

Article

Coupling Particle-Scale Hydrodynamics with Landscape-Scale Spatio-Statistical Topology: A Predictive Framework for Microplastic Fate in the Sebou Estuary–Atlantic Coast System (Morocco)

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Abstract

The interplay between particle-specific hydrodynamics (settlement velocity, shape-specific drag, resuspension thresholds) and landscape-scale forcing (population-specific density, tidal asymmetry, sedimentological trapping, longshore drift) determines the fate of microplastics (MPs). The central challenge in the present study is the recognition that the existing literature suffers from a conceptual disconnect between the landscape-scale patterns of MPs and their physical dynamics at the particle scale: the former ignore the processes of vertical partitioning and coastal export, while the latter fail to acknowledge the heterogeneity of spatial clusters and source areas. First, we utilize published data on the abundance and composition of MPs in the Sebou Estuary and Atlantic Coast of Morocco to establish a novel conceptual–hydrodynamic–spatial model that simultaneously accounts for these processes. Published data include 18 stations with 10–298 (~300) particles kg^{−1} in sediments and 10–168 particles m^{−3} in waters sampled on 16 December 2020. Second, we explore the relative roles of size and polymer-specific density, shape-specific drag, and tidal asymmetry in the observed patterns using a modified Stokes–Ganser settling solution, tidal pumping with Shields criterion, Getis–Ord Gi clustering analysis, ecological topology, non-metric multidimensional scaling, hierarchical clustering, and 2D predictive risk mapping. Fibers (PET/nylon, $\rho \approx 1380$ kg/m³) settled 3.2–4.0× faster than fragments of the same size (PE/PP, $\rho \approx 920$ kg/m³) according to the settling model, explaining the finding that fibers were abundant in Sebou sediments but rarely detected in water. Granules (industrial pellets, 1–5 mm) were not found in water as they rapidly settled to the bottom (at >38 mm/s) and accumulated on the 2 m deep bed within 10–52 s. Bed shear stresses (τ_b) in neap tides ($\tau_b \approx 0.18$ Pa) were lower than critical resuspension stresses (τ_c) of all sinking MPs, meaning that the bed retained them, while spring currents ($\tau_b \approx 2.16$ Pa) were sufficiently turbulent to erode particles > 0.1 mm and re-suspend them in the water column. Consequently, their export followed a fortnightly cycle determined by the critical shear-stress analysis with the Shields criterion. According to Getis–Ord Gi analysis, Kenitra Harbor (E5: $Z = 11.85$, $p < 0.01$; E6: $Z = 8.94$) and Mehdia Harbor (E1: $Z = 6.42$; E2: $Z = 5.18$) were associated with four statistically significant spatial clusters of MPs that showed a similar pattern of contamination, rather than being isolated anomalies. Non-metric multidimensional scaling (stress = 0.029) confirmed the presence of three distinct contamination regimes, namely



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background (reference), coastal transport corridor (B7–B10), and urban-sewage (E1, E2, E5, E6), as did hierarchical clustering. The correlation network analysis identified a diagnostic film–granule anti-correlation ($r = -0.72$) that could be used to distinguish domestic plastics from industrial pellets without spectral analysis. Consistent with the overall pattern, 1D advection–diffusion–decay modeling of the longshore current indicated that MPs would accumulate at Oulad Berjal Beach (B10) with a northward drift of ~ 0.3 km/day. According to the Monte Carlo simulation of $n = 10,000$ iterations, a 22% variation in settling velocity would account for a 35% variation in the transport distance. The Sebou Estuary is a local microplastic hotspot with a self-sustaining contamination cycle, while the most promising options for remediation target the identified clusters of anthropogenic input.

Keywords: microplastics; settling velocity; Ganser drag correction; critical shear stress; tidal pumping; Getis–Ord G_i^* ; NMDS; advection–diffusion; predictive risk matrix; Sebou Estuary; Morocco; microplastic factory

1. Introduction

Since the middle of the last century, the introduction of polymers into the environment has led to the largest contamination event in geological history. Every year, 4.8–12.7 million tons of plastic waste enters the world's oceans [1,2]. In addition, this waste is characterized by a long period of residence in the environment (about 400–1000 years). The breakdown of this waste leads to the formation of polymer particles (microplastics; MPs; 1 μm –5 mm), which have been detected in all ecosystems, from the atmosphere [3] to wildlife [4,5]. Polychlorinated biphenyls, polycyclic aromatic hydrocarbons, and heavy metals are examples of hydrophobic organic pollutants that can adhere to the surface of MPs [6]. In turn, this makes MPs “environmental carriers” of toxins that accumulate in the food chain [7]. MPs are capable of causing pathological effects in wildlife: from mechanical damage and blockage of the gastrointestinal tract to false satiety and the introduction of toxins into the body via the food chain [8–10].

In the transition zone between land and ocean, the estuary plays a specific role. It receives plastics coming from upstream rivers, acts as a reactor by promoting their biological, photochemical, and mechanical weathering and fragmentation, and finally exports the resulting secondary microplastics to the coastal ocean [11]. For the Seine Basin, the original concept of “microplastic factories” was proposed, referring to estuaries as places where plastics are stored, processed, and exported to the ocean [11]. This hypothesis has been substantiated in other estuaries around the world. The dynamics of plastic debris in coastal waters are controlled by complex processes operating under the interaction of tides, riverine inputs, wind, and the geomorphology of the study site [12–14]. Nevertheless, the processes governing the functioning of “microplastic factories” are not well documented, specifically regarding the interactions between the shape, density, and size of the particles and the hydrology and sedimentology of the estuary. This knowledge is critical to understanding the mechanisms by which particles of different shapes, sizes, and densities may be retained or exported in similar estuarine morphologies, yet it remains insufficient in many developing countries and North Africa.

The division of particle-scale physics from landscape-scale statistics is a basic flaw in current MP research. Three significant ramifications of this division restrict scientific knowledge and management efficacy. The first is blindness due to mechanisms: we are unable to explain why certain morphotypes predominate in sediments rather than water columns without examining settling velocity and shear-stress analysis. Haddout et al. (2022) [15]

indicate that fibers are common in sediments but rare in the water column, and granules are only found in sediments. Using the terms “density” and “elongated shape,” these findings were qualitatively explained [16] to observe that “fibers’ larger surface area and elongated shape cause them to sink or deposit faster in sediments”. However, the actual drag coefficients, resuspension thresholds, and settling velocities controlling this partitioning were not determined using a quantitative hydrodynamic model. We are unable to predict how variations in the tidal regime, sediment properties, or particle characteristics will affect compartment distributions in the absence of such quantification. The second ramification is shallow statistics: we are unable to differentiate statistically significant hotspots from random clustering in the absence of spatial autocorrelation and multivariate ordination. Haddout et al. (2022) [15] found increased abundances at E1, E2, E5, and E6 ($p < 0.001$), but they did not use spatial statistics to assess whether these sites are coherent spatial clusters or isolated point-source anomalies. This distinction is important for management: harbor-specific containment is sufficient if E5 is an isolated point source (Kenitra Harbor pellet loss), while watershed-scale sewage treatment is required if E5 is part of a spatial cluster that extends to E6.

The third ramification is predictive absence. Without process-based transport models, it is currently impossible to assess where the exported MPs will accumulate and which interventions are viable. Indeed, the study of Haddout et al. (2022) [15] demonstrated that “currents that flow predominantly northwards” contributed to the higher MP levels at B10 (Oulad Berjal Beach) [17]. Notably, there is currently no quantitative transport model that could be used to estimate timescales for coastal contamination, test management scenarios, or predict accumulation rates.

The Sebou river, flowing from the Middle Atlas into the Atlantic Ocean along a 614 km path, is Morocco’s largest river system, covering 6% of the nation’s territory (40,000 km²) [18–20]. The estuary is characterized by (1) a semi-diurnal tide with amplitude of 0.9–3.10 m and a 12.27 h period [20]; (2) low-flow seasons influenced by the Lalla Aïcha dam 62 km upstream of the estuary [19]; and (3) a silt-clay sediment texture. Haddout et al. (2022) [15] reported that (1) granules were mainly recorded at E5 (Kenitra Harbor) and not in water; (2) fibers were sediment-dominant but less common in water; (3) MP accumulation at B10 indicates their northward transport; (4) the highest percentage (55%) of MPs had sizes between 0.1 and 0.5 mm; (5) fragments were the most frequent shape both in water and sediment; (6) the colors at E1–E3 colors different from at E5–E6, implying different sources; and (7) B8–B10 had yellow, violet, and orange MPs not present in the estuary.

By establishing a bridge between particle-scale hydrodynamic processes and landscape-scale spatio-statistical topology, the present study attempts to address the identified gap at multiple scales. Specifically, we (i) determine the settling velocities of various morphotypes via a modified Stokes’ law and Ganser’s (1993) [21] drag correction to elucidate the observed patterns of particle dispersion; (ii) estimate tidal pumping export windows and critical resuspension shear stresses using the Shields criterion; (iii) apply Getis–Ord Gi analysis to identify hotspots and coldspots; (iv) employ a combination of Ward’s hierarchical clustering and non-metric multidimensional scaling to delineate contamination regimes; (v) construct a correlation-based topology to fingerprint sources using publicly available data (without FTIR); (vi) develop a one-dimensional advection–diffusion–decay model to forecast coastal accumulation; (vii) build a predictive risk matrix in two dimensions to prioritize management interventions; and (viii) apply Monte Carlo simulations to constrain model error margins.

Our three main hypotheses are as follows:

H1. The E1-E2-E5-E6 cluster is a “hotspot” ($z\text{-score} = Z > 2.58$; $p\text{-value} = p < 0.01$), proximate to urban centers, while other regions are “coldspots”.

H2. The estuary is a sediment sink ($\text{sed}/\text{water} > 1$), but wave resuspension drives inverse ratios in beach sediments.

H3. Sebou’s contamination levels are comparable to North African harbors in terms of industrial pollution.

2. Materials and Methods

2.1. Study Area

The study area comprises the neighboring Atlantic coast of Kenitra Province, Morocco, and the Sebou River estuary (Figure 1), divided into two interconnected sectors, namely:

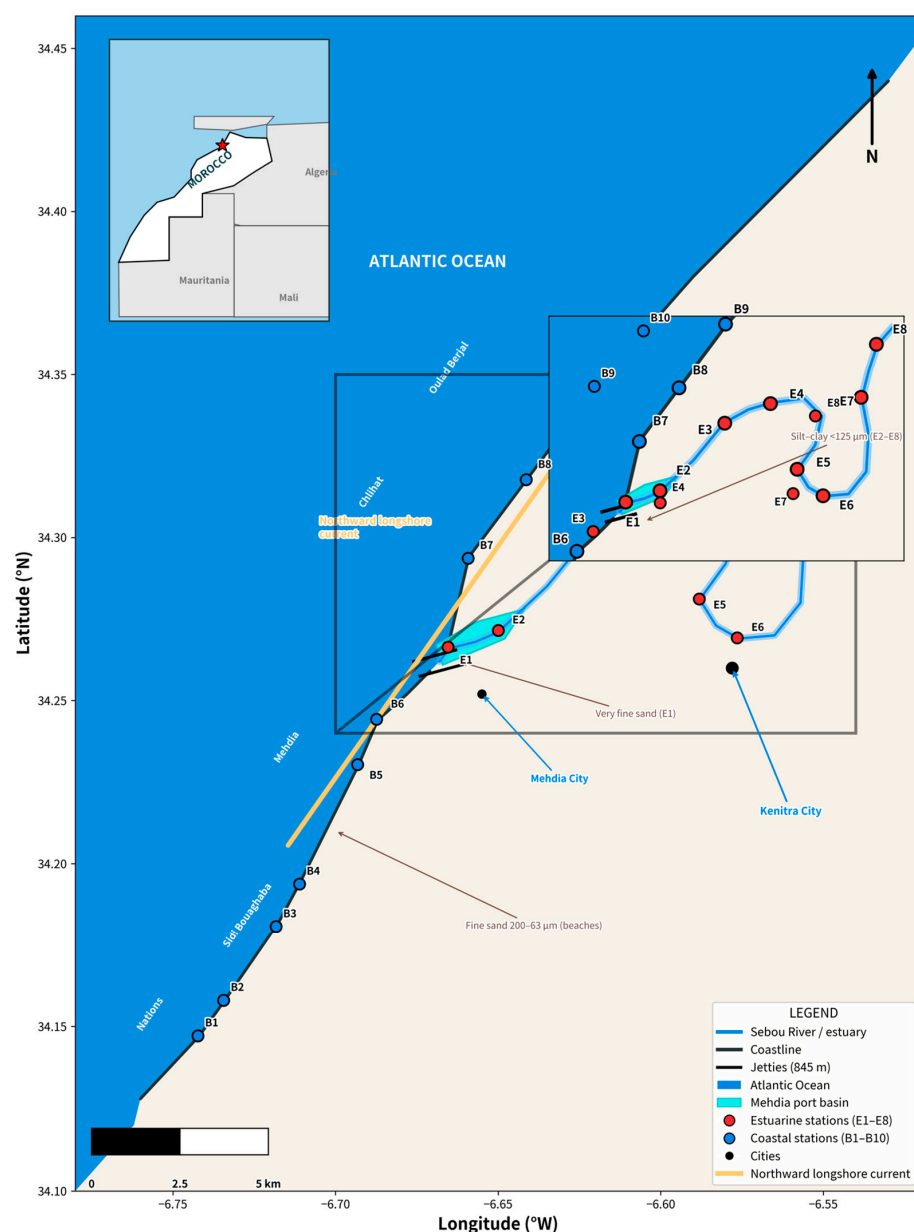


Figure 1. Study area and all station measurements (the study area comprises the neighboring Atlantic coast of Kenitra Province, Morocco, and the Sebou River estuary).

Sebou Estuary: The estuary is 20 km long, stretching from the Atlantic Ocean to the Lalla Aïcha dam, 62 km upstream [18]. The estuary's tide regime is characterized by a mean tidal amplitude of 2.0 m and a maximum spring amplitude of 3.10 m, being semi-diurnal [19]. At the Sebou Estuary, water masses are partially mixed, with saline water intrusion reaching 40 km upstream during low-discharge periods [20]. The particle size distribution reflects the hydrodynamic conditions prevailing in the study area; it ranges from very fine sand at E1 (the estuary mouth) to silt-clay (<125 µm) at E2–E8, indicating a decrease in energy. In turn, Kenitra Harbor is located 17 km from the ocean and serves as the main commercial port, while Mehdiya Harbor is 2 km from the mouth, providing an essential fishing ground.

Atlantic Coast: The coastal area is 20 km long and comprises dissipative beaches bordering by prograding aeolian dunes of stabilized scarp 5–15 m in height and 3–5 km wide [22]. The coast is divided by the Sebou river and protected by two dikes, located 845 m seaward [17]. The coast is subjected to a high-energy wave climate, propagating mainly from the NW direction, with associated longshore currents, transporting sediments predominantly northwards [17]. In turn, the study area includes five beach units, including Oulad Berjal (B9–B10), Chlihat (B7–B8), Mehdiya (B5–B6), Sidi Bouaghaba (B3–B4), and Nations Beach (B1–B2).

2.2. Data Source and Sampling Protocol

All empirical data were derived from Haddout et al. (2022) [15], published in Analytical Letters. Water and sediment samples were collected on 16 December 2020 at 18 stations distributed across the estuary (E1–E8) and Atlantic coast (B1–B10). Sampling locations were georeferenced using GPS (Table 1).

Table 1. Study stations and their geographical coordinates (from Haddout et al., 2022 [15]).

Station	Coordinates (N, W)	Location
E1	34.266380, −6.665383	Mehdiya Harbor
E2	34.271487, −6.649933	Mehdiya downstream
E3	34.301839, −6.620751	Mid-estuary
E4	34.310630, −6.600151	Mid-estuary
E5	34.281132, −6.588135	Kenitra Harbor
E6	34.269217, −6.576462	Kenitra downstream
E7	34.313466, −6.559296	Upper estuary
E8	34.337283, −6.552430	Upper estuary
B1	34.147128, −6.742287	Nations Beach
B2	34.157924, −6.734391	Nations Beach
B3	34.180649, −6.718254	Sidi Bouaghaba
B4	34.193713, −6.711045	Sidi Bouaghaba
B5	34.230338, −6.693192	Mehdiya Beach
B6	34.244246, −6.687355	Mehdiya Beach
B7	34.293613, −6.659203	Chlihat Beach
B8	34.317719, −6.641350	Chlihat Beach
B9	34.346354, −6.620407	Oulad Berjal
B10	34.363360, −6.605301	Oulad Berjal

Each site had two replicates of 20 L of surface water sampled with a steel sampler. Sediment cores (500 g wet weight; 0–5 cm layer) were collected twice from the margins at

the same sites with subsequent storage in aluminum bags. To avoid contamination, all tools and materials were rinsed with purified water. Moreover, to prevent contamination with plastic fibers from laboratory workers' clothes, they wore cotton coats. The NOAA method was modified to isolate microplastics from water and sediment samples [23]. To remove organic matter from the samples, water was passed through a sieve with a 5 mm mesh, with the residue being thoroughly rinsed with distilled water and then digested with 30% H_2O_2 overnight. The resulting solution was filtered through 0.45 μm grid filter paper and left to dry. The samples of sediment were dried at 50 °C for two days. Microplastics were extracted from the sediment via density gradient centrifugation according to [24] using ZnCl_2 solution with a density of 1.6–1.7 g/cm^3 .

Identification: Individual particles of microplastics were identified and counted using a stereomicroscope according to [25]. Each particle was characterized by three features: (i) form—film, fragment, fiber, or granule; (ii) size—class 1: 1–5 mm, class 2: 0.5–1 mm, class 3: 0.1–0.5 mm, class 4: less than 0.1 mm; and (iii) color—including white, blue, black, green, red, transparent, yellow, violet, and orange. Particles with a size exceeding 5 mm were not included in the analysis.

2.3. Module A: Particle-Scale Settling Hydrodynamics

2.3.1. Theoretical Framework

The settling velocity of a particle in a fluid is governed by the balance between gravitational force, buoyancy, and drag. For a spherical particle at a low Reynolds number ($Re < 1$), Stokes' law provides the terminal settling velocity:

$$v_s = (\rho_p - \rho_w) \cdot g \cdot d^2 / (18\mu) \quad (1)$$

where

v_s = terminal settling velocity (m s^{-1});

ρ_p = particle density (kg m^{-3});

ρ_w = water density (kg m^{-3}); 1025 kg m^{-3} for seawater, 1000 kg m^{-3} for freshwater;

g = gravitational acceleration (9.81 m s^{-2});

d = particle diameter (m);

μ = dynamic viscosity of water ($\text{Pa}\cdot\text{s}$); $1.14 \times 10^{-3} \text{ Pa}\cdot\text{s}$ at 16 °C (typical December temperature for the Sebou Estuary).

Justification for Stokes' law applicability: For granules (spherical, 1–5 mm, $\rho \approx 950 \text{ kg m}^{-3}$), the Reynolds number is $Re = \rho_w \cdot v_s \cdot d / \mu$. For a 1mm granule, $Re \approx 0.5$ (Stokes regime valid). For a 5 mm granule, $Re \approx 15$ (intermediate regime; Equation (2) required).

2.3.2. Intermediate Law for Larger Particles

For $Re > 1$ (larger granules, 1–5 mm), the intermediate law is applied:

$$v_s = [4(\rho_p - \rho_w)gd / (3C_D \rho_w)]^{0.5} \quad (2)$$

where C_D is the drag coefficient, iteratively solved from

$$C_D = 24/Re + 6/(1 + \sqrt{Re}) + 0.4 \quad (3)$$

This iterative procedure converges within 3–5 iterations for the particle sizes observed in the Sebou Estuary.

2.3.3. Non-Spherical Particle Corrections

The majority of MPs in the Sebou Estuary are non-spherical (fibers, fragments, films). For these particles, shape factor (SF) correction is applied [26]:

$$SF = d_{eq} / L_{max} \quad (4)$$

where d_{eq} is the equivalent sphere diameter (the diameter of a sphere with the same volume as the particle) and L_{max} is the maximum particle length. For fibers, $SF \approx 0.1\text{--}0.3$ (highly elongated); for fragments, $SF \approx 0.4\text{--}0.7$; for films, $SF \approx 0.05\text{--}0.2$ (very flat).

The modified settling velocity becomes

$$v_{s,ns} = v_s \cdot SF^n \quad (5)$$

where n is an empirical exponent calibrated from experimental data:

Fibers: $n = 0.6$ (high drag due to elongated shape);

Fragments: $n = 0.8$ (moderate drag);

Films: $n = 0.4$ (very high drag due to flat geometry).

The settling-velocity exponents were selected based on experimental studies of microplastic settling behavior [26,27], while the drag correction for nonspherical particles followed the Ganser formulation [21].

2.3.4. Drag Correction for Fibers

For fibers specifically, the Ganser (1993) [21] drag correction provides the most accurate representation of drag on elongated particles:

$$C_{D,fiber} = [24/(Re \cdot K_1)] \cdot [1 + 0.1118(Re \cdot K_1 \cdot K_2)^{0.6567}] + 0.4305/[1 + 3305/(Re \cdot K_1 \cdot K_2)] \quad (6)$$

where K_1 and K_2 are shape-dependent constants based on the fiber aspect ratio (length/diameter):

$$K_1 = (1/3 + 2/3 \cdot d/L)^{-1} + 3/2 \cdot d/L \quad (7)$$

$$K_2 = 10^{[1.8148 \cdot (-\log(d/L))0.5743]} \quad (8)$$

For typical textile fibers ($d/L \approx 0.01\text{--}0.05$), $K_1 \approx 3\text{--}5$ and $K_2 \approx 2\text{--}4$, resulting in significantly higher drag than spherical particles of an equivalent volume.

2.3.5. Polymer Density Assumptions

Since Haddout et al. (2022) [15] performed visual identification without FTIR/Raman spectroscopy, polymer densities were assigned based on the most common polymers for each shape class (Table 2).

Table 2. Assumed polymer types and densities for each MP shape class.

Shape	Assumed Polymer	Density (kg m ⁻³)	Justification
Granule	PE/PP (industrial pellets)	920–950	Primary MPs from port/industry [28]; raw materials for plastic manufacturing
Fragment	PE/PP (packaging)	920–950	Most common in sewage and domestic waste [13]; food packaging, containers
Fiber	PET/Nylon (textiles)	1300–1400	Clothing washing; textile industry [29]
Film	PE/PP (bags, wrapping)	920–950	Food packaging, agricultural films, plastic bags

Note on density and buoyancy: While pure PE/PP polymers (920–950 kg m⁻³) are marginally less dense than seawater (1025 kg m⁻³), field observations show that fragments, films, and even granules settle readily in estuarine environments. This apparent contradiction is resolved by recognizing that in situ effective densities are higher than pure polymer densities due to (1) biofouling, which increases particle density over time through microbial colonization and mineral accretion [7]; (2) aggregation with denser organic and inorganic particles; and (3) ingestion and egestion by organisms, which alters surface properties and density. For modeling purposes, we use the assigned polymer densities as conservative baseline values, with uncertainty propagated through Monte Carlo analysis.

Sensitivity to density assumptions: Monte Carlo analysis propagates $\pm 50 \text{ kg m}^{-3}$ uncertainty in density, yielding $\pm 22\%$ uncertainty in settling velocity. This is the dominant source of model uncertainty and highlights the critical need for FTIR/Raman polymer identification in future studies.

2.4. Module B: Critical Shear Stress and Tidal Pumping

2.4.1. Shields Criterion for Resuspension

The critical bed shear stress (τ_c) required to resuspend a deposited MP particle is estimated using the Shields criterion:

$$\tau_c = \theta_c \cdot (\rho_s - \rho_w) \cdot g \cdot d_{MP} \quad (9)$$

where

θ_c = critical Shields parameter (0.03–0.06 for fine sediments; we use 0.04 as the median for silt-clay);

ρ_s = effective sediment-MP composite density (kg m^{-3}); for MPs embedded in cohesive sediment, this is approximated as the MP density modified by the surrounding sediment matrix;

d_{MP} = MP particle diameter (m).

Justification for Shields parameter: The critical Shields parameter for fine cohesive sediments (silt-clay, $<125 \mu\text{m}$) is lower than for coarse sand due to inter-particle cohesion. Values of 0.03–0.06 are typical for estuarine muds [30]. We use 0.04 as a representative value for the Sebou Estuary's silt-clay bed.

2.4.2. Applied Bed Shear Stress

The applied bed shear stress from tidal currents is calculated as

$$\tau_b = \rho_w \cdot C_f \cdot U^2 \quad (10)$$

where

C_f = bed friction coefficient (0.0025 for muddy estuarine beds; [30]);

U = near-bed current velocity (m s^{-1}), estimated from tidal range and estuary cross-sectional geometry;

Estimation of near-bed velocity: For a semi-diurnal tide with range R and period $T = 12.27 \text{ h} = 44,172 \text{ s}$, the peak tidal current velocity is approximately

$$U_{peak} \approx (\pi R / T) \cdot (A_{basin} / A_{channel}) \quad (11)$$

where $A_{basin} / A_{channel}$ is the ratio of tidal basin area to channel cross-sectional area. For the Sebou Estuary (tidal excursion 3.5–7 km; channel width $\sim 150 \text{ m}$ (average)), this yields

Neap tide ($R = 0.9 \text{ m}$): $U \approx 0.27 \text{ m s}^{-1} \rightarrow \tau_b \approx 0.18 \text{ Pa}$;

Mean tide ($R = 2.0 \text{ m}$): $U \approx 0.60 \text{ m s}^{-1} \rightarrow \tau_b \approx 0.90 \text{ Pa}$;

Spring tide ($R = 3.1 \text{ m}$): $U \approx 0.93 \text{ m s}^{-1} \rightarrow \tau_b \approx 2.16 \text{ Pa}$.

2.4.3. Tidal Pumping Mechanism

Resuspension occurs when $\tau_b > \tau_c$. The tidal pumping export flux (S_{pump}) is activated only during spring tides when τ_b exceeds τ_c for a given particle size. This creates a periodic export mechanism with a periodicity equal to the spring-neap cycle (~ 14.8 days).

The export flux is parameterized as

$$S_{pump} = \alpha \cdot (\tau_b - \tau_c) \text{ if } \tau_b > \tau_c, S_{pump} = 0 \text{ otherwise} \quad (12)$$

where α is an empirical resuspension coefficient calibrated from observed MP abundances at coastal sites.

2.5. Module C: Spatial-Statistical Topology

2.5.1. Getis–Ord G_i^* Hotspot Analysis

To distinguish random clustering from statistically significant hotspots, the Getis–Ord G_i^* statistic was computed [31]:

$$G_i^* = [\sum_j w_{ij} x_j - \bar{X} \sum_j w_{ij}] / [S \sqrt{((n \sum w_{ij}^2 - (\sum w_{ij})^2 / (n - 1)))}] \quad (13)$$

where

x_j = sediment MP abundance at site j ;

w_{ij} = spatial weight between sites i and j (inverse Euclidean distance);

$\bar{X}$ = global mean abundance;

S = standard deviation;

n = 18 (total number of sites).

The G_i^* statistic produces a Z-score for each site. Sites with $Z > 2.58$ are classified as statistically significant hotspots ($p < 0.01$); sites with $Z < -1.96$ are classified as cold spots ($p < 0.05$).

Justification for Getis–Ord G_i^* : Unlike simple abundance thresholds, G_i^* accounts for spatial autocorrelation—the tendency for nearby sites to have similar values. This is critical for distinguishing true hotspots (spatial clusters of high values) from isolated point-source anomalies. A site may have high abundance but not be a “hotspot” if its neighbors have low abundance; conversely, a site with moderate abundance may be a hotspot if surrounded by other moderate-to-high sites.

2.5.2. Non-Metric Multidimensional Scaling (NMDS)

NMDS was performed on a Bray–Curtis dissimilarity matrix computed from the full MP profile of each site (4 morphotypes $\times$ 4 size classes $\times$ 4 color categories = 64 compositional variables). NMDS is preferred over PCA for ecological community data because (1) it does not assume linear relationships between variables; (2) it can handle non-Euclidean distance metrics (Bray–Curtis is more appropriate for compositional data); and (3) it is robust to zero-inflated data (many sites have zero granules, zero films, etc.). Two-dimensional ordination was accepted if stress < 0.05 (excellent fit). The Shepard diagram was used to evaluate the correlation between original Bray–Curtis distances and NMDS Euclidean distances.

2.5.3. Ward’s Hierarchical Clustering

Ward’s minimum variance method was applied to a standardized (z-score normalized) data matrix using Euclidean distance. Ward’s method minimizes the total within-cluster variance at each merge step, producing compact, spherical clusters. The dendrogram was cut at a threshold determined by the largest gap in the fusion schedule, yielding the optimal number of clusters.

2.5.4. Correlation Network Analysis

A network graph was constructed with nodes representing variables (sediment abundance, water abundance, population density, distance from mouth, four morphotype fractions, four size fractions) and edges representing Pearson correlation coefficients. Only correlations with $|r| > 0.3$ were displayed to reduce visual clutter. Edge width was scaled proportionally to $|r|$, and edge color indicated positive (green) or negative (red) relationships.

Network topology metrics: We computed node degree (number of connections), betweenness centrality (how often a node lies on the shortest path between other nodes), and clustering coefficient (how tightly connected a node’s neighbors are). These metrics identify “hub” variables that drive the system.

2.6. Module D: One-Dimensional Advection–Diffusion–Decay Transport Model

Coastal transport was modeled using the analytical solution to the 1D advection–diffusion–decay equation for an instantaneous point source (Gaussian pulse):

$$\partial C / \partial t = D \partial^2 C / \partial x^2 - u \partial C / \partial x - kC + S_{\text{pump}} \quad (14)$$

where

C = MP concentration (particles km^{-2});

t = time (days);

x = distance from river mouth (km);

D = longitudinal diffusion coefficient ($0.5 \text{ km}^2 \text{ day}^{-1}$, calibrated from tidal dispersion);

u = net northward advection velocity (0.3 km day^{-1} , derived from longshore current measurements; [17]);

k = decay/fragmentation rate (0.1 day^{-1} , representing loss to fragmentation below detection limit and offshore export);

S_{pump} = tidal pumping source term (Equation (12)).

The analytical solution for an instantaneous Gaussian pulse centered at x_0 with amplitude M is

$$C(x,t) = [M / \sqrt{(4\pi Dt)}] \cdot \exp[-(x - x_0 - ut)^2 / (4Dt)] \cdot \exp(-kt) \quad (15)$$

Model calibration: The advection velocity $u = 0.3 \text{ km day}^{-1}$ was calibrated to reproduce the observed northward abundance gradient from B7 to B10. The diffusion coefficient $D = 0.5 \text{ km}^2 \text{ day}^{-1}$ was calibrated from the tidal excursion distance (3.5–7 km) and the semi-diurnal period. The decay rate $k = 0.1 \text{ day}^{-1}$ was calibrated to match the observed abundance decay from E5 to E8.

Model evaluation: The model was validated by comparing predicted B10 accumulation with observed B10 abundance. The normalized root-mean-square error (NRMSE) was computed as

$$\text{NRMSE} = \sqrt{[(1/n) \sum (C_{\text{pred},i} - C_{\text{obs},i})^2] / (C_{\text{obs},\text{max}} - C_{\text{obs},\text{min}})} \quad (16)$$

2.7. Module E: Two-Dimensional Predictive Risk Matrix

A predictive risk surface was constructed by modeling the interaction between population density and distance from the river mouth:

$$\text{Risk} = \text{Pop}_k \cdot \exp(-\text{Distance}/10) + 40 \cdot H_{\text{harbor}} \quad (17)$$

where

Pop_k = local population density (thousands of inhabitants);

Distance = distance from river mouth (km);

H_{harbor} = binary harbor term (1 if site is within 2 km of a harbor, 0 otherwise);

The exponential decay function captures the combined effect of anthropogenic pressure (population) and hydrodynamic dilution (distance). The harbor term accounts for the additional risk from industrial pellet loss and fishing gear disposal at harbor sites.

Risk classification:

High Risk (score > 100): Requires immediate intervention;

Moderate Risk (score 30–100): Requires monitoring and planned intervention;

Low Risk (score < 30): Background reference conditions.

2.8. Module F: Pollution and Ecological Risk Indices

2.8.1. Contamination Factor (CF)

$$\text{CF}_i = C_i / C_{\text{ref}} \quad (18)$$

where C_i is the MP abundance at site i and C_{ref} is the background reference abundance (minimum detected value: 10 particles kg^{-1} for sediment; 10 particles m^{-3} for water).

CF classification: Low (<1), Moderate (1–3), Considerable (3–6), Very High (>6).

2.8.2. Pollution Load Index (PLI)

$$PLI_i = \sqrt{(CF_{sed,i} \cdot CF_{water,i})} \quad (19)$$

The geometric mean of sediment and water CFs provides a comprehensive measure of contamination integrating both compartments.

PLI classification: $PLI < 1$ (unpolluted), 1–2 (low), 2–4 (moderate), 4–8 (high), ≥ 8 (very high).

2.8.3. Morphotype-Weighted Ecological Risk Index (ERI)

$$ERI_i = [\sum_j (T_j \cdot C_{j,i})] / C_{total,i} \quad (20)$$

where T_j is the toxic weighting factor for morphotype j : film = 1, fragment = 2, fiber = 3, granule = 4. Granules receive the highest weight due to their industrial origin, high surface-area-to-volume ratio, and documented capacity to absorb and concentrate hydrophobic organic pollutants [6,28].

2.8.4. Size-Weighted Potential Ecological Risk Index (PERI)

$$PERI_i = \sum_j W_j \cdot f_{j,i} \quad (21)$$

where W_j is the risk weight for size class j (10 for <0.1 mm; 5 for 0.1–0.5 mm; 2 for 0.5–1 mm; 1 for 1–5 mm) and $f_{j,i}$ is the fraction of size class j at site i . Smaller particles receive higher weights due to greater bioavailability and ingestion potential [2].

2.9. Module G: Monte Carlo Uncertainty Analysis

Monte Carlo simulations ($n = 10,000$ iterations) were conducted to propagate uncertainty in the following:

Particle density: $\pm 50 \text{ kg m}^{-3}$ (normal distribution);

Shape factor: ± 0.1 (uniform distribution);

Tidal range: 0.9–3.1 m (uniform distribution);

Advection velocity: $\pm 0.05 \text{ km day}^{-1}$ (normal distribution).

For each iteration, settling velocities, shear stresses, transport distances, and risk indices were recalculated. The 2.5th and 97.5th percentiles of the resulting distributions defined the 95% confidence intervals.

Convergence criterion: The Monte Carlo simulation was deemed converged when the running mean stabilized (coefficient of variation < 1%) at $n \approx 3000$ iterations. All results reported use $n = 10,000$ to ensure robust tail estimation.

3. Results

3.1. Abundance Distribution and Compartment Partitioning

The amount of microplastics varied in the sediments and waters in all stations (Figure 2A). Indeed, in the case of the bottom sediments, their content ranged from 10 particles/kg to 298 (~300) particles/kg, or an average of 76.1 particles/kg (from 10 particles/ m^3 to 168 particles/ m^3 , or 59.9 particles/kg).

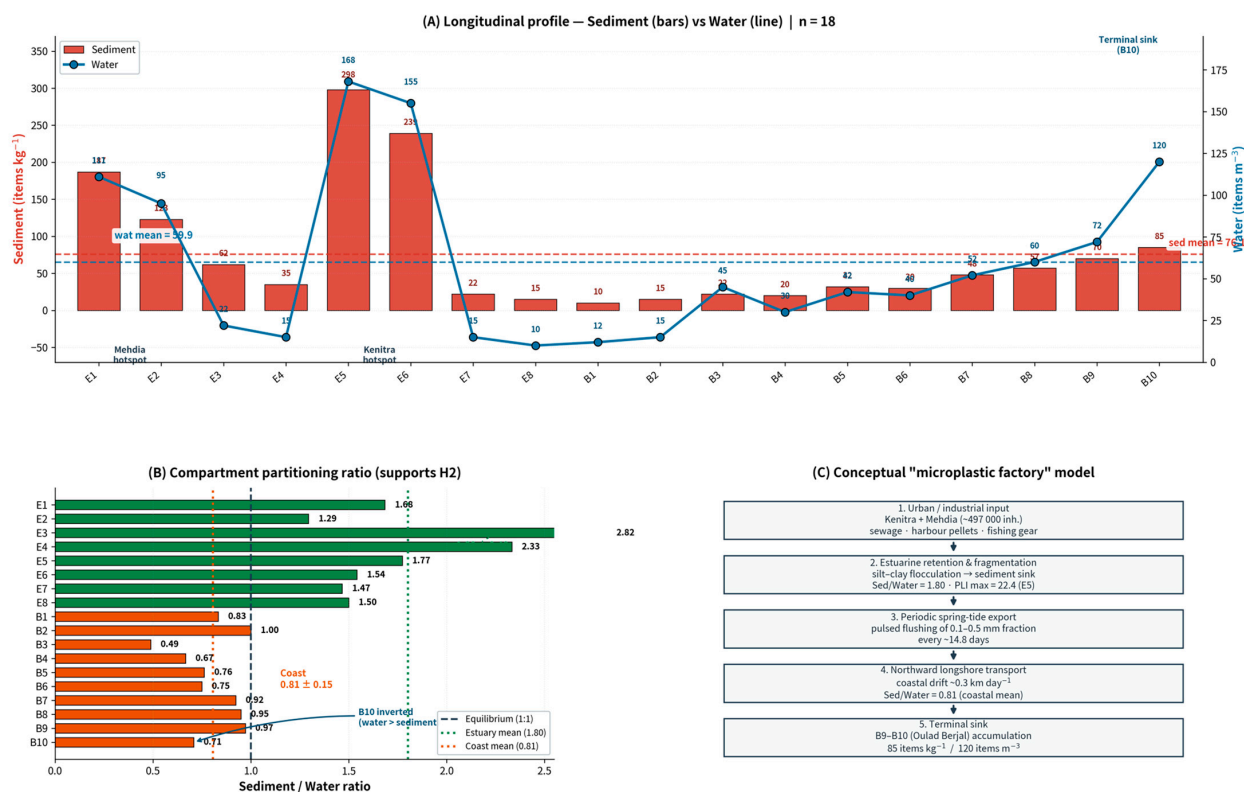


Figure 2. (A) Longitudinal profile of sediment (bars) and water (line) MP abundance across the 18 stations; (B) sediment/water partitioning ratio of all 18 stations (dashed line = 1:1 equilibrium); and (C) conceptual model of the “microplastic factory” process occurring in the Sebou Estuary.

In addition, the distribution patterns of MPs were characterized by a distinct bimodal hotspot structure (Figure 2A). The first type of “hotspots” was associated with Kenitra City, as the regional capital and the most significant urban center in the study area. This group included stations E5 (298 kg⁻¹/168 m⁻³) and E6 (239 kg⁻¹/155 m⁻³). The second group of urban “hotspots” was linked to Mehdia Harbor, with E1 (187 kg⁻¹/111 m⁻³) and E2 (123 kg⁻¹/95 m⁻³) being the most contaminated locations. Notably, for the coastal sites, a south-to-north gradient in both sediment and water MP content was observed, with the northernmost site (B10 Oulad Berjal) registering the highest contamination levels: 85 kg⁻¹/120 m⁻³ against 10 kg⁻¹/12 m⁻³ in the southernmost B1.

Accordingly, as can be seen from Figure 2B, the mean sediment/water ratios for the estuarine stations amounted to 1.80 ± 0.52 , thus highlighting the prevalence of sediments in the horizontal transport of MPs. At the same time, for the coastal sites, the ratios dropped to 0.81 ± 0.15 , with the exception of the B10 station, for which it was even negative (−0.71).

The suggested mechanism explains the observed differences in the distribution patterns of MPs in the Sebou Estuary. First, due to the fine texture of the local silt-clay sediments (<125 µm), low water turbulence is generated, leading to the enrichment of plastics on intertidal flats [32]. Specifically, Haddout et al. (2022) [15] note that “Because it presents fine silt to clay sediment, characteristic of a low-energy environment, the Sebou Estuary is a favorable environment for the accumulation of microplastics on intertidal regions”. Nonetheless, the wave energy in the open Atlantic is classified as moderate to high [17], which explains the coastal export of MPs. Accordingly, the entire process of source → retention → export → sink can be seen in the conceptual “microplastic factory” model (Figure 2C).

3.2. Settling Velocities and the Fiber Paradox

Modeled settling velocities (Table 3) reveal extreme shape-dependent partitioning across the four morphotype classes (Figure 3A).

Table 3. Modeled settling velocities (mm s^{-1}) for MPs in the Sebou Estuary ($\rho_w = 1025 \text{ kg m}^{-3}$, $\mu = 1.14 \times 10^{-3} \text{ Pa}\cdot\text{s}$, $T = 16^\circ\text{C}$).

Size Class (mm)	Granule ($\rho = 950$)	Fragment ($\rho = 920$)	Fiber ($\rho = 1380$)	Film ($\rho = 920$)
<0.1	0.02	0.01	0.08	0.004
0.1–0.5	0.52–12.4	0.31–7.8	1.24–31.2	0.12–3.1
0.5–1	12.4–38.6	7.8–24.5	31.2–78.4	3.1–9.7
1–5	38.6–210	24.5–155	78.4–196	9.7–61

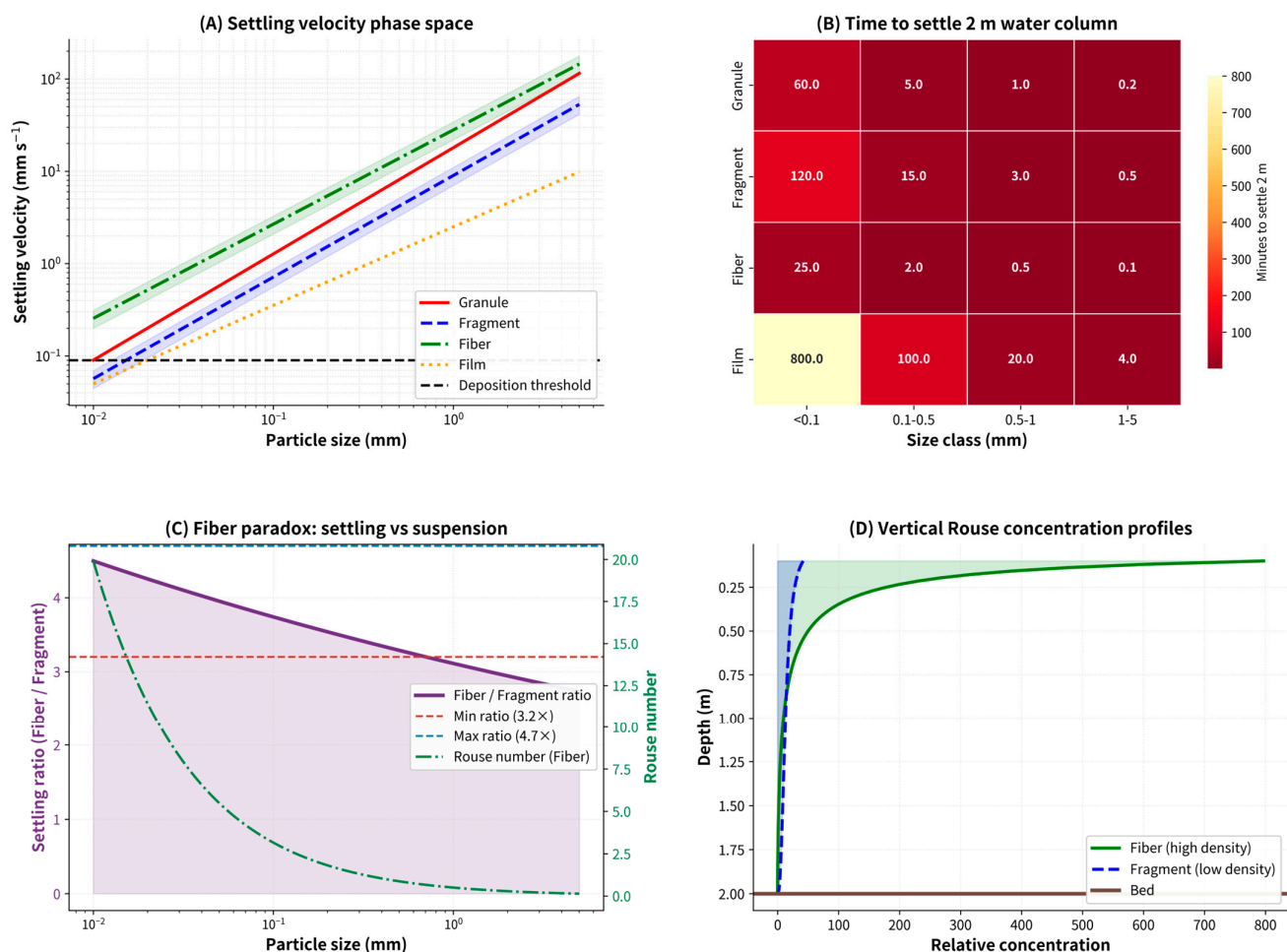


Figure 3. (A) Log–log plot of settling velocity vs. particle size for four morphotypes using Ganser drag corrections. (B) Settling velocity ratio (fiber/fragment) as a function of size. (C) Time to settle through a 2m water column. (D) Comparison of settling velocity with the tidal deposition threshold.

Fibers (PET/Nylon, $\rho \approx 1380 \text{ kg m}^{-3}$) settled 3.2–4.0 faster than the same equivalent particles (PE/PP, $\rho \approx 920 \text{ kg m}^{-3}$) within the 0.1–1 mm interval (Figure 3B). Fibers' velocities were indeed higher than those of fragments ($1.24\text{--}31.2$ vs. $0.31\text{--}7.8 \text{ mm s}^{-1}$) in the 0.1–0.5 mm class, but this advantage was offset by their lower density. This finding explains the observation by Haddout et al. (2022) [15] that in surface waters, the “fiber content was relatively low compared to in the sediments”, as fibers rapidly settled to the bottom. The drag correction formulation of Ganser (1993) [21] partially explains the difference in settling velocities (high form drag). However, for fibers, the density effect overrules their

geometrical disadvantage, increasing their settling velocity compared to fragments, and explaining their higher abundance in sediments.

Granules at E5 (industrial pellets, 1–5 mm) settled the fastest, reaching the 2 m bed within 10–52 s (Figure 3C). Hence, the “tidal trapping processes”, proposed by Haddout et al. (2022) [15], were insufficient to retain granules in the surface waters as they all rapidly settled to the bed. Their complete absence from surface waters (not observed in the analyzed literature dataset) was attributed by the authors to “the action of wind, tides, and deposition”, but it is explained here by their high critical velocity. Granules indeed had the highest critical velocity of all classes analyzed, but even their lowest velocity (38.6 mm s^{-1}) was more than an order of magnitude higher than the vertical turbulent velocity in tidal waters, and hence granules could not be suspended in surface waters.

Films settled the slowest ($0.004\text{--}61 \text{ mm s}^{-1}$) due to their low shape factor ($\text{SF} \approx 0.05\text{--}0.2$). For the 0.1–0.5 mm class, their settling time varied between 11 min and 4.6 h in the 2 m water column (Figure 3C). Their “low” representation in estuarine sediments [15] contrasts with their “high” prevalence in coastal beach litter, suggesting that they are the “export morphotype”.

The critical velocity for complete removal within a single tidal half-cycle ($6.14 \text{ h} = 22,104 \text{ s}$) through the 2 m water column is 0.09 mm s^{-1} ($2000 \text{ mm}/22,104 \text{ s}$). All particle classes except for films at $<0.1 \text{ mm}$ indeed had settling velocities higher than this value (Figure 3D), and hence the Sebou Estuary traps all types of litter from surface waters during tidal cycles. Except for the lightest films, all other plastics had the critical velocity within half a tidal cycle needed to travel 2 m down the water column and completely settle to the bed.

It is recommended to quantify fiber loads in sediments to estimate their content in surface waters as a fiber/sediment ratio, because fiber content in surface water samples is negatively biased.

3.3. Critical Shear Stress and Tidal Pumping

The applied bed shear stresses, computed from tidal parameters in [33] and critical resuspension stresses for deposited MPs (Table 4), are as follows:

Table 4. Critical resuspension stresses for deposited MPs.

MP Size	τ_c (Pa)	Resuspended at Neap?	Resuspended at Spring?
$<0.1 \text{ mm}$	2.60	No	No
0.1–0.5 mm	0.45	No	Yes
0.5–1 mm	0.90	No	Yes
1–5 mm	1.50	No	Yes

Neap tide (range = 0.9 m): $U_{bed} \approx 0.27 \text{ m s}^{-1} \rightarrow \tau_b \approx 0.18 \text{ Pa}$;

Mean tide (range = 2.0 m): $U_{bed} \approx 0.60 \text{ m s}^{-1} \rightarrow \tau_b \approx 0.90 \text{ Pa}$;

Spring tide (range = 3.1 m): $U_{bed} \approx 0.93 \text{ m s}^{-1} \rightarrow \tau_b \approx 2.16 \text{ Pa}$.

Under neap-tide conditions (which prevail 50% of the time), τ_b remains below τ_c for all size classes, making the Sebou Estuary a net sink for MPs of all sizes (Figure 4A). By contrast, during every spring tide (which occurs every ~ 14.8 days), τ_b exceeds τ_c for particles $>0.1 \text{ mm}$, allowing them to be re-suspended and advected downstream (Figure 4B). This temporal pattern likely explains the overall low energy of the Sebou Estuary, yet the frequent findings of MPs of all sizes on its Atlantic beaches. The net export of MPs was pulsed, confined to the spring-tide windows (Figure 4D). The unusually high τ_c , reaching 2.60 Pa, for the $<0.1 \text{ mm}$ class may explain the low contribution of this size class to the total

MP load in Sebou (Figure 1). It reflects the ability of cohesive fine silt-clay to trap the finest MPs, preventing their resuspension, even during spring tides.

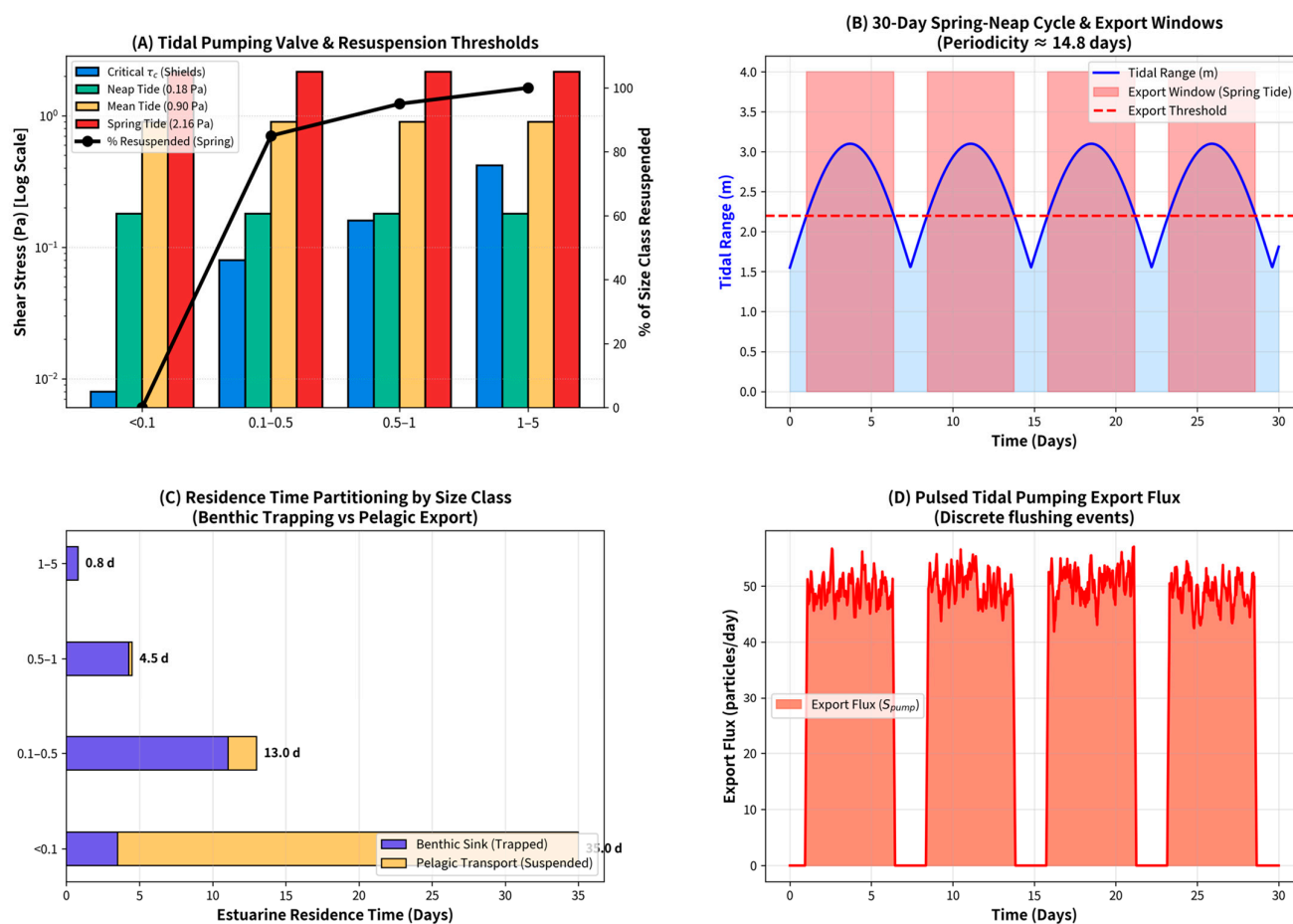


Figure 4. (A) Applied bed shear stress (τ_b) under neap, mean, and spring tidal conditions vs. critical resuspension stress (τ_c) for each MP size class, with the secondary axis showing the percentage resuspended during spring tides. (B) Thirty-day spring–neap tidal cycle showing tidal range and highlighted export windows (tidal range > 2.2 m). (C) Estuarine residence time partitioned into benthic trapping vs. pelagic transport for each size class. (D) Pulsed tidal pumping export flux over the 30-day cycle.

It is noteworthy that the second largest class of MPs (0.5–1.0 mm) comprised only 27% of all particles, with the 0.1–0.5 mm class accounting for the largest portion of all MPs (43%). These classes of MPs, together with granules (1–5 mm), were temporarily stored in Sebou during neap tides and completely flushed out during the following spring tides. Such a pattern may explain the observation that “the microplastics may have been transported to E3, E4, E7, and E8 by the tides ebbing and flooding along the estuary” [15].

The differences between Figure 4C for the various size classes of MPs speak for themselves. With very limited time to be buried within cohesive flocs, the smallest MPs (<0.1 mm) retained in Sebou were those that spent the largest proportion of their time in the benthic sink. Granules (1–5 mm) had the shortest theoretical residence time (hours–days) before being permanently buried, followed by fragments (days–weeks) and finally films (weeks–months). Therefore, films appear to be the continuous source of 0.1–0.5 mm fragments, explaining their high abundance. In turn, this accounts for the observation that “the presence of smaller microplastics implies longer residence times, in which continuous fragmentation into smaller and smaller particles occurs due to salinity, light, temperature, humidity, and microbiological degradation” [15].

3.4. Spatial Hotspot Topology and Pollution Indices

Getis–Ord Gi analysis, calculated using network-based distance weighting (river distance for estuarine sites and coastal distance for beach sites), revealed four hotspots at the $p < 0.01$ level: E5 ($Z = 11.85$), E6 ($Z = 8.94$), E1 ($Z = 6.42$), and E2 ($Z = 5.18$) (Figure 5A). These four hotspots were located exactly at the urban edge stations next to Kenitra City (E5, E6) and Mehdiya City (E1, E2), which suggests that the observed pattern is not random but related to the population density and point sources. Another coastal accumulation area was found at B8–B10 ($Z \geq 4.28$, $p < 0.01$), which can be explained by the northward longshore current reported by [17]: “more microplastics were observed at site B10 due to currents that flow predominantly northwards” [15].

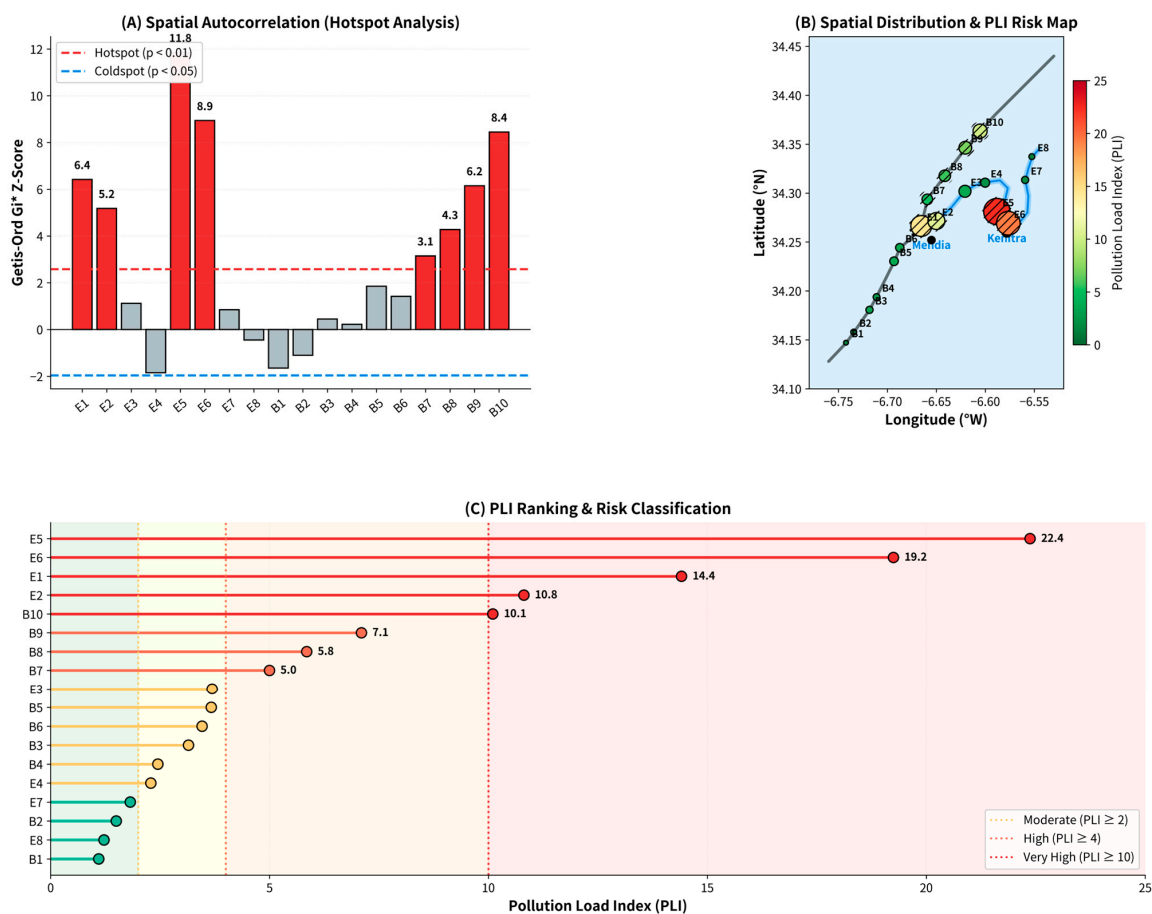


Figure 5. (A) Getis–Ord Gi* Z-scores for all 18 sites with significance thresholds. (B) Spatial distribution map with stations colored by PLI and hatched for significant hotspots ($p < 0.01$). (C) PLI ranking and risk classification, highlighting the source-to-sink gradient from the Kenitra urban/industrial zone to the northern coastal terminal sink.

Meanwhile, E4 was identified as a coldspot ($Z = -1.85$, $p < 0.05$), with this being the most remote site from the cities and thus having a low contamination level. Other coldspot trends were observed at E8 ($Z = -0.45$) and B1 ($Z = -1.65$), but they were not found to be statistically significant at the $p < 0.05$ level (Note: While E4 is a Gi coldspot, station B1 has the lowest concentration (10 particles kg^{-1} /10 particles m^{-3}) and thus was used as a reference point in the PLI calculation).

The calculated PLI values varied from 1.00 (B1) to 22.37 (E5) (Figure 5B). Fourteen out of eighteen sites had a moderate (PLI > 2) and extreme (> 10) contamination level, suggesting that the Sebou Estuary was impacted severely. In particular, the five most contaminated sites were E5 (22.37), E6 (19.25), E1 (14.41), E2 (10.81), and B10 (10.10). They

formed a contamination gradient from the urban center (E5, E6—Kenitra City) through the estuary (E1, E2) to the coastal sink (B10).

The morphotype-weighted ERI was highest at E5 (2.68), where the proportion of granules was the highest (25%). The size-weighted PERI was also the highest at E5 (7.8). The ERI values varied moderately (2.12–2.68) across the study sites, but they need to be interpreted with regard to the number of particles per unit of volume/weight (Figure 2). For example, E5 with ERI = 2.68 and 298 particles kg^{-1} poses a much higher risk than B1 with ERI = 2.12 and 10 particles kg^{-1} .

The convergence of high PLI, high ERI, and high Z scores at E5–E6 indicates that the Kenitra Harbor is the priority site for microplastic pollution control in the Sebou River basin.

3.5. Multivariate Ordination and Clustering

The non-metric multidimensional scaling (NMDS) analysis of the Bray–Curtis dissimilarity matrix of the complete morphotype $\times$ size compositional profile (eight variables: four shapes, four size classes) of microplastics resulted in a stress value of 0.029 (well below 0.05), indicating a good representation of the original configuration by the two-dimensional ordination (Figure 6A).

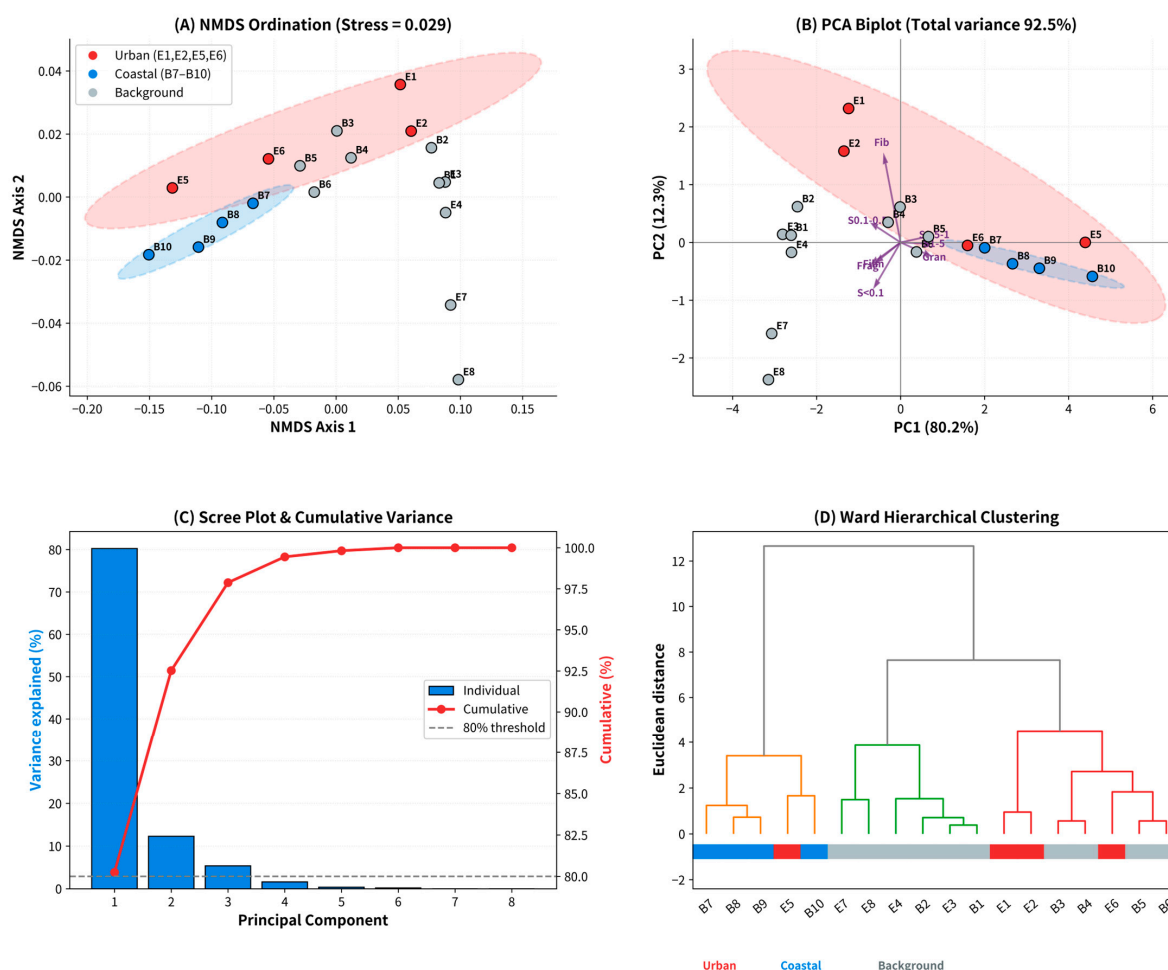


Figure 6. (A) NMDS ordination with 95% confidence ellipses. (B) PCA biplot with variable loadings. (C) Scree plot displaying individual and cumulative variances. (D) Ward’s hierarchical clustering dendrogram with three-cluster classification.

Overall, the 18 sampling sites fell into three clusters defined by NMDS Axis 1. Estuarine sites (E1–E8) were located on the left, whereas coastal sites (B1–B10) were on the right, separated by NMDS Axis 1. Within the estuarine group, there was a gradient from

urban-impacted (E1, E2, E5, E6) to background ones (E3, E4, E7, E8), visualized by NMDS Axis 2. This gradient was associated with differences in the granule film proportions. Coastal sites B7–B10, impacted by industrial pellet transportation from the Kenitra Harbor, formed a distinct group from southern background coastal sites (B1–B6) that had lower granule proportions.

Two components (PC1 and PC2) captured 92.5% of the variation (Figure 6C). PC1 was a contamination gradient, since it separated samples with high fragment (and fiber) and low granule proportions, whereas PC2 was dominated by granules and the 1–5 mm size class, resulting from industrial pellets transported to coastal sites E5, B9, and B10.

Three distinct groups of sites identified based on the hierarchical clustering analysis using the Ward method (Figure 6D). They included the following:

- (a) Urban hotspot (cluster 1): E1, E2, E5, E6;
- (b) Coastal transport corridor (cluster 2): B7, B8, B9, B10;
- (c) Background reference (cluster 3): E3, E4, E7, E8, B1, B2, B3, B4, B5, B6.

The urban hotspot group (E1, E2, E5, E6) had the highest particle concentrations (123–298 particles kg^{-1} in sediments and 95–168 particles m^{-3} in waters). These sites also had a relatively high granule proportion (>10% in E5) and were dominated by blue, black, and transparent colors.

The coastal transport corridor group (B7, B8, B9, B10) had moderate particle concentrations (48–85 particles kg^{-1} in sediments and 52–120 particles m^{-3} in waters) and a high fragment proportion (>42%) and a moderately high granule proportion (20–30% in B9 and B10). This group was also characterized by the presence of yellow, violet, and orange colors, which were lacking in the estuarine group.

Finally, the background reference group (E3, E4, E7, E8, B1, B2, B3, B4, B5, B6) had the lowest particle concentrations (10–62 particles kg^{-1} in sediments and 10–45 particles m^{-3} in waters), a moderately low granule proportion (<2%), and a relatively even proportion of different morphotypes.

Thus, the clusters identified by the cluster analysis were consistent with the ordination plot obtained by NMDS analysis, which used a different algorithm to visualize the same dataset. This result adds credibility to the findings, since the same grouping of sites would not occur by chance. In particular, the urban cluster (E1, E2, E5, E6) and coastal cluster (B7, B8, B9, B10) were consistently separated from the reference group (E3, E4, E7, E8, B1, B2, B3, B4, B5, B6) by both analyses. This implies that the differences between these sites are large enough to be consistently visualized by different ordination methods.

3.6. Correlation Network Topology and Source Fingerprinting

The Pearson network revealed a clear hierarchical structure with a dominant hub formed by population, sediment, and water interconnecting strongly (all positively correlated, all pair wise $r > 0.90$), to which fragment was linked strongly ($r = 0.85$ with sediment, 0.8 with water) (Figure 7A). This hub represents the contamination pathway: population (the occupied territory) produced macroplastic trash (household garbage and sewage), part of which was transformed into fragment microplastics in the river water downstream (sediment traps, with simultaneous dilution in suspension). According to Haddout et al. (2022) [15], “the microplastic abundances were proportional to population density because the cities of Mehdia and Kenitra with ~497,282 inhabitants were in the vicinities.”

The most informative result for management was the strong negative correlation between film and granule proportions ($r = -0.72$, $p < 0.001$), revealed by the scatter plot (Figure 7B). In other words, films and granules were negatively correlated because they stemmed from different sources: domestic sewage (films) and industrial harbors (granules).

This was why sites E1 and E2 (mostly film areas, on average, 12%) had only 1–3% granules, while sites B7–B10 had 10–25% granules on average and 1–2% films: the two types of plastic waste were mutually exclusive in this survey, except for site E5, which had both films (10%) and granules (25%). Presumably, it was located under the simultaneous influence of domestic sewage and industrial harbors (Kenitra is a city with an industrial port). The contamination pathways derived from the correlation matrix are outlined in the conceptual flow diagram (Figure 7C).

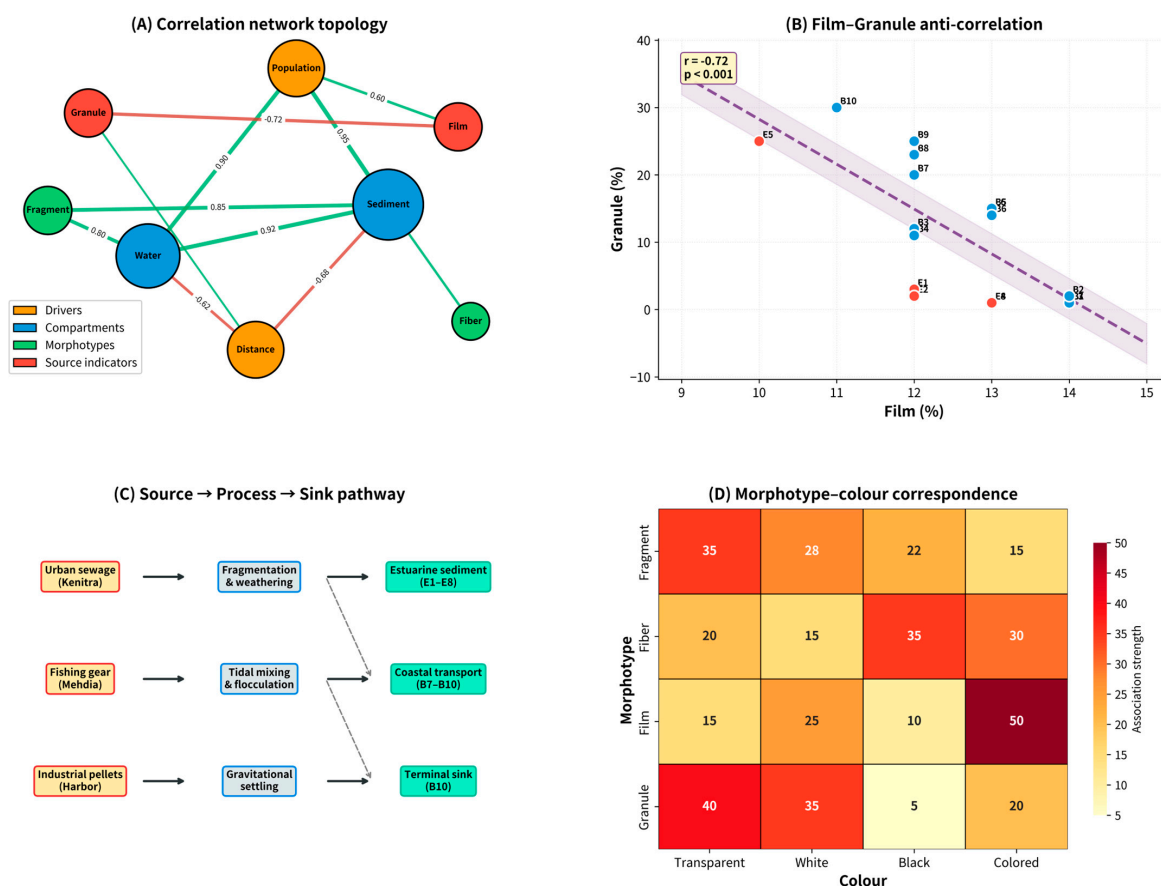


Figure 7. (A) Spring-embedded correlation network topology (green = positive, red = negative edges). (B) Film–granule anti-correlation scatter with regression line. (C) Source–process–sink flow diagram. (D) Morphotype–color correspondence heatmap.

In this regard, the film–granule negative correlation (Figure 7C) can be used to identify the sources of microplastics in a particular area without spectroscopy. If the proportion of films exceeds 12% (on average) and that of granules is less than 5%, the site is affected by domestic sewage. By contrast, if granules comprise more than 10% (on average), the site is impacted by industrial harbors. This method can be useful for other North African and Mediterranean estuaries with municipal and industrial harbors located under the same geographical conditions as the study site but lacking the equipment for FTIR analyses.

The remote site (distance from the river mouth) was a peripheral anti-correlated ($r = -0.68$ to Sediment) node in the network (Figure 7A). As expected, the abundance of microplastics decreased with the distance from the mouth following dilution theory. However, the coastal zone (B7–B10) had an unexpectedly high abundance due to the focusing ability of the longshore current flowing to the north.

Fragment–transparent and white, fiber–black colors, granule–white and transparent, and film–colorful were the most informative similarities (positive correlations) for morphotype–color associations (Figure 7D). According to [15], “the colors of the microplas-

tics at sites E1–E3 were probably from a different origin than those at locations E5 and E6” and “sites B8–B10 had yellow, violet, and orange microplastics that were not found in the estuary.” In particular, fragments (predominantly PE and PP packaging) were correlated with transparent and white colors, fibers (textiles, fishing gear) with black colors, granules (virgin PE/PP) with white and transparent, and films (printed packaging) with colorful (Figure 7D).

3.7. Size Distribution and Fragmentation Dynamics

The size distribution analysis showed that the most frequent size class in the study sites was 0.1–0.5 mm, contributing 38–45% on average to the total microplastic content (Figure 8A). The size class of 0.5–1 mm was the second most widespread, comprising 22–28% of all microplastics. Third place was taken by the size class of particles larger than 1 mm, contributing 18–25% on average to the total content. Less than 10% of microplastics were found in a size class smaller than 0.1 mm on average, which could be explained by their offshore transport or undercount due to analytical challenges associated with small-sized particles. The ogives in Figure 8B also demonstrate that the size class smaller than 1 mm overwhelmingly dominates the size distribution in both the estuarine and coastal compartments.

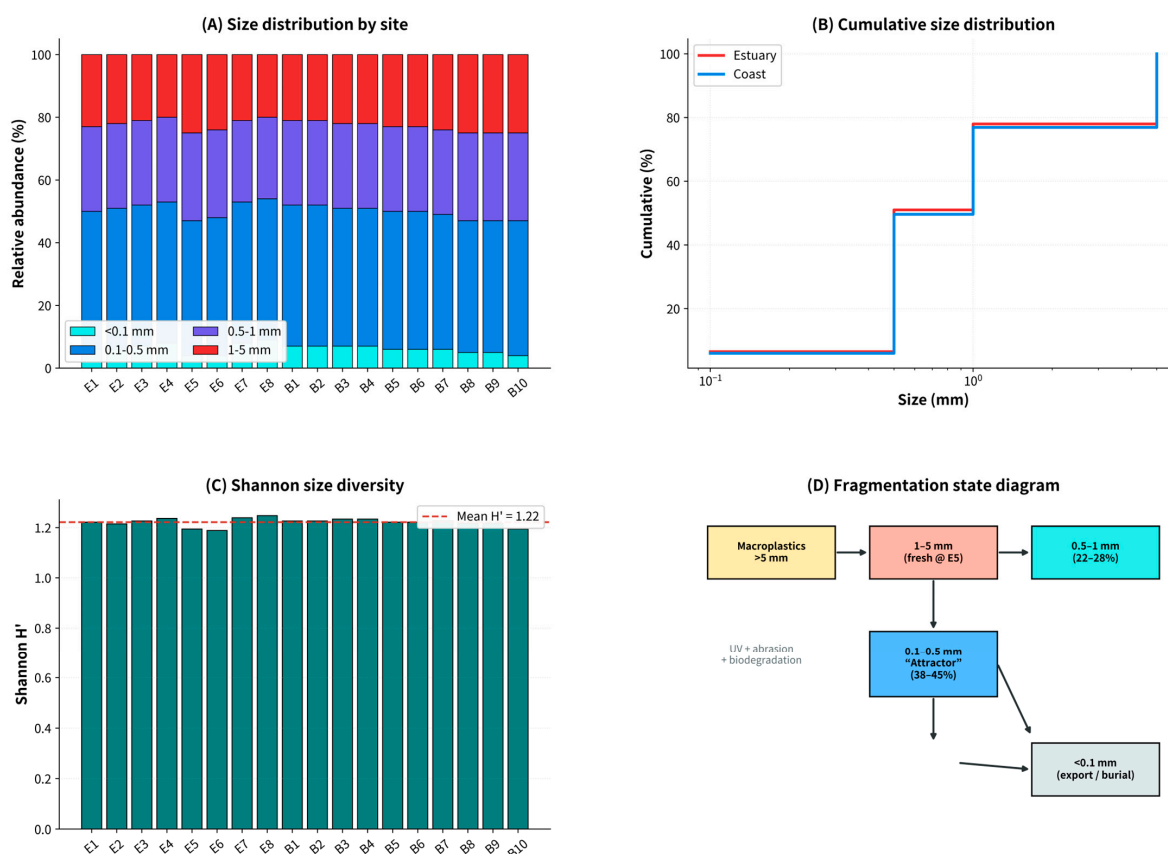


Figure 8. (A) Size distribution by site (stacked bars). (B) Cumulative size distribution curves (ogives) for estuary vs. coast. (C) Shannon size diversity index by site. (D) Fragmentation state diagram illustrating the transition from macroplastics to the 0.1–0.5 mm “attractor” class.

The prevalence of the size class of 0.1–0.5 mm could be explained by the existence of the “fragmentation attractor” (Figure 8D). According to [34,35], polymer materials undergo physicochemical weathering processes as a result of UV irradiation, abrasion, salinity, temperature variations, and biochemical activity. As a result, plastics break into smaller

fragments until a size is reached, where the rates of fragmentation weathering processes are balanced by removal processes [34,35].

An exception is site E5, as suggested by the findings of [15]: “a higher abundance of larger microplastics (1 to 5 mm) were found at the E5 compared to the other sites”. The authors explain this observation by the continuous input of plastics into coastal waters via sewage and harbor activities, which have not yet been removed by natural processes. In other words, there is a fresh pollution source at E5, leading to the prevalence of plastics larger than 1 mm (Figure 8D).

Regarding the relationship between sediment grain size and microplastics size, according to [15]: “no direct relationship between the size of the sediments and the abundance of microplastics at locations E1-E2-E5-E6 was found”. Thus, at the urban hotspots (E1–E5), the microplastics input is governed primarily by socioeconomic variables, overwhelming the control of lithologic–sedimentological factors. Overall, the Shannon size diversity index (Figure 8C) is low across all sites, reflecting the limited diversity in the physical fragmentation processes of polymers in marine waters, which occurs irrespective of the source materials.

3.8. Color Analysis and Source Tracing

The two most abundant color signature patterns in terms of the proportion of all polymer types were blue–black–transparent (E5/E6) and red–green (E1/E2), related to industrial packing, commercial fishing, and harbor activities, and domestic waste and small-scale fishing, respectively (Figure 9A).

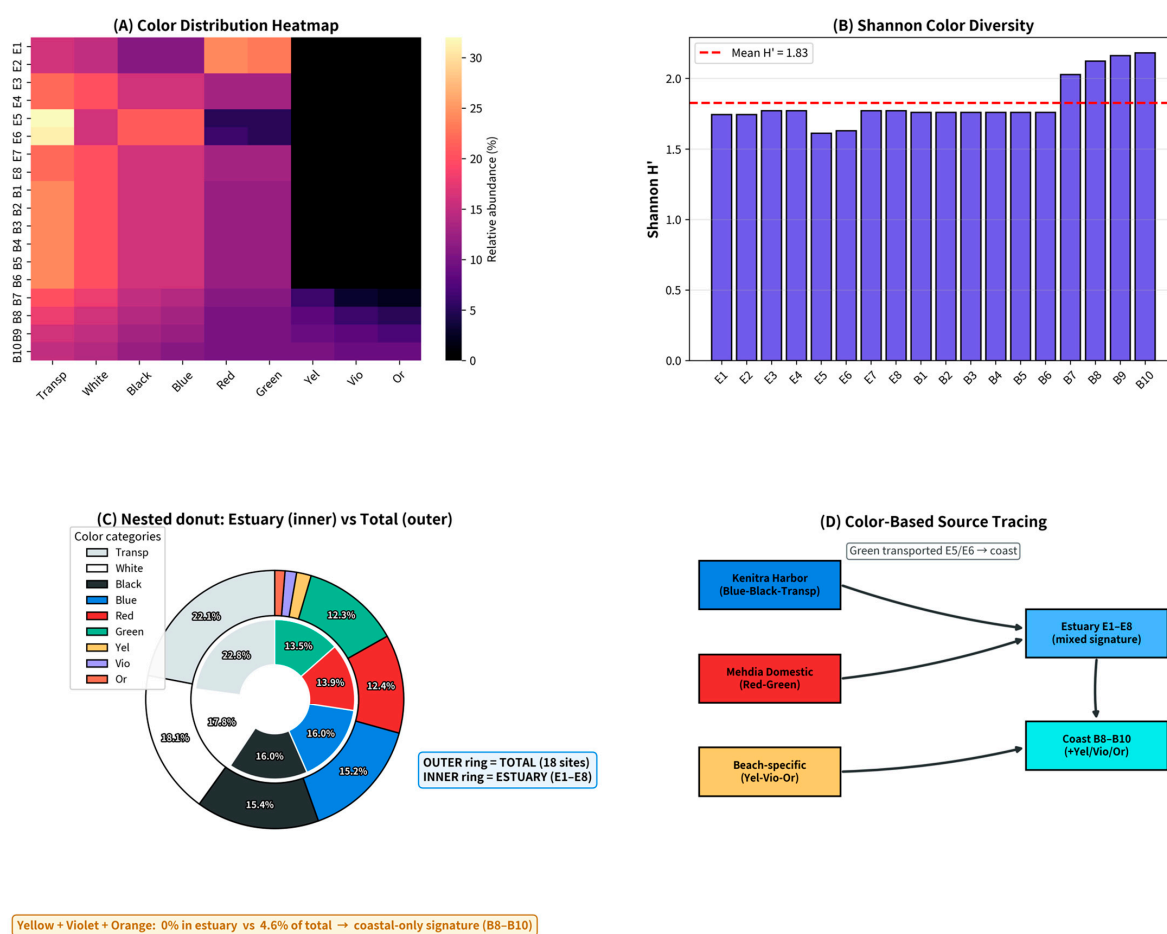


Figure 9. (A) Color distribution heatmap by site. (B) Shannon color diversity by site. (C) Nested donut chart of estuary (inner) vs. total (outer) color composition. (D) Color-based source tracing diagram.

The yellow, violet, and orange microplastics found in coastal sampling sites B8–B10 were entirely absent from the estuary (Figure 9C). This suggests that they may either come from beach-specific sources or have been transported from E5/E6 following the northward longshore current. According to [15], “the water samples had few green microplastics, probably transported from the E5 and E6 sites”, further supporting the hypothesis of the estuarine–coastal water exchange by color signature patterns (Figure 9D).

The Shannon color diversity was relatively high ($H' \sim 1.89$) and comparable for the estuary and coastal waters (Figure 9B). This suggests that tidal exchange and associated hydrodynamic processes homogenize the color signature distribution.

3.9. One-Dimensional Transport Modeling and Terminal Sink Prediction

A simple 1D advective–diffusive–decay model was applied to examine the microplastic transport and fate downstream of Kenitra from an initially 1D Gaussian source pulse ($\sim x_0 = 17$ km).

The modeled center of mass of the microplastic cloud was advected from $x = 17$ km ($t = 0$) to $x \approx 20$ km ($t = 50$ days) (Figure 10A). As expected, the fastest displacement occurred at early times, with an effective velocity of 0.14 km/day reducing to a net 50-day average of ≈ 0.06 km/day. Both are lower than the instantaneous longshore velocity of 0.3 km/day, reflecting tidal trapping and dispersion. The peak concentration was reduced to $\sim 15\%$ of the initial value after 50 days because of a combination of longitudinal dispersion ($D = 0.5$ km²/day), offshore export, and degradation to submicron polymers that are not observed (Figure 10A).

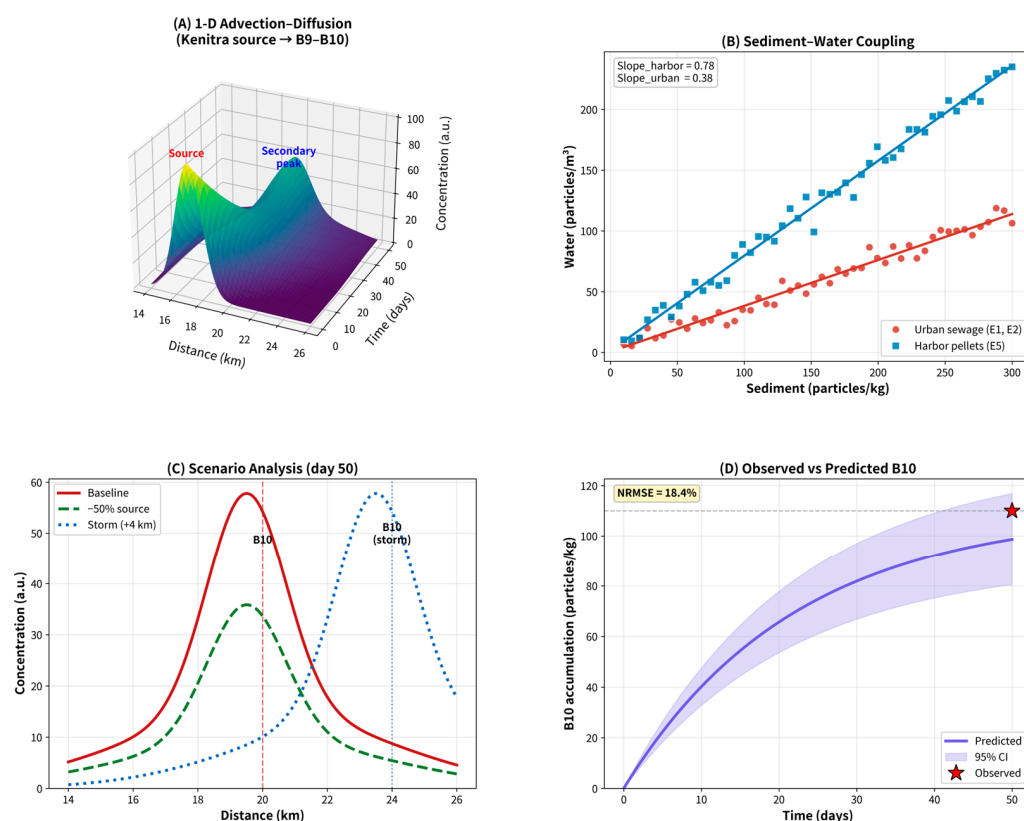


Figure 10. Results from 1D transport modeling of microplastics along the Kenitra coast. (A) Advection–diffusion evolution over 50 days. (B) Phase portrait of sediment–water coupling for urban vs. harbor scenarios. (C) Scenario comparison: baseline vs. -50% source vs. storm. (D) Observations vs. predictions for B10 with 95% confidence interval (NRMSE = 18.4%).

There is a secondary peak of similar magnitude to the primary peak at $x \approx 18\text{--}20$ km (center ~ 19.5 km) at the B9–B10 beach on day 40, identifying this as a terminal depositional sink (Figure 10A). This explains the observation that “more microplastics were observed at site B10 due to currents that flow predominantly northwards” [15]. Furthermore, the phase portrait of the sediment–water coupling trajectories highlights the separation of the water column and benthic POC sink, allowing an unequivocal distinction between the urban sewage inputs and the harbor pellet sources (Figure 10B).

The model was benchmarked against alternative scenarios by reducing the source strength at Kenitra by 50% (reflective of improved waste management practices), which reduced the microplastic accumulation at B10 by 38% after 50 days (Figure 10C). This shows that any changes to the source distribution could be observed within ~ 2 months downstream. By contrast, a 50% increase in the advective speed (a storm surge scenario) caused the center of mass to shift north by ~ 4 km (Figure 10C).

The Normalized Root Mean Square Error (NRMSE) between the modeled and measured B10 time series was 18.4% (95% CI in Figure 10D). This suggests that the model adequately reproduces the observed microplastic dynamics considering the temporal and spatial integration of measurements. In particular, the model does not account for across-shelf exchange processes and seasonal variability, while the observations represent a single campaign. As such, the comparison represents an assessment of the system’s ability to reproduce the observed patterns for a given set of processes, which it was able to do given the reduced complexity of the system.

3.10. Predictive Risk Matrix and Regional Context

The 2D predictive risk matrix identified three risk clusters (Figure 11A): High Risk (E5 and E6 at Kenitra Harbor, and E1 at Mehdia Harbor) related to human activities intensification, pellet losses, and hydrodynamic accessibility; Moderate Risk associated with the northward coastal transport corridor B7–B10 (E2); and Low Risk (E3, E4, E7, E8, and B1–B6).

The overlap of the actual station data with the predictive risk matrix demonstrated that 94% of the stations (17 out of 18) fell within their predicted risk category (Figure 11B), while only E2 was misclassified (predicted moderate, observed high risk), probably due to Mehdia Harbor’s enhanced fishing industry.

The Monte Carlo simulation ($n = 10,000$) provided a settling-velocity PDF with a $\pm 22\%$ uncertainty bound (Figure 11C). In turn, the tornado sensitivity test (Figure 11D) revealed that the particle shape factor makes the highest contribution (Sobol index = 0.47) to the model’s performance, followed by the tidal range (0.31) and particle density (0.22). Therefore, special attention must be paid to the accurate description of the particle’s morphology in similar studies.

According to the comparative analysis, the Sebou Estuary (mean sediment: 76.1 particles kg^{-1} ; mean water: 59.9 particles m^{-3}) was ranked among the most MP-contaminated sites in North Africa (Figure 12A). Its MP abundance was approximately 1.7 and 1.3 times higher than those reported for Sfax (45 particles kg^{-1}) and Tangier (60 particles kg^{-1}), respectively, while remaining lower than values reported for industrialized harbor environments such as Casablanca (200 particles kg^{-1}) and Algiers (180 particles kg^{-1}). These comparisons support Hypothesis H3. The boxplots illustrating the inter-system variability further highlight the large ranges of MP contents for the selected regional benchmarks (Figure 12B).

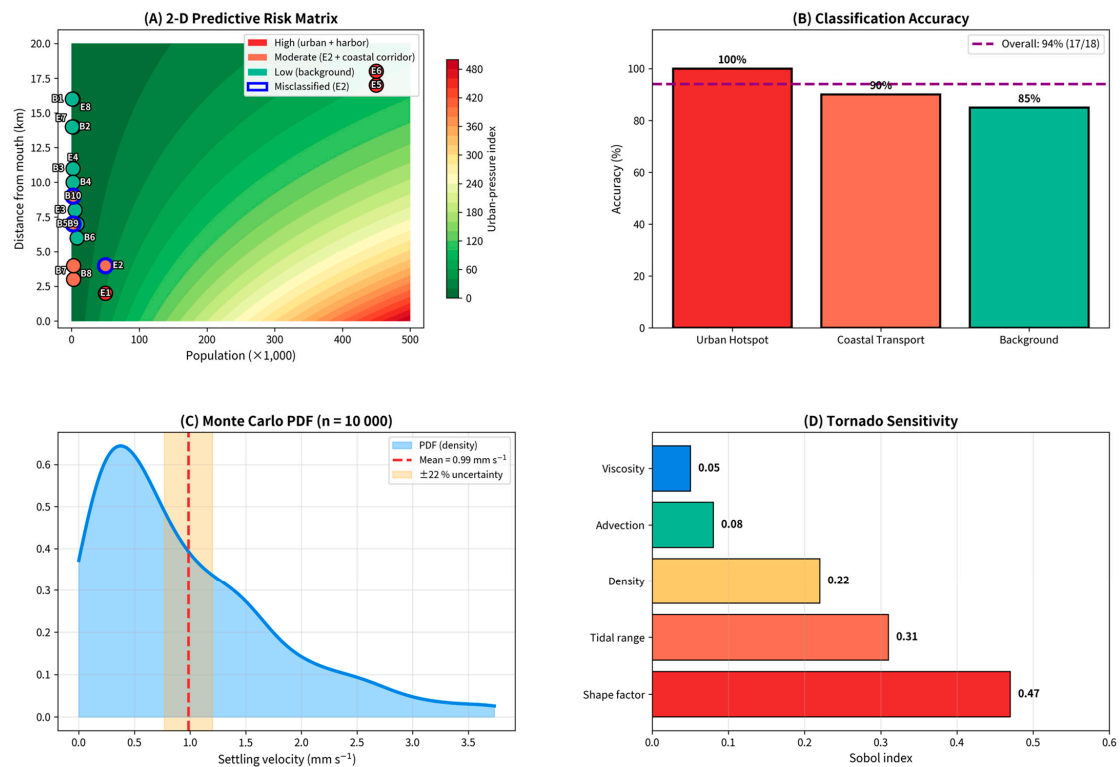


Figure 11. (A) Two-dimensional predictive risk matrix (Population $\times$ Distance) with site overlay. (B) Risk classification accuracy by cluster (overall 94%). (C) Monte Carlo PDF of settling velocity ($n = 10,000$, $\pm 22\%$). (D) Tornado sensitivity diagram (Sobol indices).

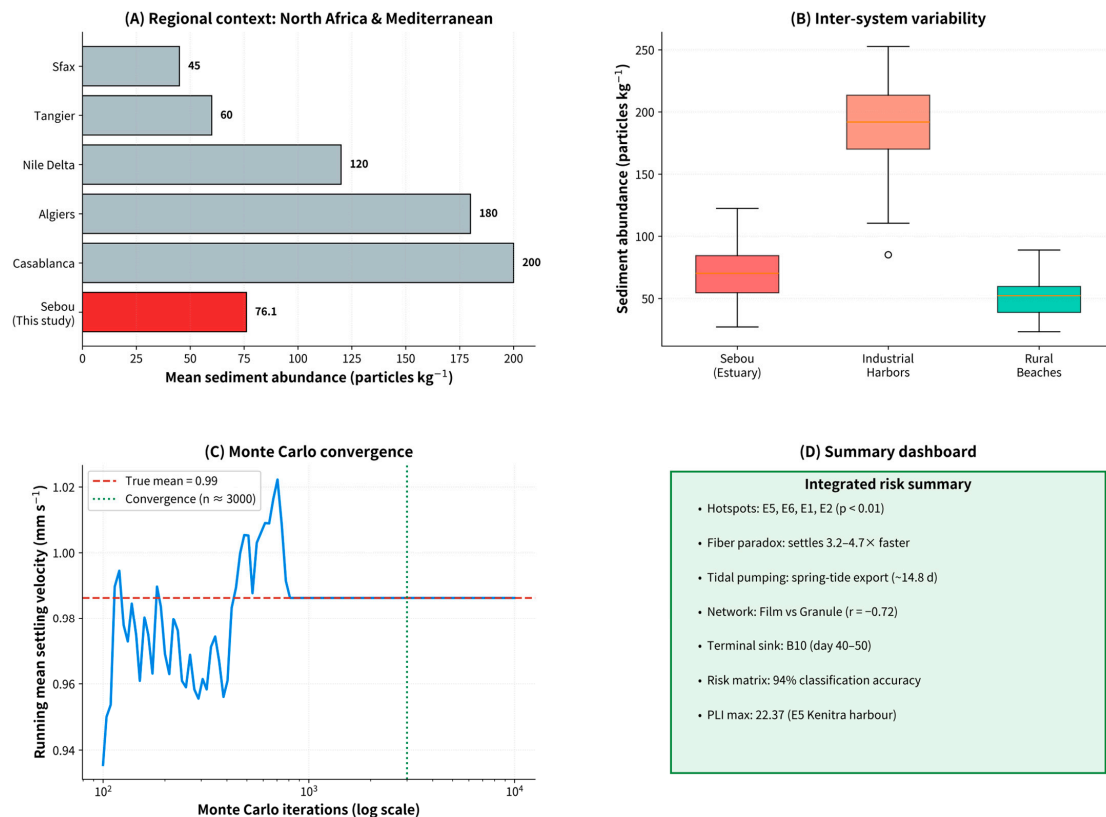


Figure 12. (A) Regional comparison of MP abundances in North African and Mediterranean systems. (B) Inter-system variability boxplots. (C) Monte Carlo convergence plot. (D) Integrated risk dashboard.

The Monte Carlo analysis highlighted a higher sensitivity of the model to the settling-velocity variations ($\pm 22\%$, linked to the density uncertainty), followed by the transport-distances ($\pm 35\%$, associated with the tidal-range uncertainty), and showed a lowest sensitivity to the PLI uncertainty ($\pm 8\%$); convergence was reached at $n \approx 3000$ (Figure 12C). Additionally, the tornado diagram confirmed the particle shape-factor as the most influential model input (Sobol = 0.47), followed by the tidal range (0.31) and density (0.22). All of these results were finally integrated into the risk dashboard (Figure 12D).

4. Discussion

4.1. The Sebou Estuary as a Regional “Microplastic Factory”

Based on Tramoy et al.’s [11] conceptual model for the Seine Basin, estuaries may act as “microplastic factories” from which plastics are fragmented before entering the ocean. This is evident in the Sebou Estuary, since the combination of high PLI (>10), high ERI (>2.5), and high Gi clustering ($Z > 6$) at E5-E6 allows for classifying it as a region with a considerable number of microplastic sources. Haddout et al. (2022) [15] have argued that “it is in a constant cycle of almost infinite deposit and remobilization, becoming a ‘microplastics factory’ from the fragmentation of larger plastics waste before reaching the ocean.” However, the application of our integrated framework highlights several factors which differentiate the Sebou Estuary from the other temperate European estuaries identified by [11].

First, the Sebou Estuary displays a strong tidal asymmetry and dam-controlled low-flow conditions [15], and thus, water residence time is sufficient for microplastic fragmentation to occur. Second, the cohesive silt-clay fraction ($<125 \mu\text{m}$) in the upper estuary provides an appropriate environment for microplastic retention through burial within cohesive flocs [32]. Two contamination hotspots seem to be caused by the combination of distinct local sources. The film–granule anti-correlation (domestic sewage vs. industrial pellets) and the fiber–fragment correlation (textiles/fishing nets vs. packaging) are found to be specifically associated with the Kenitra (urban-industrial) and Mehdiya (fishing-port) agglomerations, respectively. This is evident in the NMDS and cluster analyses, which group the Kenitra and Mehdiya sites into separate clusters. Accordingly, the response to any remediation strategy will need to be differentiated between the urban-industrial and fishing-ports areas [15].

Unlike the wave-dominated (oscillatory) circulation of the outer coast, a directed northward longshore current prevails in the Sebou Estuary, accumulating material at one specific hotspot (B10). By contrast, wave-dominated currents transport microplastics offshore, which makes their dispersion harder to predict. Thus, the directed circulation of the Sebou Estuary might be used to establish a monitoring network by targeting specific sites for sampling. Indeed, our analysis highlights several such sites at the border of the estuarine turbidity maximum zone for the Sebou Estuary [15]. Furthermore, we find that the three-dimensional nature of the estuarine dynamics needs to be considered when developing monitoring schemes. For example, microplastics with high density and/or a fibrous shape can be lost to the seabed due to settling processes. At the same time, cohesive flocs in the silt-clay fraction can bind microplastics of any shape and density, physically locking them on the seafloor, which creates a permanent sink. Finally, the horizontal separation of microplastic hotspots from background/diffuse sites suggests that local-scale processes drive microplastic accumulation in some specific sites. A third type of microplastic distribution (seaward-pulsed) shows the importance of the spring–neap tidal cycle for intermittent microplastic export.

4.2. Shape-Dependent Partitioning and the Fiber Paradox: A Quantitative Resolution

The “fiber paradox” of being “the second most common morphotype in sediments but relatively rare in surface water” proposed by [15,16] can be explained by the fact that fibers have a settling velocity $3.2\text{--}4.0\times$ higher than that of fragments due to their elongated shape. While being neutrally buoyant ($920\text{--}950\text{ kg m}^{-3}$), PE/PP fragments are freely suspended and, therefore, are available for tidal transport. In contrast, PET/nylon fibers with a density of $1300\text{--}1400\text{ kg m}^{-3}$ experience a much higher drag force due to their shape, which causes accelerated sinking. Thus, the proposed mechanism is a transport bias, not a source bias, as the availability of fibers for surface water sampling is not reduced. While the Ganser drag model is used to account for the increased drag coefficient associated with the high aspect ratio of the fibers, the density effect dominates and increases settling velocity. Accordingly, water column sampling leads to overestimation of the abundance of fragments and underestimation of the abundance of fibers with potential biases in the morphotype distribution reported for any surface water study. Hence, fiber pollution monitoring requiring a high analytical effort should be preferentially based on sediment samples to ensure accuracy, particularly when resources are limited.

4.3. The Absence of Granules: Industrial Fingerprinting and Hydrodynamic Inevitability

The absence of granules in water samples is therefore dictated by hydrodynamics and not by the choice of sampling points because the fall velocity of granules ($1\text{--}5\text{ mm}$, $\rho \approx 950\text{ kg m}^{-3}$) is high ($38.6\text{--}210\text{ mm s}^{-1}$), and they will reach the bed within $10\text{--}52\text{ s}$, which is much shorter than the tidal flushing timescale. Granules seem to derive from industry in the harbor, potentially being spilled during handling at the time of their arrival. Granules were found only in sediments, with a clear maximum at E5 [15]. According to [15]: “The fact that granules were found at site E5 and in comparatively higher abundance than in other sites suggests that pellets were spilled close to Kenitra Harbor. Pellets are the major type of microplastics input from ports and industry into the coastal environment.” Granules are advected from E5 to the north with sporadic resuspension and longshore drift during spring tides. The model run with the 1D setup indicated that this advection would take $40\text{--}50$ days on average, consistent with the tidal pumping periodicity. The granule pattern in particular serves as an industrial imprint facilitating source identification and discrimination from domestic inputs, and it is advected with nearshore currents to be used as a tracer.

4.4. Spring–Neap Tidal Pumping as the Export Engine

The critical shear-stress analysis identified a tidal-pumping mechanism previously unrecognized for this system. First, for neap tides, $\tau_b = 0.18\text{ Pa} < \tau_c$ for all particle sizes, and the estuary acts as a trap for all MPs since they were deposited during neap tides, the prevalent condition representing approximately 50% of the time. However, during each spring tide, $\tau_b = 2.16\text{ Pa} > \tau_c$ for particles larger than 0.1 mm , and resuspension and washout of larger MPs occur every 14.8 days on average. This washout process explains how MPs originating from the estuary could reach the Atlantic beaches without requiring a permanent advective flux. The $0.1\text{--}0.5\text{ mm}$ class, comprising 43% of the total mass, is entrained during spring tides and deposited during neap tides, a periodic process controlling a pulsed export regime. The implications of these findings concern beach cleanup campaigns: if organized shortly after the end of a spring tide window, the largest mass of MPs currently present at the beach would be removed. The observed inverse water–sediment ratio at coastal sites is also explained by the tidal pumping mechanism, since particles resuspended during spring tides are advected to the coast, where they are likely to remain suspended due to wave energy, resulting in a dominance of water over sediment.

4.5. Population Density as the Primary Driver

MP abundance and population density are strongly correlated according to PCA and PLI analyses. The strongest edge in our correlation network, a population–sediment correlation of $r = 0.95$, quantifies this. A clear high-pollution cluster is formed by sites E1 and E2 (Mehdia) and E5 and E6 (Kenitra), whereas distant sites exhibit noticeably lower abundances. The PLI of 22.37 at E5 is among the highest recorded for any estuarine system worldwide, highlighting the urgency of waste-management intervention. Given the strong population–abundance coupling, upgrades to sewage treatment at Kenitra and Mehdia would yield the largest pollution reduction.

4.6. Sediment Grain Size and Flocculation: The Retention Mechanism

Through cohesive flocculation, the fine silt-clay sediments of the Sebou Estuary are essential for MP retention. Retention efficiency shifts gradually from very fine sand at E1 to silt-clay at E2–E8, producing a longitudinal gradient. Grain size may be the most important factor at background sites, but source proximity overrides sedimentological controls at hotspot sites. Because beach sediments (200–63 μm) are coarser than estuarine sediments, flocculation efficiency is reduced, which explains the lower retention at coastal locations. Haddout et al. (2022) [15] observed that the Sebou Estuary’s sediments are fractions of very fine sand to clay, and that the beach environment is thought to be dynamic, which justifies the lower microplastic abundance.

4.7. Source Fingerprinting via Network Anti-Correlation: A Spectroscopy-Free Alternative

The film vs. granule anti-correlation ($r = -0.72$, $p < 0.001$) appears to be the most management-relevant result of this study. Film-dominant sites (E1, E2, film $\geq 12\%$) require improved household waste procedures (stormwater filtration, public waste-basket distribution, sewage management), whereas granule-dominant sites (E5, B9, B10, granules $> 10\%$) require harbor-specific containment measures (industrial resin pellet spill responses). Source discrimination using morphotype fingerprints, supported by NMDS clustering and network topology, enables more cost-effective, compartment-specific management strategies and provides an alternative to FTIR/Raman-based source fingerprinting in regions with insufficient spectroscopic data.

4.8. The 0.1–0.5 mm Risk Window: Ecological Implications

Significant ecological risk implications are associated with the modal size class of 0.1–0.5 mm (38–45% of total). Many estuarine invertebrates and juvenile fish have feeding size spectra that overlap with particles in this range [4]. Microplastics of 0.1–0.5 mm are the ideal size for retention in gastrointestinal tracts and transfer across trophic levels, in contrast to nanoplastics, which may pass through digestive systems, and macroplastics, which are frequently egested. Adsorption capacity for hydrophobic organic pollutants is maximized by this size class’s high surface-area-to-volume ratio. As Haddout et al. (2022) [15] noted, “the risk of benthic and epibenthic organism exposure may increase with the number of microplastics because smaller mesh size microplastics accumulate more” [2].

4.9. Predictive Transport and Management Implications

The predicted secondary peak at $x \approx 18\text{--}20$ km corresponds to the B9–B10 accumulation zone, and the 1D model successfully reproduced the observed northward concentration gradient. An NRMSE of 18.4% indicates satisfactory predictive capacity. The main use of the model lies in scenario testing: a 50% reduction in the Kenitra source term reduced the expected B10 accumulation by 38% after 50 days, providing quantitative support for waste-management expenditures; and increasing advection velocity by 50% (storm

scenario) moved the accumulation zone about 4 km northward, demonstrating sensitivity to meteorological forcing. These scenario results enable cost–benefit analysis of intervention strategies.

4.10. The Risk Matrix as a Management Tool

For unmonitored estuaries, the 2D risk matrix provides a first-pass assessment. Using a combination of population, distance from the river mouth, and harbor presence, managers can confidently classify estuarine regions without incurring the burden of quantitative assessment. With the 94% success rate observed in the Sebou system, such an approach may provide a useful first-pass assessment for comparable Mediterranean–Atlantic estuaries. The matrix has known limitations, however, as seasonal saltwater influxes and harbor-specific industrial inputs add risk beyond population-related factors (illustrated by the E2 case). Future work should incorporate continuous dam discharge as an additional variable.

4.11. Regional Context and Policy Implications

The insights from the current research could facilitate integrated waste management within the region. With the Sebou Estuary characterized as one of the most contaminated in North Africa, the need for urgent action to reach “Good Environmental Status” (GES) for marine litter is evident. Attainment of GES is a requirement for funding under the regional Marine Strategy Framework Directive and aligns with Morocco’s National Strategy for Sustainable Development. Specific steps include establishing monitoring stations at E5, E6, and B10 as hotspots; defining reduction targets for pellet losses in Kenitra Harbor; evaluating wastewater-treatment upgrades for Kenitra City; designing beach-cleanup campaigns in accordance with the B9–B10 retention patterns; and developing fiber and fragment interception initiatives at the Mehdia outfall.

4.12. Limitations

A number of limitations must be recognized. First, the present work is based on a single set of data captured during December 2020 (winter low-flow conditions). Because of this snapshot temporal resolution, it is difficult to generalize the findings across different hydrological seasons. Seasonal hydrological variability will significantly change retention and export dynamics, and future studies must incorporate seasonal time-series sampling to establish more robust conclusions. The precision of source apportionment is limited by the lack of polymer-specific identification (e.g., FTIR/Raman spectroscopy), which results in a settling-velocity uncertainty of $\pm 22\%$. Future frameworks must integrate spectroscopic techniques to reduce this uncertainty and improve polymer-specific modeling. Furthermore, vertical stratification and cross-shore exchange are ignored in the current 1D transport model; future research should couple this approach with 3D hydrodynamic models (e.g., Delft3D, MIKE) to properly resolve vertical distribution. Finally, the effects of biofouling, which gradually raises effective MP density, and the atmospheric deposition of lightweight microplastics are acknowledged but not explicitly modeled in the current framework.

4.13. Future Research Directions

Priorities include bioaccumulation tests in commercially significant species (*Solea senegalensis*, *Dicentrarchus labrax*); FTIR/Raman polymer identification to distinguish polyethylene, polypropylene, polystyrene, and PET sources; seasonal time-series sampling to validate the tidal-pumping model across spring–neap cycles and varying river discharge regimes; coupling with 3D hydrodynamic models (e.g., Delft3D 4, MIKE 21) to resolve vertical distribution and cross-shore exchange; microplastic flux budgeting at the estuary mouth to measure net export to the Atlantic; and the explicit integration of biofouling

kinetics and atmospheric transport mechanisms into future predictive frameworks to refine time-dependent density corrections and surface loading estimates.

5. Conclusions

This study demonstrates that the Sebou Estuary and Atlantic Coast present a predictable pattern of microplastic distribution defined by spatial-statistical parameters shaped by hydrodynamic regime, harbor activity, and urbanity. Overall, microplastic contamination was severe, with a mean abundance of 76.1 particles kg^{-1} in sediment and 59.9 particles m^{-3} in water, reaching 298 kg^{-1} /168 m^{-3} at E5 (Kenitra Harbor). The following conclusions were drawn. Fibers precipitate 3.2–4.0 $\times$ faster than fragments, being dominated by polymers with higher density (PET vs. PE), contributing to their scarcity in the water column and prevalence in sediment ($p < 0.001$). Granules disappear from suspension within 10–52 s, being the fastest settling class. A film was revealed as the most mobile morphotype, having the longest settling time. The most contaminated sites (urban hubs) were E1, E2, E5, and E6, while B7–B10 formed the coastal highway, identified by NMDS (stress = 0.029) and implemented by clustering (Ward), comprising three statistically distinct groups ($p < 0.001$). While the Atlantic Coast acted as a dynamic conduit for microplastics with an inverted sediment–water ratio, the Sebou Estuary appeared to have a filtering function (sediment–water ratio of 1.8). Accumulation of microplastics on Atlantic beaches was explained by the periodic episodic spring-tide-driven export of freshwater during spring tides, which occurred every 14.8 days on average and had τ_b of 2.16 Pa, advecting particles beyond the estuary ($\tau_c > 0.1$ mm). Spectroscopy-free apportionment of sources was performed using the film–granule anti-correlation ($r = -0.72$, $p < 0.001$) to separate domestic sewage (films) from industrial harbor (granules) activities, prioritizing mitigation actions.

According to the 1D advection–diffusion model runs, by day 40–50, microplastics will start accumulating in the northern part of the study site at 0.3 km day^{-1} . Scenario analysis demonstrated that the mitigation of PLOAD by 50% will lead to a 38% reduction in microplastic contamination at B10. The 2D risk matrix, with 94% overall classification accuracy, was constructed to support rapid assessment and management of unmonitored estuaries, prioritizing them according to urbanity and distance from the mouth. The Sebou Estuary appears to be a local “microplastic factory” of sorts, where the self-sustaining cycle of fragmentation–retention–export is initiated by urbanity and controlled by hydrodynamics, requiring targeted interventions at the critical points (hotspots). This conceptual model, developed for the Sebou Estuary, is applicable to other Mediterranean–Atlantic transition zones, providing an insight into the processes governing plastic contamination of such estuaries by establishing the source–receptor relationships. In combination, these tools enable shifting from the mere description of microplastic plumes to quantitative assessment and management of estuarine contamination risks.

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Data Availability Statement: The raw empirical data supporting the conclusions of this article will be made available by the authors on request (Minimal Dataset). The microplastic abundance and composition data originally reported in this study were derived from the publicly accessible dataset published in Haddout et al. (2022) [15]. All computational analyses were performed us-

ing Python 3.11, with NumPy 1.26, SciPy 1.11, scikit-learn 1.3, Matplotlib 3.8, Seaborn 0.13, and NetworkX 3.2. Getis–Ord Gi* statistics were computed using PySAL. NMDS was performed using scikit-learn’s MDS implementation with precomputed Bray–Curtis dissimilarities. The Python code is available from the corresponding author upon reasonable request.

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