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Environmental Pollution 395 (2026) 127810

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Quantification of microplastics in ship-generated greywater and their contribution to Baltic marine pollution☆

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ARTICLE INFO

Keywords:
Micropollutant
Loads
Vessels
Wastewater
Treatment

ABSTRACT

The Baltic Sea remains one of the most contaminated marine seas globally, receiving diverse pollutant inputs from land-based and maritime sources. This study quantifies the concentrations and loads of microplastic (MP) in ship-generated greywater (GW) and evaluates their potential contribution to Baltic marine MP pollution. Eight GW streams from five vessels were sampled, and fifteen MP polymer types were identified and characterized. MP concentrations ranged from ≈38,000 MP/m^3 in mixed accommodation-laundry-galley (ALG) stream to ≈ 602,000 MP/m^3 in laundry GW. Polyethylene terephthalate (PET) was the dominant polymer (58%), detected in all GW samples, while polypropylene (PP, 16%) appeared in only three streams from two vessels. Estimated annual MP loads from the studied Roll on - Roll off - Passenger (RoPax) vessels ranged between ≈ 1.24 and 7.59 billion particles, which are typically delivered to municipal wastewater treatment plants (MWTPs) via Port Reception Facilities (PRFs). Considering the total Baltic fleet's greywater discharge of ≈5.4 million m^3/yr in 2022, up to 1.1 trillion MP/yr could have been released directly to the sea, with ≈93% originating from passenger ships. If this volume generated in 2022 were instead delivered to PRFs and treated at MWTPs, ≈6 million - 651 billion MP/yr could still enter the Baltic environment, depending on the treatment efficiencies and technology configurations employed at the MWTPs. These results demonstrate that ship-generated GW is a significant yet understudied source of microplastic (MP) to the Baltic Sea. Moreover, while advanced systems with tertiary treatment technologies on board and ashore can remove ≈95 – 99.9% of MP, residual emissions remain substantial given the large wastewater volumes generated. Effective mitigation strategies should therefore focus on source identification and prevention within shipboard systems, particularly in laundry, galley and accommodation operations, to minimize MP inputs into GW streams and, ultimately, the marine environment.

1. Introduction

Shipping is a key maritime activity with considerable economic importance, yet it contributes to the release of numerous pollutants into the marine environment, including plastics (GESAMP, 2021). Plastics could be classified by size into nanoplastic (NP): 1 to <1000 nm; MP: 1 to <1000 µm; mesoplastic: 1 to <10 mm; and macroplastic: 1 cm and larger (Hartmann et al., 2019). MP may be intentionally manufactured in microscopic sizes (primary MP) such as pellets (Kumar et al., 2022; Boucher and Friot, 2017) and microbeads or may originate from the physical, chemical, and/or biological disintegration of large plastic materials (secondary MP) such as microfibers from car tires, textiles (Song et al., 2024; Mattsson et al., 2023) and paint fragments (Gaylarde et al., 2021).

The Baltic Sea is among the marine ecosystems polluted by MP subject to multiple origins (Schernewski et al., 2021; Esiukova et al., 2020; Murawski et al., 2022, Kuddithamby, 2023). Well-recognized pathways include urban sources such as wastewater treatment plant

☆ This paper has been recommended for acceptance by Maria Cristina Fossi

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https://doi.org/10.1016/j.envpol.2026.127810
Received 18 November 2025; Received in revised form 4 February 2026; Accepted 10 February 2026
Available online 11 February 2026

effluents, raw sewage (Magnusson and Noren, 2014; Magnusson et al., 2016; Schernewski et al., 2021; Talvitie et al., 2017a; Talvitie et al., 2017b; Xu et al., 2019), sewage sludge leachate (Nguyen et al., 2022; Ruggero et al., 2020), river export (van Wijnen et al., 2019; Schernewski et al., 2021) and beach sediments (Lots et al., 2017), storm water run-off (Cho et al., 2023; Piñon-Colin et al., 2019), sewer overflows (Duis and Coors, 2016; Baresel and Olshammar, 2019), agriculture (van Schothorst et al., 2021; Weithmann et al., 2018) and atmospheric deposition (Klein and Fischer, 2019). Schernewski et al. (2021) estimated counts of about 67 trillion MP particles/year flowing into the Baltic Sea from wastewater treatment plants (25%), unconnected wastewater (13%) and sewer overflows (62%). Maritime activities such as extensive fishing, aquaculture, port and shipping operations are significant contributors to MP pollution in the region (Chen and Liu, 2013; Dereszewska et al., 2023). Commercial shipping introduces MP into the marine environment through various pathways (Čulin and Bielić, 2016), including improper waste management onboard (GESAMP, 2021), ship-breaking activity (Reddy et al., 2006), container losses during freight transport of plastic pellets (IMO, 2024), oil and gas platform operations, and shipping accidents (Saliba et al., 2022). Additional sources include hull scrapping of antifouling paints (IMO, 2019; Tamburri et al., 2022), ballast water exchange (Naik et al., 2019; Naik et al., 2021; Matiddi et al., 2017; Zendehboudi et al., 2024), ships cooling system water uptake and exchange (Kim et al., 2023), onboard sewage treatment plants effluents (Kalnina et al., 2022) and biosludge (Folbert et al., 2022), and greywater discharges (Jang et al., 2024; Kalnina et al., 2022; Mikkola, 2020).

Despite being a well-recognized but under-researched pollution pathway, ship-generated GW is a significant contributor to MP loads in the Baltic Sea (Kalnina et al., 2022; Jang et al., 2024; Mikkola, 2020; Peng et al., 2021; Nuka Research and Planning Group, 2019). GW may contain MP from dishwashers, where plastic pellets are used to enhance abrasion (Mujingni et al., 2024), and from personal care and cosmetic products (PCCP) such as toothpaste, body lotions, soaps, and scrubs, which contain microbeads (Carr et al., 2016; Cheung and Fok, 2016; Duis and Coors, 2016). Additionally, washing of textiles releases microfibers, and indoor ship environments contribute airborne MP that ultimately enter GW (Dius and Coors, 2016; Lin et al., 2023; Mikkola, 2020). Previous studies have recorded MP abundances of approximately 135,000 MP/m^3 in GW on a research vessel (Jang et al., 2024), 72,000 MP/m^3 on transport ships (Kalnina et al., 2022) and 100,000 – 700,000 MP/m^3 on cruise ships (Mikkola, 2020). The types and concentrations of MP in different wastewater streams vary widely. This variability is driven by several factors, including ship type and onboard activities, the number of passengers and crew, laundry practices, sample volume, the origin of the analyzed GW streams, filter mesh pore size, and the analytical methods used for MP detection (Jang et al., 2024; Mikkola, 2020; Kalnina et al., 2022).

Ship-generated GW may be discharged directly into the sea or treated together with sewage using onboard Advanced Wastewater Treatment Plant (AWTP) before discharged or delivered to PRFs for further treatment at MWTP (Mujingni et al., 2024). The extent of MP removal at the MWTPs depends on the treatment technology (Baresel and Olshammar, 2019; Talvitie et al., 2017a). Onboard ships, a biological sewage treatment system has achieved limited MP removal, with efficiencies as low as 29% for combined GW and sewage treatment (Kalnina et al., 2022). The efficiency of MWTPs in removing MP from ship-generated GW remains uncertain, as GW is processed alongside wastewater from terrestrial sources. Nevertheless, MWTPs generally achieve high MP removal rates, with efficiencies of 85% in primary treatment, 90% in secondary treatment, 95% in tertiary treatment in sand filtration, and 98% in microfiltration (Baresel and Olshammar, 2019; Schernewski et al., 2021). Studies have reported overall MP removal rates between 90% and 99.9% (Magnusson and Noren, 2014; Rasmussen et al., 2021; Talvitie et al., 2017a; Talvitie et al., 2017b; Xu et al., 2019). Despite these high efficiencies, MWTPs remain a key pathway for MP entering the marine environment due to the sheer volume of effluent discharged. For example, Magnusson and Noren (2014) found that even with >99% MP retention in sewage sludge, approximately 1770 MP particles per hour were still released into the sea from a MWTP with 14000 population equivalents (pe). Consequently, ship-generated GW discharged to MWTPs potentially contributes to MP pollution in marine ecosystems.

MP pollution in the marine environment is a growing concern that has prompted attention from researchers and policymakers as it poses severe risks to aquatic organisms and human health (GESAMP et al., 2016; Amobonye et al., 2021; Yuan et al., 2022; Naik et al., 2019). However, there are currently no legally binding international regulations specifically addressing ship-source MP pollution. The International Maritime Organization (IMO) regulates shipborne garbage under MARPOL Annex V, but the Convention lacks explicit provisions for MP. Recent IMO discussions have focused on managing plastic pellet transport at sea (IMO, 2024). At the regional level, policy developlent addressing plastic pollution began with the European Strategy for Plastics in a Circular Economy (EC, 2018), adopted under the Circular Economy framework to promote sustainable plastic use and reduce leakage to the environment. This strategy subsequently informed later policy initiatives under the European Green Deal (EC, 2019b) which provides overarching framework for achieving climate neutrality and zero pollution ambitions within the EU. Within this broader policy context, the Single-Use Plastics Directive (Directive (EU) 2019/904) was adopted to reduce the environmental impacts of specific plastic products, particularly marine litter (EC, 2019a). Further implementation of the Green Deal's environmental objectives include the EU Zero Pollution Action Plan (ZPAP) (EC, 2021) which include a banon intentionally added MP in cosmetics, PCCPs, and detergents. ZPAP aims to reduce intentional and unintentional MP emissions by 30% by 2030. Complementary regulatory measures include restrictions under the REACH regulation (EC, 2023a) and proposals to prevent plastic pellet losses across the value chain (EC, 2023b).

Although research on land-based MP sources has advanced, ship-based MP sources remain underexplored. As scientific awareness of MP pollution increases, the lack of extensive and detailed data impairs policymakers' ability to develop targeted mitigation strategies. A comprehensive strategy is essential, encompassing the mapping, quantification and classification of all MP sources and dissemination pathways on board ships. This includes both direct ship-to-sea and indirect ship-to-shore-to-sea routes, to inform effective investigation and remediation measures. This study, in addition to quantifying and characterizing ship-generated greywater-borne MP, assesses the contribution of two greywater discharge approaches to Baltic Sea MP loads.

2. Materials and methods

The ships and ports included in this paper are those examined in Mujingni et al. (2024). It consists of one special purpose ship (S1) and four RoPax ships (S2 – S4) operating in the Baltic Sea. RoPax ships were used as proxies for cruise ships due to their comparable passenger and wastewater generation capacities. All the RoPax vessels outsource most of their laundry services; primarily bed linens and table clothes, which are laundered ashore. The profiles of the investigated vessels, their routes, ports of call and a summary of the GW management system and practice as detailed in the supplementary material and in Mujingni et al. (2024).

2.1. Greywater sampling

For sampling, 5L glass bottles with rubber corks were pre-cleaned with 90% alcohol and Milli-Q water was used to sample greywater from five ships at the ports of Gothenburg, Trelleborg and Stockholm. The bottles were kept refrigerated and dark after sampling. To mitigate potential contamination from the rubber corks, the sample bottles were covered with aluminum foil before sealing. Moreover, a 1 L procedural blank containing Milli-Q water was left open in the sampling environment to capture potential airborne MP contamination during sampling (Jang et al., 2024). On S1 sampling was performed from a sampling point towards the bottom of the GW holding tank. On S2 – S5, sampling points were located along the GW pipes, and sampling was performed during GW delivery to PRF. The 5L bulk samples were agitated to ensure homogeneity, and from each 5L sample, 500 mL subsamples were extracted and sent to the laboratory for analysis (Mujingni et al., 2024).

Due to the challenges associated with accessing the designated sampling points on S4 and S5, additional piping was installed to facilitate the sampling process. On S4, a metal pipe was utilized whereas a polyvinylchloride (PVC) tube was employed on S5. A total of eight GW samples were collected from five vessels, representing different wastewater streams. These included three mixed accommodation-laundry-galley (ALG) GW streams from S1, S4 and S5 ($S1_{ALG}$, $S4_{ALG}$ and $S5_{ALG}$, respectively), as well as three sub flows from S2, comprising laundry ($S2_L$), accommodation ($S2_A$) and galley ($S2_G$) GW. Additionally, samples from S3 included a mixed accommodation-laundry GW stream ($S3_{AL}$), and a galley GW stream ($S3_G$).

For clarity in data reporting, the collected samples were classified into five categories: accommodation (C_A, n = 1), laundry (C_L, n = 1), galley (C_G, n = 2), accommodation-laundry (C_{AL}, n = 1) and accommodation-laundry-galley (C_{ALG}, n = 3). The number of samples obtained was based on the types of GW streams collected and discharged from the participating ships. Fig. 1 presents a schematic diagram showing the studied ships, the origins of the eight GW streams and the five GW categories.

2.2. Laboratory analysis of MPs in greywater from ships

The analysis of GW samples was conducted by ALS Global, a commercial laboratory headquartered in Brisbane, Australia, using an internal laboratory method. Prior to analysis, the samples were stored at 4 (±2) °C to preserve sample integrity. A blank sample was included to monitor potential background contamination. The analysis was done according to the following procedure: To remove biological material, the GW samples were treated with hydrogen peroxide (H_2O_2), (Merck KGaA) or a combination of hydrogen peroxide and Ferrous Sulphate ($FeSO_4$) as a catalyst (Fenton reagent) (Sigma Aldrich) depending on the load of biological material (Tarte et al., 2025). Fenton reagent ensures efficient removal of organic matter when clogged together with MP without degrading the plastic. The process was performed under constant stirring to ensure homogenization, with the temperature maintained below 37 °C. If needed, subsequent treatment with urea-based reagents (urea/thiourea/NaOH (Sigma Aldrich)) was applied to eliminate any remaining biological material residues following the Fenton reaction. To prevent MP contamination, all chemicals were filtered before use using a 20 μm steel filter.

For MP analysis, 250 mL GW of the treated sample were filtered through a 40 μm stainless steel mesh, and particles with grain size >40 μm ≤ 100 μm were analyzed. Visual observation revealed that the particles in the GW samples were evenly distributed and did not exhibit clumping (Supplementary Material, Fig. 1). Fourier-transform infrared spectroscopy (FTIR) was employed for spectral analysis of the particles (Spotlight 400 FTIR Imaging System, PerkinElmer, Waltham, Massachusetts, United States) (Rathikannu et al., 2025; Villegas-Camacho et al., 2025). MP were classified into black MP (BMP) and other MPs (OMP), which includes all other MP than BMP. Particles were scanned in either reflectance "image mode" or "point mode" (excluding black particles). Black particles were analyzed separately using FTIR with an Attenuated Total Reflectance (ATR) crystal.

All particles were analyzed within a 100 mm^2 area for the blank sample (corresponding to 125 mL sample size for the blank) and within a 64 mm^2 area for the GW samples (corresponding to 80 mL sample size). Spectral scans were conducted within the 4000 – 600 cm^{-1} range, using 8 accumulated scans at a resolution of 4 cm^{-1}. Spectral results were interpreted using a match correlation factor (>75%), compared against a reference library of >40000 polymer spectra in the PerkinElmer database (Supplementary Material, Fig. 1). The analyzed polymers included polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS), polyethylene (PE), polyamide (PA), polyurethane (PUR), polyacrylonitrile (PAN), ethylene vinyl acetate (EVA), polyoxymethylene (POM), ethylene propylene diene (EPDM), styrene butadiene rubber (SBR), PVC, polytetrafluoroethylene (PTFE), polycarbonate (PC) and polymethyl acrylate (PMMA). Among these, PS exhibited a recovery rate lower than 80% due to its tendency to detach from the filter because of electrostatic loading. The limit of detection (LOD) was calculated per 1000 mL sample using a blank-based statistical approach to account for background contamination and analytical variability. Procedural blanks were processed in parallel with the GW samples following the same digestion, filtration, and FTIR analysis steps. The LOD was defined as the mean number of MP particles detected in the blank samples plus three times the standard deviation of the blank measurements $\left(LOD = \bar{B} + 3\sigma_B\right)$, consistent with established analytical and MP quantification protocols (Masura et al., 2015; Shrivastava and Gupta, 2011). The resulting particle count threshold was then normalized to a reference

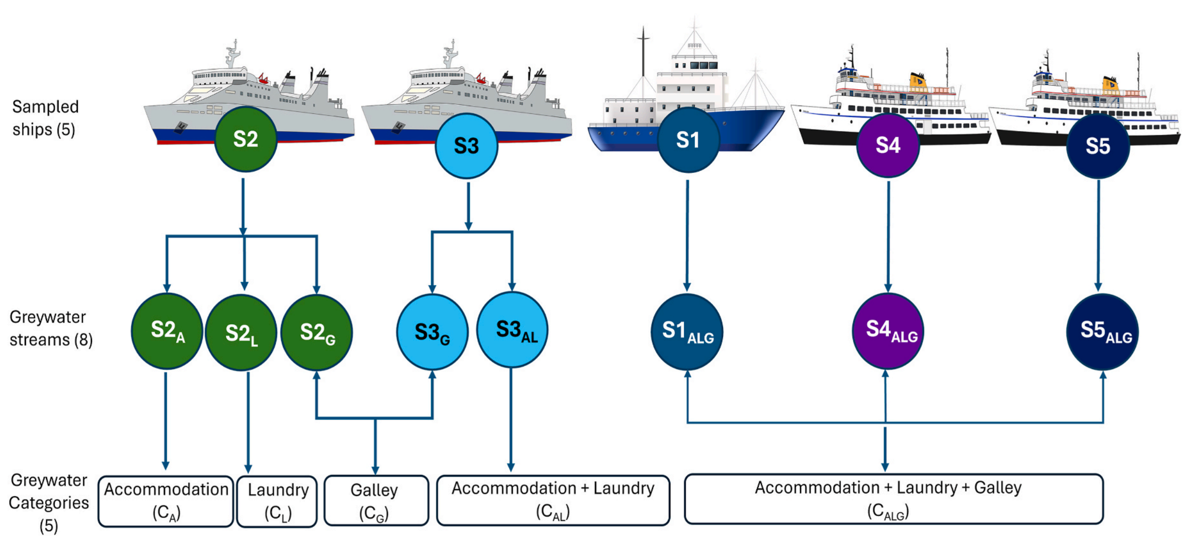

Fig. 1. Flow diagram illustrating the origins of the main and sub flows from the sampled ships.

volume of 1000 mL to allow direct comparison of MP concentrations across samples with different filtered volumes. Based on this approach, the LOD was 13 MP/L for the eight GW samples and 8 MP/L for the blank sample, with the differences attributed to variation in the effective filtered sample volumes.

2.3. Quantification of MPs in ship-generated greywater

Annual micropollutant loads discharged from the ships were defined according to Ytreberg et al. (2020), by multiplying the flow rate, $Q_{GW(yr)}$ (m^3/yr) of GW by the concentration of MP, C_{MP} (MP/m^3) in the GW stream and the discharge frequency (***f***) (equation (1)).

$$\mathbf{LOAD_{MP}} = \mathbf{Q_{GW(d)}}\left(\boldsymbol{m^3 / d}\right) \times \mathbf{C_{MP}}\ \left(\mathbf{MPs} / \boldsymbol{m^3}\right)\ \mathbf{x}\ \boldsymbol{f}\ (\boldsymbol{d / yr}) \tag{1}$$

For ships having multiple GW streams (S2 and S3), the total MP loads was obtained by summing the MP loads from individual streams (Equation (2)):

$$\mathbf{LOAD_{MP}} = (\mathbf{LOAD_A} + \mathbf{LOAD_L} + \mathbf{LOAD_G}) \tag{2}$$

Equations (1) and (2) were used to estimate the daily and annual loads of shipborne greywater MP discharged into the Baltic Sea and to PRFs. To calculate the annual per capita MP loads from the studied ships Equation (1) was divided by the annual number of persons on board (N*p*). When annual volume of GW is given equation (4) is used.

$$\mathbf{Per\ capita\ MP\ Load}\ (\mathbf{MPs} / \boldsymbol{p} / \boldsymbol{yr}) = \frac{\mathbf{Q_{GW(d)}}(\boldsymbol{m^3/d})\ \boldsymbol{X}\ \mathbf{C}_{\boldsymbol{MP}}\ (\mathbf{MP}/\boldsymbol{m^3})\ \boldsymbol{X f}\ (\boldsymbol{d/yr})}{\mathbf{N}_{\boldsymbol{p}}\ (\boldsymbol{p/yr})} \tag{3}$$

$$\mathbf{Per\ capita\ MP\ Load}\ (\mathbf{MPs} / \boldsymbol{p} / \boldsymbol{yr}) = \frac{\mathbf{Q_{GW(yr)}}(\boldsymbol{m^3/yr})\ \boldsymbol{X}\ \mathbf{C}_{\boldsymbol{MP}}\ (\mathbf{MP}/\boldsymbol{m^3})}{\mathbf{N}_{\boldsymbol{p}}\ (\boldsymbol{p/yr})} \tag{4}$$

Polymer distribution across GW streams was analyzed statistically using the Chi-Square Test of Independence (Fienberg, 1979; McHugh, 2013) to assess whether polymer composition across streams varied significantly. The null hypothesis (H_o) stated that polymer composition was uniform across all GW streams, while the alternative hypothesis (H_1) assumed that significant differences exist. Statistical significance was set at $\alpha = 0.05$.

Expected frequencies were calculated as:

$$E_{ij} = \frac{(R_i\ x\ C_j)}{N} \tag{5}$$

Where R*i* is the total count for polymer *i*, C*j* is the total count for GW stream j, and N is the grand total of all observations. The chi-square statistics were computed as:

$$X_{ij}^2 = \frac{(O_{ij} - E_{ij})^2}{E_{ij}} \tag{6}$$

Where O_{ij} is the observed frequency of polymer group *i* in stream *j*. To identify stream specific contributions, column-wise chi-square contributions were calculated by summing $\frac{(O_{ij}-E_{ij})^2}{E_{ij}}$ across polymer groups for each stream (Supplementary Material).

Critical values were calculated from the chi-square distribution for 3 degrees of freedom to provide a threshold for significance. Comparing the observed X^2 statistic with the critical value allowed determination of whether deviations from the null hypothesis were statistically significant (H_o rejected if $X^2_{obs} > X^2_{crit}$). Exact *p*-values were also computed to quantify the probability of observing the data under H_o, with $p \leq \alpha$ indicating rejection of the null hypothesis and confirming significant differences in polymer composition between GW streams. This analytical framework follows established categorical data analysis methodology (McHugh, 2013), providing a robust statistical basis for evaluating variability across GW streams.

3. Results and discussion

3.1. Concentration of MPs in ship-generated greywater streams

There was a wide variation in the MP concentrations in the eight GW streams which ranged from ≈38,000 MP/m^3 in the mixed ALG stream $S4_{ALG}$ to ≈602,000 MP/m^3 in the laundry GW stream, $S2_L$, with an average of 206,000 MP/m^3 across the eight streams. (Fig. 2A). Among the three mixed ALG streams, $S5_{ALG}$ had the highest MP concentration of 253,000 MP/m^3. Despite being sister ships with similar sizes and utilization capacities, the MP concentration in $S5_{ALG}$ was about 7 times higher than in $S4_{ALG}$. The reason for this difference was not clear; however, it might be due to dissimilar activities and greywater management strategies on board. In terms of ships, the highest concentration of MP was recorded on S2 (≈351,000 MP/m^3), followed by S5 (253,000 MP/m^3) and the lowest in S4 (38,000 MP/m^3). Approximately, the same concentration of MP was recorded on both S1 and S3 (≈101,000 MP/m^3) (Fig. 2B). The MP concentrations observed in this study fall within the range reported in comparable shipborne GW studies. Jang et al. (2024) reported an average of ≈136,000 MP/m^3 from a research vessel, while Kalnina et al. (2022) reported ≈ 72,000 MP/m^3 from five transport ships. Mikkola (2020) recorded relatively higher average MP concentration of ≈2 million – 50 million MP/m^3 from five cruise ships. All the studies in which individual GW streams (accommodation, laundry and galley) were separately analyzed reported the highest concentrations laundry GW. For instance, Jang et al. (2024) reported 178,000 MP/m^3 while Mikkola (2020) reported (C_L = 7.5 million – 250 million MP/m^3), in laundry GW constituting more than 99% of MP analyzed the study.

The relatively elevated concentrations of MP observed in cruise ship GW may be attributed to the extensive use of industrial-scale laundry systems onboard. These systems process large volumes of textiles, including bed linens, towels, tablecloths, and clothing for both passengers and crew, due to the typical long durations at sea (averaging around 7 days). Additionally, factors such as the type of washing machine, textile composition, fabric wear and aging can significantly influence microfiber release during laundering (Mikkola, 2020). However, Mikkola's study in which proxy data on microfiber emissions from land-based textile laundering was applied to cruise ship context may have introduced uncertainties, potentially contributing to the significantly higher concentrations reported. This highlights the need for direct empirical measurements of MP concentrations in GW from cruise ships to validate and refine existing estimates. The differences among these studies were also likely due to variances in the filter mesh sizes utilized, vessel type, number of persons and activities carried out on board. MP from ship-generated GW shows higher concentrations than MP from other sources analyzed from ships. For instance, an average of ≈24,100 particles/m^3 was recorded from an analysis of water from five main pipes of a cooling system on the training ship at the Korean Maritime and Ocean University (Kim et al., 2023), and ≈12,530 MP/m^3 was obtained in ballast water (Zendehboudi et al., 2024). Fig. 3B shows concentrations of MP from the current study compared with other ship-based and land-based sources.

MP pollution is recognized as a globally widespread issue, with elevated concentrations typically reported in seawater near areas of intense anthropogenic activity. For instance, relatively low MP concentrations (36,000 MP/m^3) were reported in seawater along the Swedish coastline (Magnusson and Norén, 2011). In contrast, Norén (2007) measured more than three times higher concentrations (102,000 MP/m^3) in a harbor located adjacent to a polyethene production plant. These values are comparable to the concentrations observed on S1 and S3 in the present study. Zendehboudi et al. (2024) analyzed a lower concentration (≈11,800 MP/m^3) in sea water from the Persian Gulf. Schernewski and Radtke (2022) also reported that average annual MP

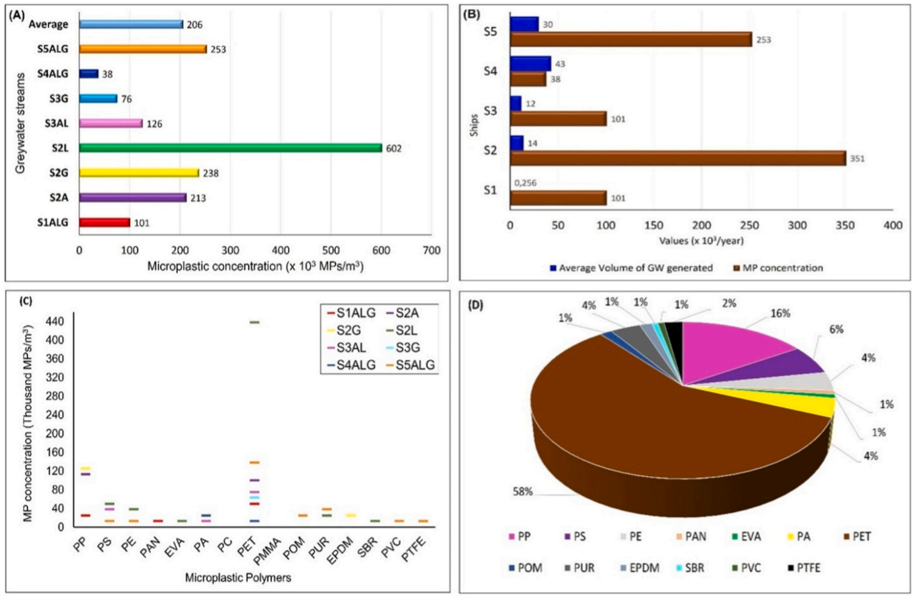

Fig. 2. (A) MP concentrations in eight ship-generated GW streams (B) MP concentrations and average annual GW generated from five ships (C) Polymer types and occurrence in eight ship-generated GW streams. (D) Polymer types and percentage occurrence in ship-generated GW streams.

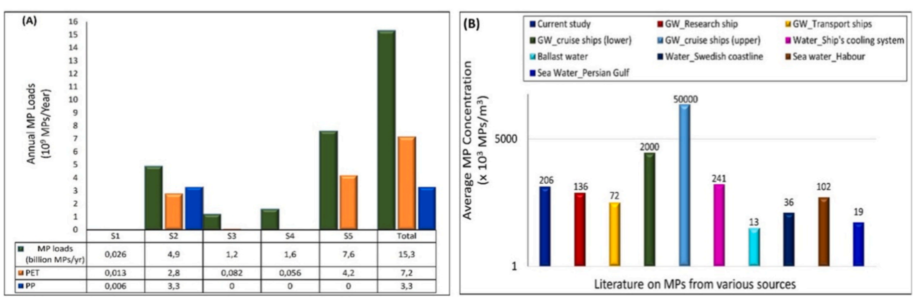

Fig. 3. (A) MP loads in GW from five ships operating in the Baltic. (B) A comparison of MP concentration in GW from the current study with other MP sources.

concentrations in the Central Baltic Sea ranged between 1 and 4 particles/m^3 at the sea surface. They further observed concentrations of ≈1 particle/m^3 within the upper 2 m of the water column. The same study indicated that MP concentrations in the upper surface layer could exceed 10 particles/m^3 in areas near the Gulf of Riga and the Gulf of Gdansk, and that MP can be transported by waves and accumulate along shorelines. Consequently, MP concentrations are often higher along coastlines located close to emission sources than in offshore areas. Densities of about 109 particles per meter of coastline were reported within a few kilometers of the source (Schernewski and Radtke, 2022). On average, the annual MP accumulation along the Baltic Sea coastlines is estimated at approximately 106 particles/m of coastline (Schernewski and Radtke, 2022).

Among the 15 targeted MP polymers, 13 were detected and included in the analysis. At least 1 polymer was detected in all eight GW streams and 2 polymers in the blank sample. PET was the only polymer that occurred in all eight GW streams (Fig. 2C). BMP constituted about 5% of the total number of MP analyzed. All BMP in the blank sample were below the limit of detection (LOD), while PP and PE were each detected at concentrations of ≈13,000 MP/m^3 in the blank sample. Despite rigorous quality control measures, the detection of these frequently occurring polymers in blank samples suggests the possibility of contamination originating from the sampling environment or procedural handling. This observation is consistent with findings by Jang et al. (2024), who reported the presence of PP, PET, PA and alkyd in three blank samples at levels exceeding the LOD. Moreover, MP analyses are inherently susceptible to contamination from ambient and laboratory environments, underscoring the importance of stringent procedural controls. The relatively low occurrence of BMP compared to OMP has been previously noted in the literature (Huang and Xu, 2022). BMP often consists of common polymers such as those derived from car tires or rubber-based materials. However, their spectroscopic identification is challenging due to the strong absorption of infrared light by carbon black additives, which are commonly used to impart black coloration (Huang and Xu, 2022). These analytical limitations may contribute to the underrepresentation of BMP in the quantified sample. The six most abundant polymers, PET, PP, PS, PE, PA and PUR, accounted for ≈92% of the total MP count. Their dominance in ship-generated GW is consistent with their high global and regional production volumes (Plastics Europe, 2019).

Moreover, the observed statistical Chi-Square Test of Independence ($X^2_{obs} = 909.9$) far exceeds the critical value ($X^2_{crit} = 32.67$, df = 21, $\alpha = 0.05$) meaning the null hypothesis (H_o), that polymer composition is the same across all GW streams, is rejected. The reported global p-value was extremely small ($p = 4.3 \times 10^{-179}$, i.e., $p < 0.001$), well below the conventional significance threshold ($\alpha = 0.05$), further confirming that differences in polymer composition across GW streams are highly statistically significant. Stream-specific contributions to the global chi-square statistics reveal which streams drive these differences. S02A and S02G were the largest contributors, accounting for 26.78% and 26.08% of the total chi-square, respectively, followed by S02L (15.05%) and S05 (10.66%). Smaller contributions came from S02 A/L (8.12%), S04 (7.82%), S01 (2.40%), and S03G (3.09%). These results indicate that the observed heterogeneity in polymer composition is largely driven by S02A, S02G, and S02L (67.9%), while other streams contribute less prominently. Therefore, not only does polymer composition differ highly significantly across GW streams, but the degree of contribution varies between streams, highlighting specific streams with disproportionately higher deviations from expected polymer distributions.

PET was the most dominant polymer type across the analyzed GW streams, exhibiting the highest concentration in the laundry GW stream ($S2_L$; ≈438,000 MP/m^3), followed by the mixed ALG stream ($S5_{ALG}$; ≈138,000 MP/m^3). The lowest PET concentration was recorded in the mixed ALG stream ($S4_{ALG}$; ≈13,000 MP/m^3), which was comparable to the blank sample level. PP was the second most abundant polymer, with its highest concentration measured in the galley GW stream ($S2_G$; ≈125,000 MP/m^3), followed by the accommodation GW stream ($S2_A$; ≈113,000 MP/m^3). The lowest PP concentration was found in the mixed ALG stream ($S1_{ALG}$; 25,000 MP/m^3) (Fig. 2C). Overall, PET was detected in all eight GW streams, representing ≈58% of the total MP count, while PP was present in $S1_{ALG}$, $S2_A$, and $S2_G$, accounting for ≈16% of the total MP. Together, these two polymers constituted ≈74% of all identified MP, indicating their predominant contribution to shipboard GW-borne MP pollution. The remaining polymer types collectively contributed ≈26% of the total, with an average individual share of ≈2.4% (Fig. 2D).

A likely source of PP is plastic chopping boards used in kitchen areas, as supported by recent findings (Yadav et al., 2023), which could explain the elevated PP levels in $S2_G$. The highest occurrence of PET in sample $S2_L$ may be attributed to its extensive use in polyester textiles, particularly as a clothing fiber in the textile industry (Akyildiz et al., 2024; Palacios-Mateo et al., 2021). Consequently, the laundering of textiles contributes significantly to the release of PET fibers in GW. Since most of the textiles from the studied RoPax ships are laundered ashore, it is plausible that PET concentrations would have been significantly higher than the values obtained if all laundering had been conducted on board. The inference is supported by findings from cruise ships showing that onboard laundry activities result in substantially elevated concentrations of PET fibers in wastewater (Mikkola, 2020).

PET is widely used in the form of fibers, films and sheets (Webb et al., 2012), and more explicitly in post-consumer products such as plastic bottles, cosmetic packaging, shampoo and soap bottles and food containers (Paladhi et al., 2022). It is also used in electronics, automative parts, houseware, lighting products, power tools, sports goods, photographic applications and X-ray sheets (Webb et al., 2012). However, elevated PET concentration observed in S2A (100,000 MP/m^3) is plausibly explained by abrasion and cleaning of PET-based cabin materials, rather than by the fragmentation of rigid packaging. Mechanical friction and surface wear are known to release MP fibers from synthetic and personal care textiles such as polyester carpets, upholstery, bedding (Hernandez et al., 2017; De Falco et al., 2019) and disposable wipes (Lee et al., 2021; Ó Briain et al., 2020). These fibers can directly enter GW under normal use and during indoor cleaning conditions, but they may also become airborne and deposited from the indoor atmospheric environment (Dris et al., 2015). Another potential contribution to S2A MP could be toothpaste (Verschoor et al., 2014; Ustabasi and Baysal, 2019). Together, these processes provide a rapid and credible pathway for PET MP into accommodation GW.

Moreover, its occurrence in samples $S2_G$ and $S3_G$ may originate from bottled water, soft drink containers, and PET food packaging. Although Hann et al. (2018) suggested that MP concentrations in galley GW are negligible, Mikkola (2020) supported this view while noting the analytical challenges associated with quantifying MP in such samples. Nonetheless, Mikkola emphasized the need for further investigation of galley GW as a potential source of secondary MP, given that the degradation of plastic kitchen utensils and equipment during regular use may contribute to MP contamination in both GW and food preparation environments. Studies have demonstrated that chopping boards and plastic bowls made from PP and PE can release MP into food during preparation, with particle release influenced by stress, chopping style and board condition (Yadav et al., 2023; Luo et al., 2022; Jander et al., 2022). Yadav et al. (2023) estimated annual human exposure to be 7.40 – 50.7 g/person/year from PE and 49.5 g/person/year from PP chopping boards. Luo et al. (2022) further estimated that cutting along grooves in chopping boards can release ≈ 100 – 300 MP/nanoplastics per mm per cut, and up to ≈ 3000 particles/mm^2/cut in the scratched areas. Similarly, Jander et al. (2022) found that approximately 662 PP particles could be released from a plastic bowl when subjected to mechanical stress with a hand mixer. These sources could plausibly contribute to the highest PP concentration (125,000 MP/m^3) observed in the galley GW sample, $S2_G$.

Although early studies downplayed the possible occurrence of MP in galley GW, the above-cited results, including the findings of this study, indicate otherwise. Moreover, the Baltic Sea is designated as a Special Area under MARPOL Annex V but this Convention legally authorizes the discharge of ground food waste with particles smaller than 25 mm into the sea.

Therefore, the mixing of food waste with GW as practiced on board some passenger ships (Kalnina et al., 2021) and discharging under MARPOL Annex V standards, constitutes another pathway for the entry of MP from the ship's galley GW into the marine environment. While none of the vessels investigated in the present study practiced such mixing, this pathway warrants further examination.

Consistent with previous research (Jang et al., 2024; Mikkola, 2020; Kalnina et al., 2022), PET and PP were identified as the most dominant polymer types in ship-generated GW, reinforcing their widespread occurrence and persistence in marine-related wastewater systems. Furthermore, POM, a polymer commonly employed in granulated form for coarse dishwashing applications, was detected exclusively in the $S5_{ALG}$ GW stream with a concentration of ≈25,000 particles/m^3. This finding was unexpected, as POM is reportedly used for dishwashing aboard ships S2 and S3 to enhance abrasion (Fig. 4) suggesting that its presence might have been anticipated in the corresponding galley GW streams.

POM is considered one of the most significant engineering thermoplastics due to its extensive applications in mechanical, automotive, electrical, electronic, and biomedical sectors (Pielichowska, 2015). Its superior mechanical strength, toughness, and rigidity across a wide temperature range, and resistance to chemical and mechanical degradation (Pious and Sabu, 2016) likely explain its absence in S2G and S3G, as the polymer may withstand typical galley operations without fragmenting. The particles observed in $S5_{ALG}$ may therefore have originated from other alternative, unidentified sources unrelated to dishwashing activities.

The MP polymers PE and PMMA are typically expected to appear in accommodation GW which serves as the primary discharge pathway for PCCPs (Mikkola, 2020) due to their use in the formulation of PCCPs, including facial exfoliants and hand cleansers (ECHA, 2019; Plastics Europe, 2019; Amec Foster Wheeler and Infrastructure, 2017). However, the absence of PE in the accommodation GW observed in the present study suggests the possibility of alternative sources of PE contamination beyond PCCPs. This finding may also reflect the growing regulatory restrictions on MP in PCCPs, particularly the European Chemicals Agency's (ECHA) proposed ban (ECHA, 2019), which was anticipated to significantly reduce their prevalence in GW after the ban and beyond. The detection of PP, PTFE, EPDM, and SBR as the only BMP polymers is likely linked to their application in onboard infrastructure. These polymers are commonly used in components such as hose lining, sealants, cable insulation, and industrial belting due to their excellent abrasion resistance and chemical stability (Priya et al., 2022). Their occurrence in the GW streams may thus result from material degradation through mechanical wear or leaching from shipboard systems. The concentration of PS in all the samples was below the LOD. However, there's the possibility that this was underestimated due to lower

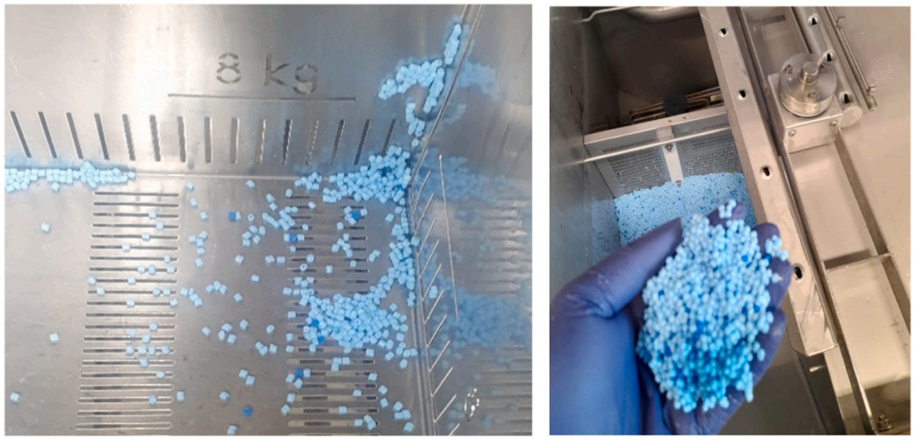

Fig. 4. Plastic pellets for granular coarse dishwashing on ship S2 and S3.

recovery rate relative to the other polymers as mentioned in Section 2.2. PVC was found at very low concentration in one sample ($S5_{ALG}$). The fact that a polymeric tube made from PVC was used on S5 for sampling could have contributed to this. The results of this study contribute to the growing understanding of MP contamination from shipboard wastewater streams and their potential transfer to MWTPs via deliveries to PRFs ashore.

3.2. Loads of MPs from ship-generated greywater into the Baltic Sea

The average annual volumes of GW generated by studied ships, S1 - S5 for the four-year period 2019 to 2022 were ≈256 m^3, ≈13,000 m^3, ≈14,000 m^3, ≈37,000 m^3 and ≈30,000 m^3, respectively, resulting in a total of ≈94,000 m^3/yr from all the studied ships (Mujingni et al., 2024). These volumes generate an average annual MP load of ≈25.9 million, 4.56 billion, 1.41 billion, 1.41 billion and 7.59 billion MPs, respectively. The load from S1, which is a relatively small quota, was discharged into the Baltic Sea, while the loads from the RoPax ships (≈15 billion MP) were delivered to PRFs with GW (Fig. 3A). Based on the estimated average number of people carried on board these ships annually, per capita annual MP loads were estimated for S1 to S5 as 11,500 MP, 19, 000 MP, 14,300 MP, 2321 MP and 12,500 MP, respectively. These loads depend on the number of people on board and the discharge frequency. The discharge frequency was assumed to be equal to the sailing days.

The lowest annual MP load from S1 was due to both the limited average number of people carried on board annually (≈2240 persons) comprising mainly crew and the lowest annual navigation days (160 days). The highest annual MP load was from S5 (7.6 million MP/yr) representing ≈ 50% of the total MP loads, followed by S2 (≈4.6 billion MP/yr) constituting ≈ 32% of the total MP loads. Despite the highest concentration of MP recorded from ship S2, the MP load from S5 was almost double that of S2. This is due to their differences in average daily utilization capacities (the average number of people carried on board daily; ≈727 and ≈ 1669 persons for S2 and S5, respectively) and their annual average volume of GW discharged. All GW from S2 to S5 was delivered to PRFs, hence, the RE of the respective MWTPs receiving GW from these ports would determine the loads reaching the Baltic Sea from the studied ships. Fig. 3A shows the annual GW-borne MP loads discharged from the studied ships.

Moreover, in 2022 the total volume of GW discharged by ships operating in the Baltic Sea was ≈5.4 million m^3 (Jalkanen et al., 2023). Based on the average MP concentrations observed in the current study, the entire Baltic Sea fleet could potentially release ≈1.1 trillion MP annually (Supplementary Material, Table 3). Of the total GW volume, ≈84.4% originated from RoPax vessels (64.5%), passenger ships (2.1%) and cruise ships (17.8%) (Jalkanen et al., 2023), which together account for ≈1.0 trillion MP (93%) potentially discharged in 2022. Within this category, RoPax ships were the dominant contributors, releasing an estimated 770 billion MP per year (≈71% of the total), followed by cruise ships with ≈212 billion MP (≈20% of the total). The highest contribution from RoPax ships shows their potential significant contribution to MP pollution due to huge volumes of GW they generate annually, if all GW generated is discharged into the sea, as well as their operations carried out all year round. Considering that RoPax vessels collectively carried ≈1.62 million people on board in 2022 (Jalkanen et al., 2023), the corresponding per capita annual MP load was estimated as ≈15,000 MP/person/yr from RoPax ships operations in the Baltic Sea.

If ship-generated GW were to be treated on board ships before discharge, the RE of the AWTP would play an important role in determining the MP load released into the Baltic Sea. For instance, an AWTP with a RE of 29% prior to discharge (Kalnina et al., 2022), would eliminate ≈ 4.45 billion MP/yr from the effluent of the studied ships, yet still contribute ≈ 10.9 billion MP annually to the Baltic Sea. IMO Regulation MEPC 227 (64) specifies a minimum phosphorus RE of 80% for AWTPs operating in MARPOL Annex IV Special Areas. As phosphorus RE has been shown to be a suitable proxy for MP removal in sewage treatment processes (Baresel and Olshammar, 2019), we estimate that ≈ 3.07 billion MP/yr could potentially be discharged into the Baltic Sea from the studied ships (see Supplementary Material, Table 2). This indicates that even if ships operating AWTPs that meet MARPOL Annex IV Special Area requirements were required to treat GW before discharge, substantial quantities of MP would still be released into the Baltic Sea. These results further show that, in addition to vessel size, utilization capacity, and the volume of GW produced, MP loads discharged directly into the sea are strongly influenced by the efficiency of onboard treatment systems.

MP loads potentially discharged directly from the studied ships were significantly lower than the estimates (≈2 million and ≈50 million MP/d) reported by Mikkola (2020) for five cruise ships. These findings indicate that cruise vessels may contribute significantly to MP pollution when GW is discharged untreated into the marine environment. Although cruise ships are generally larger and accommodate more passengers than RoPax vessels, some RoPax ships are comparable in size to smaller cruise ships, suggesting that RoPax vessels can serve as reasonable proxies for assessing MP discharges from the passenger fleet. However, the use of proxy data and assumptions in Mikkola (2020), partly based on land-based analogues, may have resulted in an overestimation of actual MP loads.

The annual MP load estimated for a research vessel (≈120 million MP/yr; Jang et al., 2024) was more than four times higher than that from the special-purpose ship S1 in this study (25.9 million MP/yr), reflecting differences in operational profiles and onboard water use. Baresel and Olshammar (2019) estimated median annual MP releases of

2.2 trillion (>20 μm) and 3.15 trillion (>10 μm) particles from technical sanitary sewer overflows (TSSOs), and 16.8 trillion (>20 μm) and 59.0 trillion (>10 μm) particles from weather-induced sewer overflows (WSSOs). These land-based loads are about two to three times higher than the total annual MP load (≈1.1 trillion MP) estimated from direct ship-generated GW discharged to the Baltic Sea in 2022. Moreover, this entire Baltic Sea passenger-ship fleet's potential annual MP release, is smaller than recent land-based input estimates for the Baltic Sea region, but still non-negligible. For example, Schernewski et al. (2021) estimate that 67 trillion MP particles/yr enter the Baltic Sea from urban land-based pathways, with the largest share originating from stormwater runoff including sewer overflows (62%), followed by MWTP effluents (25%) and untreated wastewater (13%). This implies that the fleet-wide ship GW contribution estimated corresponds to ≈1.6% of the total urban pathway load (67 trillion), but ≈ 6.6% of the MWTP-effluent component alone (17 trillion). Stormwater emissions are episodic and strongly driven by rainfall. In contrast, ship discharges are more continuous and concentrated along major shipping routes and in passenger-ship-dense sub-basins. As a result, spatial exposure patterns and local environmental impacts may differ even when the total annual loads are lower. This comparison supports the view, also highlighted in recent Baltic-focused pathway synthesis, that land-based inputs are dominated by stormwater overflow and wastewater systems (Baresel and Olshammar, 2019), but that shipborne sources can still represent a significant additional pressure in high-traffic areas and should be considered in regional scientific assessments and mitigation strategies.

3.3. The effect of various treatment technologies on MP loads into the Baltic Sea

MWTPs have a high efficiency in removing MP from wastewater even if treatment ends at the secondary treatment stage (Talvitie et al., 2017b; Baresel and Olshammar, 2019). However, advanced final treatment technologies have proven to be highly efficient. Some advanced tertiary treatment technologies employed in MWTPs within the Baltic Region include: Discfilter (DF) technology with 10 μm mesh size ($DF_{10\mu m}$) and 20 μm mesh size ($DF_{20\mu m}$) used at the Viikinmääki and Ryaverket MWTPs in Helsinki - Finland and Gothenburg - Sweden, respectively, Rapid sand filters (RSF) used at Kakolanmäki and Henriksdal MWTPs in Turku - Finland and Stockholm - Sweden, respectively, Dissolved Air Flotation (DAF) used at Paroinen MWTP in Southern Finland, Membrane Bioreactor (MBR) used at Kenkäveronniemi and Henriksdal MWTPs in Mikkeli - Finland and Stockholm - Sweden, respectively (Talvitie et al., 2017a). According to Talvitie et al., 2017a all these technologies removed >95% of MPs (>20 μm) from wastewater effluent. The highest removal efficiency was achieved with MBR (99.9%), the next in descending order were $DF_{10\mu m}$ (98.5%), RSF (97.1%) and DAF (95%). Additionally, Magnusson and Noren (2014) reported up to 99% MP retention rate in sewage sludge from a MWTP at Långeviksverket, a small MWTP in Lysekil, Sweden. Wastewater management systems like in Trelleborg where pre-treatment is carried out at PWTP could benefit from substantial MP reduction due to removal efficiencies of about 99.45% achieved by coagulation-flocculation technology (Iordachescu et al., 2024). Despite these high removal efficiencies, MP still successfully pass through various treatment stages into the effluent that is discharged into the sea. For instance, more than 99% of MP were retained at the Lysekil MWTP, but 1770 MP per hour were analyzed in the effluent leaving the plant. Considering these removal efficiencies and the concentration of MP in the GW streams obtained from the current study, ≈6 million – 651 billion MP could potentially reach the Baltic Sea if all the GW discharged into the Sea in 2022 from the entire Baltic fleet is discharged to PRFs and treated in MWTPs. This is obviously massive, and it shows that huge quantities of MP still get into the sea via treatment plants since treatment efficiencies at the MWTPs are not yet 100%. Therefore, mitigation strategies should target source separation to prevent the entry of MP into the GW streams.

3.4. Potential measures to reduce greywater-borne MP loads from Baltic shipping

From the results of this study, it could be concluded that the most effective strategy to mitigate MP emissions from ship-generated GW is to address the problem at its source by identifying major release pathways and implementing preventive measures before MP enters wastewater streams. For instance, source reduction of microfibers begins with informed choices in textile selection and optimization of onboard laundry practices. Previous research has demonstrated that fabric characteristics such as fiber type, yarn twist, weave density, and finishing treatments, as well as washing conditions and detergent formulations, have a pronounced influence on microfiber shedding during laundering (Napper et al., 2015). Therefore, targeted interventions at the source, such as the use of low-shedding fabrics and optimized washing parameters, represent a foundational step toward minimizing MP discharges from ships into the Baltic Sea.

Moreover, further research is required to distinguish and report MP separately as primary and secondary particles to better understand their sources and behavior in wastewater systems. Talvitie et al., 2017b demonstrated that, following conventional activated sludge (CAS) treatment, primary MP accounted for ≈ 19% and secondary MP for 81% of the total MP load, whereas after tertiary treatment these proportions shifted to 9% and 91%, respectively. Primary MP mainly consisted of microbeads originating from PCCPs, while secondary MP were predominantly synthetic fibers and fragmented plastic debris derived from textiles and other polymeric materials. Differentiating between these categories is essential as it enables source identification and facilitates the development of targeted mitigation strategies. For instance, policy measures such as the European Union's restriction on the intentional use of microbeads in PCCPs (EC, 2023a) represent effective interventions for reducing primary MP emissions at the source. Extending similar source-control approaches to secondary MP, through textile design innovations, fiber retention technologies, and improved wastewater treatment, would further strengthen efforts to reduce overall MP pollution in aquatic environments.

Another potential mitigation strategy at the source is the installation and use of microfiber filters in laundry machines. Like the current study with highest MP concentrations in laundry GW, prior studies, for example, Mikkola (2020) also identified laundry operations as the dominant source of MP in cruise ship GW. This finding emphasizes the critical role of onboard laundry systems as primary contributors to GW-derived MP emissions. Consequently, mitigation efforts targeting shipborne MP should prioritize the control of laundry-derived fibers before their release into GW streams. The installation of microfiber filtration units or equivalent retention technologies on laundry machines aboard passenger vessels represents a practical and effective first step toward reducing MP emissions from maritime sources.

GW originating from galley and accommodation areas also contributes to the overall MP load, although to a lesser extent than laundry effluents. These streams typically contain MP from detergents released during handwashing, toothpaste (Verschoor et al., 2014; Ustabasi and Baysal, 2019), as well as fragments from cleaning products, PCCPs, plastic kitchen utensils (Yadav et al., 2023; Luo et al., 2022; Jander et al., 2022) and other plastic materials used in daily operations. The mixed accommodation-laundry-galley (ALG) GW in this study exhibits MP concentrations within the range reported in comparable shipboard and land-based studies. While their relative contribution is lower, the larger volumes generated from accommodation and galley activities make these streams non-negligible sources of MP. Incorporating targeted source control, such as substitution of plastic-containing cleaning agents and PCCPs, and pre-filtration of galley wastewater, could complement laundry-focused measures and further reduce total MP emissions from shipboard GW systems.

Taken together, the source-specific findings underscore the need for a hierarchical mitigation strategy addressing MP emissions from ship-generated greywater. Priority should be given to high-yield sources such as laundry effluents, where end-of-pipe filtration and fiber retention technologies can deliver immediate reductions. Complementary actions in accommodation and galley systems, such as substitution of polymer-based cleaning agents, and installation of fine-mesh prefilters, can further minimize MP inputs to greywater streams. Integrating these measures within shipboard water management systems aligns with the objectives of MARPOL Annex IV, HELCOM's Baltic Sea Action Plan, and the EU Zero Pollution frameworks. Ultimately, combining source prevention with advanced port or onboard treatment technologies offers a practical pathway toward reducing MP discharges from the maritime sector, supporting the attainment of Good Environmental Status in the Baltic Sea.

4. Conclusion

This study evaluated MP polymer concentrations and annual loads in eight GW streams from five passenger ships operating in the Baltic Sea. It also assessed the contribution of ship-generated MP to the Baltic MP loads. Laundry-derived streams exhibited the highest MP concentrations, with PET and PP accounting for ≈74% of total MP abundance. Polymer distributions differed significantly among GW streams, highlighting strong source-dependent variability in MP profiles. Estimated annual MP loads from the studies RoPax vessels ranged from ≈1.24 to 7.59 billion particles, which are typically routed to MWTPs via PRFs. At the Baltic fleet scale, GW discharges of ≈5.4 million m^3/yr in 2022 could have released up to 1.1 trillion MP annually to the sea, with about 93% originating from passenger ships. Even if all GW were delivered to PRFs and treated at MWTPs, between ≈6 million and 651 billion MP/yr could still enter the Baltic environment, depending on treatment efficiency and process configuration. The findings from this study demonstrate that ship-generated GW represents a significant yet understudied pathway for MP discharges to the Baltic Sea, a Particularly Sensitive Sea Area.

Despite increasing international efforts to mitigate MP loads from maritime sources, the overall abundance of MP in the Baltic Sea is likely to rise given ongoing global growth in plastic production and the high persistence of polymers in aquatic environments. Consequently, MP might continue to pose ecological risks and socio-economic challenges for coastal communities in the region. The IMO and regional actors are aligning these efforts with the United Nations Sustainable Development Goals (SDGs), particularly SDG14: *Life Below Water*. Nevertheless, the absence of international regulations prohibiting the discharge of GW at sea means that some vessels continue to release untreated GW, further increasing MP loads in the marine environment. A key pathway for the shipping industry to contribute to SDG 14 is to adopt a comprehensive, preventive strategy targeting all potential onboard origins of MP, and addressing the problem from the source. This includes revising procurement policies for products, cleaning agents, kitchen utensils, maintenance materials, and professional textiles, and prioritizing source reduction measures. Among available mitigation options, the installation of microfiber filtration systems onboard laundry machines emerges as a practical and immediate measure to substantially reduce textile-derived MP at source and support more effective regional pollution mitigation strategies.

CRediT authorship contribution statement

J.T. Mujingni: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **E. Ytreberg:** Writing – review & editing, Validation, Supervision, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **I.-M. Hassellöv:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Funding acquisition, Formal analysis, Data curation, Conceptualization. **G.B.M. Rathnamali:** Writing – review & editing, Visualization, Validation, Investigation, Formal analysis, Data curation. **M. Hassellöv:** Writing – review & editing, Validation, Formal analysis. **K. Salo:** Writing – review & editing, Validation, Supervision, Resources, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Funding

This study was funded by the Swedish Transport Agency and the Swedish Agency for Marine and Water Management, as part of the project “Gråvatten i Östersjön” HaV Dnr 865-22.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Kent Salo reports financial support was provided by Swedish Transport Agency. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The authors acknowledge the cooperation of the shipping companies who authorized boarding and sampling of GW from their ships. Special gratitude to Jennie Folkunger, Environmental Manager at the Trelleborg Port, and Hans Friberg, Operational Manager/LDP at the Port of Stockholm, for liaising the team with relevant shipping companies.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2026.127810.

Data availability

Data will be made available on request.

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