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Microplastics (10 μm-5 mm) in European Atlantic Coastal Waters

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ABSTRACT

Microplastics (MPs) are pervasive pollutants in coastal waters, raising significant ecological concerns. This study assessed the abundance and characteristics of small-sized MPs (down to 10 μm) across European Atlantic coastal sites using harmonized sampling and analytical methods. A filter-pump microplastic sampler, the "Universal Filtering Object" (UFO), was employed at all sites, with Manta net (300-μm mesh) sampling conducted at selected locations. Microplastic concentrations ranged from <10 MPs m^{-3} to >1600 MPs m^{-3}, with the Gulf of Finland showing the lowest concentration (5 MPs m^{-3}) and the Scheldt estuary in Belgium the highest (1603 MPs m^{-3}). Most MPs (80%) were <300 μm, primarily consisting of polyester, polypropylene, and polyethylene fragments. Manta net sampling consistently underestimated both total microplastic concentrations and microplastics larger than 300 μm compared to UFO sampling. Estuaries and wastewater effluents were identified as pollution hotspots, strongly influencing local MP distributions. The median microplastic concentration found in European Atlantic waters in this study was lower than the global median for coastal waters measured using pump-based sampling devices. Although current MP levels are unlikely to pose an immediate risk to the marine pelagic food web, the projected increase in plastic production, combined with its low degradability and chemical leaching, underscores the urgency of implementing mitigation measures to prevent future environmental impacts.

1. Introduction

Plastics are the most common type of debris in marine ecosystems, accounting for 50 to 90% of the litter on shorelines and seafloors and up to 95% of floating debris (Agamuthu et al., 2019). Microplastics (MPs, 1 μm - 5 mm, Frias and Nash, 2019) have become widespread pollutants in aquatic environments, raising serious concerns due to their persistence and potential ecological impacts (UNEP, 2021). In Europe, monitoring MPs in surface waters is a standard practice under the Marine Strategy Framework Directive (MSFD, Technical Group on Marine Litter 2013;

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2023), with Manta nets being the predominant tool recommended for sampling floating microlitter (e.g., Cózar et al., 2014; Cutroneo et al., 2020, Pasquier et al. 2022). While Manta nets are advantageous for their ability to quickly sample large volumes of water and cover vast areas without requiring electricity or extensive expertise, they mainly capture MPs larger than the mesh size, usually above 300 μm (Montoto-Martinez et al., 2022). However, several studies highlight the importance of MPs smaller than 300 μm in various aquatic environments such as the North Atlantic Subtropical Gyre (Enders et al., 2015), the Greenlandic fjord Nuup Kangerlua (Rist et al., 2020), and rivers such as the Tsurumi River in Japan. (Kameda et al., 2021). MPs < 300 μm are similar in size to plankton, thereby increasing the risk of ingestion by planktivorous organisms (e.g., Cole et al., 2011, Rodriguez-Torres et al., 2024). Laboratory studies indicate that the ingestion of these small-sized MPs at high exposure concentrations can alter the physiology of marine organisms and induce potential toxicity from associated contaminants, thereby exerting adverse effects on marine biota (e.g., Botterell et al., 2019). Therefore, assessing the prevalence, concentration, mass, and characteristics of small-size MPs (<300 μm) is essential for evaluating the potential impacts of plastic pollution on marine ecosystems.

To address the limitations of nets in capturing smaller MPs, pump-based sampling devices for MPs down to 10 μm have been recently developed and employed in certain areas (Rist et al., 2020; Karlsson et al., 2020; Gunaalan et al., 2023a; Wu et al., 2024). Despite challenges such as lower filtered volume and limited sample area, pump-filter sampling devices are crucial for gaining a more comprehensive understanding of microplastic pollution. Significant advances in MPs identification and characterization have been made through various techniques, from simple light microscopy and particle-by-particle infrared spectroscopy to more complex methods such as FPA μ-FTIR and thermal degradation techniques (Mariano et al., 2021). However, the rapid diversification of analytical methods has led to challenges in comparability across studies, complicating the development of regulatory frameworks (Adhikari et al., 2022). Moreover, the high costs and complexity of some analytical methods for environmental MPs can limit their accessibility, particularly in resource-constrained regions, leading to a geographic bias in MP pollution data. As recommended by recent studies, a harmonized approach to sampling and analysis is critical to ensuring consistency in data across different geographic locations (Meyers et al., 2024; Gerigny et al., 2024).

The European Atlantic coast, characterized by dense populations and extensive human activities, is particularly vulnerable to MP pollution. In this study, we investigated the concentrations, mass, and characteristics of small-sized MPs (>10 μm) across diverse coastal environments in European Atlantic coastal waters, including bays, estuaries, areas near wastewater treatment plant (WWTP) effluent discharges, river mouths, harbors, and straits. A filter-pump MP sampler ("Universal Filtering Object" UFO-sampler) was used as a common device for sampling small-sized MPs in all studied sites. Additionally, Manta net sampling was simultaneously conducted at selected sites to compare its efficiency with the UFO sampler for collecting MPs. Furthermore, we analyzed the effects of varying processing and analytical techniques on MP concentrations and characteristics, offering critical insights to support the harmonization and standardization of methodologies in MP research. Finally, we compared MP pollution levels in European Atlantic waters with those reported in other regions and discussed the potential ecological implications of our findings, specifically on the risk for the pelagic food web.

2. Material and methods

2.1. Sampling

2.1.1. UFO filter-pump sampler

Thirty-nine samples for the analyses of MPs (>10 μm) were collected from Atlantic coastal waters across seven European countries (Fig. 1)

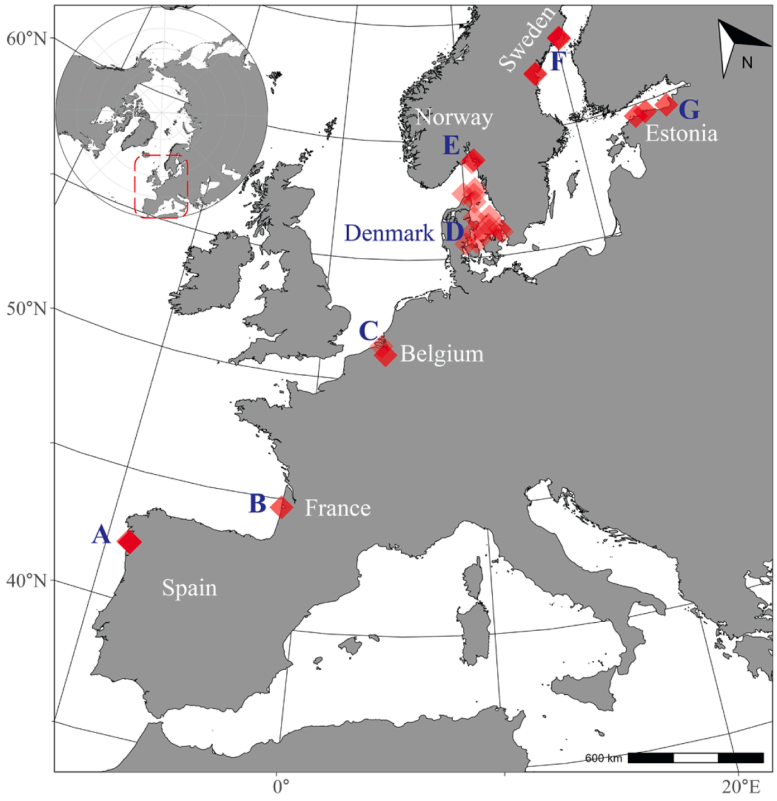

Fig. 1. Study areas and sampling locations. A- Bay of Vigo, Spain; B- Arcachon Bay, France; C- Scheldt estuary, Belgium and the Netherlands; D- Kattegat and southern Skagerrak, Denmark (Gunaalan et al., 2023); E- Northern Skagerrak, Norway; F- Gulf of Bothnia (Ugwu et al., 2024), Sweden; G- Gulf of Finland, Estonia.

using the UFO sampler developed at Aalborg University, Denmark (S.I. Fig. S1). The study was conducted in collaboration with 11 research institutions as part of the JPI OCEANS initiative, under the RESPONSE and FACTS consortia.

The sampling sites encompassed coastal bays, estuaries, areas near WWTP effluent discharge, river mouths, harbor vicinities, and straits (Table 1). Specifically, sampling in the Bay of Vigo ("Ría De Vigo") and Cíes Islands (Galicia, Spain) involved triplicate samples near an underwater WWTP outlet, duplicate samples from the Cíes Islands, a protected nature reserve, as well as samples from a coastal site unaffected by WWTP effluent and a pier (Table 1). Sampling in Arcachon Bay (France) was performed in one location, La Vigne, in duplicate.

(Table 1). This site is influenced by strong tidal currents and anthropogenic activities (small marina, e.g., 300 leisure boats, beach). The sampling in the Scheldt estuary (Belgium and the Netherlands) was conducted at three locations along the tidal-influenced part of the estuary in May 2021 (Table 1). Samples were collected around high tide (approximately 1.5 hours before to 1.5 hours after). Two sampling sites, Branst and Mariekerke (Belgium), are in villages along the tidal freshwater part of the Sea Scheldt, while the third sampling location, Hoedekenskerke (The Netherlands), is located in the Western Scheldt (Table 1). Samples were collected from a recreational fishing pontoon in Mariekerke and a private boating pier in Branst. In Hoedekenskerke, the UFO was deployed from the bank at the entrance of the small municipal port/marina (2 boats were observed when sampling). All sampling sites have a biking/walking path along the Scheldt. In three villages, there are agricultural activities. Fourteen samples were collected from various coastal and offshore stations in Kattegat and South Skagerrak (Denmark) (Table 1), as outlined by Gunaalan et al. (2023). Sampling in the northern Skagerrak (Norway) comprises two depths at a site in the Hvaler estuary, the largest estuary in Norway, close to the bottom (50 m) and at the pycnocline (5 m), as well as a sample taken proximity to the bottom at 430 m at a site outside of the estuary, near to the Island Torbjørnskjær (Table 1). Sampling in the Baltic Sea included sites in the Gulf of Bothnia (Sweden) and the Gulf of Finland (Estonia) (Table 1). The sampled sites in the Gulf of Bothnia are described in Ugwu et al. (2024). A total of six samples were collected from the Gulf of Finland from three locations: Muuga Bay, a harbor area with high anthropogenic impact; Narva Bay, an area under significant human pressure; and Eru Bay, a site close to a national park with low anthropogenic stress (Table 1). The data published by Gunaalan et al. (2023a) and Ugwu et al. (2024) are included here for comparison and integration purposes.

The components and operation of the UFO pump (S.I. Fig. S1) are described in detail by Rist et al. (2020) and Gunaalan et al. (2023a). In brief, seawater is pumped through a metal hose, initially passing through a 300 μm pore steel filter (ø = 167 mm) to capture larger particles, thereby preventing the finer mesh downstream clogging. The flow is then split into two parallel units, each equipped with 10 μm pore steel filters (ø = 167 mm). The device outlets are connected to a flowmeter to measure the filtered volume, typically approximately $1m^3$. At most sampling sites, the inlet hose was deployed directly into surface waters, except in Kattegat and South Skagerrak (Denmark), where the pump was connected to the ship's seawater intake (5 m depth, Gunaalan et al., 2023a). For samples collected in the North Skagerrak (Norway), deeper waters were sampled using an all-metal specially designed 50-L Niskin bottle, which was then directly connected to the UFO sampler. After sampling, each single filter cartridge was opened, and the 300 μm and the two 10 μm filters collected at each station were transferred to a single glass petri dish that had been pre-muffled. The petri dish was wrapped in aluminum foil and frozen at -20 °C until sample processing and analysis.

2.1.2. Manta net

In parallel with the UFO pump sampling, partners also utilized Manta net sampling. The Manta net had a frame opening of 70 × 27 cm in the Bay of Vigo (Spain), 56 × 13 cm in the Gulf of Bothnia (Sweden), and 60 × 17.5 cm in the Gulf of Finland (Estonia). These samplings were carried out in the Bay of Vigo, the Gulf of Bothnia, and the Gulf of Finland to enhance data comparability and capture MPs from the water surface. Altogether, ten samples were collected (S.I., Table S1). The net (mesh size 300-330 μm) was deployed 2-5 m from the ship's side and towed at the water surface for 20-30 min at a speed of approximately 2 knots. The samples were collected in the cod end of the net. The content of the cod end was rinsed with filtered (50 μm) tap water in glass jars and kept at a temperature of +4 °C until analysis in the lab.

2.2. Sample analyses

2.2.1. Sample preparation for analysis of MPs from the UFO sampler

The samples collected on steel filters with the UFO sampler were processed using an enzymatic-oxidative approach (S.I. Fig. S2). These procedures, adapted from Löder et al. (2017) and refined by Gunaalan et al. (2023a), were applied to all samples with minor modifications (S.I. Table S2). For more information about sample preparation from the UFO sampler, see the description in S.I. Text S1.

2.2.2. MP extraction from Manta net samples

The Manta net samples were filtered through a 100 μm stainless steel filter. Subsequently, the sample was rinsed off the filter into a glass beaker with a 5% SDS solution and kept at 50°C with stirring (100 rpm) for at least 48 h. Then, the samples were filtered through a 100 μm stainless steel filter, rinsed with filtered water, and washed back into the beaker with 200 mL of Tris-HCl buffer (pH 8.1), followed by the addition of 500 μL of protease and 500 μL of lipase. The sample was incubated at 50°C with gentle stirring for 72 h. Afterward, samples were filtered onto a 1.6-μm pore size (VWR) glass fiber filter (47 mm diameter). The filters were dried in glass petri dishes in a drying oven (BINDER FD 115) at 60°C for 15 min. Filters were stored in glass petri dishes until analysis.

2.2.3. Detection and analyses of MPs in the UFO samples

μFTIR imaging was used for most of the UFO samples (S.I. Table S2). The UFO samples from the Bay of Vigo (Spain), Kattegat-Skagerrak (Denmark), northern Skagerrak (Norway), and the Gulf of Bothnia (Sweden) were processed, and analyzed in the same laboratory (AAU) using FPA-μFTIR and identical procedures. More detailed description of the detection and analyses of MPs is in S.I. text. S2 In all samples, particles were classified as either fibers or fragments by dividing the major (longest) dimension by the minor (shortest) dimension. If the resulting ratio was greater than 3, the particle was categorized as a fiber; otherwise, it was classified as a fragment (Gunaalan et al., 2023a).

2.2.4. Analyses of MPs (>300 μm) from the Manta net samples

For all Manta net samples, MP particles (>300 μm) were analyzed using the following standard methods: organic matter was digested with a 30% hydrogen peroxide solution for 24- 48 h. After filtration on GF/C filters, samples were dried at 50 °C and particles were visually examined and picked with a hot needle to test for melting, distinguishing plastics from other materials (Devriese et al., 2015). After that, MP particles were photographed using a stereomicroscope camera. Plastic-like particles were manually picked using tweezers and placed on appropriate substrates (e.g., ZnSe transmission windows or glass Petri dishes). Spectral analyses were performed with Fourier-Transform Infrared Spectroscopy (FTIR): Spotlight 400, PerkinElmer for the Gulf of Finland or Attenuated Total Reflection (ATR): Thermo Scientific Nicolet 6700 for the Bay of Vigo, Spectrum two™, Perkin Elmer for the Gulf of Bothnia. Spectra were recorded at a resolution of 4 cm^{-1}, with a spectral range of 4000- 600 cm^{-1}. Commercial and custom-made libraries were used for polymer identification, with a correlation threshold of 0.7-0.75 for validation. The water volume filtered with a Manta net was calculated by multiplying the trawl mouth area by the shipping speed and towing time, accounting for submergence where relevant (see Mishra et al., 2022 for details). MP concentrations were expressed as the

Table 1
Sampling locations with coordinates, sampling date, sampled water volume (= V) and characteristics of the sampling site for UFO pump samples. Concentration of MPs without blank corrections = C, concentration of MPs in the blanks = B, corrected MP concentrations after considering blank concentrations = *C, estimated mass of MPs =M, and sampling depth = D (m). The concentration data from the Kattegat/Southern Skagerrak are from Gunaalan et al. (2023), and those from the Gulf of Bothnia are from Ugwu et al. (2024).

sample #	Location/Region	Latitude	Longitude	date	(MPs m^{-3})			M (µg m^{-3})	D (m)	V (m^3)	Type of Environment	Characteristics of the sampling site
					C	B	*C					
1	Bay of Vigo (Atlantic Ocean)	42.22921944	-8.796944	26/07/2021	399	11	388	25.79	0.5	1.364	WWTP	Close to underwater WWTP outlet
2	Bay of Vigo (Atlantic Ocean)	42.22921944	-8.796944	26/07/2021	587	11	576	42.53	0.5	1.448	WWTP	Close to underwater WWTP outlet
3	Bay of Vigo (Atlantic Ocean)	42.22921944	-8.796944	26/07/2021	331	11	120	15.88	0.5	1.290	WWTP	Close to underwater WWTP outlet
4	Bay of Vigo (Atlantic Ocean)	42.20666667	-8.808333	26/07/2021	116	12	104	15.30	0.5	1.150	Estuary	Coastal site with no MPs near sources
5	Bay of Vigo (Atlantic Ocean)	42.23820278	-8.890764	28/07/2021	145	11	134	18.60	0.5	1.211	Estuary	Protected nature reserve (Cies Islands)
6	Bay of Vigo (Atlantic Ocean)	42.23820278	-8.890764	28/07/2021	178	11	167	2.69	0.5	1.223	Estuary	Protected nature reserve (Cies Islands)
7	Bay of Vigo (Atlantic Ocean)	42.20206	-8.79847	29/07/2021	365	12	353	16.80	0.5	1.370	Harbour	ECIMAT pier
8	Gulf of Bothnia (Baltic Sea)	63.72777778	20.327222	03/05/2021	423	18.5	405	N/A	0.5	0.8	Estuary (river mouth)	Österfjärden, close to center
9	Gulf of Bothnia (Baltic Sea)	63.71194444	20.342222	04/05/2021	37	18.5	19	N/A	0.5	1.0	Estuary (river mouth)	Österfjärden, East of Holmen
10	Gulf of Bothnia (Baltic Sea)	62.42305556	17.401944	06/05/2021	180	18.5	162	N/A	0.5	1.0	Harbour	Alnösundet
11	Gulf of Bothnia (Baltic Sea)	62.42305556	17.401944	06/05/2021	83	18.5	65	N/A	5	1.0	Harbour	Alnösundet, 5 m
12	Scheldt estuary (North Sea)	51.074133	4.190506	3/05/2021	258	16	242	27.98	0.5	0.062	Estuary	Branst
13	Scheldt estuary (North Sea)	51.062464	4.187892	27/05/2021	1629	26	1603	346.91	0.5	0.116	Estuary	Mariekerke
14	Scheldt estuary (North Sea)	51.425457	3.918039	3/05/2021	389	3	386	317.80	0.5	0.293	Estuary	Hoedekenskerke
15	Arcachon Bay (Atlantic ocean)	44.673496	-1.23873	01/07/2021	65	8	57	1.6	0.5	0.91	Harbour	Ponton la Vigne
16	Arcachon Bay (Atlantic ocean)	44.673496	-1.23873	01/07/2021	19	7	12	0.7	0.5	0.90	Harbour	Ponton la Vigne
17	Gulf of Finland (Baltic Sea)	59.52655	24.977807	13/08/2021	16	4	12	19.7	1	1.1	Bay	Muuga bay
18	Gulf of Finland (Baltic Sea)	59.52655	24.977807	13/08/2021	9	4	5	1.3	5	1.2	Bay	Muuga Bay
19	Gulf of Finland (Baltic Sea)	59.58278167	25.84744	11/08/2021	22	8	14	0.8	1	1.01	Bay	Eru bay
20	Gulf of Finland (Baltic Sea)	59.58278167	25.84744	11/08/2021	18	4	14	2.0	5	1.01	Bay	Eru bay
21	Gulf of Finland (Baltic Sea)	59.52453	27.706277	12/08/2021	31	4	27	8.4	1	1.08	Bay	Narva bay
22	Gulf of Finland (Baltic Sea)	59.52453	27.706277	12/08/2021	16	5	11	0.6	5	1.01	Bay	Narva bay
23	Kattegat	57.66505	10.75958333	23/10/2020	89	1.2	87	69.9	5	1.082	Bay	North Kattegat
24	Kattegat	57.36006667	10.76963333	24/10/2020	60	1.2	59	32.6	5	1.255	Strait	North Kattegat
25	Kattegat	56.69215	10.63878333	24/10/2020	12	1.2	11	1.3	5	0.845	Strait	Central Kattegat
26	Kattegat	56.08805	10.4199	25/10/2020	24	1.2	22	1.9	5	1.056	Strait	Aarhus bay
27	Kattegat	55.60945	9.886033333	25/10/2020	24	1.2	22	9.4	5	1.327	Strait	Little belt
28	Kattegat	55.64651667	10.699	26/10/2020	41	1.2	39	193.1	5	1.18	Strait	Great belt
29	Kattegat	56.05665	11.15778333	26/10/2020	30	1.2	29	6.0	5	0.973	Strait	Southern Kattegat
30	Kattegat	56.3514	11.64176667	26/10/2020	21	1.2	20	4.7	5	1.169	Strait	Southern Kattegat
31	Kattegat	56.11031667	11.6921	27/10/2020	42	1.2	40	6.5	5	1.048	Strait	Southern Kattegat
32	Kattegat	56.12131667	12.45256667	27/10/2020	58	1.2	57	2.3	5	1.006	Strait	Ørsund
33	Kattegat	55.98415	12.61466667	28/10/2020	25	1.2	24	52.9	5	0.915	Strait	Ørsund
34	Kattegat	56.68095	11.81463333	28/10/2020	18	1.2	16	0.2	5	0.904	Strait	Central Kattegat
35	Southern Skagerrak	57.89068333	10.87206667	29/10/2020	65	1.2	63	3.9	5	1.032	Strait	Skagerrak
36	Southern Skagerrak	57.72195	9.942566667	30/10/2020	63	1.2	62	6.2	5	0.727	Strait	Skagerrak
37	Northern Skagerrak	59.0415282	10.7963577	15/03/2023	252	41.7	210	8.21	430	1.0	Estuary (fjord)	Torbjørnskjær
38	Northern Skagerrak	59.12104	10.97493	14/03/2023	155	41.7	113	8.49	5	1.0	Estuary	Hvaler estuary
39	Northern Skagerrak	59.12104	10.97493	14/03/2023	692	41.7	650	82.82	50	1.0	Estuary	Hvaler estuary

number of particles per cubic meter (particles m^{-3}).

2.2.5. *Normalization of Manta net and UFO sampler data for comparative analysis*

Microplastic concentrations obtained using the Manta net and the UFO sampler were normalized to facilitate descriptive comparison of relative variations within each region. Due to differences in mesh size, sampling efficiency, and targeted particle size ranges, these methods are considered complementary rather than directly comparable in absolute terms.

To highlight site-specific deviations from the regional average, we applied a relative normalization of MP concentrations (conc.) using the following formula:

$$Normalized\ MP\ conc. = (Sampled\ MP\ conc./Regional\ average\ MP\ conc.) - 1 \quad (1)$$

This normalization centers the data around zero and expresses each sampling point as a proportion above or below the regional mean, allowing for visual interpretation of intra-regional variability and spatial patterns of microplastic pollution. This is a descriptive method, and no inferential statistical tests (e.g., t-tests or ANOVA) were applied.

2.3. *Quality control and quality assurance*

Plastic materials were avoided during sampling and sample processing wherever possible. Strict quality control (QC) and quality assurance (QA) measures were implemented to mitigate contamination by external MPs both onboard (field blanks) and in the lab (procedural blanks) during sample processing and analysis (Gunaalan et al., 2023a; Liu et al., 2019). For contamination control during the sampling with the UFO pump, an air blank sample was taken at each station by exposing an empty glass petri dish to the surrounding environment during the filter-changing phase of the UFO pump. Additionally, paint fragments were collected onboard the ship to assess the potential contamination of

this specific source. All blanks underwent the same processing and analysis procedures as the pump samples. Rubber from the UFO gaskets and PTFE were excluded from the MPs quantification, as avoiding them during sampling and preparation was impossible. Potential contamination from the polyamide (PA) mesh used for Manta samples was acknowledged. Matching paint particles found in the samples were excluded. All materials were rinsed three times with Milli-Q or filtered water and covered with aluminum foil until use. Steel filters were pre-treated by muffling at 500 °C before use. Samples were covered with aluminum foil or the glass petri dish lids whenever possible. All solutions used for sample processing were filtered through GF/F filters (Whatman, pore size 0.7 μm, diameter 47 mm). Sample processing was conducted in a laminar flow cabinet where feasible, and cotton lab coats were always worn. Blank particle counts were normalized to the sampled volume. The number of MPs detected at each station was corrected to account for contamination based on the field and procedural blanks (Table 1).

2.4. Statistical analysis

To assess the influence of environmental variables on abundance of MPs (response variable), we examined four major predictors: location, type of environment (Bay, Estuary, Strait, Wastewater Treatment Plant (WWTP) effluent, and Harbor), distance from coast (an approximate nearest distance to the shore (perpendicular) from sampling point was calculated using Google Earth Pro (version 7.3.6.10201)) and depth of the sample collected. A general linear model (GLM) was employed to evaluate the effects of these predictors on the response variable. The response variable was log-transformed to meet the assumptions of normality. Post-hoc pairwise comparisons were performed using Tukey's test to compare the differences among types of environments and locations. Significance was determined at a = 0.05. All statistical analyses were computed using R software (version 4.3.2).

3. Results

3.1. Abundance, mass, and distribution of MPs collected with the UFO-pump

MPs were detected at all sampled sites, varying by up to two orders of magnitude (Fig. 2; Table 1). In the Bay of Vigo, the highest concentration was 576 MPs m^{-3} near an underwater WWTP outlet, whereas a coastal site with no nearby MP sources exhibited significantly lower levels at 104 MPs m^{-3} (Fig. 2; Table 1). Sites at the Cies Islands had moderate concentrations, with replicates measuring 134 and 167 MPs m^{-3}, whereas a concentration of 353 MPs m^{-3} was found in the sampled pier (Fig. 2; Table 1). The Gulf of Bothnia, Baltic Sea, exhibited concentrations ranging from 19 MPs m^{-3} at Österfjärden, East of Holmen, to 405 MPs m^{-3} at Österfjärden close to the center. In Kattegat and southern Skagerrak waters, the MPs concentrations varied between 11 and 87 MPs m^{-3} (Fig. 2; Table 1). The lowest MPs concentration (11 MPs m^{-3}) was found in the coastal area in the middle of Kattegat, and the highest (87 MPs m^{-3}) was in the northern part of Kattegat (Fig. 2; Table 1). The MPs concentrations were significantly higher in the northern Skagerrak than in the southern Skagerrak (Fig. 2; Table 1). At Torbjørnskjær (430 m depth), the concentration reached 210 MPs m^{-3}, while at the Hvaler estuary, the concentrations were 113 MPs m^{-3} at 5 m depth and peaked at 650 MPs m^{-3} at 50 m depth (Fig. 2; Table 1). Additionally, the Scheldt estuary in the North Sea had notably high MP levels, with Mariekerke recording 1603 MPs m^{-3} (Fig. 2; Table 1). In contrast, Muuga Bay in the Gulf of Finland showed the lowest concentrations among the studied sites, with 5 MPs m^{-3} (Fig. 2; Table 1). In Arcachon Bay, where only MPs > 100 um were quantified, the median concentration was 35 MPs m^{-3} (Fig. 2; Table 1).

The estimated total mass of MPs varied significantly across the study sites, ranging from 0.2 μg m^{-3} in the middle of the Kattegat to 346.91 μg m μg m^{-3} in the Scheldt estuary (Table 1). The lowest MP mass in the Gulf of Finland was 0.6 μg m^{-3} in Narva Bay, almost identical to the lowest mass recorded in Arcachon Bay (0.7 μg m^{-3}) (Table 1). The maximum MPs mass in the Bay of Vigo was 42.53 μg m^{-3}, near an underwater WWTP outlet (Table 1). The Scheldt estuary presented the two highest estimated MP masses in the study, with values >300 μg m^{-3} in Mariekerke (Belgium) and Hoedekenskerke (The Netherlands) (Table 1).

When comparing locations, we found that the Scheldt estuary, the Ria de Vigo (WWTP effluent), and Skagerrak showed significantly higher concentrations of MPs compared to the Gulf of Finland ($p<0.05$) (Fig. 3A, S.I. Table S3). Analyses of MP abundance by environmental type (Fig. 3B) showed that estuaries and WWTP effluents had the highest median levels of MP pollution, with estuaries exhibiting significantly higher concentrations than the other types of environment ($p<0.05$). Using a linear model to examine the effects of types of environments, depth, and distance from the coast on the MP abundance (S.I. Table S3), the analyses revealed that the type of environment had a significant effect ($p<0.05$), whereas depth and distance from the coast did not contribute significantly to the variation in MP abundance ($p>0.05$).

3.2. Size, shape and polymer composition of MPs collected with the UFO sampler

Most MPs were shorter than 300 μm in length, except in Arcachon Bay and the Scheldt estuary, and narrower than 300 μm in width, with the Scheldt estuary being the exception, where 73% of particles exceeded this width (S.I. Fig. S3). Median lengths and widths varied: Bay of Vigo (63.6 μm, 27 μm), Arcachon Bay (577 μm, 19 μm), Scheldt estuary (2032 μm, 569 μm), Kattegat and southern Skagerrak (79.6 μm, 27.6 μm), northern Skagerrak (55.8 μm, 29.8 μm), Gulf of Bothnia (68.9 μm, 13.3 μm), and Gulf of Finland (98.4 μm, 32.1 μm) (S.I. Fig. S3).

Length of fibers varied among study sites (S.I. Fig. S3). Fibers contributed 56% in the Gulf of Finland and 97% in Arcachon Bay but were absent in the Scheldt estuary for < 300 μm lengths. Fragments were prevalent in the northern Skagerrak (88%) and Scheldt estuary (99%), while 94–99% of fragments across most sites were shorter than 300 μm (S.I. Fig. S3).

The polymer composition of MPs was predominantly polyester (42.5%), followed by polypropylene (17.5%), polyethylene (8%), polyamide (6%), polystyrene (2%), and polyvinyl chloride (1.5%) (S.I. Fig. S3). Other polymers, such as polyurethane, polycarbonate, acrylic paint, and polyethyleneimine, were present, collectively making up 21% of the samples (S.I. Fig. S3).

The polymer composition was inconsistent across the study sites, as shown in Fig. 4A, with noticeable differences in proportions of polymer types. For instance, polyester accounted for 43-46% of the polymers in Kattegat-Skagerrak (46%), and Bay of Vigo (44%). In contrast, polyester made up 17-35% in the Scheldt estuary (27%), Arcachon Bay (17%), and the Gulf of Finland (35%). In northern Skagerrak, polyamide was the dominant and accounted for 35% while polyester constituted 22%. Notably, in the Scheldt estuary, polypropylene was the most abundant polymer (41%), which was completely absent in Arcachon Bay. Polyvinyl Chloride (PVC) was not found in the Scheldt estuary nor in Arcachon Bay but was present at all other sites (<3%), including Northern Skagerrak, Kattegat and southern Skagerrak, Bay of Vigo, and the Gulf of Finland. When combining the MPs found across all sites, the majority of plastic particles were fragments (71%), while fibers accounted for 29% (Fig. 4B). In both categories, polyester and polypropylene were the most prevalent polymers, with polyester fibers making up 61% of the total MPs (Fig. 4B). Overall, 80% of MPs range between 10 μm to 300 μm regardless of the site location (Fig. 4C).

3.3. Abundance and distribution of MPs collected with the Manta net

In the Bay of Vigo, MP concentrations collected with the Manta net varied notably depending on the sampling site characteristics (S.I.

Table S4). The site near an underwater WWTP outlet recorded a concentration of 0.41 MPs m^{-3}, while the control site near the Cíes Islands showed the lowest concentration at 0.05 MPs m^{-3}. Similarly, in the Gulf of Finland, MP concentrations measured with the Manta net were comparable to those in the Bay of Vigo, ranging from 0.39 MPs m^{-3} in Muuga Bay to 0.18 MPs m^{-3} in Eru Bay (S.I. Table S4). In the Gulf of Bothnia, MP concentrations ranged from 0.33 to 0.59 MPs m^{-3}, remaining within the same order of magnitude as those observed in the Gulf of Finland (S.I. Table S4).

The total concentration of MPs collected using the Manta net was 2–3 orders of magnitude lower than that collected with the UFO sampler in the same water masses (Table 1 and S.I. Tables S4). A comparison of total MP concentrations sampled with the UFO pump and Manta net at the same sites was established (S.I., Fig. S4A). The correlation was not strong (R value = 0.47) and not statistically significant (p-value = 0.24). When focusing on the MP size fraction >300 μm in length, the concentration of MPs collected with the Manta net was generally one order of magnitude lower than that in samples obtained with the UFO sampler and showed no significant correlation between two datasets after normalizing the data (R= 0.72, p= 0.11; S.I., Fig. S4B).

3.4. Size, shape, and polymer composition of MPs in Manta net samples

The lengths of MPs varied significantly across the studied regions (Fig. 5A). Larger particles were more prevalent in the Bay of Vigo, reaching up to 50 mm, compared to smaller sizes observed in the Gulf of Bothnia (up to 9 mm) and the Gulf of Finland (up to 4 mm). The polymer composition of MPs also differed among these regions. In the Bay of Vigo, polyethylene (PE) and polypropylene (PP) were predominant, with fibers forming a significant portion of the MPs. The Gulf of Bothnia exhibited a higher prevalence of PE and other polymers, while the Gulf of Finland was characterized by a dominance of polyester (PES). Fibers were the most common shape in the Gulf of Finland, followed by fragments (Fig. 5).

The overall distribution of MPs by shape and polymer type is shown in Fig. 4B. Fibers accounted for the largest proportion (44%), followed by fragments (27%) and films (10%). The polymer composition of MPs (Fig. 6B) was largely dominated by PE (7–32% across different shapes) and PP. PP constituted a substantial proportion of fibers (30%), fragments (13%), and foam (9%). Polystyrene (PS) was common in fragments (7%), foam (2%), and films (33%), while PES was primarily found in fibers (15%). Additionally, a notable fraction of MPs fell under the "Others" category, which included less common polymers such as acrylonitrile butadiene styrene (ABS), polyoxymethylene (POM), and polyurethane (PU), Poly(butyl acrylate) (PBA), Polyether ether ketone (PEEK), Polyethylenimine (PEI), etc. (Fig. 5B).

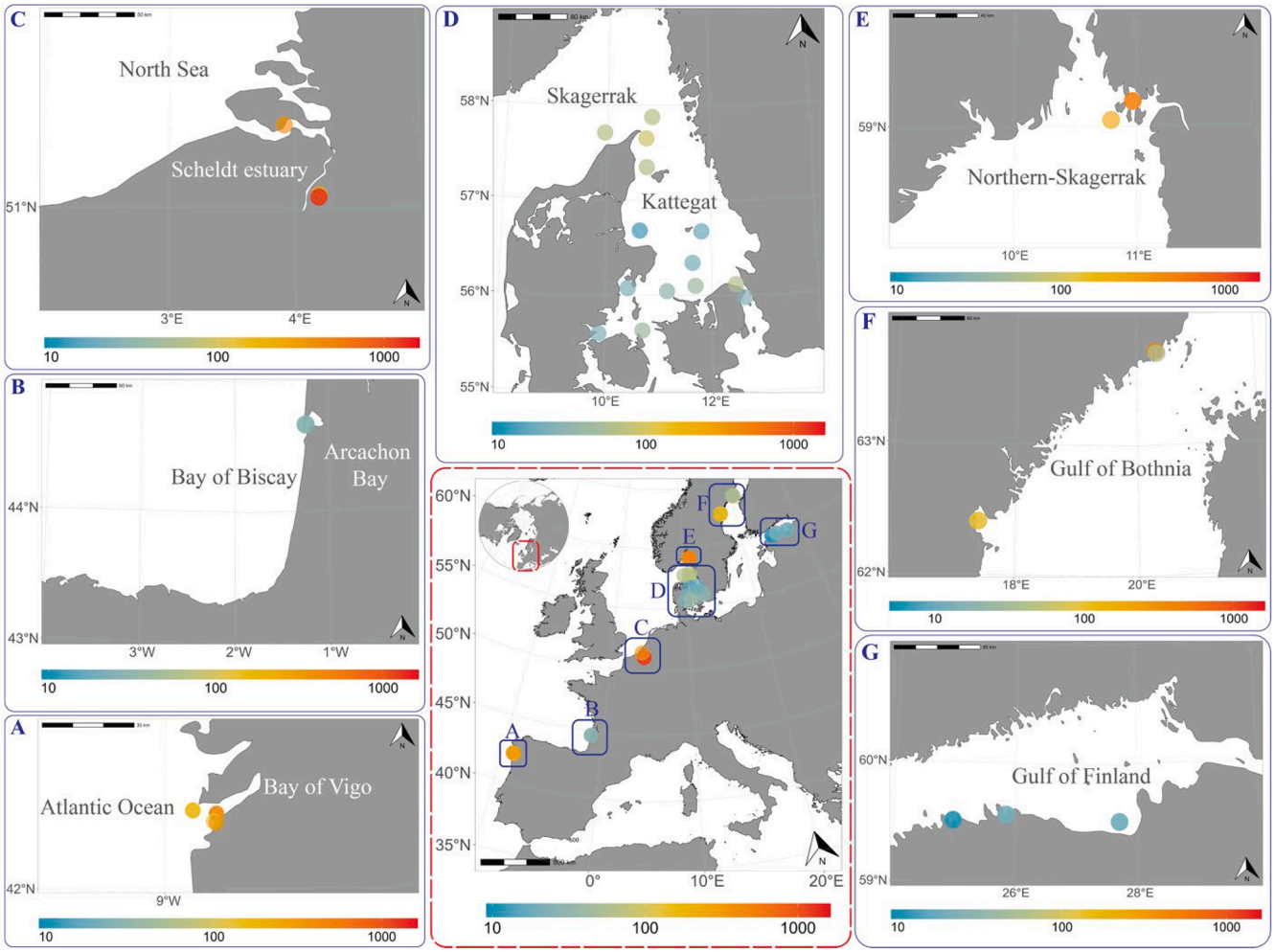

Fig. 2. Abundance of MPs collected using the UFO pump at the studied sites: A) Bay of Vigo, Spain (n=7); B) Arcachon Bay, France (n=2); C) Scheldt estuary, Belgium and The Netherlands (n=3); D) Kattegat and southern Skagerrak, Denmark (n=14) (Gunaalan et al., 2023); E) Northern Skagerrak, Norway (n=3); F) Gulf of Bothnia, Sweden (n=4) (Ugwu et al., 2024); G) Gulf of Finland, Estonia (n=6). Details about the specific sampling sites and values are provided in Table 1.

4. Discussion

4.1. Distribution, sources, and hotspots of MPs in Atlantic European coastal waters

Our findings highlight the widespread presence of MPs in European coastal waters, revealing substantial spatial variability in their concentrations and a strong influence of land-based sources. The Scheldt estuary in Belgium and the Hvaler estuary exhibited significantly higher MP concentrations compared to other coastal regions studied, highlighting the role of rivers as major conduits for transporting terrestrial plastic debris into marine environments, consistent with previous studies (e.g., Lebreton et al., 2017, Wang et al., 2022). In the Hvaler estuary, higher MP concentrations were observed in deeper waters than in surface layers, contrasting with reports from the Norwegian Coastal Current, where higher concentrations of small MPs were found in surface waters (Wu et al., 2024). This suggests that seasonal and regional hydrodynamic processes and stratification may play a key role in influencing the distribution and accumulation of MPs at different water column depths (Qian et al., 2021; Uurasjärvi et al., 2021). In Arcachon Bay, only MPs larger than 100 μm were analyzed. When comparing the observed concentrations for this size fraction with other studied bays, the level of pollution appears to be comparable to coastal sites with no MPs near sources in the Bay of Vigo.

Additionally, the effluent from wastewater treatment plants (WWTPs) in the Bay of Vigo was identified as a significant local source of MPs, with concentrations 2–3 times higher than those at other nearby sites. This indicates that current wastewater treatment processes are inadequate in removing small-sized MPs. High MP concentrations were also observed at locations affected by human activities, such as piers (e.g., ECIMAT, Bay of Vigo) and bays near urban areas in the Gulf of Bothnia. In contrast, the lowest concentrations were found farther offshore (e.g., central Kattegat) and in the less populated waters of the Gulf of Finland. Overall, these findings underscore the considerable impact of land-based sources and anthropogenic activities on the abundance and distribution of MPs in marine environments.

4.2. Characteristics of MPs in Atlantic European coastal waters

Our findings are consistent with previous studies reporting that small-sized MPs (10–300 μm) are significantly more abundant than MPs >300 μm (Wang et al., 2018, Rist et al., 2020; Wu et al., 2024). This observation is expected, as primary marine MPs are typically smaller than 300 μm (Kukkola et al., 2024), and larger plastic debris breaks into smaller particles through weathering processes (Hadiuzzaman et al., 2022, Menzel et al., 2022, Shi et al., 2023). It is important to highlight that the μFTIR technique used in this study has a lower detection limit of 10 μm (Raju et al., 2020; Veerasingam et al., 2021). Consequently, concentrations of MPs in the 1–10 μm range and nanoplastics (<1 μm; Frias and Nash, 2019) remain underrepresented despite their expected numerical dominance in the marine environment.

When examining MPs >10 μm, fragments were more abundant than fibers or other shapes, consistent with the fragmentation of plastic debris in the environment (Sorasan et al., 2022). Similar findings have been reported in other studies and aquatic compartments (Yan et al., 2021; Yu et al., 2024; Gunaalan et al., 2024). A greater diversity of plastic debris shapes was observed with the Manta net, with fibers being dominant, consistent with findings from other studies (Lusher al., 2015; Aigars et al., 2021). We identified a wide variety of synthetic polymers, with the dominant types in the studied coastal waters (PES, PP, PE, PA, PVC) reflecting the most globally produced plastic polymers (Geyer et al., 2017; Park et al., 2024) and a clear link between production and pollution. For instance, polyester, the most widely used polymer for synthetic fibers (Gupta et al., 2009), was the dominant type among fibers collected with the UFO sampler, while PP was the predominant polymer in fragments. However, differences in polymer composition among studies likely reflect variations in local sources of plastic debris and their specific uses. Polymers associated with "paints" (e.g., polyurethanes, polyacrylates) were also found in the samples, but at low concentrations, despite paint particles being increasingly recognized as a major type of environmental microplastics (European Commission, 2023), often overlooked (Gaylarde et al., 2021; Turner, 2021). It is important to note that tire wear particles are reported to be the

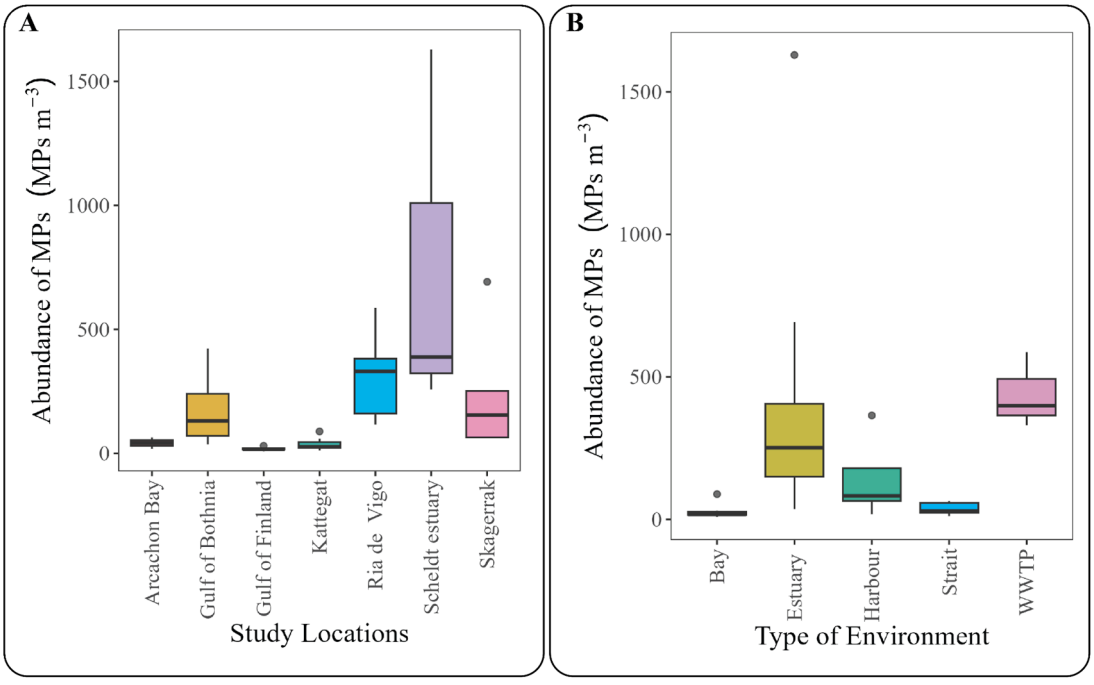

Fig. 3. Boxplot shows the abundance of MPs (MPs m-3) across our study sites (**A**) including Arcachon bay (n=2), Gulf of Bothnia (n=4), Gulf of Finland (n=6), Kattegat (n=12), Ria de Vigo (n=7), Scheldt estuary (n=3), Skagerrak (n=5), and across different environmental types (**B**) including bays (n=7), estuary (n=11), harbor (n=5), strait (n=13), and WWTP effluent (n=3).Boxes encompass the interquartile range, horizontal continues bar shows the median, whiskers are 1.5 times the interquartile range, and the dots are considered potential outliers.

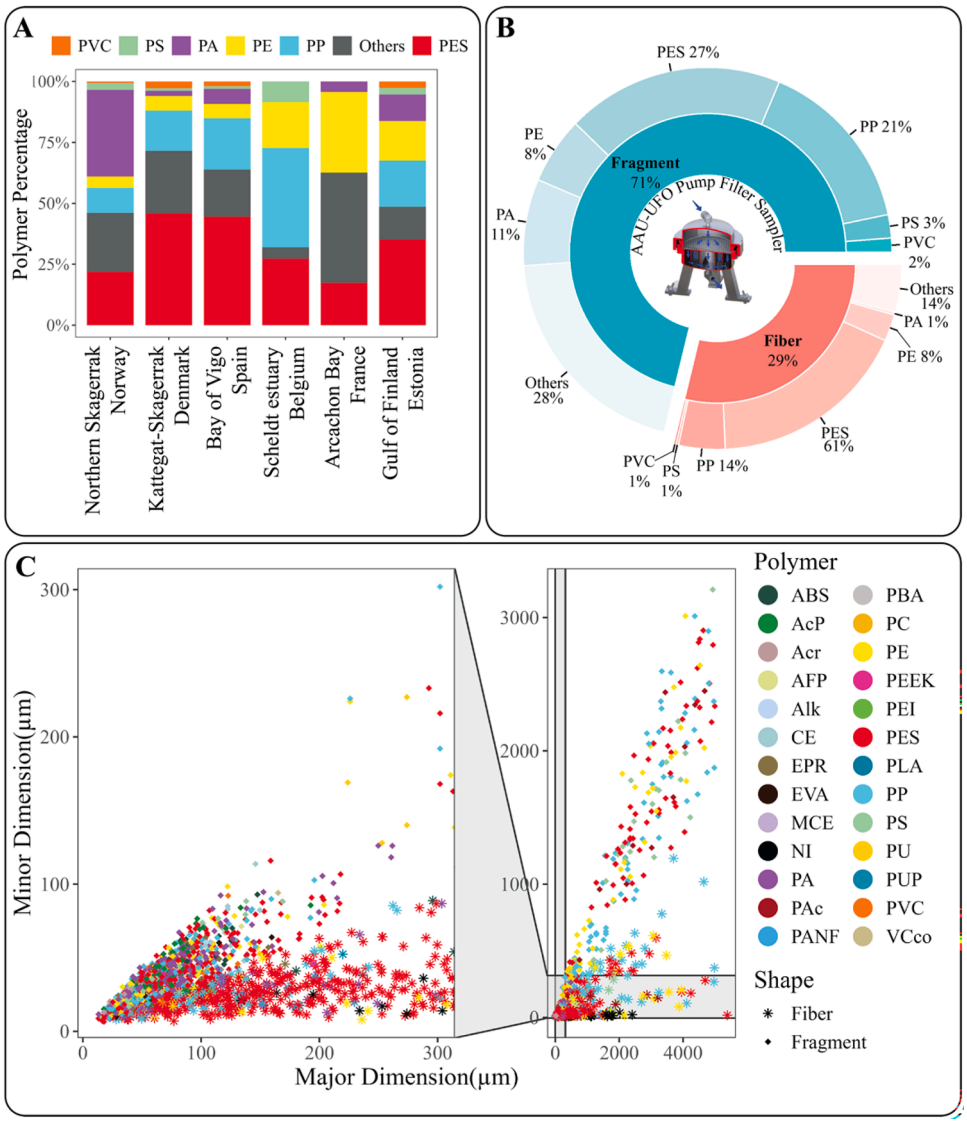

Fig. 4. A) Polymeric composition (%) of the samples collected with the UFO-sampler at each site, B) Overview of shape and primary polymer composition of MPs (%) considering MPs from all study sites (PES: Polyester/Polyethylene terephthalate, PA: Polyamide, PP: Polypropylene, PE: Polyethylene, PS: Polystyrene, PVC: Polyvinylchloride), C) Overview of the sizes of particles of different polymers, considering MPs from all study sites for MPs between 10-300 μm (the left panel) and MPs between 10 and 5000 um (right panel) collected with the UFO-sampler. The category "Others" (Panel A&B) includes all detected polymers (Panel C; ABS: Acrylonitrile butadiene styrene, AcP: Acrylic paint, Acr: Acrylic, AFP: Antifouling paint, Alk: Alkyd, CE: Cellulose ester, EPR: Epoxy phenoxy resin, EVA: Ethylene-vinyl acetate, MCE: Modified cellulose ester, PANF: PAN acrylic fiber, PBA: Poly(butyl acrylate), PC: Polycarbonate, PEEK: Polyether ether ketone, PEI: Polyethylenimine, PAc: Polyacrylamide, Polylactic acid, POM: Polyoxymethylene, PU: Polyurethane, PUP: Polyurethane paint, VCco: Vinyl chloride copolymer, NI: Unknown) except the dominant ones: PES, PA, PP, PE, PS, and PVC.

dominant type of environmental MPs (Kole et al. 2017; European Commission, 2023); however, the analysis used here did not detect synthetic thermoplastic elastomeric rubbers, such as polybutadiene (PBD) and styrene-butadiene copolymer (SBR).

The detection of both low-density polymers, such as polyethylene and polypropylene, which tend to float, and high-density polymers, including polyester, polyamide, acrylic, and polyvinyl chloride, which are more likely to sink, underscores the complex behavior of small-sized MPs in the water column. The presence of high-density small MPs in surface waters aligns with previous studies reporting that such MPs can dominate the water column (Eo et al., 2021; Gunaalan et al., 2024). This suggests that, beyond particle density, factors such as local hydrodynamics, turbulence (Siht et al., 2025; Mishra et al., 2022; Reisser et al., 2014), wind (Kukulka et al., 2012; Collignon et al., 2012), storm events (Lattin et al., 2004), salinity gradients (Uurasjärvi et al., 2021), aggregation, biofouling (Mishra et al., 2025), and interactions with biota play significant roles in redistributing small MPs throughout the water column. These findings challenge the assumption that surface waters predominantly contain only low-density MPs.

4.3. *Methodological comparisons for sampling and analyzing marine microplastics*

As expected, the total concentration of MPs larger than 10 μm collected with the UFO sampler was several orders of magnitude higher than that collected with the Manta net. This difference is primarily due to the high abundance of small-sized MPs (<300 μm) that are not effectively captured by the Manta net. While, it could be expected that a higher number of particles larger than 300 μm would correspond to a higher number of particles in the 10–300 μm range within the same water mass, suggesting a potential positive relationship between the two size fractions collected by the different methods (Manta net and UFO

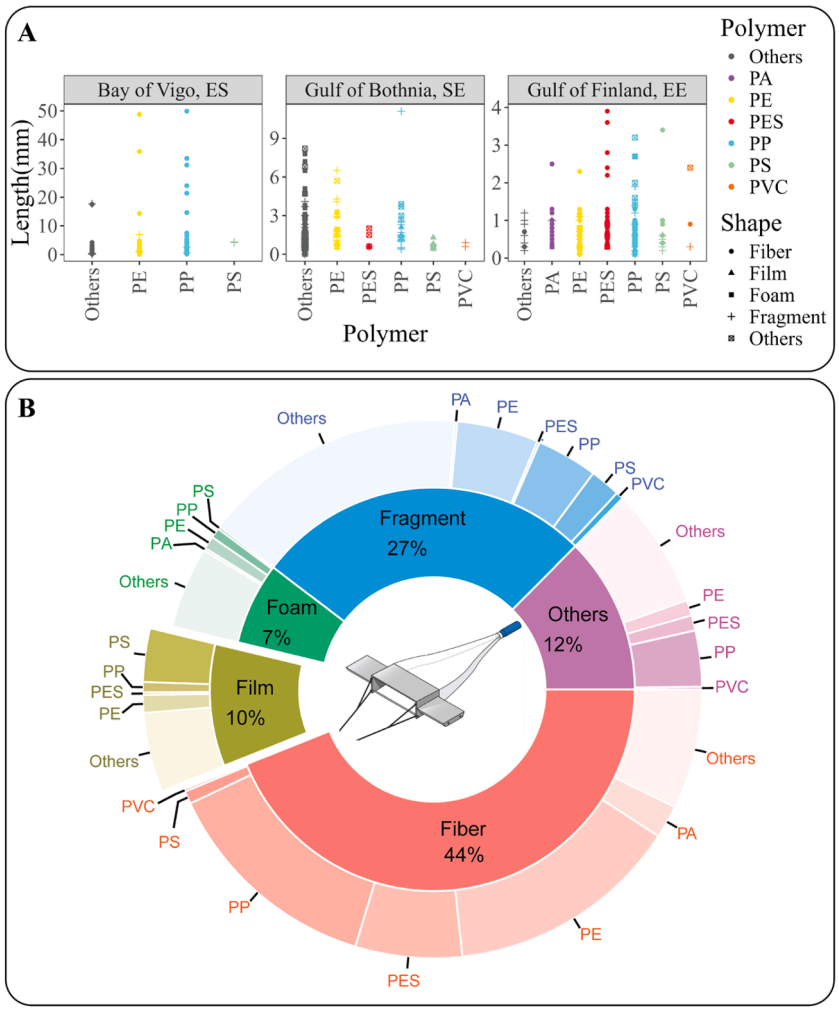

Fig. 5. A) Distribution of dominant polymer composition and shape of the meso- and microplastics collected using Manta net at each site (PES: Polyester/Polyethylene terephthalate, PP: Polypropylene, PE: Polyethylene, PS: Polystyrene, PVC: Polyvinylchloride). The y-axis scale is different for each sampling site. B) Overview of shape (%) and polymer composition of MPs collected with the Manta net combining the data from all the study sites (PES: Polyester/Polyethylene terephthalate, PP: Polypropylene, PE: Polyethylene, PS: Polystyrene, PVC: Polyvinylchloride, PA: Polyamide, and "Others" include rest of polymers identified in the samples).

sampler). However, this correlation was not observed, likely due to the limited dataset or the patchy distribution of MPs, particularly for larger particles. In terms of polymer composition, both methods identified similar dominant polymer types, with some local variations. Many fragments likely originate from the breakdown of textile fibers, potentially explaining the high proportion of PES observed among the fragments. When comparing the shapes of MPs collected by the two methods, differences were noted. Fibers dominated the Manta net samples, suggesting that longer fibers are better retained compared to fragments, which are typically smaller than 300 μm, as previously mentioned. Although Manta net sampling covers a larger surface area and may be less influenced by small-scale dynamics compared to point sampling with a filter-pump device, the combined use of both methodologies is recommended for monitoring efforts. This integrated approach provides a more comprehensive estimation of the abundance and characteristics of MPs in coastal waters.

The samples analyzed using FPA-mapping with μFTIR (Table 1, S.I. Table S2) exhibited a similarly high percentage of MPs smaller than 300 μm (88–96%) compared to those analyzed with imaging μFTIR (Scheldt Estuary, Gulf of Finland) and ATR-FTIR (Arcachon Bay). It is worth noting that ATR-FTIR is limited to quantifying MPs larger than 100 μm due to the requirement for manual visual sorting of plastic particles (Meyers et al., 2024; Chand et al., 2022; Liu et al., 2019), which introduces a potential bias toward larger MPs. In contrast, the use of automated FPA detectors with μFTIR enhances spatial resolution without preselection, enabling improved quantification of MPs down to 10 μm (Löder et al., 2015; Primpke et al., 2018; Moses et al., 2023).

From a methodological standpoint, MP detection methods, both in global studies and our study sites, were categorized into two main categories: microscopy-based and spectroscopy-based techniques (μFTIR/FTIR/Raman). The median MPs abundance detected using microscopy was notably higher (231 MPs m-3) compared to that detected through FTIR/μFTIR/Raman methods (63 MPs m MPs m^{-3}), reflecting methodological differences in detection sensitivity and particle size resolution. However, this difference was not statistically significant ($p>0.05$), likely due to the high variability among the samples (S.I. Fig. S5).

Quality assurance and quality control (QA/QC) protocols are crucial to obtaining reliable quantitative results for small-sized MPs in environmental matrices (Hung et al., 2021; Primpke et al., 2022). For instance, procedural recovery tests are vital for evaluating the efficiency of analytical processes in recovering MPs from environmental samples and preventing underestimation. The recovery efficiency in the protocol used for our samples was 89%-92% (Liu et al., 2023). Identifying, quantifying, and accounting for potential contamination sources is essential during both field sampling and laboratory analyses to ensure precise measurements. Standardized practices, such as incorporating

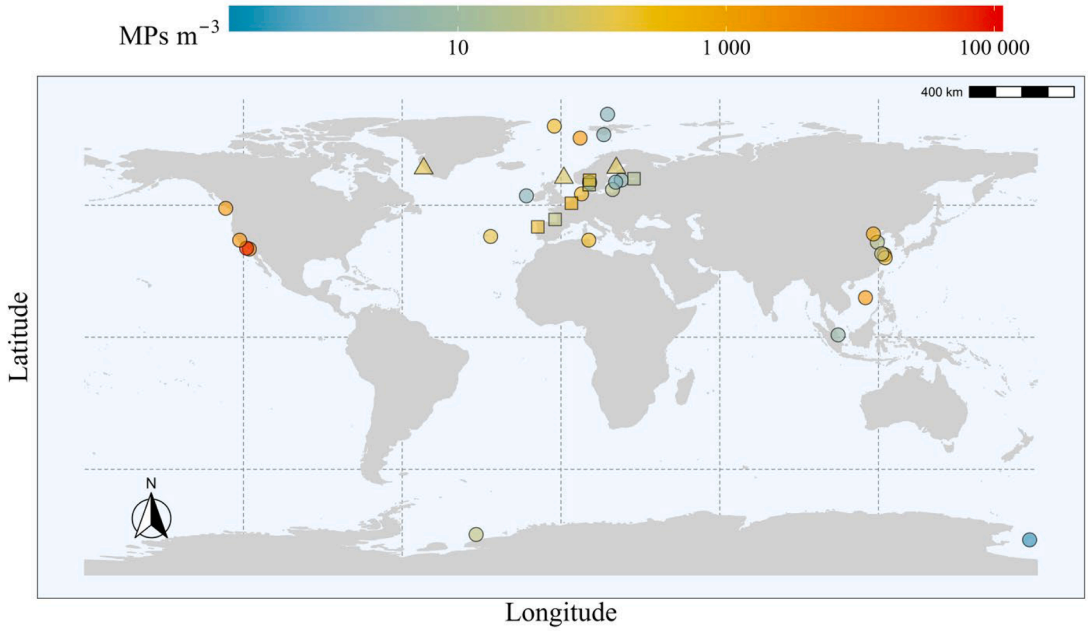

Fig. 6. Global comparison of MP abundance sampled using pump-filtration devices. Squares (□) represent data from this study obtained with the UFO sampler, triangles (△) indicate data from previous studies in other regions using the UFO sampler, and circles (○) denote data from previous studies utilizing other pump-filtration devices. The size of the symbols is not to scale and is adjusted to enhance visualization. Details of the studies included in this Fig. are provided in S. I. Table S5.

field (airborne) and procedural (laboratory) blank samples, are critical for estimating contamination levels and avoiding overestimation of MP concentrations. In our study, atmospheric and sampling contamination was generally low (<5%) but occasionally reached up to 37%. One notable contamination source during certain sampling events was paint from vessels, underscoring the importance of thorough contamination assessments to improve data reliability.

4.4. Microplastic pollution in European Atlantic waters: a global comparison

The concentrations of MPs (>10 μm) observed in this study (ranging from 5 to 1600 MPs m^{-3}) are consistent with values reported in other regions of the Atlantic Ocean. For surface waters, Enders et al. (2015) documented concentrations between 13 and 500 MPs m^{-3} across areas from the North Atlantic European Coast to the North Atlantic Subtropical Gyre. Similar levels have been observed in subsurface waters during the Atlantic Meridional Transect (Pabortsava and Lampitt, 2020) and in the Norwegian Coastal Current (0 to 1240 MPs m^{-3}, Wu et al., 2024). Higher concentrations have been noted in open waters of the Northeast Atlantic Ocean, with values from 47 to 2154 MPs m^{-3} (Karlsson et al., 2020).

For a global comparison, data on MP concentrations obtained using pump-based sampling methods from previous studies are summarized in S.I. Table S5 and graphically represented in Fig. 6. Data on MPs smaller than 300 μm from other oceans remain limited, particularly in southern hemisphere waters (Fig. 5); however, the available studies indicate comparable concentrations. For instance, 17 to 611 MPs m^{-3} were reported in the Qin River, China (Zhang et al., 2020), 10 to 50 MPs m^{-3} in Jiaozhou Bay, China (Zheng et al., 2019), and 300 to 1240 MPs m^{-3} in the Tsurumi River, Japan (Kameda et al., 2021). In the Arctic Ocean, reported concentrations range from 0 to 1287 MPs m^{-3} (Tekman et al., 2020) and 67 to 278 MPs m^{-3} (Rist et al., 2020). Globally, the highest concentration of MPs has been reported in an urban coastal environment, specifically Long Beach Harbor, California, where levels exceeded 100000 MPs m^{-3} (Wiggin and Holland, 2019; S.I. Table S6). However, when comparing studies globally, we found that unusually high concentrations were often linked to analyses that relied exclusively on microscopic identification and Nile Red staining, without validation using additional methods such as Raman or FTIR/μFTIR spectroscopy (S. I. Fig. S5). This suggests a potential overestimation of MPs in such cases (S.I. Fig. S5). Although pump-based sampling is considered, the type of pump, volume of water filtered, filtration mode, tools used, and methods for detecting MPs highly vary between studies (S.I. Table S6). In addition, most studies adhered to quality assurance and quality control (QA/QC) protocols. However, 'air blanks' (field blanks) were often absent, and some studies did not subtract blank values from the environmental samples (S.I. Table S5), which may lead to biased estimation of MP pollution (Koelmans et al., 2019; Brander et al., 2020; Primpke et al., 2022).

Compared to the global median MPs abundance of 160 MPs m^{-3}, the median abundance observed at our study sites was notably lower, at 59 MPs m^{-3} (S.I. Fig. S5). Although this difference was not statistically significant (Wilcoxon test, $p > 0.05$), likely due to high variability, it indicates generally lower levels of MP pollution in the studied sites.

4.5. Potential ecological risk for pelagic communities

The use of pump-based samplers combined with FPA-mapping μFTIR enables improved quantification of small, plankton-sized MPs, which are potentially ingestible by pelagic organisms such as zooplankton and filter feeders (e.g., Compa et al., 2018; Botterell et al., 2022; Gunaalan et al., 2023b). Laboratory studies have shown that the Effect Concentrations (ECs) of conventional MPs, typically smaller than 100 μm, for pelagic organisms generally exceed 2.0×10^8 MPs m^{-3} (equivalent to >200 MPs mL^{-1}) (e.g., Cole et al., 2013; Lee et al., 2013; Setälä et al., 2014; Vroom et al., 2017; Jeong et al., 2017; Choi et al., 2020; Rodrigues et al., 2021; Rist et al., 2020; Almeda et al., 2021, 2024). In comparison, the concentrations of MPs observed in Atlantic coastal waters are five to six orders of magnitude lower than these EC thresholds, suggesting that current MP pollution levels in these regions are unlikely to pose an acute adverse effect to the marine pelagic food web. Additionally, some pelagic organisms, such as feeding-current-generating copepods, have the ability to discriminate between prey and MPs of similar sizes, further

mitigating potential ingestion risks (Xu et al., 2022). However, further research is needed to understand the effects of MPs in combination with other stressors, such as climate change, other pollutants, and nutrient imbalances, particularly in areas with high plastic debris and MP accumulation (e.g., by the formation of marine litter windrows, Cozar et al., 2021) or coastal waters influenced by runoff events (Tian et al., 2021).

Microplastics not only pose physical risks, such as ingestion or entanglement, but also act as carriers of a cocktail of toxic chemicals. Plastics are associated with thousands of chemicals, including monomers, synthetic additives, processing aids, and non-intentionally added substances, many of which pose risks to marine ecosystems (Gunaalan et al., 2023; Costa et al. 2023; Maes et al., 2023). Once plastics enter aquatic environments, chemicals such as phthalates, alkylphenols, and bisphenols can leach into the water, potentially harming and/or bioaccumulating in marine biota (Teuten et al., 2009; Groh et al., 2019; Wagner and Schlummer, 2020; Rillig et al., 2021; Wiesinger et al., 2021, Pedersen et al 2024). Furthermore, plastics in aquatic environments can sorb harmful substances, including chlorinated pesticides, polychlorinated biphenyls, polybrominated diphenyl ethers, polycyclic aromatic hydrocarbons, antibiotics, and heavy metals, effectively transforming them into carriers and potential vectors of pollutants (Teuten et al., 2009; Tang et al., 2020), potentially amplifying the ecological risks (Mato et al., 2001; Brennecke et al., 2016, Ferreira-Filipe et al., 2024). Estimating the ecological impact of chemicals associated with plastics on aquatic life is challenging due to the complexity of leaching processes, sorption of contaminants, the multitude of plastic-associated chemicals that vary between materials (Wiesinger et al. 2021), and weathering processes that influence the degradation and transformation of these chemicals in aquatic environments. Recent studies indicate that acute effects from plastic leachates typically occur at much higher concentrations than those observed in natural waters (e.g., Cormier et al., 2021; Almeda et al., 2023; Gambardella et al., 2024). For instance, the mass of plastics detected in the water column is in the range of micrograms per cubic meter (Table 1), several orders of magnitude lower than the concentrations of micronized plastics known to produce toxic leachate effects on plankton (Gunaalan et al., 2020; Cormier et al., 2021; Almeda et al., 2023; Gambardella et al., 2024). However, even if MP concentrations in the water column are not acutely harmful, their leached chemical additives, especially persistent organic compounds, may still pose risks through bioaccumulation, trophic transfer, and chronic exposure, particularly for apex predators (Desforges et al., 2018; Hasegawa et al., 2021; Andvik et al., 2020, 2024; Pedersen et al., 2024). While techniques for measuring the additive composition of plastic litter are advancing (Corami et al., 2021, 2022; Rosso et al., 2022), the composition and concentrations of synthetic additives in small marine MPs remain largely unknown. With plastic pollution projected to triple in the following decades, more research is needed to understand the transformation, degradation, and persistence of plastic chemicals once released into aquatic environments.

5. Conclusions

The median concentration of MPs (10-5000 μm), across all studied sites in European waters was 59 MPs m^{-3}, with values ranging from 5 to over 1600 MPs m^{-3}. In terms of mass, concentrations were estimated to range from <1 μg m^{-3} to >300 μg m^{-3}. The majority of MPs detected in coastal waters were small fragments (<300 μm) primarily composed of PES, PP, PE, and PA, commonly used synthetic polymers that dominate global production. Estuaries and wastewater effluents were identified as hotspots for microplastic pollution, whereas offshore waters and sparsely populated regions exhibited significantly lower concentrations, highlighting the critical role of land-based sources in shaping MP distribution patterns. Given the numerical dominance of small MPs and the lack of correlation between data collected using Manta nets (300 μm mesh) and pump-based devices, we recommend incorporating both sampling methodologies, along with mandatory and uncompromising QA/QC procedures, into monitoring programs. Additionally, we also advise the use of FPA-mapping μFTIR for more accurate assessments of MPs down to 10 μm. Microplastic pollution levels in Atlantic European coastal waters were found to be three times lower than the median values reported for other coastal regions globally. However, this difference was not statistically significant, highlighting the widespread nature of this environmental issue. While acute exposure to the current concentrations of MPs in the water column in these regions is unlikely to pose an immediate threat to the marine pelagic food web, the projected increase in plastic production, coupled with the low degradability of plastics and chemical leaching, underscores the urgency for mitigation measures. Proactive efforts to reduce plastic pollution are essential to safeguarding marine ecosystems and mitigating impacts under future scenarios of escalating plastic waste driven by a growing global human population.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work, the author(s) used ChatGPT, a language model developed by OpenAI, in order to revise the grammar and language fluency. After using this tool/ service, the author(s) reviewed and edited the content as needed and take(s) full responsibility for the content of the publication.

CRediT authorship contribution statement

Natalja Buhhalko: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation. **Gunaalan Kuddithamby:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation. **Alvise Vianello:** Writing – review & editing, Visualization, Validation, Resources, Methodology, Investigation, Formal analysis, Conceptualization. **Anna Rotander:** Writing – review & editing, Methodology, Investigation, Formal analysis, Data curation. **Leticia Vidal-Liñán:** Writing – review & editing, Methodology, Investigation, Formal analysis, Data curation. **Ricardo Beiras:** Writing – review & editing, Project administration, Methodology, Investigation, Funding acquisition. **Mathilde Falcou-Préfol:** Writing – review & editing, Methodology, Investigation, Data curation. **Raewyn M. Town:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation. **Ketil Hylland:** Writing – review & editing, Methodology, Investigation, Formal analysis, Data curation. **Bénédicte Morin:** Writing – review & editing, Methodology, Investigation, Formal analysis. **Jérôme Cachot:** Writing – review & editing, Methodology, Investigation. **Christelle Clérandeau:** Methodology, Investigation, Formal analysis, Data curation. **Ronny Blust:** Writing – review & editing, Supervision, Funding acquisition. **Torkel Gissel Nielsen:** Writing – review & editing, Supervision, Funding acquisition. **Urmas Lips:** Writing – review & editing, Funding acquisition. **Francesca Garaventa:** Writing – review & editing, Funding acquisition. **Jes Vollertsen:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition. **Francesco Regoli:** Writing – review & editing, Project administration, Investigation, Funding acquisition. **Rodrigo Almeda:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.envadv.2025.100644.

Data availability

Data will be made available on request.

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