was placed 'as received' in a glass beaker. These were placed into a drying cabinet and the samples allowed to dry (Plate 1). The dry weight of each sample was recorded.

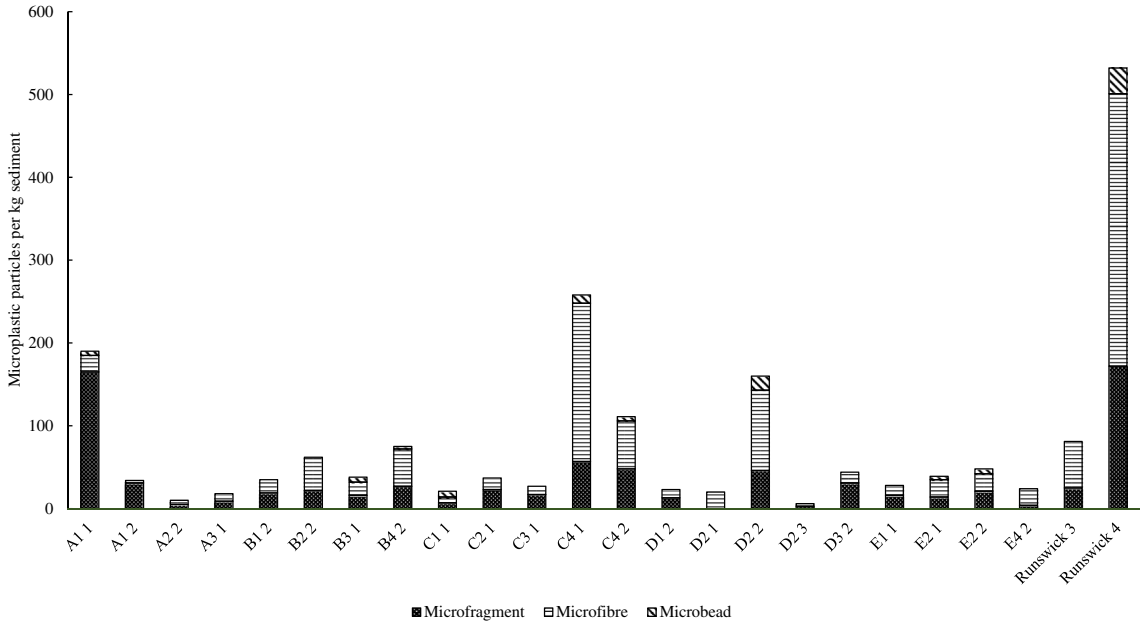

Fig. 4. Microplastic content (total and type) by site.

Wet sieving

Filtered deionised water was added to each dried sample to make it into a sludge. The sludge was passed between 5 and 0.3 mm sieves, rinsing with water until the flow ran clear (Plate 2). The 0.3–5.0 mm isolated fraction was then dried.

Density separation

The 0.3–5.0 mm dried isolated fraction was transferred into a density separator glassware. About 300 ml of sodium heteropolytungstate (d. 1.6 g/ml) density liquid was added, shaken and the contents allowed to settle. The floated material was filtered through a membrane filter, rinsed with water and dried (Plate 3).

Wet Peroxide Oxidation (WPO) extraction

The density separated material was rinsed into a glass beaker using 20 ml of iron(II) catalyst and 20 ml of 30% Hydrogen peroxide (Plate 4). The beaker was covered and placed into an orbital incubator at 50°C and 120 rpm revolution. The sample was mixed until the effervescence stopped. If natural organic material was still visible then the sample was repeatedly extracted with additional 30% hydrogen peroxide. The contents were filtered through a membrane filter, rinsed with water and allowed to dry.

Microscopic and gravimetric examination

Under a microscope using a x40 magnification (Plate 5), the solids recovered from the floatation and WPO extraction were examined to identify potential microplastics, characterising them into microfragments, microfibres and microbeads (Plate 6). The isolated material was weighed using a microbalance.

As a quality control, a blank sample was carried out using baked sand to ensure external contamination was controlled. To ensure extraction efficiency, a sample of baked sand spiked with a known level of microplastics was analysed along with samples.

Results

All samples analysed contained microplastics and values have been presented in Table 2. The number of microplastic particles (per kilogram of sediment) found in a single sample ranged from 6 particles (D2_3, Tees Bay) to 532 particles (Runswick 4, Runswick Bay). The result at Runswick 4 accounted for 28% of the total microplastics found across all samples. The average number of microplastic particles found was 80 p/kg (σ 114 p/kg); however, when excluding the relatively high Runswick 4 result the average is 60 p/kg (σ 63 p/kg). The most common microplastic type was microfibres, with some found in all samples and accounting for 54% of the microplastic particles found overall. Microfragments accounted for 41% of microplastics and were present in all samples except one (D2_1). Microbeads were the least common microplastic type (accounting for the remaining 5%) and were present in only 10 of the 24 samples.

Particle size analysis showed that the majority of sites sampled were characterised predominantly by sand, but

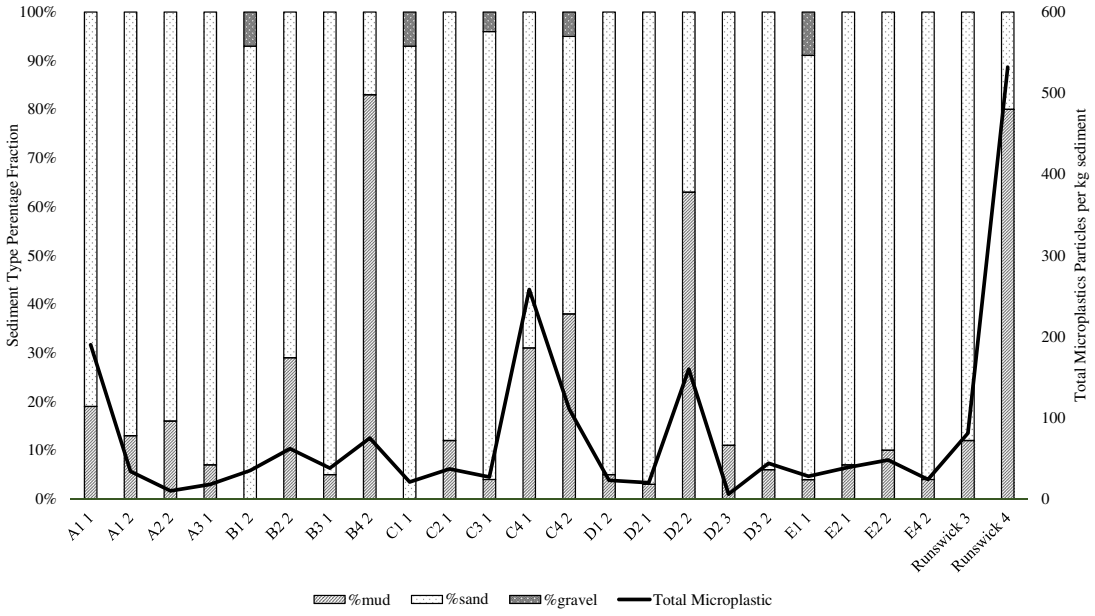

Fig. 5. Total microplastic content (particles per kg sediment) and percentage sediment fractions by site.

also containing some mud fractions (Fig. 2). Table 2 also presents the sorting coefficient of the samples, where the higher value means there was more variation of particle sizes within the sediment sample.

Figures 3 and 4 show the microplastic content by site, with values split between microplastic types, revealing that sites predominantly contained values below 50 microplastic particles per kg sediment, with some isolated sites showing hotspots of microplastic contamination. Figures 5 and 6 present some apparent correlation between the fine sediment fraction, sorting coefficient and total microplastic content, and Fig. 4 shows a similar correlation with respect to each of the microfragments, microfibres and microbeads.

Discussion

Comparing the results found in Table 2 with the summary of results from projects using the same metrics in Table 1, the concentrations sampled are comparatively low. The lack of similar UK studies, however, prevents effective comparison and as noted previously, many of the comparative studies targeted areas that were expected to have high concentrations of microplastics. Additionally, many of the studies in Table 1 sample beach sediments, whereas the northeast England study sampled subtidal sea bed sediments only.

A comparative methodology in Zobkov and Esiukova (2017) sampling subtidal coastal sediments found lower values identified in the northeast England study. The mean value of samples (80 p/kg) in the present study was much lower than the mean of samples taken in the English Channel (306 p/kg) presented in Maes *et al*. (2017), though the maximum values identified were comparable, 643 p/kg in Maes *et al*. (2017) compared to 532 p/kg in the northeast England study.

Direct comparisons should not be accepted without consideration of the further processes that impact microplastic

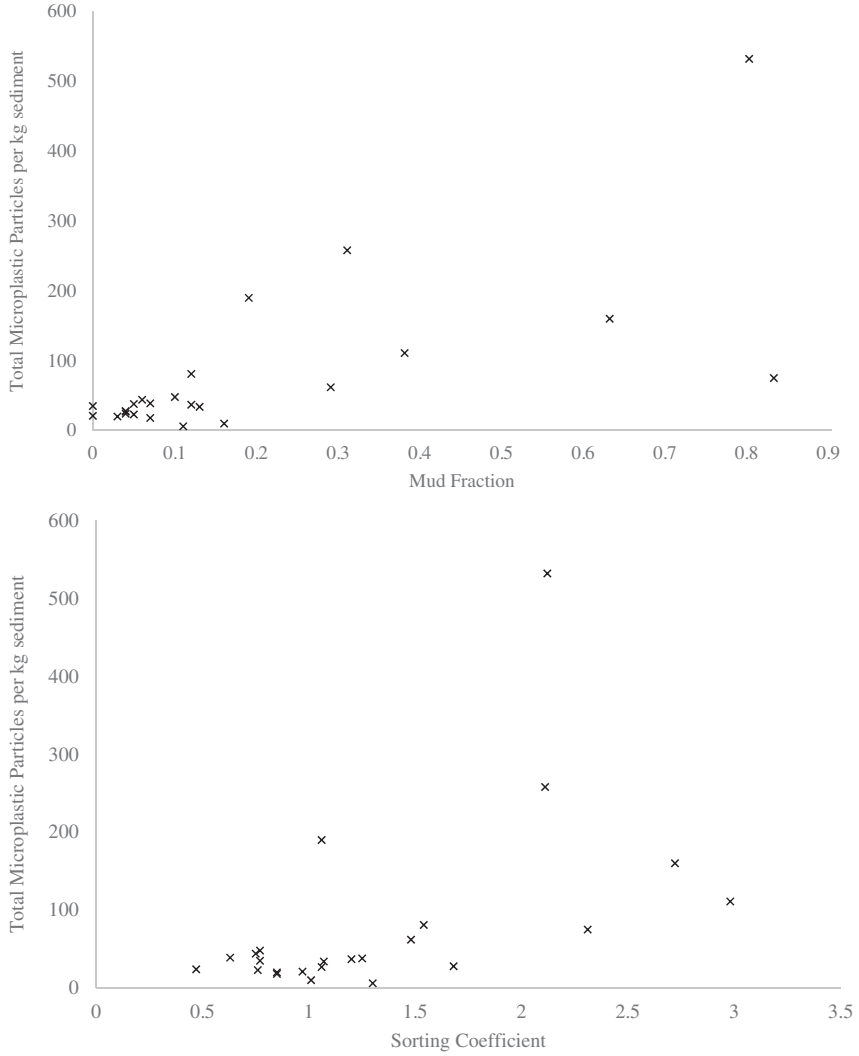

Fig. 6. Total microplastic (particles per kg sediment) vs Fine Sediment fraction (top) and Sediment sorting coefficient (bottom).

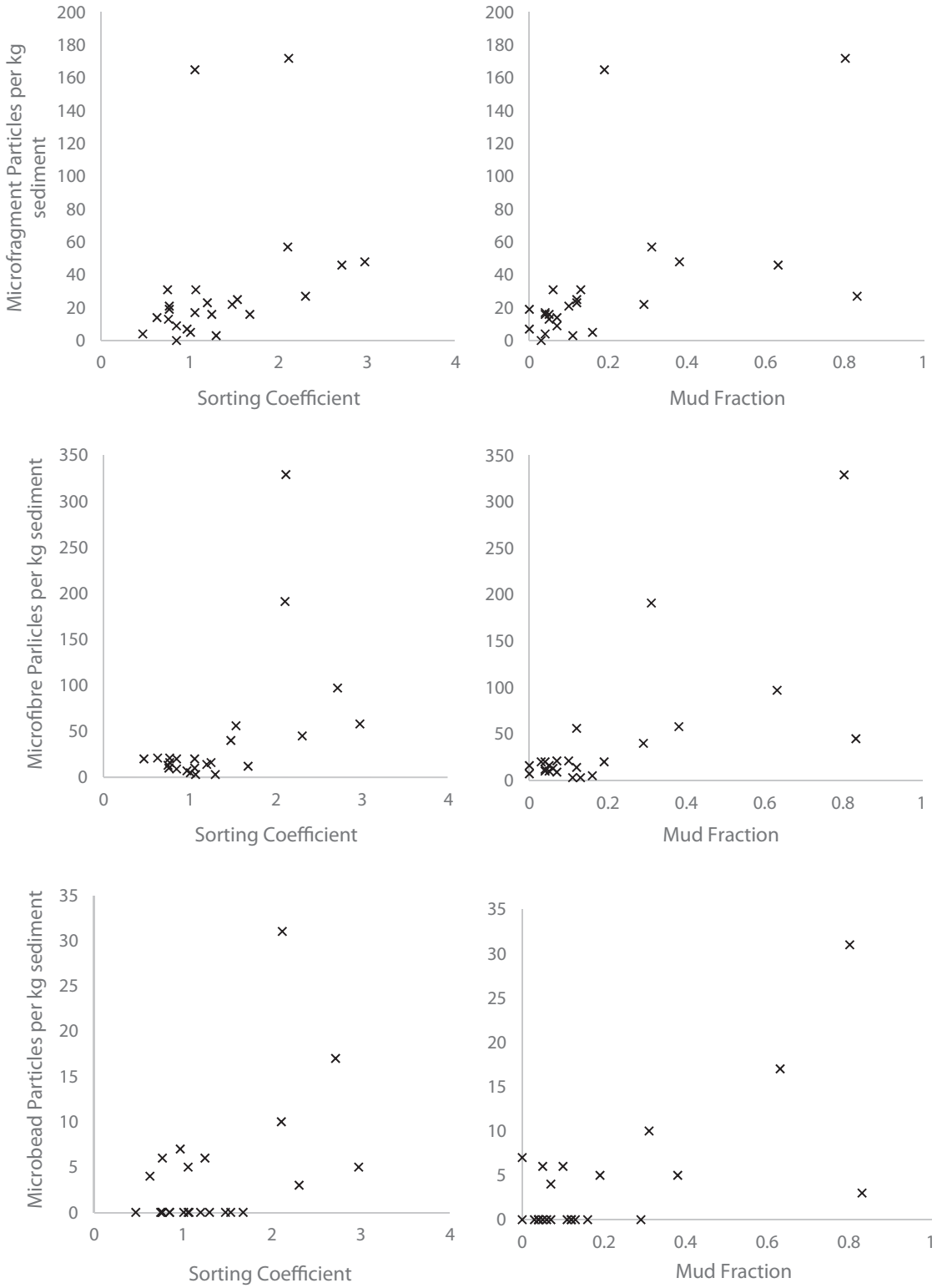

Fig. 7. Microplastic content (particles per kg sediment) by type (Microfragment – top; Microfibre – middle; Microbead – bottom) vs sediment sorting coefficient (left) and fine sediment fraction (right).

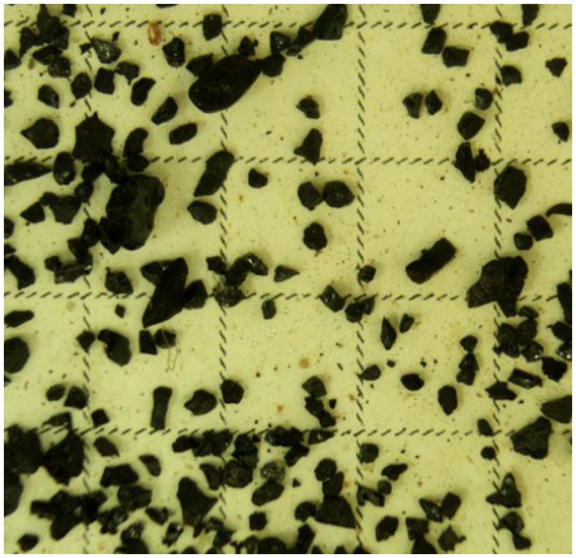

Plate 7. Coal particles (recovered from a sediment sample).

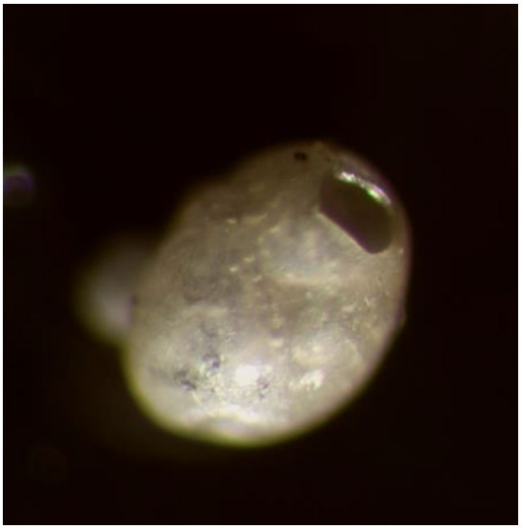

Plate 8. Fly ash particle recovered from isolated coal particles (this could be misidentified by an untrained microscopist as a microbead of plastic).

contamination. For example, the study in Table 1 with the most comparable results to the present study is from a semi-enclosed sea surrounded by the most densely populated and industrialised zones in China, with multiple riverine inputs (Dai *et al*., 2018). However, in that study it was noted that the low concentrations of microplastic particles in the sediment were not consistent with higher concentrations within the water column sediments, indicating high currents in the area prevented the settlement of microplastics within the sediment.

It appears that there is a weak positive correlation between the percentage of mud in the samples with the total microplastic content (Figs. 5 and 6 (top)). However, this is not the case in all situations. B4_2, for example, has a fine sediment fraction of 83%, but total microplastic is slightly below average (75 p/kg). Similarly, a general trend for an increased sorting coefficient (poorly sorted sediment) increasing microplastic concentration can be seen (Fig. 6 (bottom)). Poorly sorted sediment tends to relate to lower energy environments, so this may indicate that the high levels found in Runswick Bay could be due to the sheltered nature of the area. Similar patterns are seen when comparing the same plots but separating microplastics types (Fig. 7). However, without statistical analysis, it is unconfirmed whether these potential relationships have a significant correlation or not, and as noted within the literature review there are many other factors that influence the sedimentation of microplastics.

When reviewing the types of microplastics found in the sample results, the results matched wider findings of microbeads being the least common and microfibres being the most common. Samples with comparatively high microfibres included C4_1, C4_2 and D2_2, grouped to the north of Tees Bay. However, samples to the north of Seaham indicated a source of microfragments, specifically for sample points A1_1 and A1_2.

Reviewing the data, no clear spatial pattern is immediately apparent. This is likely to be due to the multiple factors that influence microplastic spatial distribution, including physical and chemical processes and anthropogenic activities. Local physical processes of influence are likely to include the riverine input from the River Tees, run-off from the urban areas of Hartlepool and Seaham, and wastewater treatment plants along the coast. Human activities are likely to include navigation dredging (capital and maintenance) and fishing activities within the area. These factors result in dispersing, mixing or homogenising the microplastic particle deposits and masking any pattern that might otherwise more directly result from the sources of the microplastics. Indeed, if microplastics can be found in high concentrations in Orkney despite a lack of local plastics sources (Blumenröder *et al.,* 2017), then there is no guarantee that sources on this coastline contribute to concentrations of microplastics in local sediments. Even microplastics within the sediments can be subject to further disturbance, with sample points D3_2 and D2_3 being located within a regularly dredged channel.

Within the samples, a high number of spherical coal particles and fly ash were also identified. In some samples over 1,000 coal particles were identified within the sample. This is mentioned here for two reasons.

Plate 9. Colliery spoil beach at Horden, County Durham.

First, sediment samples that contain significant amounts of coal fragments can cause major interference issues with the analysis of microplastics. One of the main isolation stages for microplastics is the density floatation. This is designed to float-away the microplastics from the heavier material such as sand and gravel. Plastics have a variety of densities and density floatation is carried out using a liquid solution at 1.6 g/cm^3 which will float all common plastics. However, coal has a density between 1.2 and 1.8 g/cm^3 (depending on the type) and thus also becomes floated-away with the microplastics during the floatation process if it is present within the sample. Whilst it is possible to use a floatation liquid of 1.15 g/cm^3 to float polyethylene and polypropylene away from coal particles, this will result in some other plastics being lost from the analysis. For the study, a detailed search of each sample was performed under the microscope to identify and remove individual coal particles from the microplastics within the sample (Plate 7). An important part of this process was to recognise that fly ash within the samples (derived from burnt coal) looks like spherical balls, which can be misidentified by an untrained microscopist as microbeads of plastic (Plate 8).

Second, the presence of coal particles and fly ash within the samples is highly likely to be associated with the legacy of colliery spoil tipping which occurred along the County Durham coastline and has previously been discussed in detail in the Cell 1 Sediment Transport Study (Royal HaskoningDHV, 2014). Tipping began around 1900 and continued until 1993 with the closure of the last colliery. At the peak of tipping over 2.5 million tonnes of spoil were tipped in one year (1983) and in total it is estimated that around 100 million tonnes were tipped over 90 years, both at foreshore tipping sites and at offshore disposal grounds. This tipping despoiled the beaches and resulted in significant progradation of the shore (Plate 9). Since cessation, there has been a tendency for ongoing erosion of the spoil beaches. The tipping (and cessation) of colliery spoil along this coastline has, therefore, caused significant anthropological effects on natural processes (Cooper *et al*, 2017).

It is concluded from the literature review that the high number of factors that could be impacting the concentration, distribution and type of microplastics within marine sediments within the study area does not allow for the attribution of the results found to any specific factors. However, this study does present a first baseline assessment for the area. The broad nature of the sampling means that there was no targeted approach to (artificially) identify high microplastic concentrations, contributing a more complete picture of microplastic pollution to existing literature.

Conclusions

(1) It is clear from the literature review and the results from the northeast England study that a multitude of factors govern the sources, movement and sinks of microplastics in the marine environment. Further statistical analysis and data on these factors, such as the interpretation of local currents and sediment movements, areas of erosion and accretion, fishing effort and the location of wastewater treatment plants and storm drains, may provide further detail and interpretation of the results.

(2) The present study presents a baseline for the sea bed between south Sunderland and Redcar, and in Runswick

Bay, for subtidal marine sediments off the northeast England coast. Repeat surveys in future years and decades, as well as further data from beach sediments and water column samples, would present a wider understanding of the movements of microplastics within the study area. Interpretation of available data on both macroplastic and microplastic debris on local beaches could also provide further understanding of the wider plastic influences on the environment.

Data availability statement

Data available in article supplementary materia.

To submit a comment on this article please go to http://mc.manuscriptcentral.com/wej. For further information please see the Author Guidelines at wileyonlinelibrary.com

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