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Mud and organic content are strongly correlated with microplastic contamination in a meandering riverbed

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The scale of microplastic pollution in river sediments is gradually being elucidated through an increasing number of large-scale studies. Nevertheless, microplastic distribution within a riverbed – a crucial aspect for quantification – remains poorly understood. Here we evaluate in the meandering River Lys, Belgium, how microplastic concentration varies between different sedimentary environments within the riverbed. We find that microplastic abundance is about an order of magnitude higher towards the riverbanks compared to the thalweg, corresponding with river hydrodynamics. Moreover, organic-matter and mud content are robust predictors of microplastic concentrations, apart from the outer bends, where erosion into organic-rich, muddy floodplain sediments inhibits microplastic deposition. These results increase our understanding of microplastic distribution at the small riverbed scale. They are a crucial element to guide for future sampling efforts across diverse river systems, paving the way for normalization and better quantification of microplastics trapped by river sediments and in other aquatic environments.

Substantial amounts of microplastics (MPs) are likely accumulating in river sediments[1]. Considering the global-plastic-budget, Koelmans et al.[2] calculated that 99.8% of plastics that should have entered the ocean since 1950 is not detected in the ocean surface layer. Based on a global approximation, part of this fraction (3.1−14.4 million tons of MPs) resides on the ocean floor[3–6], while an even larger part (40–90 million tons) may be residing in the water column[5,7,8]. Other mechanisms removing MPs from the ocean surface layer include accumulation in marine organisms[9], accumulation on beaches[10] as well as physical[11] and microbial degradation[12] leading to smaller size ranges that can either float or sink, yet the plastic ocean cycle remains to be fully elucidated. Additionally, another part of these plastics may never reach the oceans, indeed MPs may have been sequestered and stored in river beds on their way towards the oceans, despite the constant water flow in rivers[13]. However, due to our poor knowledge of MP distribution in river sediments[14–16], it is currently not possible to make reasonable estimates of MP accumulation in these environments.

In 2014, MPs have been discovered in river sediments for the first time[17]. Since then, MP studies in river and all freshwater systems have increased rapidly, revealing that MPs are omnipresent in all freshwater environments[18–21]. Factors that are found to affect the abundance of MPs in freshwater sediments are population density, urban centers, water flow velocity, water catchment size and position and type of sewage and waste management[22–24]. However, not all of these relations are consistent. For example, while some studies report positive correlations[25,26] between MP source regions (highly populated and industrialized areas) and MP abundance in river sediments, many studies do not find significant correlations[21,24,27–29]. This lack of correlation has been attributed to hydrodynamic effects[22,27], and increased MP abundances have been predicted[30,31] and reported[28,32–34] for areas with lower flow velocities. Such lack of correlation has also been observed during MP-investigations of sediments of several Belgian rivers, in contrast to the concentrations in the water itself, for which better links with MP source areas were found[35,36]. What the vast majority of these studies have in common, is their large-scale approach, in which sediment samples are obtained over distances of tens to even hundreds of kilometers along the path of the river; and – depending on the study – at each site sediments are than retrieved from either the thalweg or the riverbank[18]. Few studies integrate samples from across the riverbed to obtain average values[33,36,37], or do compare river bed with floodplain sediments[34], but at present no study has thoroughly investigated the depositional patterns of MPs at the scale of the riverbed[16].

Considering the highly variable hydrodynamic conditions within a river channel[38] and the observation that hydrodynamic forces may strongly influence MP abundance at a site, one could expect large variability in MP abundance across a river bed[14–16,39]. Indeed, some studies indicate elevated

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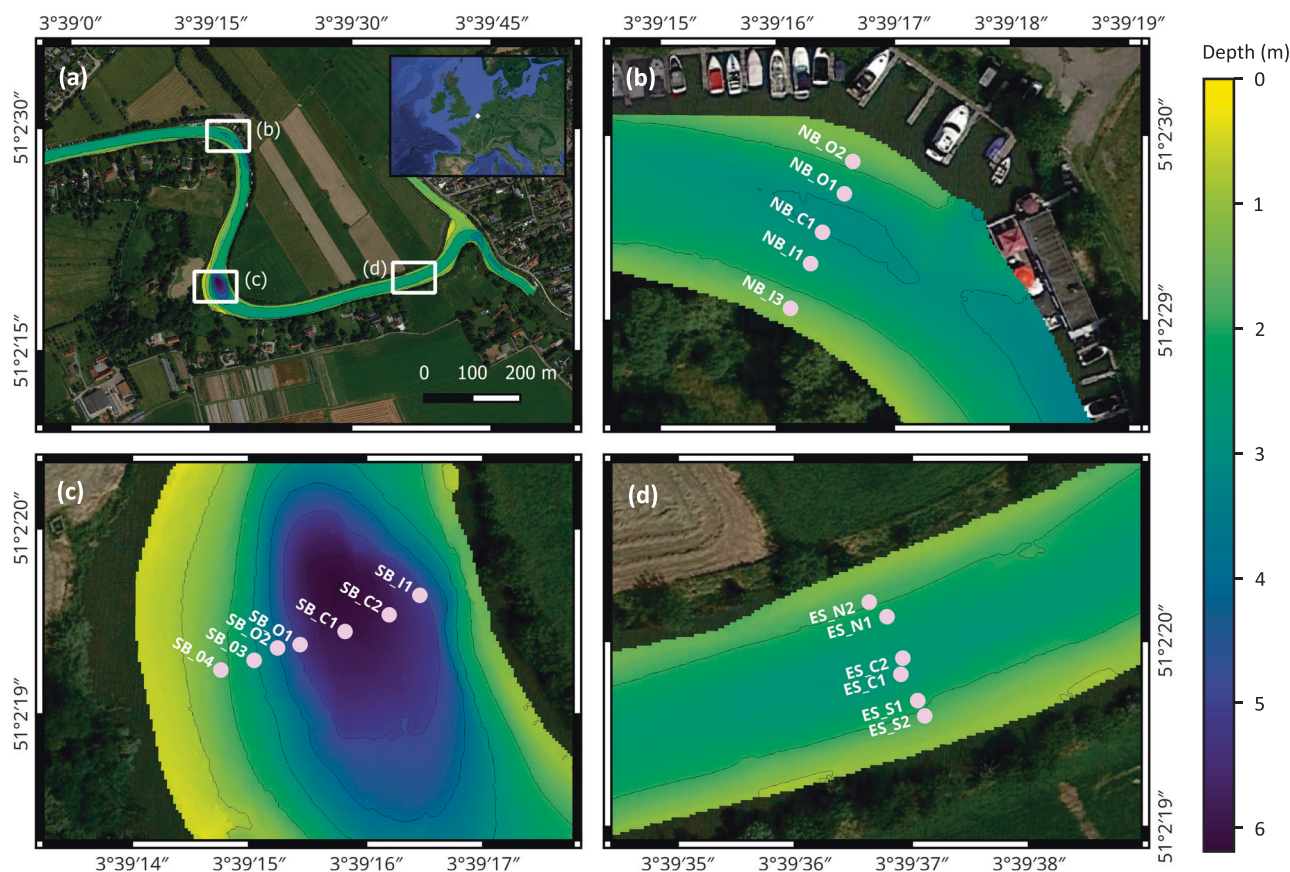✉e-mail: maarten.vandaele@ugent.be

1

Fig. 1 | Setting with aerial imagery, multibeam bathymetry obtained during this study and sample locations. a Overview of the study area with the two studied bends and straight section in the meandering Lys River. Inset with indication of study area in Europe. **b** North Bend (NB) with location of the 5 sediment samples (pink dots). **c** South Bend (SB) with location of the 7 sediment samples (pink dots). **d** East Straight (ES) with location of the 6 sediment samples (pink dots). Bathymetric contours at 1 m. Map data: Google, ©2024 Maxar Technologies; imagery date: 21 July 2021.

MP concentrations at point bars[34,40], which form in the inner bend of a river meander, where flow velocities are reduced. Since these hydrodynamic conditions also determine natural sediment transport and deposition[14], it is not surprising that in estuarine and marine environments the occurrence of MPs has been linked to finer clastic grain-size fractions[41–43]. In a river environment, several studies found good relations with sediment grain size and/or amount of organic debris[23,33,39,44], but the qualitative and large-scale approach inhibit high-quality regressions. Indeed, Enders et al. [41] argue that the MP-sediment correlation improves for small scale well-connected sampling areas, as the MP-sediment relationship is not obscured by variability in MP input. Understanding the distribution of MPs at the small scale of the riverbed, and the relation to its sedimentary environments, is crucial for developing appropriate sampling strategies for future larger-scale research and to facilitate comparisons of data between global research results[18]. Furthermore, if a strong relationship with natural sediment characteristics does exist, it may allow the prediction of MP abundances in the sediment and thereby reducing the number of costly and time-consuming MP analyses. Such understanding is paramount to obtain reliable estimates of total MP sequestration in river beds.

In this study, we investigate MP distribution within the riverbed of the meandering Lys River, near the city of Ghent, Belgium. We determined MP concentrations in the water column and surface sediments, and additionally sedimentological parameters (grain size, organic-matter content, carbonate content) of the surface sediments in the main sedimentary environments of the river. In accordance with fluvial hydrodynamics, we observe – an order of magnitude – higher MP concentrations on sites with lower flow velocities near the river banks compared to the thalweg, and the lowest abundances in the outer bend (i.e. the outerbank slopes of the thalweg in a bend). Our results further suggest a strong relationship between MP concentration, and

both organic-matter and mud content, only in the outer bend erosion inhibits permanent deposition of MPs but exposes old muddy, organic-rich floodplain sediments. The gained mechanistic understanding is broadly applicable, likely even on other types of rivers (braided, straight, anastomosing). Taken together, these results and their application represent a key ingredient for MP normalization and for quantifying the amount of MPs stored in river sediments, and even other aquatic sedimentary environments, and thus estimating to which extent river sediments store the plastic that is not detected in the oceans.

Results and discussion
Microplastic deposition from the water column
We used bathymetry and side-scan sonar imaging to select and sample surface sediments (0–3 cm) in three across-channel transects (5–7 sediment samples per transect; Fig. 1). We targeted the main sedimentary environments in the river by having two transects in the apex of a bend ("North Bend - NB" and "South Bend - SB", the latter with a deep pool zone; Fig. 1a, b) and one in a straight section ("East Straight - ES"; Fig. 1c) of the river. We found microplastics at all 18 sites where surface sediments were sampled, albeit with large variations in MP concentration between sites, and some replicates without MPs. Concentrations at single sites varied between $1.50 \pm 2.12 \times 10^2$ MPs kg^{-1} DW (dry weight), which is in the order of the field blank ($1.86 \pm 2.62 \times 10^2$ MPs kg^{-1} DW, i.e. 0–3 MPs in the ~8 g DW samples, only polypropylene), and $1.70 \pm 0.84 \times 10^5$ MPs kg^{-1} DW. These concentrations seem high compared to other rivers globally[16,19,45,46]. However, it is well known that MP concentrations increase with smaller particle sizes[19,47–49], and when correcting for our analyzed MP particle size range (lower cut-off at 25 μm), our corrected environmental concentrations (2.1×10^5–2.4×10^8 MPs kg^{-1} for a 1–5000 μm size range[49]) correspond to the

Fig. 2 | Diagrams visualizing average concentrations of microplastic (MP) types for each water depth and in the surface sediments. a Bar chart showing absolute number of each MP type (in # L^{-1}) at three depths in the water column. **b** Normalized MP abundances for each MP type at three depths in the water column (circles), with linear regression (lines). **c** Bar chart showing relative amount of each MP type (%) in the water column and in the surface sediments. **d** Box-whisker plots showing variability of relative amount of each MP type between sampling sites (center line, median; box limits, upper and lower quartiles; whiskers, 1.5× interquartile range; points, outliers). PS Polystyrene, PP Polypropylene, PET Polyethylene terephthalate, PE Polyethylene, PVC Polyvinyl chloride, PAM Polyacrilamide, PU Polyurethane.
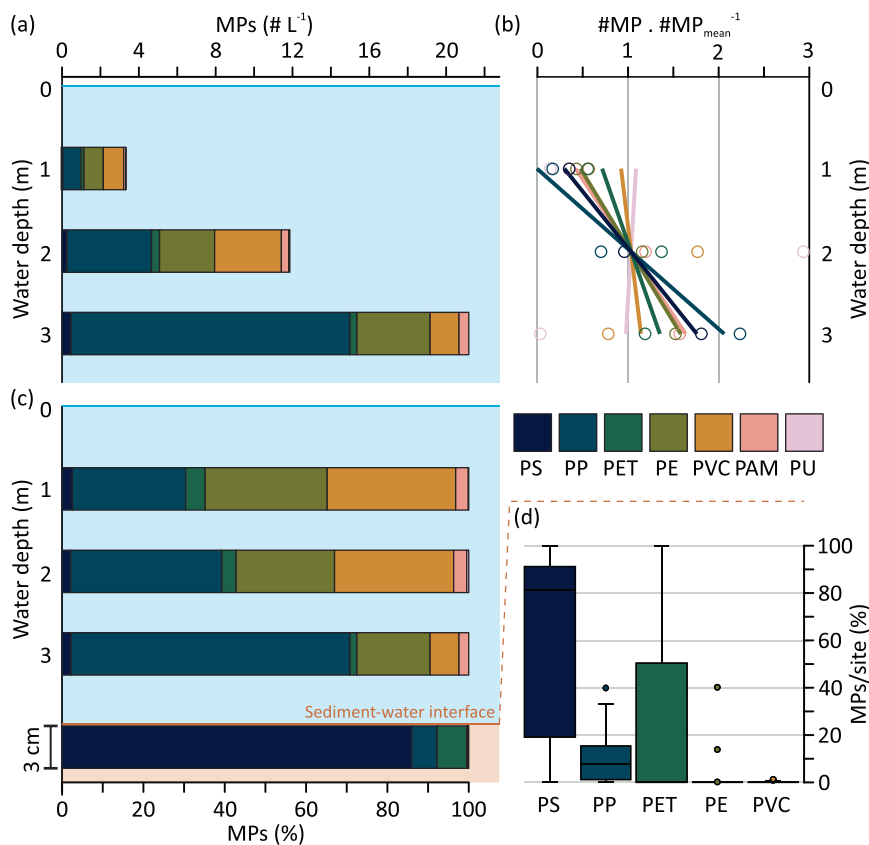
upper half of global frequency distributions[21]. Six polymer types were identified, with polystyrene (PS) being most abundant (86%), followed by polyethylene terephthalate (PET; 7 %) and polypropylene (PP; 6 %), whilst polyethylene (PE; 0.3 %), polyvinylchloride (PVC; 0.1 %) and polyacrylamide (PAM; 0.01%) were only sporadically detected (Fig. 2). PS and PP occur most regularly and were identified at nearly all sites (17 and 16 out of 18, respectively), whereas PET was only detected at half of the sites, indicating a more patchy distribution.

In the NB and ES transects we filtered water at different depths (1, 2 and 3 m) in the thalweg, and additionally at 1 m depth in the inner and outer bend (i.e. between thalweg and inner and outer bank, respectively) of the NB transect. At each station we filtered in total between 159 and 381 L (sum of 5 replicates spread over 6 weeks to reduce the influence of temporal variability of MPs in the water column) with an in-house developed filtering device for microplastic research (see (Supplementary) Methods). Also in the water, MPs were detected at all depths in all sampling locations, with concentrations ranging from 0.13 ± 0.19 to 25.75 ± 44.31 MPs L^{-1}, although in some replicates MP concentrations were 0 after LOQ corrections (see (Supplementary) Methods). In the water, the same 6 polymer types (PAM, PE, PET, PP, PS ad PVC) as in the surface sediments were detected, albeit in very different relative abundances, as well as almost negligible amounts of polyurethane (PU; <0.5%; Fig. 2c). Overall the most common polymer type was PP (47%), followed by PE (23%) and PVC (21%), while PET (3%), PAM (3%) and PS (2%) occurred in much smaller concentrations. The absolute concentration of almost every MP type increases with depth (Fig. 2a), but the normalized concentrations clearly show that the relation with water depth does depend on the type of MP (Fig. 2b). While PU, PVC and PET seem to show little to no relation with depth; especially PP, and to lesser extent PS, PAM and PE, become more abundant in deeper water. The tendency of especially PP to prefer the larger depths is remarkable as it is a low-density polymer (<0.9 kg m^{-3} [50]), and would thus be expected to float, as observed for PE, also a low-density polymer. This behavior could be related to differences in degradation, leading to smaller or larger particles for

specific polymers[51,52], or fouling rates which may influence sinking behavior[53,54]. Similarly noticeable is the tendency of the relatively high-density PVC (1.16−1.6 kg m^{-3} [50]) to show little relationship with water depth, while relatively low-density PS (1.04−1.1 kg m^{-3} [50]) prefers deeper water, again pointing to differences in degradation or biofouling of particles affecting overall behavior, the latter of which may be specifically relevant for PS[53]. Whatever the cause, there are some clear tendencies that seem to have an effect on depositional trends.

Even though there is a strong discrepancy between the most common types of MPs found in the water column compared to what is retained by the sediment, there is some convergence between both datasets. The two types of polymers that show preference for larger depths (i.e. PP and PS), are also the most widely distributed in the surface sediments (Fig. 2d), while polymer types that show no relation with water depth (i.e. PU, PVC) are poorly represented in the sediments (Fig. 2c). It is, nevertheless, striking that PS and PET, which are the most and second most abundant polymers in the sediments (87% and 7.0%, respectively), are almost absent in the water column (2.3% and 3.2%, respectively). Recent research showed a high influence of biofouling on the sedimentation of PS and PET[53,55], which we presume is, combined with an efficient retention of these MP types by the sediment, likely resulting in fast deposition when flow velocities are low, and lack of erosion – into suspension – at times of higher flow velocities. The latter could be related to the higher density of PS and PET compared to PP, which would increase the critical shear stress needed for erosion[56] resulting in transport of these particles predominantly as bedload[57], and the large contrast between PS and PET concentrations in the water column and in the sediment. In contrast, more than other polymer types, PP – which are normally hydrophobic[58] – are often treated to be more hydrophilic[59], and this could explain the high sinking rates and rapid resuspension following deposition, in turn explaining both the high concentrations in the deeper water and relatively low abundance in the sediment.

Our data thus confirms that certain MP types (here PS and to a minor extent PP and PET) are sequestered more efficiently[53,58] and those types will

thus contribute more to the plastic that does not reach the oceans. Apart from efficient sequestration, the MP concentration of the most abundant MP types in the surface sediments may further increase when subsequent remobilization is dominated by bedload transport (here PS and PET) as opposed to resuspension (here PP).

Depositional patterns of microplastics

As our sample locations are quite regularly spread along the three transects (Fig. 1), we can estimate the average MP concentration per transect and compare this to the area of the cross section, which is inversely related to average flow velocity. Our data show that even though the straight section (ES) has the lowest cross-sectional area (110 m² at the time of the bathymetric survey) and thus highest average flow velocities (i.e. ~4.5 cm s⁻¹ during the survey and ~24 cm s⁻¹ annual maximum), it has the highest average MP concentration (4.78×10^4 MPs kg⁻¹ DW), indicating that the MPs are preferentially deposited in straight parts of the river, as opposed to bends. This is likely a result of the strong (erosional) downstream flow velocities in the outer bend and the related secondary flow affecting the entire bend, further increasing shear stress at the river bed[38], which is also reflected by the more sandy sediments in especially the inner bends compared to the straight section. When comparing both bends there does seem to be an effect of the cross-sectional area. The wider and deeper SB transect (347 m²; flow velocity of ~1.4 and ~8.5 cm s⁻¹ during survey and annual maximum, respectively), has almost two times higher MP concentrations (2.87×10^4 MPs kg⁻¹ DW) than the NB transect (132 m²; ~3.8 and ~20 cm s⁻¹; 1.63×10^4 MPs kg⁻¹ DW). Hence, within a similar environment the average flow velocity does influence the MP sequestration by the river sediments.

Variability within one transect easily outpaces that observed between transects. The ES transect, with its relatively symmetric bathymetric profile, exhibits a clear trend of low MP concentrations in the thalweg and increasing MP abundances towards the banks, albeit less pronounced on the southern slope (Fig. 3a; Supplementary Fig. 2a). Such trend is indeed what can be expected based on river hydrodynamics with the core of the flow in the thalweg causing higher shear stress in this environment. However, the scale of the variability, of about one order of magnitude, is striking, especially for this calm river (average flow velocity of ~1–5 cm s⁻¹ at time of survey). Also in the bends, which both have an asymmetric bathymetric profile, the patterns of MP distribution are not very surprising. Both in the NB and SB the lowest MP concentrations are found in the outer bends (Fig. 3b, c; Supplementary Fig. 2b, c), where the highest flow velocities occur due to centrifugal forces. Especially the lower slopes from the thalweg to the outer banks seems to lack MPs, while the shallower areas closer to the outer banks again have slightly higher MP abundances, again explained by typically lower flow velocities in these areas[38]. Nevertheless there are some differences between both bends. The NB transect exhibits a gradual increase in MP abundances from the outside bend of the thalweg towards the inner bank, and a slight increase towards the outer bank (Fig. 3b). This pattern reflects the simple, classic bathymetric profile. The bathymetric profile of the SB transect is more complex, which is reflected in the MP abundances (Fig. 3c). Similar to the NB transect, MP abundances are very low on the outer bend slopes, and stay remarkably low even on the higher and flatter slopes towards the outer bank. Also similar to the NB transect, the concentrations rise from the outside bend of the thalweg towards the inner banks, but absolute values are much higher in this deeper pool zone (~6 m water depth compared to 3 m in the NB). The most striking sample is that closest to the inner bank, where very low MP abundances are recorded and may be explained by the location of this sample (SB_I1) at the foot of relatively steep "point bar" slopes, which is also the most sandy of all samples (see also 'Predicting MP abundance').

The low MP abundances in the surface sediments of the thalweg and the outer bends are likely a combined result of low deposition rates during periods of low flow regimes (spring, summer, autumn), and erosion at times of high flow regime (winter) (Supplementary Fig. 1a). As there are no sites where MPs are completely absent, we infer that during times of
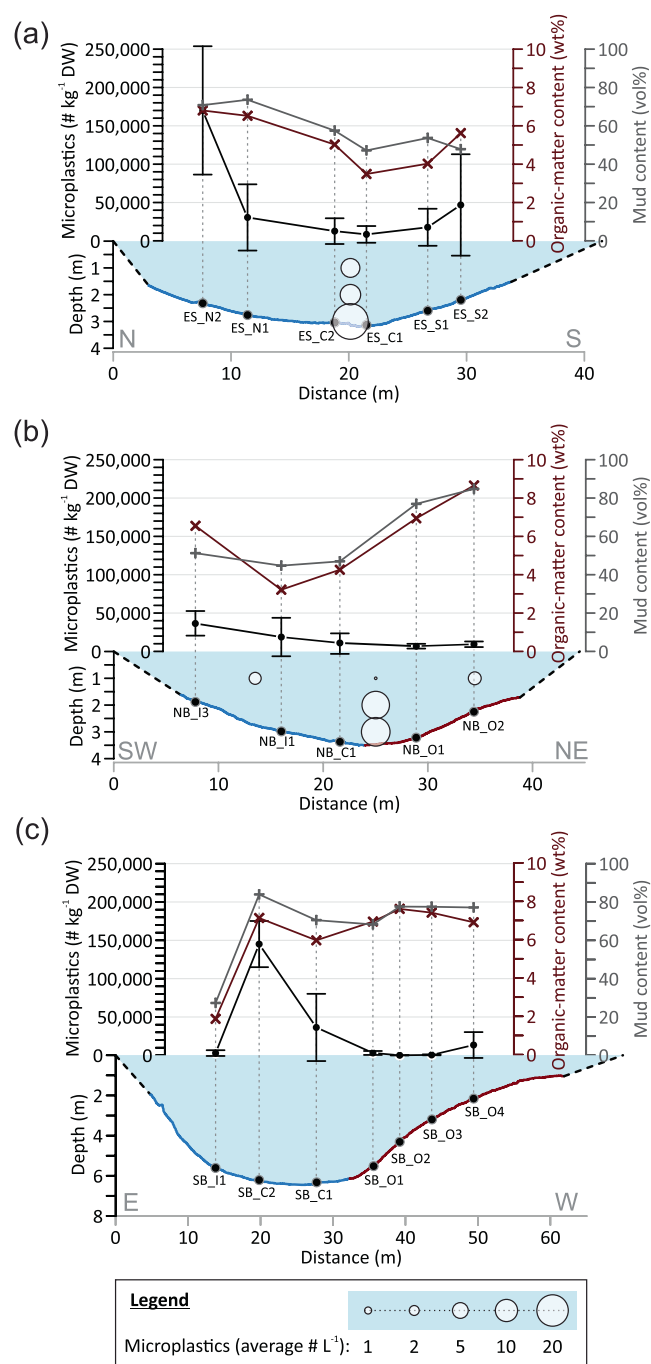

Fig. 3 | Transects across the River Lys with bathymetric profile, and for each sample location microplastic (MP) concentration, organic matter content and mud content. a East Straight, b North Bend, c South Bend. Upper part of each panel: MP concentrations (black dots) with 1 σ error bars, organic-matter content (brown x crosses) and mud content (gray + crosses) for each site. Lower part of each panel: average MP concentrations in the water column indicated by circles from which the area relates to the concentration (legend below). For both bends the orientation of the profile is such that the outer bend is to the right-hand side. Note that all axes have the same scale to allow comparison between profiles, apart from the distance across the profiles. Blue river floor corresponds to areas with positive correlation between MP concentration and organic matter and mud content (see also Fig. 4). Red river floor corresponds to areas without correlation between MP concentration and organic matter and mud content.

low discharge MPs are deposited in all environments, although variation in flow velocity across the river bed will cause initial development of the general depositional patterns described above. However, during periods of high discharge (e.g., winter, floods) the depositional pattern may be

Fig. 4 | Scatter plots between log-transformed MP concentrations (log(MP)) and sediment predictors. Log(MP) versus (**a**) mud fraction of the clastic sediment and (**b**) organic-matter content estimated through LOI$_{550}$. Red data points are outer bend samples, blue data points all other samples. Blue dots with a dark gray border are (sandier) inner bend or thalweg samples, those with a light gray border are (muddy) samples from the south bend pool zone or straight section slopes. Linear regressions (blue solid line) through blue datapoints with 2σ range in blue shading. Linear regressions of previously published datasets in dashed and dotted lines (legend in lower right corner). **c** Linear regressions through log-transformed corrected environmental MP concentrations (log(ceMP)) for full 1–5000 μm MP size range; for this study (blue solid line) and previously published datasets (gray dashed and dotted lines) in marine and lacustrine environments. Green and red shaded bands indicate zones where grain-size normalized contamination levels are one or more orders of magnitude lower and higher than in this study, respectively. **a** and **b** share the same vertical scale; **a** and **c** share the same horizontal scale.
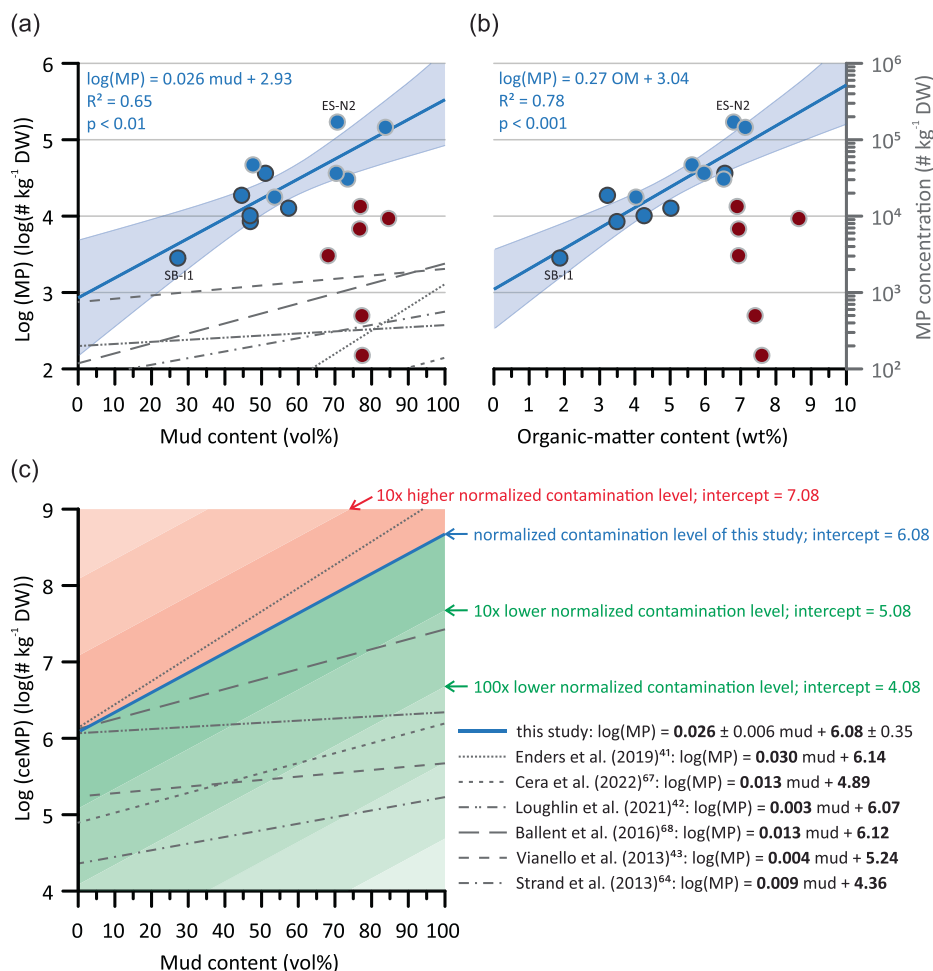
intensified, as it has been shown that flooding can effectively remobilize MPs from the sediment bed[29,60,61]. While during a flooding event deposition may (or not) continue near the (inner) banks, erosion will likely occur in the thalweg and especially on the outer bend slopes. Due to the large variability in MP particle densities, sizes and shapes, the degree to which a certain particle is susceptible to erosion is more complicated than for natural sediment[56]. If PS is moved primarily as bedload after initial deposition, while other polymer types (e.g., PP) are more easily brought back into suspension, this may not only explain the generally higher abundances of PS in the sediment, but it would also further increase the observed patterns. Indeed, if MPs are transported as bedload by secondary currents from the thalweg towards the (inner) banks, the difference in abundance between these environments will increase. However, as our study only sampled at a single moment in time, these remain hypotheses, and more monitoring studies are needed to reveal spatiotemporal variability of MP pollution. Our study does confirm previous observations of higher MP concentrations at point bars[34,40].

Perhaps counterintuitively, but MP particle sizes show very little trends across our transects (Supplementary Fig. 2). While the thalweg samples in the bends do show a relative increase in MPs >100 μm, this is not the case in the case for the straight section. This poor correlation between MP concentrations and MP size may be in part explained by the experimental observation by Waldschläger and Schüttrumpf[56], that larger particles are eroded faster than would be expected from their size, when compared to behavior of natural sediments. This observed stability in MP size distribution – across sites with variable sediment grain size – suggests that the correction factors proposed by Kooi et al.[49] for normalization to full 1–5000 μm MP size ranges[48], may remain relatively stable across samples with variable sediment grain sizes.

Predicting microplastic abundance

The observed patterns of MP distribution are in line with what can be expected from river hydrodynamics, and may thus also correlate with characteristics of natural sediments. Indeed, the slightly sandier inner bend (point bar) and thalweg samples contain less microplastics then the muddier slope samples in the straight section and the pool zone (Figs. 3 and 4a). Enders et al.[41] showed that mud percentage is a good predictor for the logarithm of #MPs kg^{-1} DW (further: log(MP)). However, when we test this for our full dataset we do not find any correlation ($R^2 = 0.0024$; $n = 18$), but when excluding the six outer bend samples we obtain a significant positive correlation ($R^2 = 0.65$; $p < 0.01$; $n = 12$; Fig. 4a). Similarly, we find no correlation ($R^2 = 0.0043$; $n = 18$) between log(MP) and organic-matter content for the full dataset, but a significant positive correlation ($R^2 = 0.78$; $p < 0.001$; $n = 12$) when excluding outer bend samples (Fig. 4b). This discrepancy is because the outer bend samples have relatively low MP concentrations (<15,000 MP kg^{-1} DW), even though the sediments are fine-grained (68–85% mud) and organic-rich (8.4–10.2% organic matter). This seems contradictory, but it is not when considering that the outer bend is a net erosional environment where floodplain sediments may be exposed (Fig. 5). Indeed, the outer bend sediments are fine-grained and rich in organic material, but have lower carbonate content, which we attribute to their floodplain origin (see Supplementary Fig. 3 and accompanying caption). As these exposed floodplain sediments are old (Holocene in age, and pre-Anthropocene), they do not contain MPs, which explains the low MP concentrations in these outer bend environments. In summary, mud and especially organic-matter content are good predictors for MP abundances in all net depositional environments, and their predicting power is even slightly increased when both are combined ($R^2 = 0.79$; $p < 0.001$; $n = 12$). On the contrary, in net erosional environments (outer bend) neither grain size nor
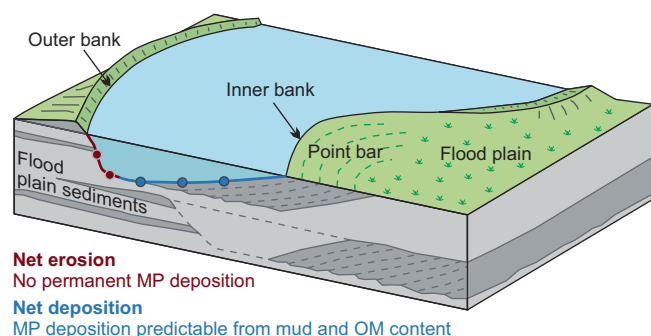
Fig. 5 | Schematic cross section across a meandering river. Inner bend and thalweg samples (blue) recover sandy point bar sediments where net deposition results in predictable MP concentrations. Outer bend samples (red) are in a net erosional environment where old flood-plain sediments are exposed, resulting in low to negligible MP concentrations. Redrawn and adjusted after Nichols[73].

organic matter can be used to predict MP abundances, as they represent the exposed old floodplain sediments. However, these abundances will be relatively low, and likely reduce to zero following periods of high discharge[29,60,61], and MPs are expected to become completely absent with depth below the river floor.

Strikingly, samples with seemingly extreme MP concentrations when compared to neighboring samples, such as the very low MP concentrations in the inner bend of the SB transect (SB-I1; Figs. 3c and 4) and the very high concentrations towards the northern banks of the ES transect (ES-N2; Figs. 3a and 4), do have low and high sediment predictors (mud and organic matter content), respectively. This further shows that mud and organic matter content are indeed good predictors, even for small scale spatial MP variability within the river bed. Indeed, it has been suggested by Enders et al.[41] that a larger sample area coverage will result in poorer correlations, because of the variable MP input that overprints MP-sediment relationships which likely exist on small spatial scales. This small-scale variability and predictability is especially relevant in rivers, where hydrodynamic conditions can change on very short distances[38].

Studies in marine, estuarine and lagoonal environments had already successfully correlated MP abundance with mud percentage[41,43,62], our study shows that also in meandering rivers with muddy sediments mud percentage is indeed the best predicting grain-size parameter (Supplementary Fig. 4). However, in the Lys River case, organic-matter content is performing even better. Indeed, Ballent et al.[63] showed that the low densities of most MP types cause them to behave similarly to low-density sediment particles such as coal, making organic matter potentially a good predictor for MP abundances.

Implications for quantifying MP contamination in meandering rivers and other aquatic environments

MP contamination studies of (river) sediments can have the goal to either compare contamination levels between different parts along the course of a river (e.g., near or distant to MP sources) or to estimate the total amount of MPs stored in the sediments. For the latter, apart from the small-scale variability discussed above, it is well-known that also the procedure for MP separation and identification is a crucial factor[18], with the lower MP size cut-off being a critical issue[48]. Therefore, in the following all MP concentrations are normalized to full 1–5000 μm ranges to achieve corrected environmental concentrations[48,49] (further *ceMP*). However, for environmental contamination levels to be really comparable, *ceMP* abundances need to be further normalized for sedimentary characteristics (e.g., grain size, organic-matter content), similar to what is done in other contaminant studies[64]. Contrary to many other MP sinks (e.g. marine environments), rivers are very heterogeneous sedimentary environments, and MP abundances can vary drastically over a distance of just a few meters (Fig. 3). Normalization of MP data from river sediments is thus even more crucial than in most other environments. Our data show that following log-transformation of the

ceMP abundances, mud and organic matter content can act as normalizer. When normalized against mud content, the mud-based normalization factor (i.e. the slope of the regression) in this study (0.026 ± 0.0060 ($1\ \sigma$); Fig. 4c) is only slightly flatter than that obtained by Enders et al.[41] in an estuarine environment (0.030). This similarity suggests that when only hydrodynamic conditions determine MP abundance, there may be a consistent (universal) mud-based normalization factor, especially because the slightly steeper slope of the Enders et al.[41] regression can be attributed to several coarse samples in their dataset that are near the mouth and outside of the estuary, which are thus more distant from potential MP sources, and have likely lower normalized *ceMP* contamination levels. In contrast, regressions from even larger-scale studies in marine[42,43,62] and lacustrine[65,66] environments show consistently flatter regression slopes (Fig. 4c). These studies typically present a dataset ranging from coastal to near/offshore samples, and as the coarser-grained coastal samples are located closer to the MP sources, they can thus be expected to have higher normalized contamination levels, compared to lower levels for the finer offshore sediments. The low regression slope in all these studies can thus be explained by their large scale, in which MP throughput is not equal across the study area. This sets our study apart, as in the studied River Lys section the throughput of MPs can be considered equal for all sites, it is an ideal case to determine such normalization factor based on the regression slope. Hence, our normalization factor of 0.026 (Fig. 4c) is currently the best available estimate of the relation between mud content and *ceMP* concentration in all aquatic environments. This normalization factor results in a factor 1.8 increase in number of MPs for every 10% increase in mud content, or a factor 390 between 0 and 100% mud. While more research (e.g., similar case studies) is needed to further finetune the universal normalization factor proposed here, until that is established, we propose to use our normalization factor to establish grain-size normalized contamination levels. The normalized contamination level (*NCL*) of a certain sample i can be represented by the intercept (mud content of zero) of the relationship passing through that datapoint while using the slope determined in this study (i.e. 0.026):

$$NCL_i = \log(ceMP_i) - 0.026\ mud_i$$

where *ceMP*$_i$ is the corrected environmental concentration of sample i, in MPs kg^{-1} DW, and *mud*$_i$ is the mud fraction of the clastic material in the same sediment sample, in percent. Applying this to the large-scale studies for which *ceMP* abundances and mud content are available[42,43,62,65,66] (Fig. 4c), shows that even though their fine-grained samples contain more MPs, the normalized contamination level of these samples is actually lower, resulting from their more distal location from MP source areas. On the other hand, in a (meandering) river such as the one studied here, MP abundances can be expected to differ up to an order of magnitude or even more depending on the depth, width or location across a river bed, even when normalized contamination levels are the same.

In the Lys River organic matter content is an even better normalizer then mud content (Fig. 4), but it seems to be much less widely applicable. Contrary to mud content, less studies demonstrate covariation with organic matter content, even though due to its low density it may react similar as MPs to certain hydrodynamic conditions. This may be because organic matter has many different sources (aquatic, terrestrial), sizes (colloids to leaves and twigs) and can reach the sediment in various ways (settling, benthic in-situ production, with hard parts of the organism)[64]. Furthermore, unlike mud percentage, organic-matter content strongly depends on local input or production, and the correlation with MP abundance may explain the results in our small study area very good but demonstrates the lower suitability for use at large spatial scales. Any correlation factor is therefore only locally applicable, but the locally strong correlation can be very useful to more accurately estimate total MP content in that part of the river. Indeed, apart from determining contamination levels, estimating total MP sequestration by rivers is needed to determine how much of the that is unaccounted for in the oceans, is stored in river beds. As MP concentrations vary drastically on small spatial scales and MP analysis is too costly and time

consuming to perform comprehensive sampling efforts, predictors such as mud and organic-matter content, which can be analyzed through dense sampling efforts, are essential. This study provides insights that can be used as a guide to perform such studies. The hardest part in estimating total MP sequestration by river sediments may be accounting for what is below the surface. Apart from many MP analyses, (relative) sedimentation rates will be necessary. However, as this study demonstrates that classical river sedimentology brings us a long way for the river-bed sediments, this will also be the case for buried sediments, and inner bends, where high sedimentation rates occur on the point bars, may be hotspots for total MP storage.

Methods
A more detailed methodology is provided as Supplementary Methods for most analyses.

Research design and study area
The studied section of the Lys River is located in the meandering part of the river between the cities of Deinze and Ghent. In Deinze, approx. 19 km upstream from the studied section, the river discharge shows annual peaks during winter reaching 30–40 $m^3 s^{-1}$, followed by decreasing discharge rates through spring, reaching minima of merely 1–2 $m^3 s^{-1}$ by the end of summer (Supplementary Fig. 1a). During the study period, daily average discharge was at intermediate levels, between 4 and 6 $m^3 s^{-1}$ (Supplementary Fig. 1b), but the effective discharge fluctuated between 0 and 15 $m^3 s^{-1}$ as a result of upstream weirs, which is typical for periods of low average discharge in this river[67]. Flow velocities reported here are average velocities for the entire river cross section at the sampling transects, calculated by dividing the discharge rate by the cross sectional areas. For flow velocities at the time of the fieldwork, we use discharges of 5 $m^3 s^{-1}$ (Supplementary Fig. 1b) and the cross-sectional area as calculated from our multibeam bathymetric dataset (Fig. 3). However, when discharge increases, the water level rises and cross-sectional area will increase as well. Hence, for calculating annual maximum flow velocities, for which we used a discharge of 35 $m^3 s^{-1}$, we estimated the cross-sectional area by increasing the water level with 0.9 m (based on the increase in water level at the hydrograph station in Deinze when discharge rises from 5 to 35 $m^3 s^{-1}$) compared to that at the time of our multibeam survey, for the same river width.

In the 19 km between Deinze – which acts as a potentially important MP source – and the study area, the river has either natural banks (majority) or is bordered by private gardens. The main potential plastic sources within this section of the river are the E40 highway crossing 1.5 km upstream of the studied section and the abundant pleasure craft, especially during summer. The three studied transects are all located within a 750 m stretch along the river course, which is primarily bordered by natural banks, and we therefore assume that the throughput of MPs is equal for each sampled transect. The specific study area was further selected based on the presence of two sharp bends in this short section, one of which relatively shallow and one with a deep pool zone, allowing for comparison of these different sedimentary environments.

Geophysical river floor mapping
A high-resolution bathymetry of the studied section of the Lys River was acquired by a portable Norbit Wide Band Multibeam Sonar (Flanders Marine Institute; VLIZ). The system was operated at a frequency of 700 kHz, a ping rate of 60 Hz and a swath coverage of 150°. Beams were steered towards the banks to obtain optimal coverage in the shallow areas.

To determine the optimal (i.e. undisturbed) location for the transects, a dual frequency Klein System 3000 side-scan sonar, operating at high (500 kHz) and low (100 kHz) resolution, was used to obtain an image of the river floor. The images were used to avoid (human-induced) disturbances or objects that may affect sampling efforts and results.

Water filtering
MP samples from the water column were obtained by filtering water using a device constructed for this research with 150 and 51 μm stainless steel filters (see Supplementary Methods). Over a period of 6 weeks in February-March 2022 (Supplementary Fig. 1b), 5 replicate samples were obtained at 8 "stations" in the studied section of the Lys River: at 1, 2 and 3 m water depth in center of the East Straight and North Bend transects and at 1 m depth in the inner and outer bend of the North Bend. In total 159–381 l of river water was filtered at each station.

Sediment sampling and sedimentological data
For this study, we targeted the upper 3 cm as representative for the sediment bed. Based on the combination of a lower MP cut-off at 25 μm and a sediment grain size of mud to fine sand (median grain size between 24 and 74 μm (outlier SB_I1 of 153 μm)), infiltration of MP particles is expected to be less than 3 cm[68]. Moreover, 3 cm corresponds to the maximum height of bedforms (current ripples) that is expected in mud to fine sand[69], and recently deposited MPs would thus not be buried more than 3 cm due to bedform migration. This sample depth thus ensures full retrieval of all sediment bed MPs, whilst avoiding signal contamination from buried MPs. Sediment samples were retrieved with a UWITEC gravity corer (USC 09000) equipped with a PVC tube. To limit the microplastic contamination, the upper 3 cm of the sediments were immediately extruded[70] and stored in closed aluminum trays after removing the outer border of sediments that were touching the liner. In total, 18 surface samples were gathered (North Bend: 5, South Bend: 7, East Straight: 6; Fig. 1). To calculate the water content all samples were well homogenized and oven-dried at 60°. Organic and carbonate content of the sediments was estimated by loss-on-ignition: LOI_{550} and LOI_{950}[71], respectively. LOI_{550} values were corrected to estimate organic matter content, by subtracting the LOI_{550} values of the field blank (1.53%), which had been pretreated with LOI_{550}, and its values can thus be attributed to water in clays. Grain-size analysis of the clastic fraction was done using a laser diffraction analyzer Malvern Master 3000 following pretreatment to remove organic matter (H_2O_2), biogenic silica (NaOH) and carbonates (HCl).

Microplastic data
Following removal of the filtered fraction from the stainless-steel filters using 5% SDS and ultrasonication, MPs were isolated following the protocol of Vercauteren et al.[35] including KOH digestion and density separation steps (see Supplementary Methods). From each surface sediment sample, two replicates were analyzed for MP content, followed by extraction of MPs following the protocol of Vercauteren et al.[35] (see Supplementary Methods for details). The identification and concentration of MPs in the filtered water and sediment samples were performed by microscopic Fourier Transform InfraRed (micro-FTIR) spectroscopy (Nicolet iN10 FT-IR Microscope; Thermo Fisher Scientific, Madison, Wi, USA) of individual MP particles following automated optical mapping of the MPs on a polytetrafluoroethylene (PTFE) filter (see Supplementary Methods).

Quality assurance and control
To prevent contamination with external MPs all work was carried out avoiding plastic equipment as much as possible and under high laboratory standards. For filtered water samples, a correction was performed using the limit of detection (LOD) and limit of quantitation (LOQ) method based on blank samples[36,72]. For sediment sampling, an MP-free field blank was prepared by heating a mix of two previously collected van Veen grab samples to 550 °C for 4 hours to obtain MP-free sediment[18]. This sediment introduced in the PVC liners during the sampling campaign and then integrated in the full sampling and analytical procedure. Further methodological details can be found in the Supplementary Methods of this paper.

Correction to environmental MP concentrations
To compare MP concentrations of this study to those of other studies we applied the normalization procedure proposed by Koelmans et al.[48] to calculate corrected environmental MP concentrations for MP size ranges of 1–5000 μm. To calculate the correction factors, we used power-law exponents α from Kooi et al.[49] of 3.25 for freshwater sediments (rivers (this study)

and lakes[65,66]) and 2.57 for marine sediments[42,43,62], both based on the power law obtained from particle length. For the estuarine sediments studied by Enders et al.[41], we used the freshwater exponent, as their presented salinities correspond to fresh or low-salinity brackish water. Because the exponents for freshwater and marine sediments differ quite strongly, and as for example exponents based on particle width are especially lower for freshwater sediments, the obtained environmental concentrations do need to be treated with care, specifically when comparing freshwater and marine environments.

Data availability

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Author contributions

Maarten Van Daele, Inka Meyer, Maaike Vercauteren and Jana Asselman designed the study, Maarten Van Daele, Ben Van Bastelaere, Jens De Clercq an Inka Meyer executed field work. Ben Van Bastelaere and Jens De Clercq performed lab analyses, supervised by Inka Meyer and Maaike Vercauteren. Maarten Van Daele wrote the manuscript and produced the figures, with input from all co-authors. Maarten Van Daele and Jana Asselman secured funding.

Competing interests

The authors declare no competing interests.

Additional information

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