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Theme

Spatial Distribution and Characterization of Microplastics along the Central Algerian Coast

Presented by:

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Defended on **29/06/2026**, before the following jury:

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بسم الله الرحمن الرحيم

﴿وَآخِرُ دَعْوَاهُمْ أَنِ الْحَمْدُ لِلَّهِ رَبِّ الْعَالَمِينَ﴾

This thesis marks far more than the completion of a degree. It embodies years of perseverance, growth, challenges overcome, and lessons learned. Every page reflects moments of doubt, determination, sacrifice, and gratitude. Above all, it is a testament to the strength, guidance, and countless blessings that accompanied me throughout this journey. It is dedicated to the child I once was, who dreamed of understanding the world, to the person I am today, still learning to navigate it, and to all those—present or absent—whose kindness, advice, and quiet encouragement have left an enduring mark on my path.

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Abstract

Microplastic (MP) pollution is a growing environmental concern due to its persistence, widespread distribution, and potential to transport hazardous contaminants. This study assessed the abundance, distribution, and characteristics of MPs in beach and marine sediments and in surface seawater, in two coastal sites along the western Algiers coastline: Sidi Fredj and El Djamila Beach (Ain Benian). Sampling followed standardized TSG-ML/MSFD protocols. MPs were extracted by density separation using saturated NaCl solution (1.20 g cm^{-3}), visually characterized under stereomicroscopy, and chemically identified by FTIR spectroscopy, and analyzed for associated trace metals by ICP-OES. MP particles were detected in all matrices and sampling stations. Beach sediment abundances were similar at both sites (709.81 ± 295.74 and 697.98 ± 175.73 MPs kg^{-1}). In contrast, marine sediment contamination was higher at Sidi Fredj (773.39 ± 679 MPs kg^{-1}) and seawater contamination was greater at Ain Benian (612 MPs L^{-1}). Fibers dominated all matrices, followed by fragments, foams, and films, while pellets were absent. Particles smaller than 1 mm and transparent particles were most abundant. FTIR analysis identified seven polymer types, including PP, PE, HDPE, PET, PVC, PS, and PA, with polyethylene (PE) being the most prevalent. ICP-OES analysis of trace metals revealed higher concentrations in MPs than in beach sediment for Pb, Zn, and Cu, confirming the role of MPs as active metal vectors, while Cr showed no matrix-specific enrichment and Ni reflected predominantly geogenic sediment inputs. The results indicate predominantly secondary MP contamination from urban, domestic, textile, and fishing-related sources, and emphasize MPs' role as carriers of trace metals, highlighting the need for long-term monitoring and improved coastal waste management along the central Algerian coast.

Keywords: Microplastics, sediment, seawater, Algerian coast, marine pollution, trace metal elements

Résumé:

La pollution par les microplastiques (MP) constitue un problème environnemental croissant en raison de leur persistance, de leur large répartition et de leur capacité à transporter des contaminants dangereux. Cette étude a évalué l'abondance, la répartition et les caractéristiques des MP dans les sédiments des plages et des fonds marins, ainsi que dans les eaux de surface, sur deux sites côtiers situés le long de la côte ouest d'Alger : Sidi Fredj et la plage d'El Djamila (Ain Benian). L'échantillonnage a été réalisé selon les protocoles normalisés TSG-ML/MSFD. Les microplastiques ont été extraits par séparation densimétrique à l'aide d'une solution saturée de NaCl (1,20 g cm^{-3}), caractérisés visuellement au stéréomicroscope, identifiés chimiquement par spectroscopie FTIR et analysés pour déterminer les métaux traces associés par ICP-OES. Des particules de microplastiques ont été détectées dans toutes les matrices et à toutes les stations d'échantillonnage. Les concentrations dans les sédiments de plage étaient similaires sur les deux sites (709,81 ± 295,74 et 697,98 ± 175,73 microparticules par kg). En revanche, la contamination des sédiments marins était plus élevée à Sidi Fredj (773,39 ± 679 microparticules par kg) et celle de l'eau de mer était plus importante à Ain Benian (612 microparticules par L). Les fibres dominaient toutes les matrices, suivies des fragments, des mousses et des films, tandis que les granulés étaient absents. Les particules de moins de 1 mm et les particules transparentes étaient les plus abondantes. L'analyse FTIR a permis d'identifier sept types de polymères, notamment le PP, le PE, le PEHD, le PET, le PVC, le PS et le PA, le polyéthylène (PE) étant le plus répandu. . L'analyse par ICP-OES des métaux à l'état de traces a révélé des concentrations plus élevées de Pb, de Zn et de Cu dans les microparticules que dans les sédiments de plage, confirmant ainsi le rôle des microparticules en tant que vecteurs actifs de métaux, tandis que le Cr ne présentait aucun enrichissement spécifique à la matrice et que le Ni reflétait principalement des apports sédimentaires d'origine géogénique. Ces résultats indiquent une contamination par les microplastiques essentiellement secondaire, provenant de sources urbaines, domestiques, textiles et liées à la pêche, et soulignent le rôle des microplastiques en tant que vecteurs de métaux traces, mettant en évidence la nécessité d'une surveillance à long terme et d'une meilleure gestion des déchets côtiers le long de la côte centrale algérienne.

Mots-clés: Microplastiques, sédiments, eau de mer, côte Algérienne, pollution marine, métaux de traces

الملخص:

مصدر قلق بيئي متزايد بسبب ثباتها، وانتشارها الواسع، وقدرتها على نقل الملوثات (MP) يُعد التلوث بالجسيمات البلاستيكية الدقيقة الخطرة. وقد قيمت هذه الدراسة وفرة الجسيمات البلاستيكية الدقيقة وتوزيعها وخصائصها في الرواسب الشاطئية والبحرية وفي مياه البحر السطحية، في موقعين ساحليين على طول الساحل الغربي للجزائر: شاطئ سيدي فريدج وشاطئ الجميلة (عين بنيان). وقد اتبعت عملية أخذ الموحدة. وتم استخلاص الجسيمات البلاستيكية الدقيقة عن طريق الفصل الكثافي باستخدام TSG-ML/MSFD العينات بروتوكولات محلول كلوريد الصوديوم المشبع (1.20 غ/سم³)، وتم تحديد خصائصها بصريًا تحت المجهر المجسم، وتم تحديدها كيميائيًا بواسطة التحليل وتم الكشف عن جسيمات البلاستيك الدقيقة .ICP-OES كما تم تحليلها للكشف عن المعادن النزرة المرتبطة بها بواسطة ،FTIR الطيفي ± في جميع المصفوفات ومحطات أخذ العينات. كانت وفرة الرواسب الشاطئية متشابهة في كلا الموقعين (709.81 ± 295.74 و697.98 جسيمًا من الجسيمات الدقيقة لكل كيلوغرام). في المقابل، كان تلوث الرواسب البحرية أعلى في سيدي فريدج (773.39 ± 679 175.73 جسيمًا ميكرويًّا لكل كيلوغرام) وكان تلوث مياه البحر أكبر في عين بنيان (612 جسيمًا ميكرويًّا لكل لتر). سيطرت الألياف على جميع المصفوفات، تليها الشظايا والرغوة والأغشية، بينما لم تكن الحبيبات موجودة. وكانت الجسيمات الأصغر من 1 مم والجسيمات الشفافة هي وكان البولي ،PAو PSو PVCو PETو HDPEو PEو PP سبعة أنواع من البوليمرات، بما في ذلك FTIR الأكثر وفرة. حدد تحليل للمعادن النزرة عن تركيزات أعلى في الجسيمات البلاستيكية مقارنة برواسب ICP-OES هو الأكثر انتشارًا. وكشف تحليل (PE) إيثيلين مما يؤكد دور الجسيمات البلاستيكية كناقلات نشطة للمعادن، ، في حين ،(Cu) والنحاس (Zn) والزنك (Pb) الشاطئ بالنسبة للرصاص بشكل أساسي مدخلات الرواسب ذات الأصل (Ni) أي إثراء مرتبط بنوع معين من المصفوفة، بينما عكس النيكل (Cr) لم يُظهر الكروم يأتي في الغالب من مصادر ثانوية، مثل المصادر الحضرية (MP) الجيولوجي. تشير النتائج إلى أن التلوث بالجسيمات البلاستيكية الدقيقة والمنزلية والمنسوجات والمصايد السمكية، وتؤكد على دور الجسيمات البلاستيكية الدقيقة كناقلات للعناصر المعدنية النزرة، مما يسلط الضوء على الحاجة إلى الرصد طويل الأمد وتحسين إدارة النفايات الساحلية على طول الساحل الجزائري الأوسط.

الكلمات المفتاحية: الجسيمات البلاستيكية الدقيقة، الرواسب، مياه البحر، الساحل الجزائري، التلوث البحري، العناصر المعدنية النزرة

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List of abbreviations

ABS	Acrylonitrile Butadiene Styrene
ASA	Acrylonitrile Styrene Acrylate
HDPE	High-Density Polyethylene
LDPE	Low-Density Polyethylene
MP(s)	Microplastic(s)
NP(s)	Nanoplastic(s)
PA	Polyamide (Nylon)
PAN	Polyacrylonitrile
PBAT	Polybutylene Adipate Terephthalate
PC	Polycarbonate
PE	Polyethylene
PES	Polyester
PET	Polyethylene Terephthalate
PLA	Polylactic Acid
PMMA	Polymethyl Methacrylate
POM	Polyoxymethylene
PP	Polypropylene
PS	Polystyrene
PTFE	Polytetrafluoroethylene
PUR	Polyurethane
PVC	Polyvinyl Chloride
ALDFG	Abandoned, Lost, or Discarded Fishing Gear
AAS	Atomic Absorption Spectrometry
FTIR	Fourier Transform Infrared Spectroscopy
LOI	Loss on Ignition
Py-GC/MS	Pyrolysis – Gas Chromatography – Mass Spectrometry
SCS	Size and Color Sorting
SEM	Scanning Electron Microscopy
UPW	Ultrapure Water
MARPOL	International Convention for the Prevention of Pollution from Ships
MSFD	Marine Strategy Framework Directive (EU)
PIC	Prior Informed Consent
UNEA	United Nations Environment Assembly
EPR	Extended Producer Responsibility
AFNOR	Association Française de Normalisation (French Standards Association)

AND	Agence Nationale des Déchets (National Waste Agency)
Eco-Jem	Système public de récupération des emballages (Public Packaging Recovery System)
EUROMAP	European Machinery Association for Plastics and Rubber Industries
IMO	International Maritime Organization
IUCN	International Union for Conservation of Nature
GESAMP	Group of Experts on the Scientific Aspects of Marine Environmental Protection
TSG-ML	Technical Subgroup on Marine Litter
IMAP	Integrated Monitoring and Assessment Programme
INC	Intergovernmental Negotiating Committee
IUPAC	International Union of Pure and Applied Chemistry
NOAA	National Oceanic and Atmospheric Administration
OECD	Organization for Economic Co-operation and Development
UNEP	United Nations Environment Programme
UNEP/MAP	United Nations Environment Programme / Mediterranean Action Plan
UNCTAD	United Nations Conference on Trade and Development
MENA	Middle East and North Africa
APEC	Asia-Pacific Economic Cooperation
OM	Organic Matter
POPs	Persistent Organic Pollutants
Cd	Cadmium
Cu	Cuivre (Copper)
Cr	Crome (Chromium)
Ni	Nickel
Pb	Plomb (Lead)
Zn	Zinc
d.w.	Dry Weight
g	Gram(s)
kg	Kilogram(s)
kt	Kiloton(s)
L	Liter(s)
mL	Milliliter(s)
mg	Milligram(s)
mm	Millimeter(s)
Mt	Million Ton(s) / Megaton(s)
µm	Micrometer(s)
CO_2-eq	Carbon Dioxide Equivalent

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Introduction

Plastic pollution has emerged as one of the most pervasive and persistent environmental challenges of the 21st century. Global plastic production surged from 2 Mt in 1950 to over 460 Mt annually by 2019, with projections estimating output could reach 1,231 Mt by 2060 under business-as-usual scenarios (Dokl et al., 2024; Geyer et al., 2017; OECD, 2022). In 2024 alone, an estimated 220 Mt of plastic waste were generated, of which one-third was mismanaged and entered the natural environment (Geyer et al., 2017; PlasticsEurope, 2024). Approximately 70–80% of this mismanaged waste originates from land-based sources and reaches marine ecosystems through rivers, stormwater drainage, and coastal littering (Jambeck et al., 2015; MacLeod et al., 2021). Once released into the marine environment, plastic debris fragments progressively through environmental processes into microplastics (MPs), which are defined as solid synthetic polymer particles ranging from 0.001 to 5 mm in their largest dimension (GESAMP, 2015; Hartmann et al., 2019), are now recognized as ubiquitous contaminants across all marine compartments, from surface waters and beach sediments to deep-sea sediments and polar regions (Andrady, 2011; Eriksen et al., 2014; Hale et al., 2020; Stubbins et al., 2021). Their small size, persistence, and capacity to adsorb trace metals and organic pollutants contribute to their potential impacts on marine ecosystems and human health (Law & Thompson, 2014; Yang et al., 2021). The Mediterranean Sea is considered one of the regions most affected by microplastic pollution worldwide. As a semi-enclosed basin with heavy maritime traffic, dense coastal populations, and limited exchange with the Atlantic Ocean, it exhibits particularly high levels of plastic accumulation. Surface concentrations have been reported to reach up to 1.25 million particles per km², exceeding those observed in several major oceanic gyres, including the North Pacific Subtropical Gyre (Cózar et al., 2014).

Located on the southern shore of the Mediterranean, Algeria faces growing challenges associated with plastic waste management. The country is among the largest plastic consumers in Africa, with imports of approximately 1,095 kt in 2024 (EUROMAP, 2025), and a recycling rate of only 5–10%, and faces severe solid waste management challenges (Plast Alger, 2025). Consequently, plastic debris accounts for nearly 87% of marine litter on Algerian beaches, with single-use plastic items dominant (AND, 2020).

Despite increasing concern regarding microplastic pollution, studies along the Algerian coastline remain limited. Existing studies have primarily focused on the eastern and western coastal regions, documenting the occurrence and distribution of MPs in selected beaches and marine environments (Amenouche et al., 2026; Bentaallah et al., 2024; Grini et al., 2024; Taïbi et al., 2021). In contrast, the central Algerian coast, particularly the highly urbanized Algiers metropolitan area, remains poorly studied despite being subjected to intense anthropogenic pressures arising from urbanization, industrial and port activities, domestic wastewater discharges, fishing activities, and seasonal tourism.

Furthermore, comprehensive investigations integrating multiple environmental matrices and combining microplastic characterization with sedimentological and trace metal analyses remain scarce. Such multidisciplinary approaches are essential for understanding the sources, distribution patterns, and environmental implications of coastal contamination.

To address these knowledge gaps, the present study provides a multi-matrix assessment of MP contamination at two representative sandy beaches along the western Algiers coastline: the West Beach of Sidi Fredj and El Djamila Beach (Ain Benian).

The aims of this study are to:

- Analyze the abundance, distribution, and physical characteristics of MPs, including their shapes, colors, and sizes, as well as their chemical properties (polymer type) across all matrices and sites.
- Determine trace metal concentrations in sediment and selected MP particles.
- Compare the contamination levels observed at both sites with published data from the Algerian coast and the wider Mediterranean basin.

The findings provide valuable baseline data on microplastic pollution along the central Algerian coast and contribute to a better understanding of the interactions between plastic debris and associated contaminants in coastal environments. Ultimately, this work aims to support future monitoring programs and inform sustainable coastal management strategies designed to mitigate environmental and ecological risks associated with plastic pollution.

To present the research coherently and progressively, this manuscript is structured into four main chapters:

Chapter I: Literature Review presents the general context of plastic pollution in coastal environments. It reviews the sources, distribution, environmental behavior, and ecological impacts of microplastics, occurrence in the Mediterranean Sea, as well as their interactions with other contaminants and the current regulatory framework.

Chapter II: Materials and Methods describes the study area, sampling strategy, and analytical procedures employed throughout the investigation. Particular attention is given to sediment characterization, microplastic extraction, identification and quantification protocols, polymer characterization, and trace metal analysis.

Chapter III: Results and Discussion presents and interprets the results obtained from the different environmental matrices. The findings are compared with published data from Algeria and other Mediterranean regions, allowing an assessment of contamination levels, spatial patterns, and the factors influencing pollutant distribution.

Chapter IV: Conclusions and Perspectives summarizes the main findings of the study and highlights their environmental significance. Recommendations for future monitoring programs and coastal management are proposed, together with research perspectives focusing on the interactions between microplastics and trace metals in coastal ecosystems.

Chapter I: Literature review

PART 1, Overview and concepts of microplastics

I. Fundamentals of plastics

I.1. Definitions of marine litter

Several definitions of marine litter can be found in the scientific literature, UNEP regional seas reports, and governmental documents. This diversity reflects the multidimensional complexity of the phenomenon, which encompasses both environmental, legislative, and socio-economic aspects.

Origin of the term

Etymologically, the word *"déchet"* (waste), which appeared in the 14th century, comes from the verb *"déchoir"*, which means the decrease in value of a material or object to the point where it becomes unusable in a given place and time (Pichat, 1995).

An integrated definition of marine debris, based on a synthesis of various interpretations from the literature and regulatory frameworks: "Marine debris, also known as marine litter, refers to any solid, persistent material of varying size, manufactured or processed, of anthropogenic origin, discarded, abandoned or lost, directly or indirectly, intentionally or accidentally, in the marine or coastal environment. This debris may originate from land-based activities via rivers, drainage systems, sewers, or winds, or result from activities at sea, such as shipping, aquaculture, or offshore platforms.

Although it includes a wide variety of materials, such as plastics, metals, glass, processed wood, textiles, rubber, paper, etc., plastics are currently the most widespread and most concerning category of marine litter worldwide.

I.2. Definition and composition of plastics

The term "plastic" comes from the Greek word "plastikos", which means "capable of being molded" or "malleable".

Plastics are defined as non-metallic polymers synthesized by humans (artificial), possessing high molecular weight and consisting of repetitive macromolecules (Bowmer & Kershaw, 2010; Faure et al., 2015; Hess et al., 2006), also known as monomers (Aucher, 2009).

It comprises a vast family of polymer materials with varied properties and structures, among which three main categories are distinguished: thermoplastics, thermosets, and elastomers. The distinction between these polymer categories is based mainly on the polymerization method and the resulting structure formed from monomers (Aucher, 2009). Thermoplastic polymers are the most common, accounting for around 90% of plastic polymer applications (PlasticsEurope, 2015). Predominantly derived from fossil resources (petroleum and natural gas), some plastics can also be produced from renewable raw materials such as cellulose and corn starch (Geyer et al., 2017; Reshmy et al., 2021). According to Andrady (1990), this definition excludes several categories, including inorganic polymers (glass, silicones), low molecular weight polymers that are not solids, as well as natural polymers derived from living organisms, particularly cellulose, natural rubber, and bacterial polyesters. Furthermore, in the case of a composite material, it is not considered plastic if the polymer it contains is a minor, non-essential component of its formulation.

I.3. Plastic life-cycle

Plastics pose a major environmental challenge due to their exponential production growth and widespread effects across all life-cycle stages, resulting in cumulative and systemic impacts on ecosystems and human health. Global plastic production surged from 2 Mt in 1950 to over 460 Mt annually by 2019, with projections estimating output could reach 1,231 Mt by 2060 under business-as-usual scenarios (Dokl et al., 2024; Geyer et al., 2017; OECD, 2022; Sonke et al., 2025). In 2024 alone, approximately 220 Mt of plastic waste were generated, of which one-third was mismanaged and ended up in the natural environment (PlasticsEurope, 2024; UNCTAD, 2025). This production relies predominantly on fossil fuel feedstocks; petroleum (90%) and natural gas (10%), generating 1.7 to 3.5 tons of CO_2-eq per ton produced, with lifecycle emissions projected to rise from 1,781 Mt CO_2-eq in 2015 to 6,500 Mt by 2050, contributing 3-4.5% of global greenhouse gas emissions (Pottinger et al., 2024; Zheng & Suh, 2019). Global waste management reveals critical failures at end-of-life: only 9% of plastic waste is effectively recycled, 46% accumulates in landfills, 17-20% is incinerated, and 22% is mismanaged through improper disposal and littering (Geyer et al., 2017; PlasticsEurope, 2024).

The transition toward circular plastic economies through improved waste collection systems, advanced recycling methods, extended producer responsibility, and enhanced regulatory frameworks represents an urgent global priority (Pottinger et al., 2024; Tang, 2025; Xu, 2025)

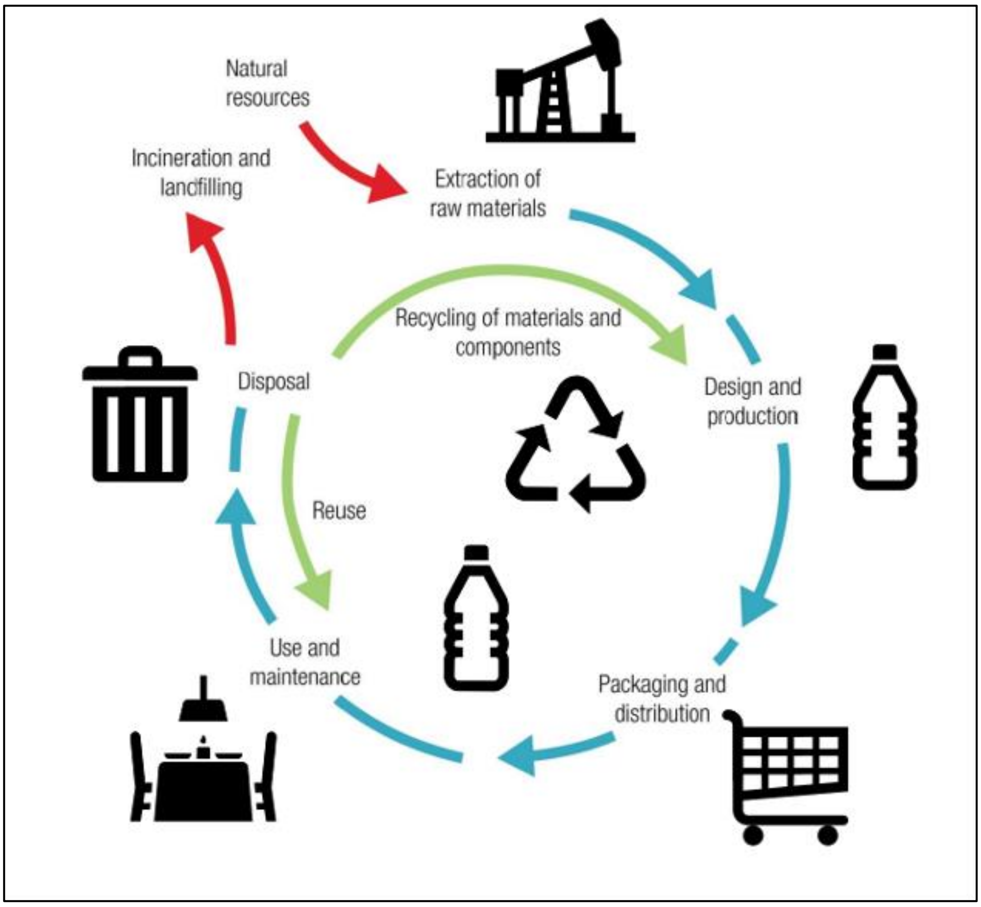


Figure 1: **Plastics life-cycle from resource extraction to end-of-life management** (Life Cycle Initiative. (n.d.))

I.4. Plastic in Algeria

I.4.1. Plastic trade and consumption in Algeria

Algeria has become one of the largest plastic consumers in Africa, with imports of plastic raw materials growing from 0.096 Mt in 1992 to approximately 1,095 kt in 2024, representing an average annual increase of 7.8% since 2007 and positioning the country as the third-largest importer in the Middle East and North Africa (MENA) region, after the UAE and Egypt. From 1992 to 2017, Algeria's cumulative plastic imports reached approximately 13.14 million tons, accounting for 11.2% of Africa's total primary plastic imports (Babayemi et al., 2019; Geyer et al., 2017; Plast Alger, 2024). Simultaneously, Algeria has become Africa's leading importer of plastics processing technology, with imports surging by 34.6% annually from 2020-2024 to €299 million, surpassing Egypt's €220 million (Plast Alger, 2025).

Despite governmental investments aimed at developing domestic production capacity of plastics, projected to reach 416 kt by 2028, and recent regulatory measures targeting single-use plastics, the country faces severe waste management challenges, with recycling rates estimated at only 5-10% and national waste collection rates of merely 5 to 7%, far below comparators such as Tunisia (20%) and South Korea (60%) (Plast Alger, 2025). This inadequate waste infrastructure, combined with annual consumption of around 5.5 billion plastic bags in 2017, results in approximately 0.52 Mt of poorly managed plastic waste each year (AND, 2020; Geyer et al., 2017; Jambeck et al., 2015), with coastal monitoring revealing that plastic debris constitutes 87% of marine litter on Algerian beaches, primarily single-use items (AND, 2020).

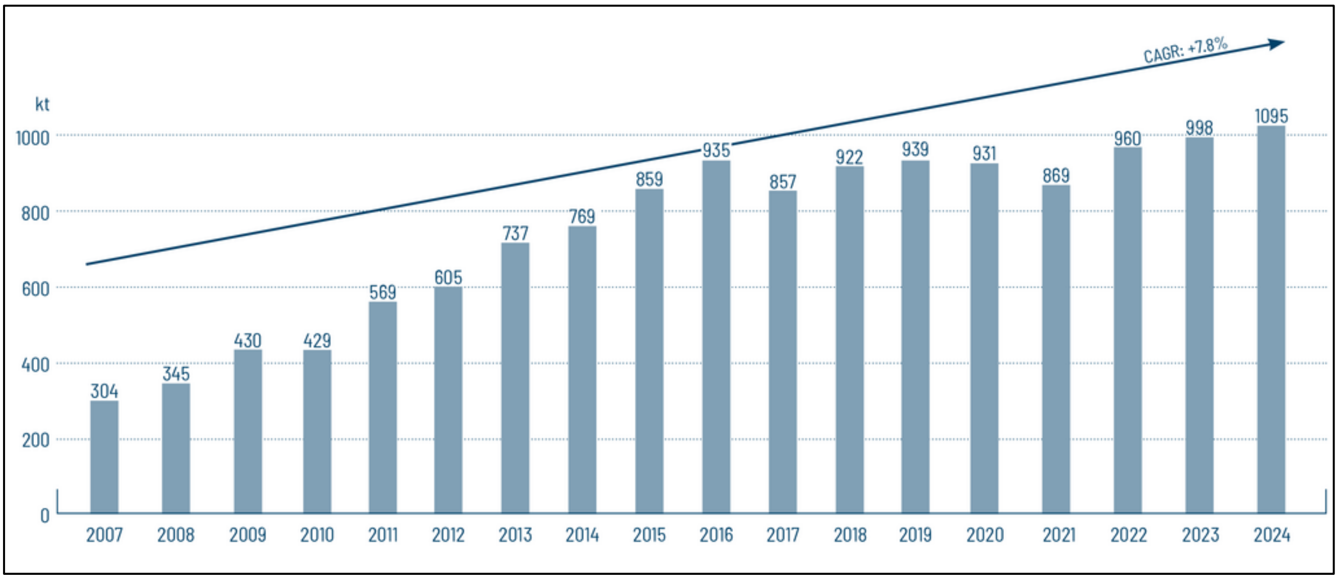


Figure 2: **Trends of plastic raw material imports in Algeria (2007–2024)** (EUROMAP, 2025)

I.4.2. Single-use plastic in Algeria

Single-use plastics constitute a significant fraction of waste streams in Algeria, both in household waste and coastal environments. A national characterization campaign conducted by the National Waste Agency (AND) between April 2018 and March 2019 across three bioclimatic zones (northern, semi-arid, and arid regions), to assess changes in the composition of similar household waste and identify its recovery potential. The study revealed that plastic waste represents 15.31% of municipal solid waste generated annually, corresponding to approximately 2.1 million tons, with LDPE dominating at 8.14% and PET accounting for 3.57% of total waste produced (AND, 2019).

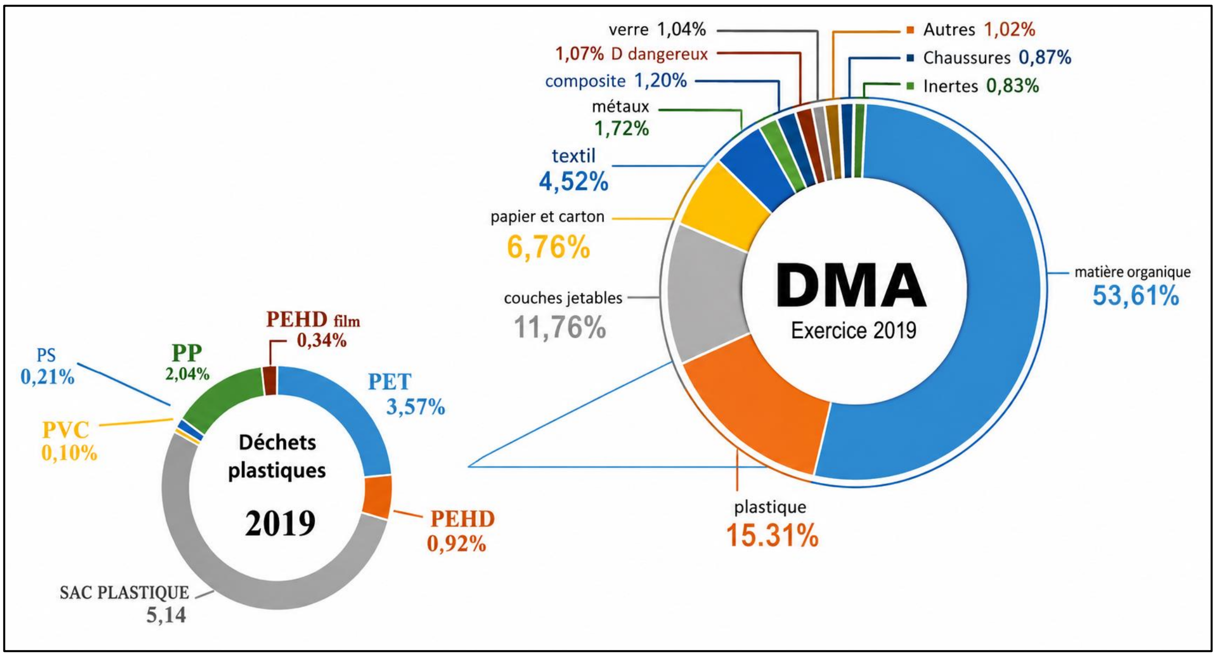


Figure 3: **Composition of plastic waste in Algeria** (AND, 2020)

The situation is even more critical in marine and coastal environments. A seasonal monitoring campaign conducted by the AND in 2020 across three pilot wilayas: Jijel "East", Tipaza "Center", and Ain Temouchent "West", with two beaches per wilaya, as part of the Horizon 2020 program, revealed that plastic debris constitutes between 80-85% of total beach litter (AND, 2020). The dominant items identified include: beverage bottles, caps, plastic bags of various forms, food packaging (biscuits, candies), nets and other fishing gear, drinking straws, disposable tableware, cotton swabs, lollipop sticks, polystyrene and plastic fragments, and cigarette butts, etc. These wastes predominantly originate from tourism activities, inadequate sanitation systems, and waste that has escaped collection and management circuits (AND, 2020) (fig 4).

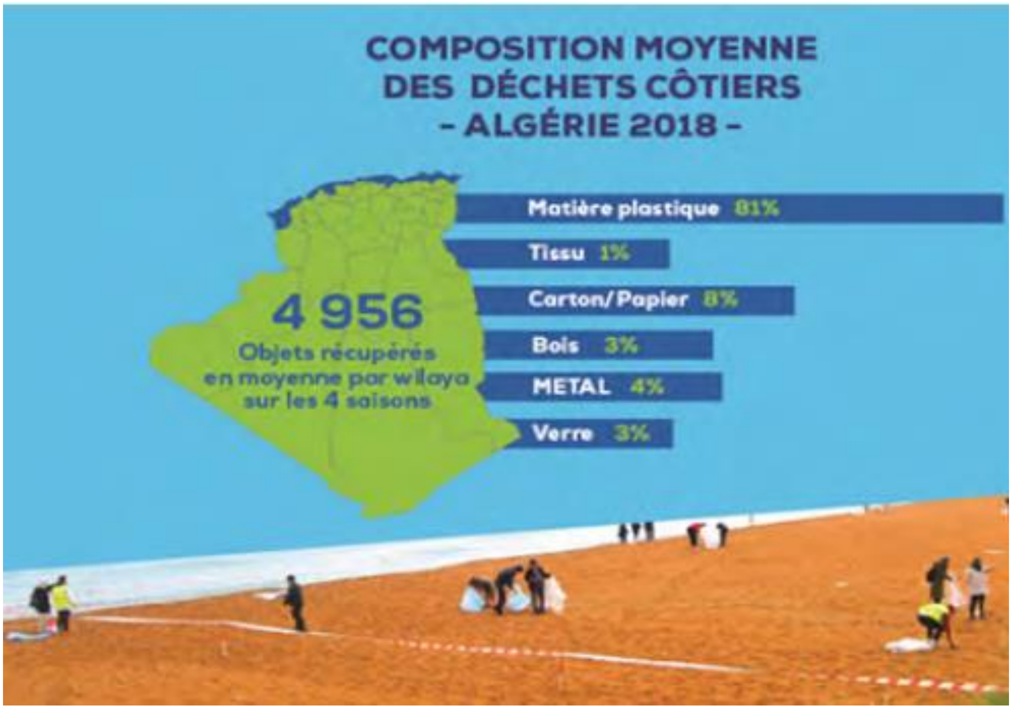


Figure 4: **Average composition of coastal waste** (AND, 2020)

I.5. Mechanisms of plastic degradation in the marine environment

Plastic degradation in marine environments is a complex, multifaceted process involving both abiotic (physicochemical) and biotic (biological) mechanisms that work in tandem to break down polymeric materials across multiple size scales, from macroplastics (>5mm) to microplastics (<5mm), and eventually nanoplastics (<1µm) (Andrady, 2011; Gewert et al., 2015; Viel et al., 2023). Unlike naturally biodegradable materials, plastics are highly resistant to complete mineralization and instead undergo gradual fragmentation, increasing their environmental persistence, surface area, and bioavailability (Chamas et al., 2020).

Degradation refers to the breakdown of a polymer due to environmental factors, including thermal energy, UV radiation, biological activity, and mechanical action, resulting in altered chemical and physical properties.
In the marine environment, degradation occurs through several pathways, which are generally classified according to the causative agent (Andrady, 2011):

1. Photodegradation: degradation by sunlight, UV-induced oxidation.
2. Mechanical degradation: physical fragmentation.
3. Hydrolytic degradation: chemical reaction with water.
4. Thermo-oxidative degradation: slow oxidative breakdown at moderate temperatures.
5. Thermal degradation: action of high temperature. (Not an environmental degradation mechanism)
6. Biodegradation: the action of living organisms, primarily microbes.

Table 1 provides a comparative overview of the main plastic degradation pathways in the marine environment, outlining their mechanisms, key influencing factors, and typical outcomes. It highlights the complexity of these processes and the interaction between abiotic and biotic factors that ultimately govern the formation of microplastics.

Table 1: Degradation pathways of plastics in marine environments: mechanisms, influencing factors, and typical outcomes (Brožová et al., 2025).

Degradation Pathway	Mechanism	Key Influencing Factors	Typical Outcomes/Limitations
Photodegradation	*UV radiation breaks polymer bonds, causing surface oxidation*	*Light intensity & spectrum, exposure duration, water depth, additives (UV stabilizers)*	*Surface embrittlement, fragmentation, enhanced microbial colonization; limited in turbid/deep waters*
Thermo-oxidation	*Heat and oxygen induce chain scission*	*Temperature, oxygen availability, and material type*	*Accelerated embrittlement under warm, oxygen-rich conditions; limited relevance in cold/deep waters*
Hydrolysis	*Cleavage of bonds by reaction with water molecules*	*Polymer chemistry (esters, amides), pH, temperature*	*Significant for biodegradable polyesters (e.g., PLA, PBAT); negligible for polyolefins*
Mechanical Fragmentation	*Physical abrasion, turbulence, sediment grinding*	*Wave action, sediment load, flow conditions*	*Rapid size reduction into microplastics; does not mineralize material*

Biodegradation	*Microbial enzymes metabolize polymer chains*	*Polymer chemistry, biofilm formation, nutrient availability, and temperature*	*Effective mainly for biodegradable polymers; slow for conventional plastics*

The persistence of plastics in the aquatic environment remains highly uncertain (Chamas et al., 2020; World Economic Forum, 2018). A general overview of approximate degradation times for common plastic items is presented in Table 2. However, these values should be treated with caution, for example, as photodegradable plastics do not undergo photodegradation; instead, they disintegrate into smaller particles, many of which may be invisible to the naked eye.

Table 2: Estimated degradation times of common plastic items in the marine environment (Chamas et al., 2020; World Economic Forum, 2018).

Plastic Item	***Polymer Type***	***Estimated Degradation/Persistence Time***
Plastic bag (single-use)	LDPE/HDPE	10–20 years
Plastic bottle	PET	450 years
	HDPE	58–1200 years
Foam plastic products	PS	50+ years
Disposable diaper	PP, PE	50-100 years (450 years noted)
Fishing line (monofilament)	Nylon (PA)	600 years
Fishing net	Nylon (PA), PE	600+ years
Cigarette butt (filter)	Cellulose acetate	1–10 years
Plastic straw	PP	200 years
Plastic cup	PS	50 years
Six-pack rings	Conventional LDPE	400 years
	Photodegradable	Months to years
Plastic utensils	PS, PP	200–450 years
PLA items	PLA	Similar to HDPE in marine water
PVC pipe	PVC	No measurable degradation after 32 years

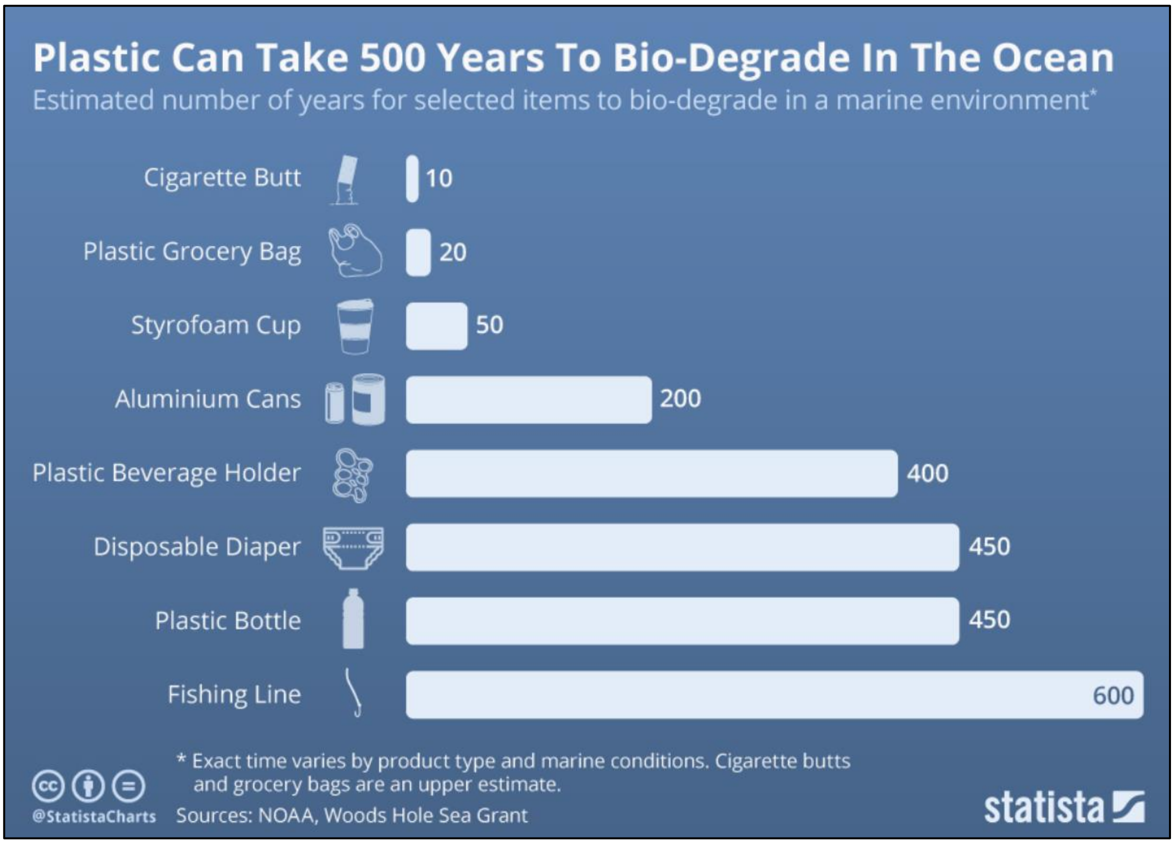


Figure 5: **Estimated biodegradation times of common plastic items in the marine environment** (NOAA, Woods Hole Sea Grant, 2018)

I.6. Plastic waste in the marine environment: sources and pathways

I.6.1. Land-based sources

Land-based plastic debris commonly found in the marine environment ranges from everyday single-use packaging products to industrial materials. Although its proportion remains uncertain, it is widely cited that up to 70-80% of marine plastic waste originates from land (Bel Hassen et al., 2025; Jambeck et al., 2015; MacLeod et al., 2021), indicating that the terrestrial contribution of plastics in the ocean has increased as a result of the growth in global plastic production and, consequently, the amount of plastic waste generated each year (Geyer et al., 2017; Law et al., 2020; PlasticsEurope, 2024). Poor waste management appears to be a key factor in this pollution, allowing plastics to migrate from landfills and cities to the ocean through various routes (Anthony et al., 2024; GESAMP, 2016; Wienrich et al., 2021). The major land-based sources of marine plastic waste include:

- **Waste management failures:** Landfills, mismanaged waste, illegal dumping (Jambeck et al., 2015; Liffmann & Boogaerts, 1997).
- **Urban sources:** Urban and stormwater runoff, untreated municipal sewage discharges, coastal and recreational activities (Werbowski et al., 2021).
- **Industrial sources:** Industrial activities, tire abrasion, construction and demolition waste (Hale et al., 2020), improper transport and handling of plastic materials (Galgani et al., 2015)
- **Agricultural sources:** Agricultural runoff (Morales-Caselles et al., 2025).

- **Consumer products:** Synthetic fibers from textiles, personal hygiene, and cosmetics products, and direct littering.

Land-based inputs contribute millions of tons of plastic annually, including approximately 15 million tons entering the oceans each year (Borrelle et al., 2020; Jambeck et al., 2015), with 1.4 million tons transported by rivers in 2020 alone. Without additional policy interventions, this riverine input is projected to more than double to 3.6 million tons by 2060 (Lebreton & Andrady, 2019; OECD, 2022).

The global distribution of marine plastic pollution varies significantly by region. As shown in fig 6 (Jambeck et al., 2015).

I.6.2. Ocean-based sources

Ocean-based sources contribute the remaining 20-30%, with the fishing industry accounting for about 18% of marine plastic debris, mainly from abandoned, lost, or discarded fishing gear

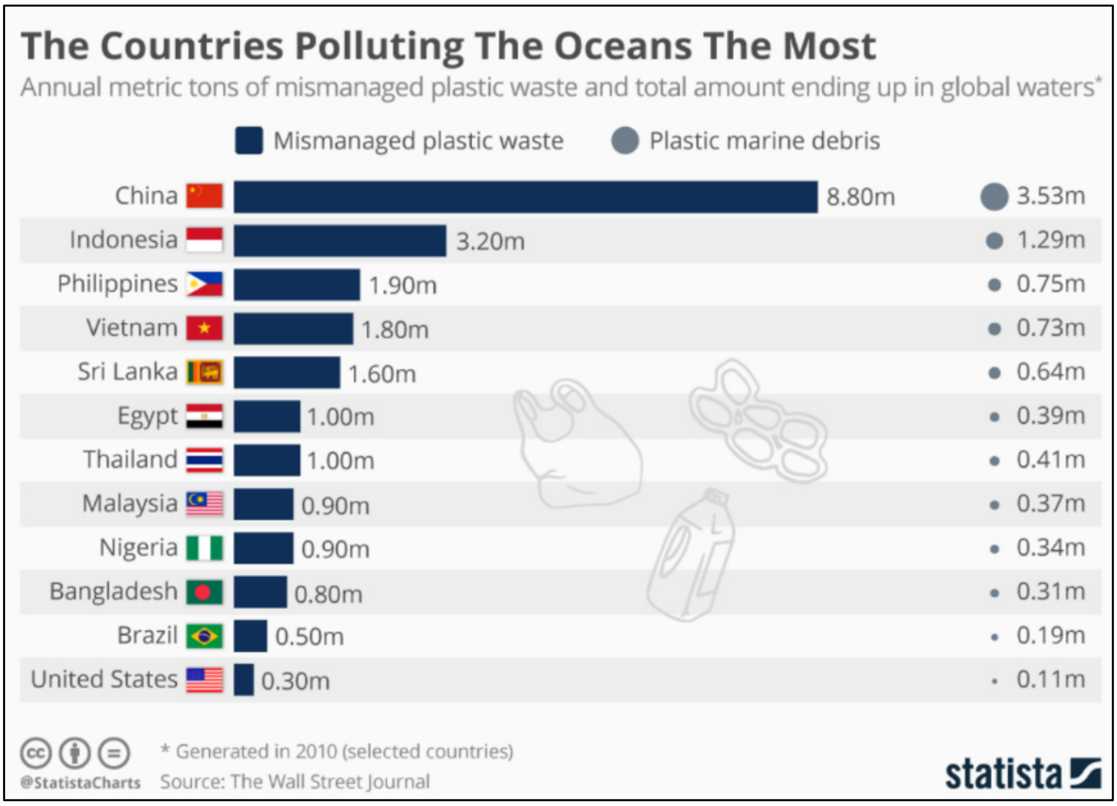


Figure 6: **The countries polluting the oceans the most: 2010 annual mismanaged plastic waste and marine debris** (Jambeck et al., 2015; Wall Street Journal)

(ALDFG), often called "ghost gear" (Lebreton et al., 2022; Richardson et al., 2019, 2022). Other marine sources include cargo losses from shipping, plastic materials from aquaculture, and virgin resin pellets lost during maritime transport (Andrady, 2011; Galgani et al., 2015).

I.7. Impacts of plastic marine pollution

I.7.1. Environmental impacts

Marine plastic pollution has emerged as a pervasive and growing environmental threat, affecting marine life, ecosystems, and the services they provide, through three main pathways: physical damage from entanglement and ingestion (Gall & Thompson, 2015; Høiberg et al., 2022; Laist, 1997; Lawson et al., 2015; Nelms et al., 2018; Vegter et al., 2014; Votier et al., 2011), chemical toxicity from plastic-associated contaminants (rafting), and habitat degradation (Shahidul Islam & Tanaka, 2004; Thompson et al., 2009). This problem has been worsened by alarming reports of plastic particles being found inside living organisms. Comparisons with the IUCN Red List show that at least 17% of species affected by plastic entanglement and ingestion are classified as threatened or near threatened (Gall & Thompson, 2015). Often mistaken for food, plastics accumulate in bodies and cause long-term health risks. Many species, including birds, fish, and turtles, are severely impacted (Kühn & Van Franeker, 2020; MacLeod et al., 2021).

The Convention on Biological Diversity reports that at least 817 marine species across all major taxonomic groups are regularly affected (Convention on Biological Diversity, 2016; Vanderreydt et al., 2024). Since the first scientific reports in the early 70s (Carpenter et al., 1972; Carpenter & Smith, 1972), studies have revealed plastic fragments in the intestines of dead marine animals, highlighting their harmful effects.

I.7.2. Socio-Economic Impacts

Beyond environmental impacts, plastic pollution imposes significant socio-economic losses, costing billions of dollars annually across key sectors such as fisheries, tourism, aquaculture, and shipping. These losses arise from direct costs, such as cleanup operations and equipment damage, and indirect costs linked to reduced revenues, and the degradation of ecosystem services, public health risks, and loss of recreational value, disproportionately affecting vulnerable coastal and low-income communities (Beaumont et al., 2019; McIlgorm et al., 2011). Adding to these burdens, plastic pollution poses a growing public health concern, with microplastics now ubiquitously detected in human tissues such as the placenta, liver, breast milk, blood, and lung tissue, particularly threatening fetuses, infants, and children (Chartres et al., 2024; Landrigan et al., 2023; Udovicki et al., 2022).

II. Microplastics: definition, origin, and classification

II.1. Introduction

Small plastic fragments were first documented in scientific literature in the early 1970s (Carpenter & Smith, 1972), with subsequent studies identifying plastic particles in seabirds during the 1980s (Harper & Fowler, 1987). It is unclear when the term “microplastic” was initially used in relation to marine debris. It was mentioned by Thompson et al. (2004) when discussing the distribution of plastic fragments in seawater, though no standardized size definition was established at the time. Since then, the term has been widely adopted to describe small plastic particles in the millimeter-to-submillimeter size range.

Today, MPs are recognized as one of the most pervasive forms of marine pollution, representing the vast majority of plastic particles by number (Eriksen et al., 2014), and are ubiquitous across all marine compartments, due to their persistence, small size, and continuous input (Amelia et al., 2021; Anderson et al., 2016). Beyond physical impacts, MPs pose chemical risks through their ability to adsorb and transport pollutants such as heavy metals and organic compounds, owing to their large specific surface area and hydrophobic properties (Law & Thompson, 2014; Yang et al., 2021). Once ingested across trophic levels, they may facilitate bioaccumulation and biomagnification. Additionally, their surfaces provide substrates for microbial colonization, potentially altering ecosystem processes (Amaral-Zettler et al., 2020). Despite growing research efforts, significant gaps persist regarding their long-term ecological and ecotoxicological consequences.

II.2. Definition of microplastics

The term ***"microplastics"*** was initially introduced to distinguish small plastic particles (micro) from larger plastic debris (macro), though early descriptions varied with the sampling and analytical methods (Andrady, 2011).

The currently accepted definition, as defined by GESAMP (2015), classifies MPs as solid synthetic organic polymer particles ranging from 0.001 to 5 mm in their largest dimension (Alimi et al., 2018; GESAMP, 2015; NOAA, 2024). Particles smaller than 0.001 mm are classified as nanoplastics, those between 5 and 25 mm as mesoplastics, and particles larger than 25 mm as macroplastics (W. C. Li et al., 2016). The 5 mm upper limit is widely adopted because particles of this size can be easily ingested by living organisms, facilitating their trophic transfer. However, some authors suggest that particles smaller than 1 mm should be defined as MPs to remain within the micrometric scale (Andrady, 2011; Lambert et al., 2014).

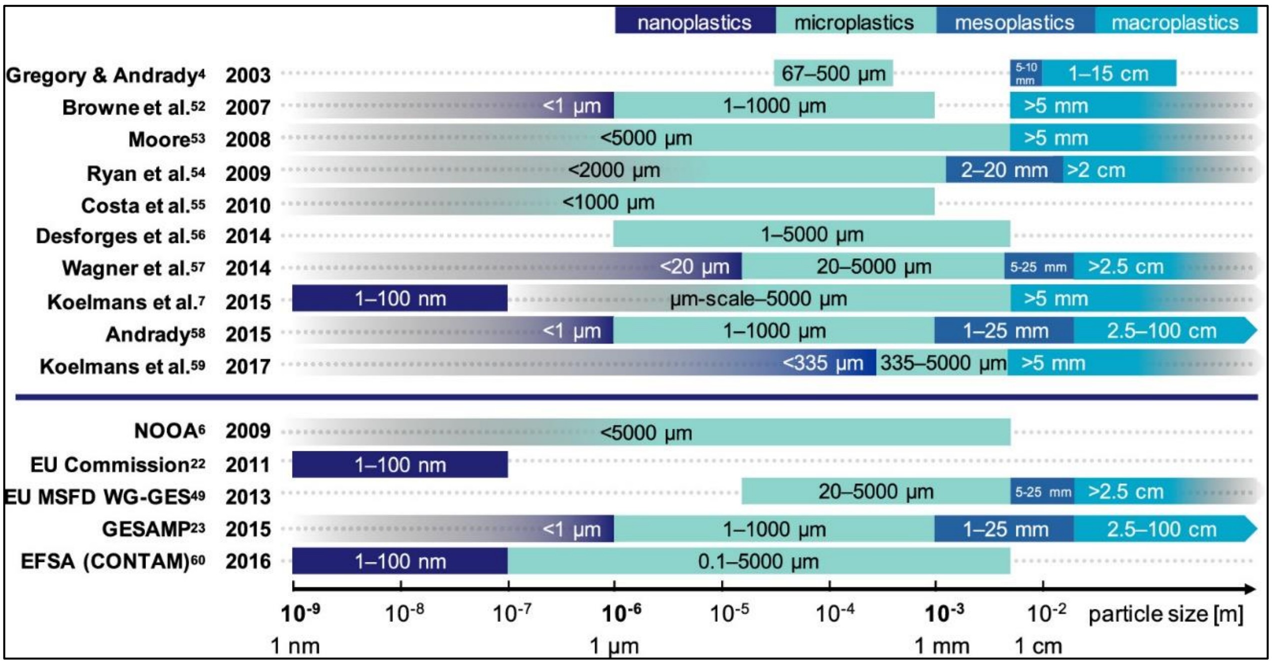


Figure 7: **Size classification limits of plastic debris according to different authors and organizations** (Hartmann et al., 2019)

II.3. Plastic composition

Polymers are macromolecules composed of repeating structural units, known as monomers, covalently linked to form long chains, branched structures, or complex networks (Causin, 2015). According to the IUPAC definition, polymers are high-molecular-weight materials characterized by the repeated occurrence of one or more constitutional units, which give them distinct physical and chemical properties; they can be natural or synthetic (IUPAC, n.d.). The first fully synthetic polymer, Bakelite, was developed in the early 20th century (Pilato, 2010); however, large-scale industrial production expanded significantly after the 1950s. Today, thousands of polymer grades are produced commercially, with the largest market dominance by low-cost, commodity thermoplastics, commonly referred to as "plastics," due to their low density, durability, malleability, and ease of production. However, these same properties contribute to their environmental persistence and widespread accumulation as plastic debris and MPs (Chamas et al., 2020; Geyer et al., 2017).

Plastic materials consist of a polymer matrix responsible for the main properties of the material (mechanical, thermal, and barrier, etc.), to which various additives or adjuvants (pigments, stabilizers, plasticizers, fillers, reinforcements) are incorporated to modulate specific characteristics (Andrady, 2011; Hahladakis et al., 2018). Most commonly identified MPs polymers include PE, LDPE, HDPE, PP, PMMA, nylon, PS, PVC, PC, PES, PUR, POM, and PAN, etc. (Fendall & Sewell, 2009; Gregory, 1996; Hidalgo-Ruz et al., 2012a).

II.4. Characterization of microplastics

The characterization of MPs integrates various analytical approaches to determine their physical and chemical properties. Physically, MPs vary in size, shape, color, and surface characteristics, which can be assessed using stereomicroscopy to infer their origin and degree of weathering (Cole et al., 2011; Hidalgo-Ruz et al., 2012b). Primary MPs typically exhibit manufactured shapes such as spheres or fibers with uniform surfaces, whereas secondary MPs present irregular, fragmented appearances resulting from environmental weathering (Andrady, 2015; Cole et al., 2011). Color is systematically documented, as it provides indirect indications of the degree of chemical contamination, as certain colors have been associated with differential adsorption of pollutants (Cole et al., 2014). The standardized Size and Color Sorting (SCS) system classifies MPs into 10 categories based on size and morphology (e.g., fragments, fibers, spheres, films, and foams) (Lusher et al., 2017), though classification schemes vary across studies, limiting the comparability of results.

Chemical characterization complements visual assessment, with FTIR or Raman spectroscopy commonly used for polymer identification (Käppler et al., 2015). Advanced methods, including pyrolysis coupled with gas chromatography-mass spectrometry (Py-GC/MS), enable the identification of plastic additives and associated chemical contaminants (Fischer & Scholz-Böttcher, 2017). Surface morphology, examined using scanning electron microscopy (SEM), reveals weathering patterns and potential adsorption sites for pollutants (Brandon et al., 2016).

II.5. Sources & origins

Microplastics are categorized into two major types based on their formation processes (GESAMP, 2015):

II.5.1. Primary origin

Primary MPs are intentionally manufactured at microscopic sizes for direct or indirect use. They include pre-production pellets or granules, flakes, spherules, and microbeads designed to be incorporated into consumer products or used as industrial abrasives (Andrady, 2011; Cole et al., 2014; Duis & Coors, 2016; Rafa et al., 2024).

The IUCN (2017) identifies seven primary sources of MPs in marine environments: synthetic textiles, vehicle tyres, road markings, personal care products and cosmetics, plastic pellets, marine coatings, and city dust. Among these, synthetic textiles contribute over a third (35%) of annual primary MP inputs to the global marine environment, making them a significant contributor to the

MPs issue (Boucher & Friot, 2017). These are not the only sources of microplastics, but they are among those for which quantification efforts have been made.

II.5.2. Secondary origin

Secondary MPs, in contrast, result from the progressive degradation of larger plastic debris through environmental weathering processes (Andrady, 2011; Cole et al., 2011; Rafa et al., 2024; Thompson et al., 2004). Major sources include mismanaged plastic waste, abandoned fishing gear, tourism-related litter, ship dismantling activities, industrial abrasives, plastics leached from waste sites, and synthetic textile microfibers released during washing. Secondary MPs constitute the most abundant type in marine environments (Andrady, 2011; Browne et al., 2011).

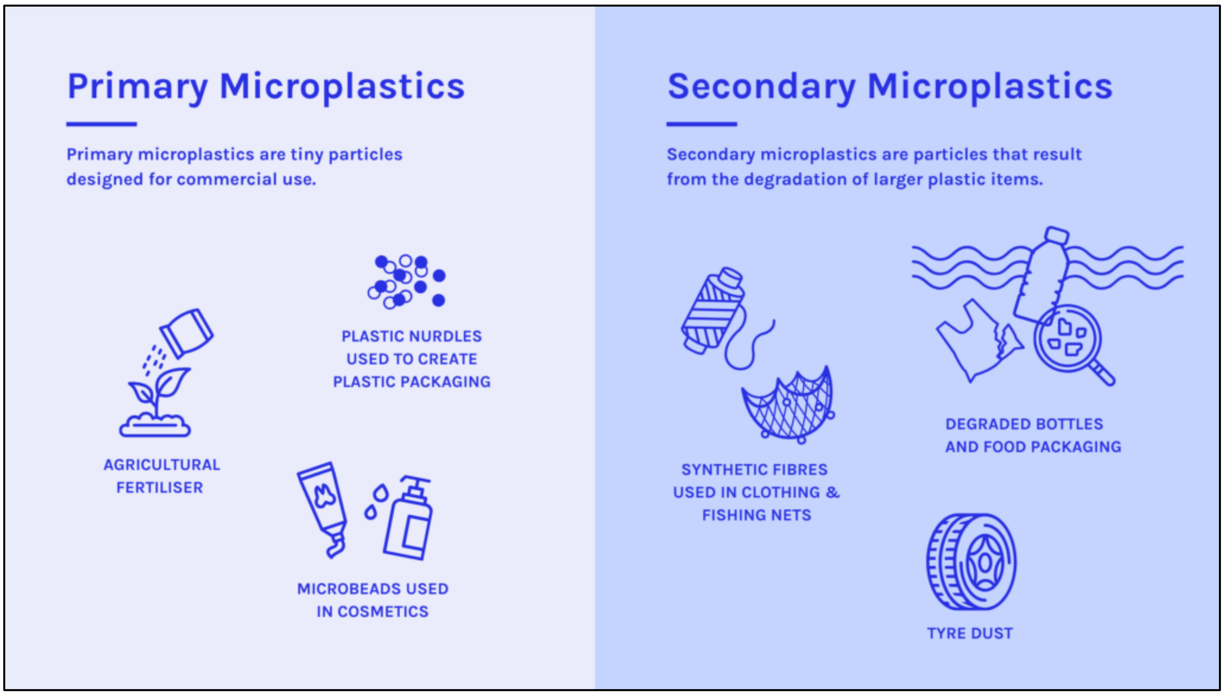


Figure 8: Classification of MPs according to their source and origin (Wessels, 2024)

II.6. Classification of microplastics

MPs are commonly classified by origin, size, morphology, and polymer composition. Based on origin, it distinguishes two main categories: primary and secondary MPs (Andrady, 2011; Cole et al., 2011). Size categories generally include large MPs (1–5 mm), small MPs (1 µm–1 mm), and nanoplastics (< 1 µm), with boundaries varying across studies (Hartmann et al., 2019). Morphologically, distinguishing fibers, fragments, films, pellets, and foams, each reflecting distinct degradation pathways and source materials (Hidalgo-Ruz et al., 2012b; Lusher et al., 2017). In terms of polymer type, PE, PP, PS, and PET are among the most frequently detected in marine environments (Andrady, 2011; Cole et al., 2011; Geyer et al., 2017). This multi-criteria classification framework is essential for standardizing data collection and ensuring meaningful comparisons across studies.

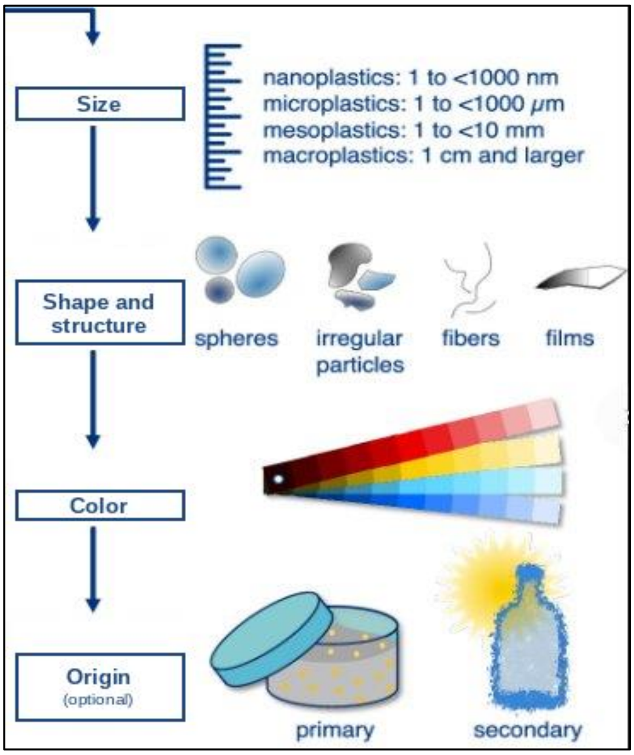


Figure 10: **Different classification criteria of MPs** (Hartmann et al. 2017)

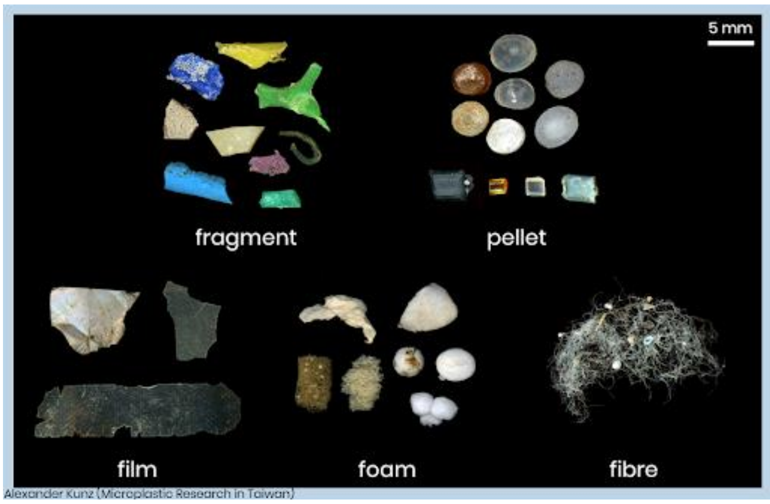


Figure 9: **Different types of MPs** (Kunz, 2020)

PART 2, Distribution and environmental impacts of MPs in marine ecosystems

I. Environmental occurrence and fate

MPs are ubiquitous across all environmental compartments, including surface and subsurface waters, sediments, soils, freshwater systems, polar regions, the atmosphere, as well as in living organisms across trophic levels, demonstrating a global distribution that extends far beyond areas of direct human activity (Galgani et al., 2015; Hale et al., 2020; Obbard et al., 2014; Stubbins et al., 2021). Once introduced into aquatic environments, MP fate is governed by polymer density, biofouling, and environmental degradation processes. Low-density MPs initially remain buoyant at the surface, whereas higher-density polymers sink and accumulate in sediments; however, biofouling progressively increases particle density, ultimately driving even low-density MPs toward the seafloor, making sediments the dominant long-term sink (Cózar et al., 2014; Zettler et al., 2013). Beyond aquatic systems, atmospheric transport enables MPs' deposition in remote regions, with Arctic deep-sea sediments confirmed as a major global sink fed by thermohaline circulation (Bergmann et al., 2019), highlighting that MPs follow complex, multi-compartment transport pathways spanning the entire Earth system, leaving no environment truly isolated from this contamination.

II. Spatial distribution of MPs in the marine environment

MPs distribution is governed by a complex interplay of ocean circulation, wind-driven surface currents, gyre systems, and thermohaline processes, which transport and concentrate particles into predictable convergence zones. Five major subtropical gyres serve as primary accumulation areas, with the North Pacific Gyre being the most emblematic (Eriksen et al., 2013; Moore et al., 2001; Van Sebille et al., 2015). Spatial heterogeneity is also pronounced at smaller scales, with coastal zones, river plumes, ports, and convergence areas consistently recording higher concentrations than offshore waters, reflecting the combined influence of local circulation, atmospheric forcing, and proximity to sources (Cózar et al., 2014; L. C.-M. Lebreton et al., 2012). Climate change further reshapes these patterns by intensifying ocean stratification, altering wind regimes and current velocities, and increasing extreme precipitation events, thereby enhancing riverine plastic inputs (Kelly et al., 2025; L. C. M. Lebreton et al., 2017). The Mediterranean Sea, a semi-enclosed basin characterized by anti-estuarine circulation, limited water renewal, and densely populated coastlines, acts as a major long-term convergence zone (Millot & Taupier-Letage, 2005), recording MP surface concentrations up to 1.25 million fragments per km², four times higher than those measured in the North Pacific Gyre (Cózar et al., 2014), with documented hotspots occur in the Northwestern basin, the Adriatic, and near the Strait of Sicily, driven by regional circulation patterns and riverine inputs such as the Rhône and Po (Collignon et al., 2012; Faure et al., 2015; Galgani et al., 2015; Suaria et al., 2016).

Patterns in spatial distribution and debris characteristics help identify pollution origins: accumulation density decreases with increasing distance from urban areas, while increasing with beach visitation rates and tourism, and the level of biofouling on plastic items indicates whether the debris has traveled long distances in the open ocean or originates from local sources (Bel Hassen et al., 2025; Thiel et al., 2018). Other forensic indicators include foreign labeling, suggesting maritime origins, and item types, such as beverage containers and personal care products, which are indicative of tourism and recreational activities (Derraik, 2002; GESAMP, 2019; L. Lebreton et al., 2018).

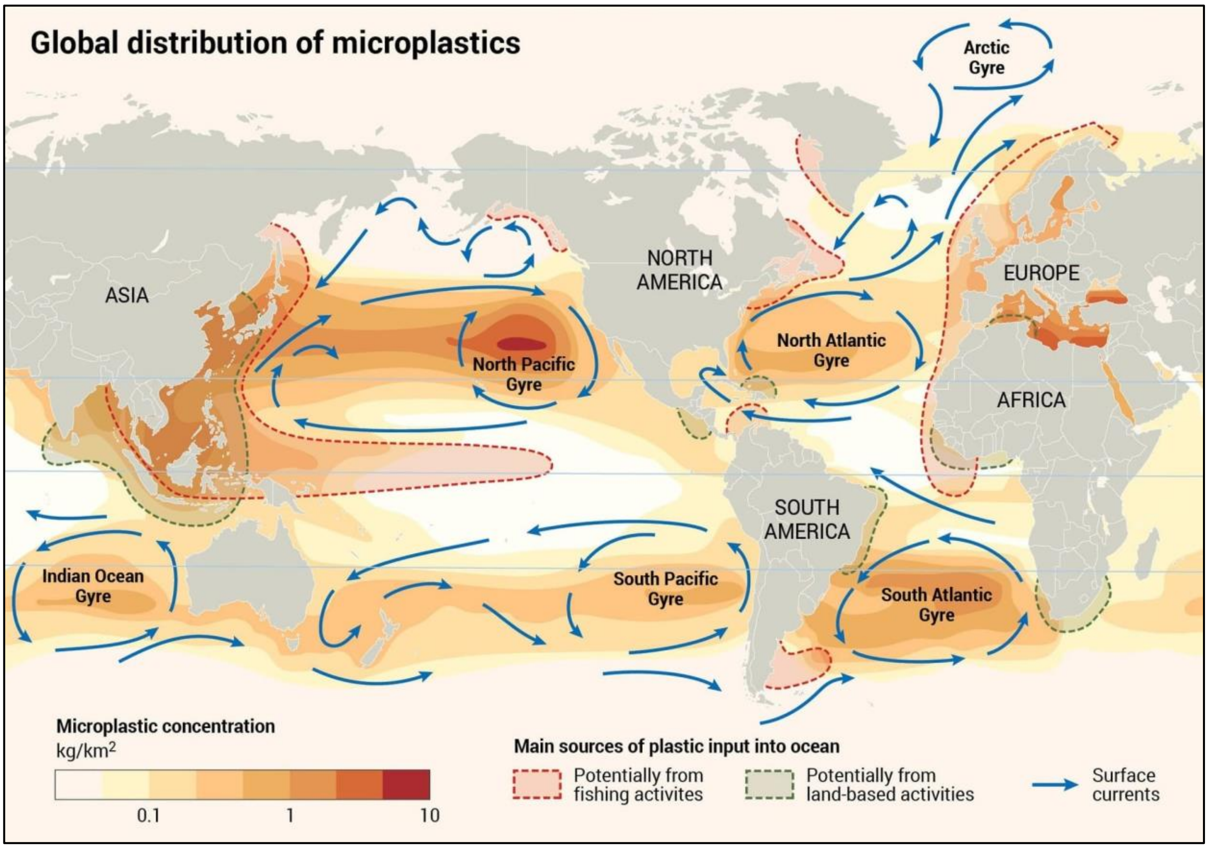


Figure 11: **Spatial distribution of subsurface MPs in the ocean** (GRID-Arendal, n.d.)

III. Microplastics as vectors of environmental pollutants

MPs are increasingly recognized as vectors of environmental pollutants in aquatic and coastal ecosystems. Due to their small size, high surface-area-to-volume ratio, and capacity to undergo environmental aging, they can adsorb and transport a wide range of contaminants, including persistent organic pollutants (POPs), trace metals, pharmaceuticals, plastic additives, and pathogenic microorganisms (Amelia et al., 2021; Caruso, 2019; Rafa et al., 2024; Squadrone et al., 2022; Tumwesigye et al., 2023). Pollutant sorption occurs through several mechanisms, such as surface adsorption, ionic interactions, hydrogen bonding, and accumulation within surface cracks formed during aging, with the efficiency of these processes being controlled by several factors, notably polymer type, particle size, surface roughness, degree of weathering, biofilm formation, and environmental conditions (Hartmann et al., 2017; Lim et al., 2022; P. Liu et al., 2019; Torres et al., 2021; Zhou et al., 2025). In marine environments, MPs may act as mobile substrates that facilitate the dispersal of contaminants through water, sediments, and food webs, thereby increasing the risk of ecological exposure and potential trophic transfer (Amelia et al., 2021; Brennecke et al., 2016; Carbery et al., 2018; Rochman et al., 2013). Beyond their direct physical impacts, MPs should consequently be considered dynamic vectors that can modify the mobility, persistence, and bioavailability of environmental contaminants (Du et al., 2025; Xiong et al., 2024).

PART 3, Regulatory framework

I. International legal framework on plastics and marine litter

Until the 70s, oceans were perceived as an easy solution for waste disposal, with their vastness suggesting an infinite capacity to absorb anthropogenic pollution. The absence of regulatory frameworks reinforced this perception, allowing negligent practices toward the marine environment to persist. Major environmental disasters such as the oil spills from the Torrey Canyon, Amoco Cadiz, and Ixtoc I in the Gulf of Mexico, as well as incidents like Minamata, triggered global awareness and prompted the progressive adoption of international rules to regulate ocean pollution. Plastics, initially ignored, were gradually included in this regulatory framework in response to the growing magnitude of their impact. Thus, the fight against marine plastic pollution has progressively been structured around a set of international conventions, regulating plastic discharge and improving waste management. Table 3 summarizes the principal regulatory framework governing plastic pollution.

II. Algerian national regulations

Algeria has developed a national legal framework in line with its international commitments and oriented towards a circular economy. Several fundamental laws govern waste management, preservation of coastal and marine environments, and environmental protection. However, provisions specifically targeting plastics, and more particularly microplastics, remain underdeveloped in current legislation. Table 4 summarizes the main national instruments.

Table 3: **Overview of the international legal framework on plastics and marine litter**

Instrument	*Adopted / In force*	*Scope*	*Main objective*	*Relevance to plastics/MPs pollution*	*Algeria: ratification status & legal instrument*
London Convention + 1996 Protocol	1972 /1975 under the auspices of the IMO (Protocol: 2006)	Global; deliberate waste dumping at sea	Prevent marine pollution by prohibiting and regulating the deliberate disposal of waste at sea from ships, aircraft, platforms, or other artificial structures	Prohibits dumping of dangerous substances (Annex I); the protocol introduces a general ban with a reverse-list exemption; applies precautionary and polluter-pays principles; and prevents the deliberate disposal of solid plastic waste at sea.	/
MARPOL 73/78	1973 / 1988 IMO	Global; all ships, fixed and floating platforms	Prevent marine pollution, whether accidental or related to marine operations.	Absolute prohibition of discharging all types of plastic waste at sea (bags, nets, ropes, etc.); classification of waste and regulation of discharge distances from the coast; mandatory Garbage Record Book and Garbage Management Plan on board.	Ratified **31 January 1989**, by **Decree n° 88-108**; entered into force **1 May 1989**. Full Party to Annexes I–V. **MARPOL Annex VI:** not yet ratified as of 2021.
Barcelona Convention + Regional Plan on Marine Litter, 2013	1976 /1978 (Plan: 2014, update 2021° UNEP/MAP	Regional: Mediterranean Sea	Protect the Mediterranean Sea against all forms of pollution and promote sustainable management of marine resources	Legally binding Regional Plan on Marine Litter Management promotes the reduction of single-use plastics and port charges; IMAP Candidate Indicator 24 establishes harmonized microplastics monitoring protocols for beach sediment and surface waters; Marine Litter MED II (2020-2023) supports Algeria in implementing these obligations.	Original Convention: accession 26 January 1980, by **Presidential decree n° 80-14**. 1995 Amendments: ratified 28 April 2004, by **Presidential decree n° 04-141**. **Prevention & Emergency Protocol**: ratified **14 November 2016**, entered into force **14 December 2016**. Full Contracting Party.
The United Nations Convention on the Law of the Sea (UNCLOS)	1982 /1994	Global	Establish a comprehensive legal framework governing the use of seas and oceans, including obligations on marine environment protection.	**Art 194:** obliges States to take all necessary measures to prevent, reduce, and control marine pollution from all sources. **Art 207:** requires national legislation on land-based pollution sources (rivers, estuaries, coastal installations). **Art 210:** enforces legislation on waste dumping	Signed **10 December 1982.** Ratified **11 June 1996,** by **Presidential decree n° 96-53;** entered into force **11 July 1996**
Basel Convention	1989 /1992 20	Global; transboundary movements of	Control transboundary movements of hazardous and other wastes and ensure their	First legally binding global treaty on plastic waste. Annex II (Y48) subjects mixed or contaminated plastics to the Prior Informed Consent (PIC);	Accession with reservation **15 September 1998**, by **Presidential decree n° 98-158**.

(2021 plastic amendment)		hazardous and other wastes	environmentally sustainable management	Annex VIII (A3210) covers hazardous plastics. Annex IX (B3011): sorted plastics intended for clean recycling, exempt from the PIC procedure; encourages source reduction and local recycling to prevent plastics from entering the environment and becoming MPs.	**Ban Amendment (1995)**: ratified by **Presidential decree n° 06-170**. **2021 plastic waste amendments**: /
Rotterdam Convention	1998 /2004	Global; hazardous chemicals and pesticides in international trade	Promote shared responsibility in the international trade of hazardous chemicals through informed decision-making	The PIC procedure applies to plastic additives classified as hazardous (plasticizers, flame retardants, and UV stabilizers); indirectly limits the trade of chemicals that leach from plastics and MPs and accumulate in the marine compartments.	Accession **21 July 2020**, by **Presidential decree n° 19-265**; entered into force **19 October 2020**.
Stockholm Convention (POPs)	2001 /2004	Global; persistent organic pollutants	Protect human health and the environment against the harmful effects of POPs	Targets POPs used as plastic additives; COP 11 (2023) added UV-328 and Dechlorane Plus to annex A for global elimination; recognizes MPs as vectors of POPs in the aquatic environment, facilitating their bioaccumulation in marine fauna	Signed **5 September 2001**. Ratified **22 September 2006**, by **Presidential decree n° 06-206**. **National Implementation Plan** published **22 April 2015**.
Global Plastic Treaty (INC process, UNEA)	Launched in 2022; not yet adopted	Global; full plastic lifecycle	Develop the first legally binding international treaty on plastic pollution, covering its entire life cycle:	INC-5.2 (Geneva, August 2025) concluded without consensus; unsolved issues include mandatory production caps, regulation of toxic chemicals in plastics, binding vs voluntary obligations, and financing for developing countries.	/

Table 4: National legislative and regulatory framework on plastic waste and marine pollution in Algeria

Laws		
Instrument	**Main objective**	**Relevance to plastics/MPs pollution**
Law No. 01-19 (amend Law No. 25-02) related to Waste Management, Control, and Elimination	Establish a comprehensive framework for waste prevention, management, recovery, and elimination in line with circular economy principles	Provides the first legal definition of single-use plastic products; establishes extended producer responsibility (EPR) and eco-contributions as economic disincentives; mandates improved collection and recovery systems; requires a

		national waste data and traceability system. Gaps in enforcement are the proximate cause of plastic accumulation in coastal environment.
Law No. 02-02 related to the Protection and Enhancement of the Coastline	Preserve the ecological integrity of the coastline, combat anthropogenic pollution, and regulate coastal development	Explicitly prohibits waste discharge in sensitive coastal zones; requires environmental impact assessments for any coastal development project; weak field application due to a lack of institutional coordination and control means.
Law No. 03-10 relative to the Protection of the Environment within the Framework of Sustainable Development	Establish the fundamental principles of Algerian environmental policy within a sustainable development framework.	Establishes the polluter-pays, precautionary, and prevention-at-source principles; provides a cross-cutting framework for all public and private environmental action; covers aquatic environments. However, no specific address the microplastics issue.
Executive Decrees		
No. 94-279 of September 17, 1994, related to the Organization of the Fight Against Marine Pollution and Establishment of Contingency Plans	Organize the prevention and response to marine pollution incidents in Algerian waters	Establishes national and regional contingency plans for marine pollution response, defines coordination mechanisms between relevant authorities (the Navy, Coast Guard, and Ministry of Environment); creates the Tel Bahr regional maritime committee; implements Algeria's MARPOL obligations; and provides a legal and operational basis for monitoring, preventing, and responding to marine litter, including plastic debris.
No. 02-175 of May 20, 2002, on the Creation, Organization, and Operation of the National Waste Agency (AND)	Establish a dedicated public institution to support and coordinate national waste management	AND mandated to build a national waste database and conduct monitoring campaigns; launched microplastic standardization and identification protocols in 2021 in partnership with Probiom and Expedition MED France.
No. 02-372 of November 11, 2002, related to Packaging Waste	Establish producer responsibility for packaging waste recovery and create a public system for its collection, sorting, and valorization	Defines producer/holder obligations for packaging waste recovery
No. 04-199 of July 19, 2004, Setting Methods for Creation, Organization, Operation, and Financing of the Public System for Packaging Waste Treatment		Creates the Eco-Jem public recovery system, managed by AND. In practice, Eco-Jem is largely ineffective due to unclear financial responsibilities, weak enforcement, and dominance of informal waste collectors, leaving most packaging plastics unmanaged and accumulating in the environment.
No. 04-210 of July 28, 2004, Defining Methods for Determining Technical Characteristics of Packaging Intended to Directly Contain Food Products or Objects Intended to be Handled by Children	Define technical and safety standards for packaging that may directly contaminate food or be handled by children	Regulates the composition and characteristics of plastic packaging for food and children's products; restricts certain materials and additives; contributes to reducing the use of potentially hazardous plastic formulations in high-risk consumer applications.

Chapter II Materials & Methods

I. Presentation of the study area

The Algiers coastline, situated in the central part of the Algerian littoral at approximately 36.75° N and 3.05° E, represents one of the most highly anthropized coastal sectors of the southern Mediterranean. Occupying a strategic position along Algeria's maritime frontage, it is characterized by intense urbanization, major port infrastructures, dense residential areas, transport networks, and industrial activities, particularly around the metropolitan area of Algiers (OTMANI et al., 2019; RABEHI et al., 2019). Tourism and recreational activities further constitute important socioeconomic drivers, especially during the summer. These combined factors generate significant environmental stress on the coastal zone, affecting sediment dynamics, shoreline stability, water quality, and the integrity of coastal habitats (Mejjad et al., 2022; RABEHI et al., 2019).

The present study was conducted on two pilot sandy beaches located along the western Algiers coastline: West Beach of Sidi Fredj and El Djamila Beach, also known as La Madrague, in Ain Benian. These sites were selected for their geomorphological representativeness and accessibility, as well as their compliance with the sampling protocol, which required the presence of wide sandy beaches allowing the establishment of a 100 m transect parallel to the shoreline and positioned above the daily high-tide line.

Figure 12: **Map of the study area.**

I.1. West Beach of Sidi Fredj

West Beach of Sidi Fredj is situated on the Sidi Fredj Peninsula, approximately 30 km west of Algiers, within the municipality of Staouéli (36°45′40″ N, 2°50′43″ E). It is a sandy beach composed mainly of medium-grained sand and pebbles, located at the interface between the open marine environment and the urban coastal fringe, which is strongly influenced by tourism, recreational use, and nearby urban and port-related infrastructure. Its geographical position makes it particularly exposed to anthropogenic pressures (Mejjad et al., 2022; Rabehi et al., 2019). The beach faces northwest and benefits from partial protection from the peninsula's eastern headland. Its hydrodynamic regime is governed by wave action, longshore currents, the influence of the Algerian Current, and local shoreline morphology, which together govern sediment transport and redistribution (Millot, 1999; Puillat et al., 2002). The beach is particularly exposed to northwesterly swells and winter storm events, which enhance sediment resuspension, coastal erosion, and the redistribution of floating debris (Puillat et al., 2002). Despite the absence of major watercourses, urban runoff and stormwater discharge may provide episodic inputs of sediments and contaminants during rainfall events (OTMANI et al., 2019). The area is subject to a typical Mediterranean semi-arid climate, characterized by mild, wet winters and hot, dry summers, which contributes to marked seasonal variability in both environmental conditions and human pressure .

Sidi Fredj is a major tourist destination along the Algerian coast, hosting numerous infrastructures, including a marina, hotels, restaurants, private beaches, water-sport facilities, a thalassotherapy and spa center (ONT, 2022). Consequently, potential sources of pollution include tourism-related activities, recreational boating, maritime traffic, urban runoff, solid waste deposition, and port operations.

Figure 13: **West Beach of Sidi Fredj**

I.2. El Djamila Beach

El Djamila Beach, commonly known as La Madrague, located in the municipality of Ain Benian, approximately 15 km west of Algiers, at 36°47′ 54” N and 2°54′ 35” E, lies within a highly urbanized recreational zone on the western Algiers coast. The beach is subject to significant anthropogenic influence from nearby residential areas and the wider urban expansion of the capital (Rabehi et al., 2019). Characterized by an open coastal setting, it is more exposed to wave action than the West Beach of Sidi Fredj. Its hydrodynamic regime is governed by wave exposure, wind patterns, the eastward-flowing Algerian Current, and local beach morphology, which together control sediment transport and shoreline deposition (Millot, 1999; Puillat et al., 2002). Although no major permanent watercourses are present, the area is influenced by seasonal Wadis and urban drainage systems that may deliver sediments, organic matter, and contaminants during rainfall events. The site is subject to the same Mediterranean semi-arid climatic conditions. Seasonal increases in tourist attendance during summer further amplify anthropogenic pressure on the coastal environment. The surrounding area is densely urbanized, comprising residential zones, villas, tourism facilities, road networks, and recreational infrastructures. Additional potential sources of environmental pressure include a nearby wastewater treatment plant, stormwater runoff, maritime recreational activities, and inadequate waste management.

Together, West Beach of Sidi Fredj and El Djamila Beach present two contrasting yet complementary coastal environments for assessing the combined influence of urbanization, tourism, hydrodynamic conditions, and watershed inputs on the distribution and accumulation of environmental contaminants along the western Algiers coastline. Both are state-managed public beaches, rather than unmanaged or wild beaches, and are subject to regular cleaning operations, providing a relevant framework for evaluating contamination under managed coastal conditions.

Figure 14: **El Djamila Beach, Ain Benian**

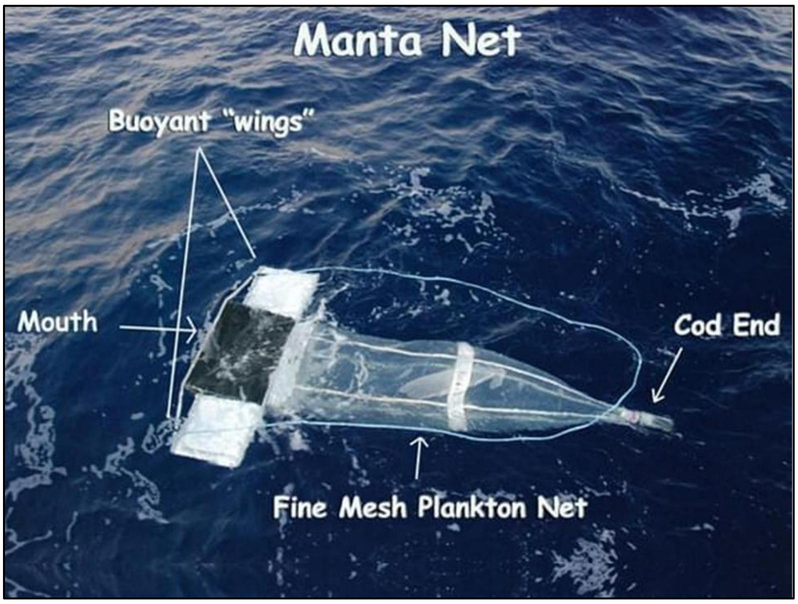
II. Sampling Materials & Methods

Materials in direct contact with samples consisted exclusively of stainless steel and glass to minimize plastic and metallic contamination and ensure reproducible results. All equipment was rinsed three times with ultrapure water (UPW) before use. Samples from the western beach of Sidi Fredj were collected on 24/02/2026, whereas samples from El Djamila beach (Ain Benian) were collected on 28/04/2026.

II.1. Surface Seawater

Surface seawater sampling followed the standardized protocol of Kovač Viršek et al. (2016), aligned with the recommendations of the TSG ML/ MSFD (Karlsson et al., 2020; Vianello et al., 2018).

A manta trawl net was deployed 10 m outside the boat's wake zone to prevent turbulence interference, and towed in a straight line at a steady speed of 2 to 3 knots for 30 minutes, with GPS coordinates, towing duration, trajectory, and weather conditions recorded throughout. At the end of sampling, the net was rinsed externally to concentrate all adhering particles in the cod-end. The collected material was transferred into pre-cleaned glass containers, rinsed with distilled, filtered water to ensure complete recovery, and preserved in 70% ethanol (for preservation and enhancing particle visibility). All containers were sealed, labeled, and then stored on ice until laboratory analysis. Sampling was conducted during calm sea conditions



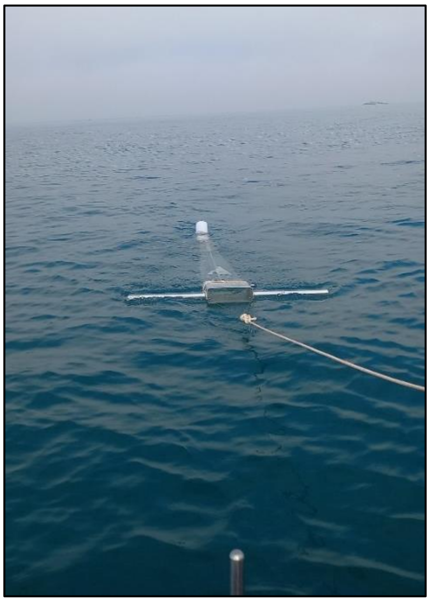


Figure 15: **Deployment of the Manta Net**

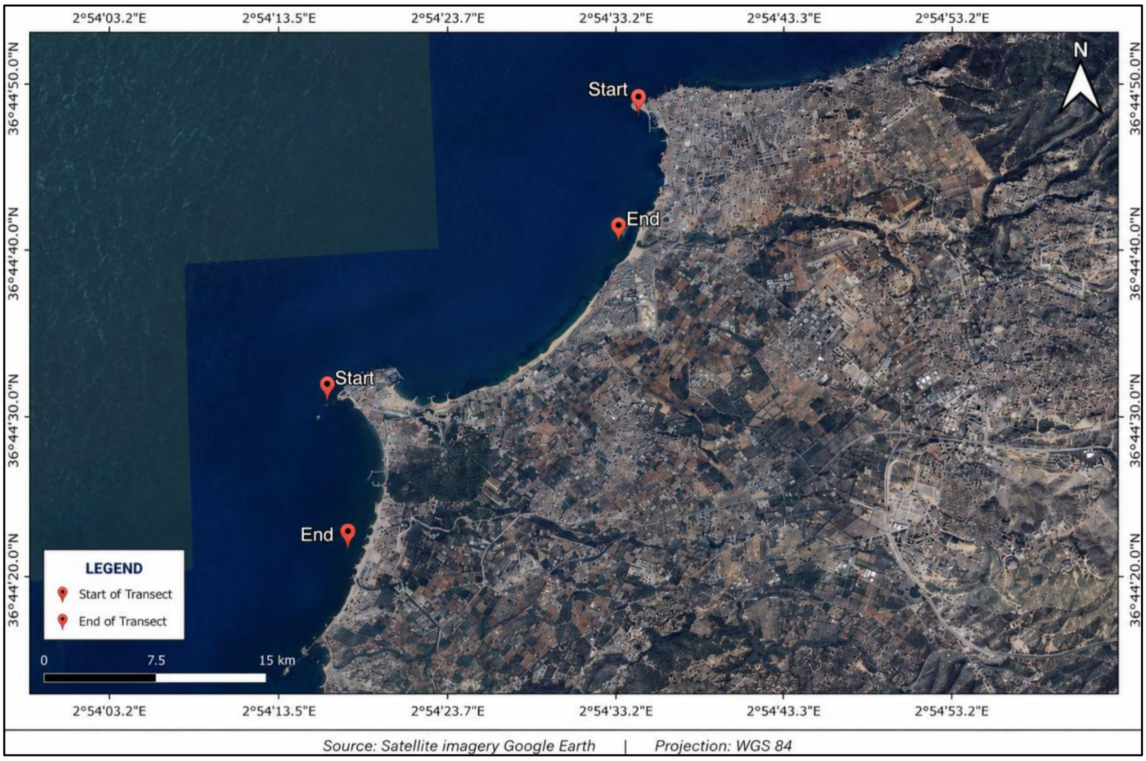


Figure 16: **Map of seawater sampling points at our study zone**

II.2. Sediment

II.2.1. Beach Sediment

Beach Sediment sampling followed the standardized protocol of Frias et al. (2018), in accordance with the international recommendations of the TSG-ML/MSFD for assessing coastal microplastic accumulation. The procedure begins with selecting the debris line, along which a 100 m transect was established, with 50×50 cm quadrats placed systematically every 10 m, with five replicates per site. Within each quadrat, surface sediment (0-5 cm deep) was collected using a stainless-steel spoon, transferred to labeled glass bottles, and stored at < 6 °C until laboratory analysis.

Figure 17: **Quadrat used**

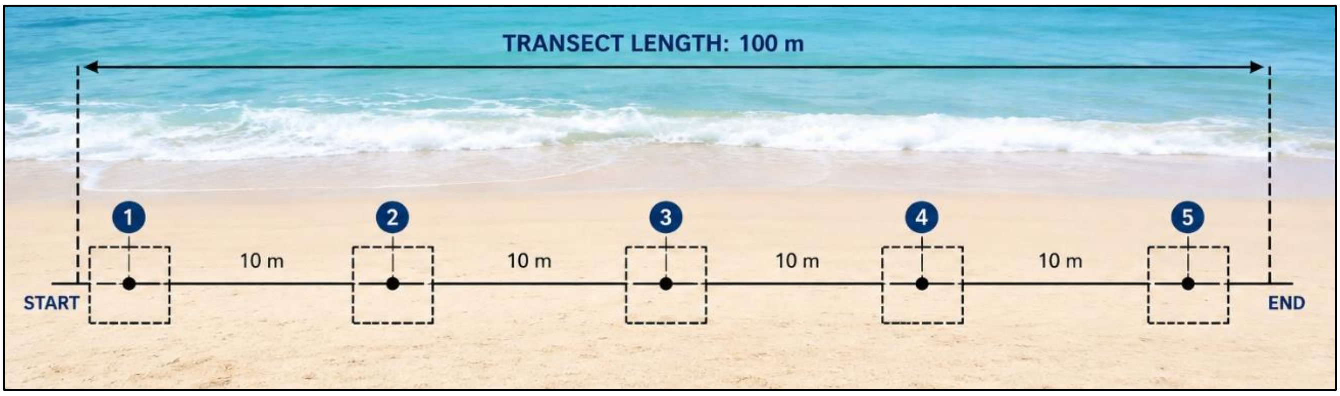


Figure 18: **Schematic representation of the sampling procedure**

II.2.2. Marine Surface Sediment

Marine surface sediment sampling was conducted following the protocol described by Frias et al. (2018). Sampling was carried out at three points per site, collected using a stainless steel Van Veen grab. Sediment was subsampled from the central portion of the grab using a stainless-steel spoon, carefully to avoid edge contamination. Samples were then transferred into glass containers and subsequently stored at −20 °C until laboratory processing and analysis.

Figure 19: **Map of beach & marine sediment sampling points at Ain Benian**

Figure 20: **Map of beach & marine sediment sampling points at Sidi Fredj**

III. Laboratory Processing & Analysis

III.1. Contamination controls

Contamination prevention measures included using glass equipment and covering samples when not in use. All work was conducted in a dedicated, clean laboratory space.

III.2. Microplastics extraction

III.2.1. Seawater samples

Samples were processed following Kovač Viršek et al. (2016), with slight modifications. Given the high organic matter content (primarily plankton), a digestion step was applied using H_2O_2 (30%) at 60°C for 48 hours. Samples were then sieved through a 315 µm mesh, as a 300 µm sieve was not available. Macro- and meso-litter (>5 mm) were removed using tweezers. The retained material was rinsed with distilled water to recover any adhering microplastics, concentrated on the sieve, and transferred into a Petri dish. The sample was then filtered through GF/C glass fiber filters (1.2 µm).

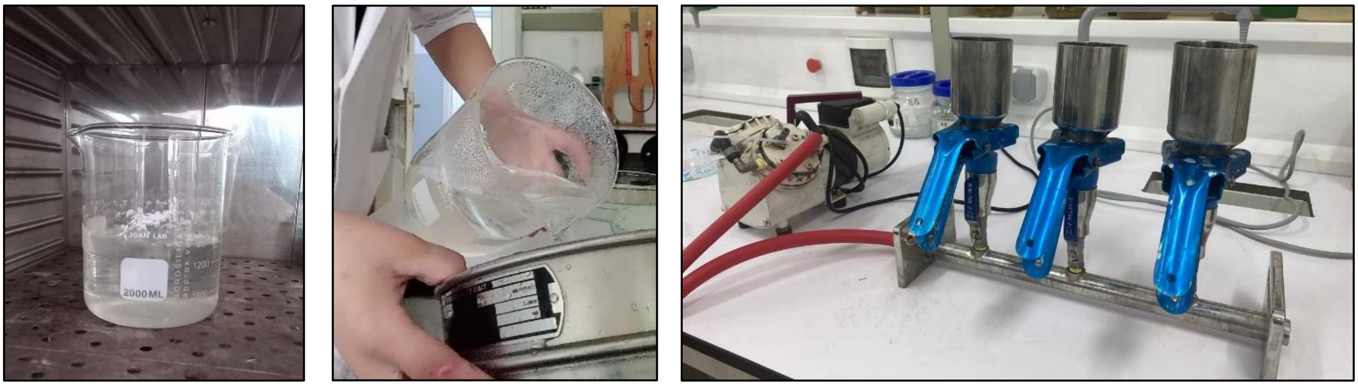


Figure 21: **Laboratory processing of seawater samples**

III.2.2. Sediment samples

Microplastic extraction followed the **TSG-ML** guidance developed under the **EU MSFD** (Galgani et al., 2013; Hanke, G. et al., 2013).

Drying: Sediment samples were placed in glass Petri dishes to avoid contamination and were dried in an oven at 65°C for 24 hours until a constant weight was reached. Once dry, samples were weighed using a precision balance (fig 22).

Figure 22: **Determination of sample dry weight**

Sieving: Dried samples were sieved using a stack of sieves (5 mm, 1 mm, and 315 µm). Particles > 5 mm were discarded. Two size fractions were obtained: 1-5 mm (material retained on the 1 mm sieve) and 315µm-1 mm (material retained on the 315 µm sieve). Each fraction was collected and weighed separately (fig 23).

Figure 23: **Size fractionation and weighing of samples**

Density separation: A saturated sodium chloride solution (NaCl, 1.20 g/cm^3) was prepared by dissolving 360 g of NaCl in 1000 mL of distilled water. The solution was stirred using a magnetic stirrer and filtered (1.2µm) before use. Its density was verified using a densimeter (fig 24) and by weighing 1 mL of solution. This density (1.20 g/cm^3) allows the separation of low-density MPs from denser sediment particles. Sediment subsamples (30-50 g) were mixed with the NaCl solution at a 1:5 (w/v) ratio. The mixture was then agitated for 2-3 hours using a magnetic stirrer and left to settle overnight (fig 25). The supernatant was subsequently filtered through GF/C glass fiber filters (1.2 µm) (fig 26). The procedure was repeated 2 to 3 times for each sample to maximize MP recovery and extraction efficiency.

NaCl was selected for its low cost, low toxicity, and ease of preparation, making it well suited to routine laboratory conditions. However, its density doesn't allow the extraction of denser polymers (e.g., PVC, POM, PET), which are difficult to recover using this method (Hidalgo-Ruz et al., 2012b; Larrea et al., 2025). More efficient density separation solutions exist, such as sodium polytungstate (SPT, up to 1.4–1.6 g/cm^3) or zinc chloride ($ZnCl_2$, up to 1.5–1.7 g/cm^3), which offer higher recovery rates across a broader range of polymer densities, though at a higher cost and with greater safety and disposal considerations (Coppock et al., 2017; Imhof et al., 2012; Prata et al., 2019).

Figure 24: **Verification of NaCl solution density using a densimeter**

Figure 25: **Density separation of sediment samples**

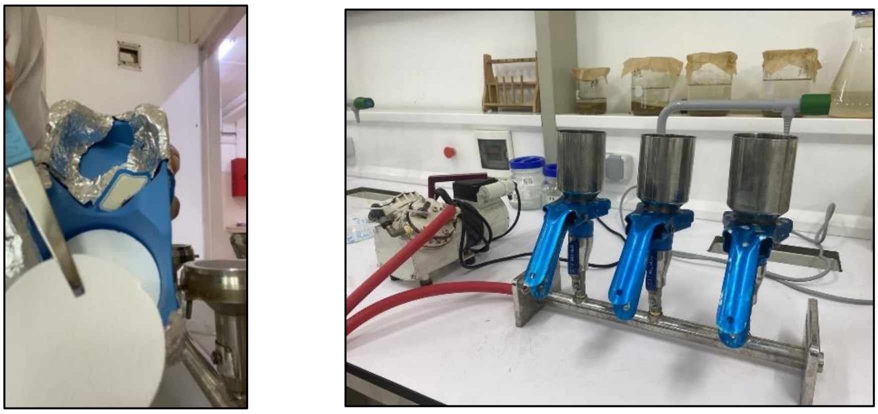


Figure 26: **Vacuum filtration of the supernatant through GF/C glass fiber filters (1.2 µm)**

III.3. Microplastics identification

MPs' visual identification was conducted according to the criteria established by Lusher et al. (2020). The filters were examined under a stereomicroscope at magnifications between 10x and 40x to analyze their shapes and colors (Pradit et al., 2022). MPs were sorted into five groups: fragments, films, fibers, foams, and pellets, following the method described by Hidalgo-Ruz et al., 2012a. Suspected particles were isolated, measured based on their maximum length, and their color was recorded. All confirmed particles were finally stored in labeled, closed glass containers before chemical analysis.

Figure 27: **Visual identification & classification of MPs particles according to their morphology**

Chemical characterization of MPs was performed using Fourier Transform Infrared (FTIR) spectroscopy on a JASCO FTIR-4X spectrometer at a spectral resolution of 4 cm^{-1} over a wavelength range of 3500–500 cm^{-1}. Each particle was analyzed individually, and the resulting spectra were compared with reference spectral libraries to identify the corresponding polymer types (table 5). This approach confirmed the synthetic nature of the suspected particles.

Only visually identifiable particles were analyzed. Very small MPs particles that were not visible to the naked eye were excluded due to methodological limitations.

Table 5: **FTIR characteristic peaks of different polymers** (Roy et al., 2025)**.**

Polymer type	Characteristic peaks
PS	3050 (aromatic C–H stretching), 2840 (C–H stretching), 1601, 1440 (aromatic ring stretching), 1022 (CH_2 band), 670, and 530 cm^{-1} (aromatic CH out-of-plane bending)
Nylon	3290 cm^{-1} (N–H stretching), 2932 and 2858 cm^{-1} (C–H stretching), 1662 cm^{-1} (C O stretching), 1533 and 1464 cm^{-1} (CH_2 band), 1284 (N–H band), 687 cm^{-1} (N–H out-of-plane bending)
PET	1713 cm^{-1} (C O stretching), 1260 cm^{-1} (C–O stretching), 1096 cm−1 (C–O stretching), 720 cm^{-1} (aromatic CH out-of-plane bending)
PP	2922 cm 2854 cm^{-1} (C–H stretching vibration) 1456 (CH_2 bending) and 1375 cm (CH_3 bending) 1166 cm^{-1} (CH_3 rocking), 981 cm^{-1} (C–C stretching) and 847 cm^{-1} (CH_2 rocking)

PVC	2922 and 2854 cm^{-1} (C–H stretching),1427 cm^{-1} (–CH_2 bending), 1250 cm^{-1} (–CH bending) 600 and 650 cm^{-1} (C–Cl gouche bond)
PE	2914 cm^{-1} (C–H asymmetric stretching), 2846 cm^{-1} (C–H symmetric stretching), 1462 cm^{-1} (CH_2 bending vibration), 717 cm^{-1} (CH_2 rocking vibration)

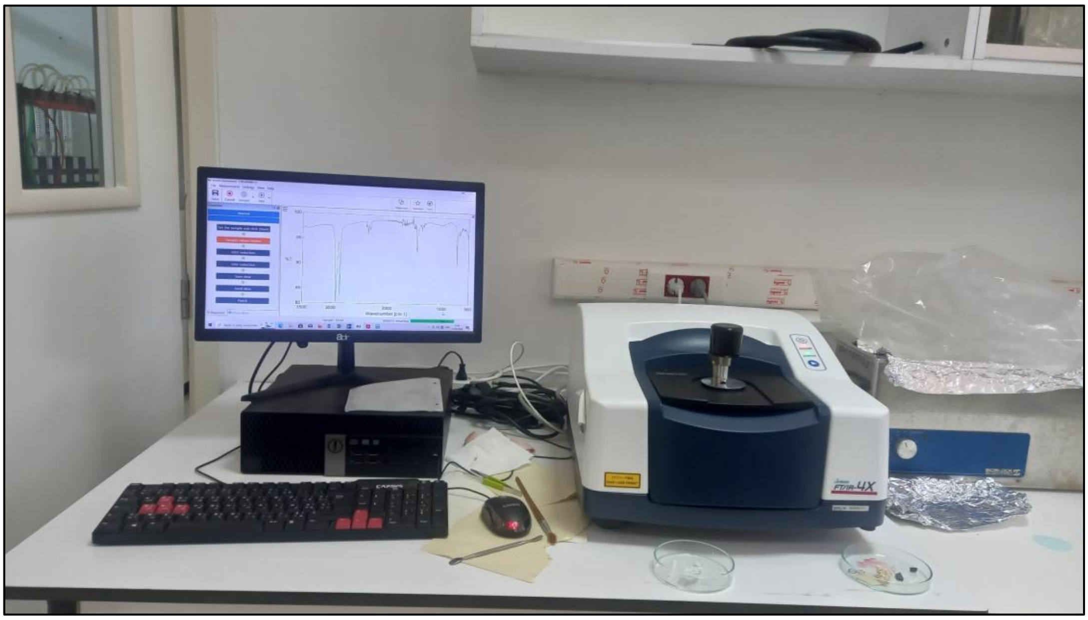


Figure 28: **FTIR spectroscopy system used for the chemical characterization of MPs**

III.4. Trace metal elements analysis

Trace metal analysis was conducted to determine metal concentrations in two environmental matrices, namely beach sediment and plastics.

III.4.1. Beach sediment

The methodology follows Quevauviller (1998) and US EPA (2015) recommendations.

Sample preparation: The sediment sample was freeze-dried for 72h to remove moisture without altering physicochemical properties or losing volatile metallic compounds. The sample was then homogenized, and 0.5 g of dry sediment was accurately weighed.

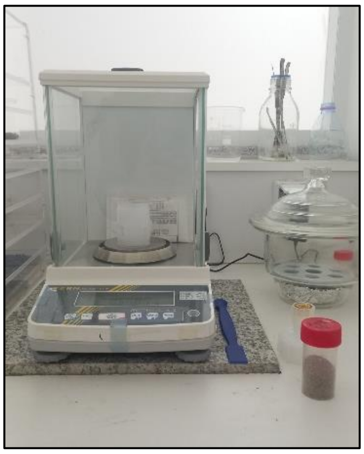


Figure 29: **Weighing of dried beach sediment sample**

Acid digestion: trace metals were extracted using aqua regia, a mixture of concentrated nitric acid (HNO_3; 68%) and hydrochloric acid (HCl; 37%) in a 1:3 (v/v) ratio. PTFE digestion containers (godet) were pre-cleaned by filling with 2 mL of aqua regia, hermetically sealing, and heating on a hot plate under a fume hood at 120 °C for 2 h 30 min. After cooling, containers were carefully vented, rinsed three times with ultrapure water (UPW), and oven-dried.

For sample mineralization, 0.5 g of dry sediment was transferred into a pre-cleaned digestion container, then 6 mL of HCl and 2 mL of HNO_3 were added. A procedural blank containing the same acid mixture was prepared and processed alongside the sample. Containers were hermetically sealed and left overnight at room temperature for cold pre-digestion, preventing abrupt exothermic reaction. Digestion was carried out on a hot plate under a fume hood at 120°C for 2 h 30 min. After cooling for 30 min, the digests were quantitatively transferred into 50 mL Teflon tubes. The digestion containers were rinsed at least three times with UPW, adjusted to a final volume of 50 mL, homogenized, and stored at + 4°C until analysis. Before ICP-OES analysis, the digests were filtered through a Millipore Millex-LCR PTFE syringe filter (0.45 µm) to remove residual particulates and prevent instrumental interference.

Figure 30: **Acids used for the preparation of aqua regia during trace metal extraction**

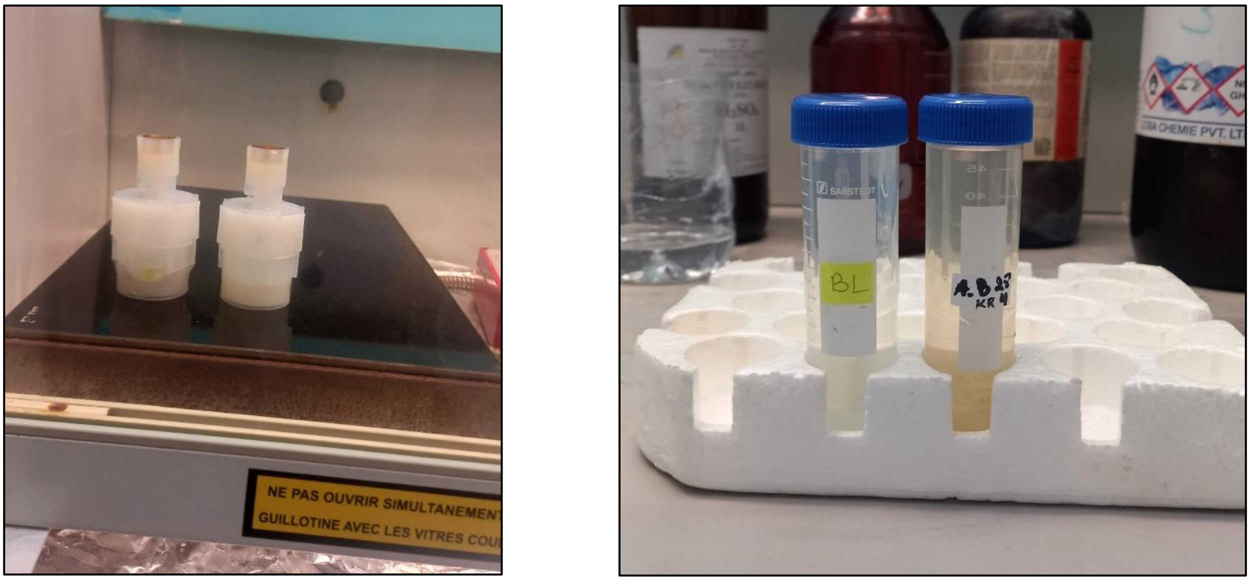


Figure 31: **Hot-plate mineralization of samples and the final diluted digest solutions prepared for analysis**

Figure 32: **Filtration step-up used before ICP-OES analysis**

III.4.2. Plastics

Figure 33: **MPs sonication step**

Surface cleaning & drying: To remove surface contaminants, plastic particles were transferred into a Teflon tube containing 10 mL of UPW and sonicated for 15 min at 40 kHz in an ultrasonic bath. The suspension was subsequently filtered through a 0.45µm membrane to recover the MPs. The operation was repeated three times until the filtrate appeared visually clear. The cleaned plastics were transferred to a petri dish and dried in an oven at 40°C for 24h. The temperature was deliberately kept below 60°C to prevent melting or deformation of thermoplastic particles (Hidalgo-Ruz et al., 2012b). Samples were subsequently placed in a desiccator for 30 min before weighing.

Acid digestion: Approximately 0.5 g of dried plastics was weighed and transferred into a clean PTFE digestion container. A mixture of 9 mL of HNO_3 (68%) and 3 mL of HCl (37%) was added, the container was hermetically sealed and heated on a hot plate under a fume hood at 200°C for 2 h 30 min. A laboratory blank containing the same acid mixture was included to account for reagent contamination. After cooling for 30 min, digests were transferred into 50 mL Teflon tubes, rinsed at least three times with UPW, brought to a final volume of 50 mL, filtered, and stored at + 4°C until analysis.

Acid digestion of MPs followed the general HNO_3/HCl-based approach established for plastic-metal analysis (Ashton et al., 2010; Brennecke et al., 2016; Holmes et al., 2012; Turner & Holmes, 2015), adapted to a hot-plate digestion system (200°C, 2h30min) given the absence of microwave digestion equipment, rather than the microwave-assisted protocols described by Hildebrandt et al. (2020).

III.4.3. Trace metal quantification

Metal concentrations were determined by Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES), and expressed in mg/Kg dry weight (d.w.).

ICP-OES principle: a multi-element analytical technique used to determine the elemental composition of samples. Its principle relies on atomic emission. The digest sample is aspirated by a peristaltic pump to a nebulizer, where it is converted into a fine aerosol and transported by argon gas into a high-temperature plasma (6000–10,000 K). In the plasma, the aerosol undergoes desolvation, vaporization, and atomization. The resulting atoms and ions absorb energy and become excited. As they return to their ground state, they emit light at characteristic wavelengths specific to each element. The emitted radiation is dispersed by the spectrometer and detected; its intensity is compared with calibration standards, and the software converts them into elemental concentrations (Boss & Fredeen, 1999; Skoog et al., 2007; Thermo Fisher Scientific, 2023).

Figure 34: **ICP-OES system used for trace metal analysis at CRNA laboratory**

Calibration range: ICP- OES calibration used five multi-element standard solutions at concentrations of 1, 2, 4, 5, and 6 ppm, each containing Nickel (Ni), Chromium (Cr), Copper (Cu), Zinc (Zn), Lead (Pb), and Cadmium (Cd), together with a calibration blank used as the zero point. For each element, signal intensity (Y) was regressed against standard concentration (X) as Y = aX + b, where a is the sensitivity (slope) and b the background signal (intercept); unknown sample concentrations were determined by interpolation on this line. The coefficients of determination (R^2) obtained were close to 1, confirming the excellent linearity and reliability of the analytical method.

Table 6: **Calibration curves and analytical characteristics of the standard solution (Sediment)**

Element	Calibration equation	R^2	LOD	LOQ	Wave length
Pb	Y=96.412754X+0.202526	0.999993	0.0063	0.0201	220.353
Ni	Y=568.344663X+0.376616	0.999997	0.0014	0.0046	231.604
Cd	Y=168.576631X+42.051380	0.999904	0.0440	0.1466	361.051
Cr	Y=1920.43835X+7.302144	0.999998	0.0015	0.0049	283.563
Cu	Y=2485.329705X+4.664338	0.999997	0.0019	0.0063	324.754
Zn	Y=1516.664038X+74.477387	0.999993	0.0003	0.0011	202.548

Table 7: **Calibration curves and analytical characteristics of the standard solution (Plastics)**

element	Calibration equation	R^2	LOD	LOQ	Wave length
Pb	Y=86.645457X+0.116072	0.999978	0.0069	0.0229	220.353
Ni	Y=573.576572X+4.323728	0.999972	0.0013	0.0045	231.604
Cd	Y=169.932055X+25.782131	0.999922	0.0427	0.1423	361.051
Cr	Y=2029.049658X+54.567069	0.999957	0.0014	0.0045	283.563
Cu	Y=2488.084903X+9.407406	0.999933	0.0018	0.0061	324.754
Zn	Y=1100.472495X+66.715084	0.999969	0.0004	0.0015	202.548

Figure 35: **Standard calibration range used**

The concentration of each metal was determined by interpolation of the measured emission intensities on the corresponding calibration curves. Final metal concentrations in the sediment samples were calculated after correction for blank contribution and dilution, according to the following equation:

$$C = \frac{Cc \times V}{Pc}$$

Where:

- **C:** metal concentration in sediment (mg/kg d.w),
- **Cc:** measured digest concentration (mg/L)
- **V:** final digest volume (L)
- **Pc:** corrected dry weight of digested sample (Kg)

III.5. Sedimentological characterization

III.5.1. Granulometric analysis

Sediment granulometric analysis was carried out using two techniques of wet and dry sieving to determine sediment textural parameters.

Experimental Protocol

- Beach sediment samples were placed in Petri dishes and dried in an oven at 105 °C for 24 h.
- After drying, a representative subsample of 200 g was weighed for analysis (P_1).

- The samples were then rinsed with tap water through a 63 µm mesh sieve, with manual agitation to facilitate the separation of the coarse fraction (>63 µm) and the fine (pelitic) fraction (<63 µm).
- After rinsing, the retained coarse fraction was collected, transferred back into the Petri dishes, and oven-dried again at 105 °C for 24 h, then weighed to determine its dry mass (P_2).
- The difference between P_1 and P_2 corresponds to the mass of the pelitic fraction removed.
- The mass of pelitic particles (P) was calculated using:

$$\boldsymbol{P\ (g) = P_1 - P_2}$$

Where:

  - **P**: mass of pelitic particles (g)
  - **P_1:** initial dry sediment mass (before rinsing)
  - **P_2:** final dry mass (after rinsing)

- The percentage of pelitic particles was calculated using:

$$\%\ \boldsymbol{Pelitic\ fraction} = \left[{}^{(P_1 - P_2)}/_{P_1}\right] \times \mathbf{100}$$

Figure 36: **Sequential steps for pelitic fraction determination**

Subsequently, dry sieving was performed according to the AFNOR standard NF P 94-056, using a series of sieves with decreasing mesh sizes (2 mm, 1 mm, 500 µm, 250 µm, 100 µm, 63 µm, and 45 µm).

- The dried sample was placed in a mechanical sieve shaker and subjected to vibration at 60 Hz for 10 minutes to ensure efficient particle separation.
- After sieving, the material retained on each sieve (sieve refusals) was carefully collected and weighed using a precision balance.
- The total recovered mass was verified by summing all fractions to ensure mass balance consistency and data quality control, and the grain-size data were used to calculate

sedimentological parameters (mean grain size (Mz), Sorting (So), Skewness (Sk_I), and Kurtosis (KG)) following the methodology proposed by Folk & Ward (1957). It is important to know the grain-size distribution of sediments as it reflects transport processes, weathering intensity, and erosional features (Boggs, 2006; Szcześniak et al., 2023; Tanabe et al., 2023).

Mean grain size (Mz):

The Mz is the average sediment particle size and the most representative parameter of the grain-size distribution, because it reflects the overall energy of the depositional environment. Coarser sediments are typically associated with high-energy settings, where strong waves and currents remove fine particles, whereas finer sediments tend to accumulate in low-energy environments. It's calculated as:

$$Mz = \frac{\phi_{16} + \phi_{50} + \phi_{84}}{3}$$

where φ16, φ50, and φ84 correspond to the 16th, 50th, and 84th percentiles of the cumulative grain-size distribution.

Value of Mz	Description
0 - 1	Coarse sand
1 - 2	Medium sand
2 - 3	Fine sand

Sorting index (So):

The sorting index describes the degree of uniformity of the grain-size distribution and evaluates the distribution of grain size around the mean. Well-sorted sediments exhibit a homogeneous granulometry, indicating relatively stable hydrodynamic conditions. In contrast, poorly sorted sediments reflect variable energy conditions. It is defined as:

$$So = \frac{\phi_{84} - \phi_{16}}{4} + \frac{\phi_{95} - \phi_5}{6.6}$$

Sorting (So) (θ)	Description
< 0.35	Very well sorted
0.35 – 0.50	Well sorted
0.50 – 0.70	Moderately well sorted
0.70 – 1.00	Moderately sorted
1.00 – 2.00	Poorly sorted
2.00 – 4.00	Very poorly sorted
> 4.00	Extremely poorly sorted

Skewness (Sk_I):

Skewness measures the symmetry of the grain-size distribution:

$$Sk_I = \frac{\phi_{16} + \phi_{84} - \phi_{50}}{2(\phi_{84} - \phi_{16})} + \frac{\phi_5 + \phi_{95} - 2\phi_{50}}{2(\phi_{95} - \phi_5)}$$

Positive values indicate a fine-tail distribution, whereas negative values indicate a coarse-tail distribution. Near-symmetrical distribution suggests a balanced sediment supply.

Skewness (Sk_I):	**Description**
+0.30 to +1.00	Very finely skewed
+0.10 to +0.30	Finely skewed
+0.10 to –0.10	Symmetrical
–0.10 to –0.30	Coarsely skewed
–0.30 to –1.00	Very coarsely skewed

Kurtosis (KG):

Kurtosis describes the degree of concentration of grain sizes around the mean (peakedness). A leptokurtic distribution indicates that the central fraction is better sorted than the tails. A platykurtic distribution reflects a more uniform spread across all grain sizes:

$$K_G = \frac{\phi_{95} - \phi_5}{2.44(\phi_{75} - \phi_{25})}$$

Kurtosis (KG)	**Description**
<0.67	Very platykurtic
0.67 – 0.90	Platykurtic
0.90 – 1.11	Mesokurtic
1.11 – 1.50	Leptokurtic
150 – 3.00	Very leptokurtic
<3.00	Extremely leptokurtic

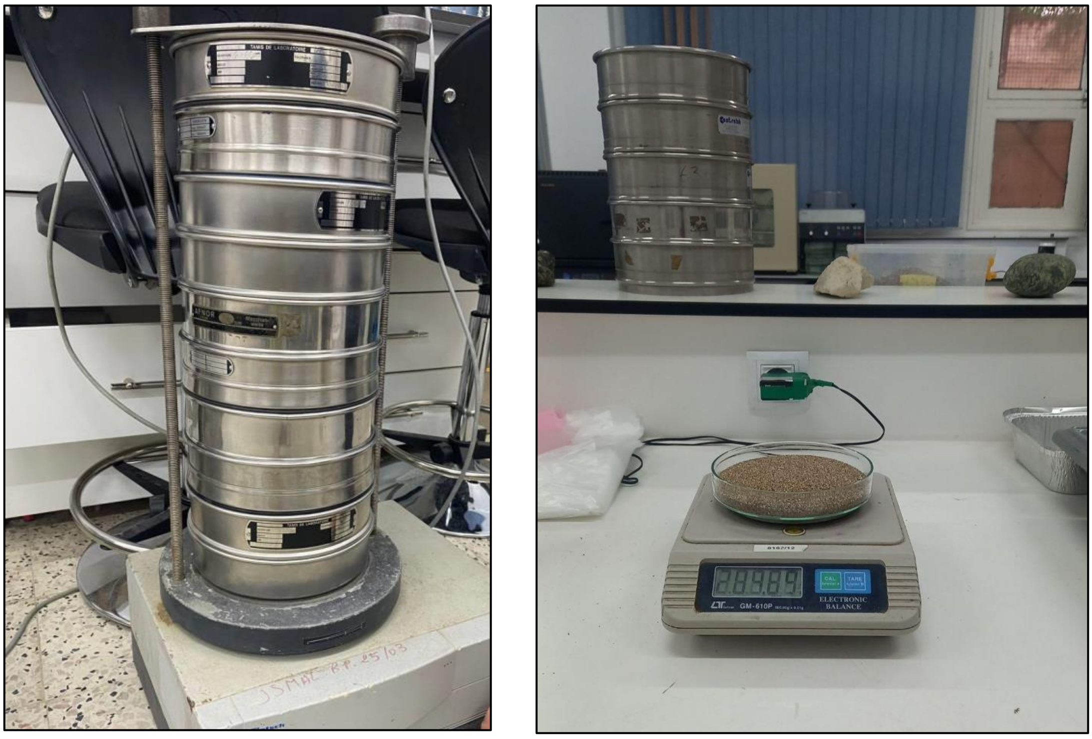


Figure 37: Granulometric analysis of sediment samples

III.5.2. Organic matter analysis – Loss on Ignition (LOI) method

Analysis procedure:

- Sediment samples were first dried in an oven at 105 °C for 24 h to remove all moisture content.
- After drying, samples were homogenized, and approximately 5 g of dry sediment was weighed for analysis.
- Clean porcelain crucibles were pre-heated in a muffle furnace at 550 °C for 1 h to eliminate any residual organic contamination, then cooled in a desiccator and weighed (M_1).
- The dried sediment subsamples were placed into the crucibles and weighed to obtain the initial mass of crucible + dry sediment (M_2).
- The samples were then combusted in a muffle furnace at 550 °C for 4 h to oxidize the organic matter.
- After ignition, the crucibles were removed using metal tongs and cooled in a desiccator to prevent moisture absorption.
- The crucibles containing the calcined sediment were then weighed to obtain the final mass (M_3).
- The Loss on Ignition (LOI), corresponding to the organic matter fraction, was calculated using the following equation:

$$\boldsymbol{OM}(\%) = \left[{(\boldsymbol{M_2} - \boldsymbol{M_3})}\big/{(\boldsymbol{M_2} - \boldsymbol{M_1})}\right] \times \mathbf{100}$$

Where:

- **M_1:** mass of empty crucible (g)
- **M_2:** mass of crucible + dry sediment before ignition (g)
- **M_3:** mass of crucible + sediment after ignition (g)

Figure 38: **Organic matter determination by LOI procedure**

Chapter III Results & Discussion

This chapter presents and discusses the results obtained from the analysis of microplastics and trace metal elements in sediment and seawater samples collected along the central Algerian coast. The findings are interpreted through statistical analyses and compared with previous studies to assess the extent of coastal contamination, identify potential pollution sources, and evaluate their environmental implications.

Given the intense anthropogenic pressures affecting the Algiers coastal zone, including urbanization, port activities, domestic and industrial discharges, and intensive seasonal tourism, this environment is particularly susceptible to contaminant accumulation. The combined investigation of MPs and trace metals provides a comprehensive understanding of coastal pollution, highlighting potential interaction between these contaminants, particularly the adsorption of trace elements onto plastic particles.

I. Sedimentary characteristics

I.1. Granulometric analysis

I.1.1. Pelitic fraction

The results of the pelitic fraction analysis for the study sites are presented in table 8.

Table 8: **Pelitic fraction (%) recorded at each study site**

Station	Pelitic fraction (%)
Sidi Fredj	0.26
Ain Benian	0.16

The pelitic fraction recorded in the beach sands of both study sites is relatively low. The maximum value is observed at Sidi Fredj West Beach (0.26%), while the minimum value is recorded at El Djamila, Ain Benian (0.16%). The very low pelitic fractions recorded at both sites indicate the predominance of a coarse-grained, sandy sedimentary system along the Algiers coastal, where fine particles appear to be either supplied in limited quantities or selectively removed by hydrodynamic sorting and transported toward deeper offshore depositional environments (De Falco et al., 2003, 2022; Ergin et al., 2007; Salem Cherif et al., 2019).

I.1.2. Granulometry analysis

The calculated textural parameters for the two study sites are summarized in table 9.

Table 9: **Textural characteristics of beach sediments at both study sites [mean (Mz), Sorting (θ), skewness (Ski), and kurtosis (KG)]** (Folk & Ward, 1957)**.**

Station	Mz (θ)	So (θ)	Sk	KG	Mz class	So class	Sk class	KG class
Sidi Fredj	1.482	0.814	+0.063	1.325	Medium sand	Moderately sorted	Near symmetrical	Leptokurtic
Ain Benian	1.397	0.475	–0.041	1.257	Medium sand	Well sorted	Near symmetrical	Leptokurtic

The Mz values of 1.482 θ at Sidi Fredj and 1.397 θ at Ain Benian classify the sediments at both sites as medium sand. This similarity suggests that the two beaches have comparable depositional environments and hydrodynamic conditions along the Algiers coastline. The predominance of medium sand also reflects moderate transport energy, which favors the deposition and accumulation of sediments within this grain-size range (Blott & Pye, 2001, 2012; Folk & Ward, 1957).

Regarding sorting, a notable difference is observed between the two sites. Sediments from Ain Benian are well sorted (So = 0.475), whereas those from Sidi Fredj are moderately sorted (So = 0.814). This suggests that sediment transport and deposition at Ain Benian occur under more uniform and stable hydrodynamic conditions, while Sidi Fredj experiences slightly greater variability in energy conditions, resulting in a broader range of grain sizes (Friedman, 1962).

The skewness values obtained for Ain Benian Sk_I = –0.041 and Sidi Fredj Sk_I = +0.063 indicate near-symmetrical grain-size distributions. This reflects a balanced contribution of fine and coarse particles, with no significant enrichment of either fraction (Folk, 1980). The slightly negative skewness at Ain Benian reflects a very weak tendency toward coarser particles, whereas the slightly positive skewness at Sidi Fredj indicates a minor enrichment in finer material (Friedman, 1967).

Furthermore, both sites display leptokurtic distributions (KG = 1.257 at Ain Benian and 1.325 at Sidi Fredj), indicating that most sediment grains are concentrated around the mean grain size. This suggests a relatively homogeneous sediment population and the predominance of similar transport and depositional processes at both beaches (Blott & Pye, 2001; Folk & Ward, 1957).

Overall, the granulometric characteristics of both sites are consistent with sandy beach environments subject to moderate to high hydrodynamic energy, with spatial variability in sorting reflecting differences in coastal morphology, degree of exposure, and the nature of local sediment inputs (Marin & Vah, 2024; Masselink & Hughes, 2014).

I.2. Organic matter content (OM)

The organic matter content measured in the beach sediments of the two study sites shows moderate values with slight spatial variation, indicating differences in OM accumulation along the Algiers coastline (fig 39). Sidi Fredj West Beach exhibited the highest OM content (8.59%), while El Djamila Beach (Ain Benian) recorded a slightly lower value (6.86%). The relatively higher OM at Sidi Fredj may reflect localized anthropogenic inputs, particularly from nearby tourist and recreational infrastructure, including the adjacent thalassotherapy center, which can enhance the deposition and accumulation of organic-rich particles within the coastal sediment (Sampaio et al., 2010). At El Djamila, although OM values were lower, they remained relatively high, suggesting persistent organic inputs from urban runoff originating from the densely populated municipality of Ain Benian, especially during rainfall events.

Overall, the OM values measured at both sites indicate moderate organic enrichment of the sediments. The observed variability is likely governed by a set of factors, including the nature of organic inputs (marine, terrigenous, and anthropogenic), local hydrodynamic conditions, and grain size distribution, the latter being a well-established control on OM retention capacity, as finer sediments offer a greater specific surface area for organic matter adsorption and preservation (Keil et al., 1994; Keil & Hedges, 1993; Kleber et al., 2021).

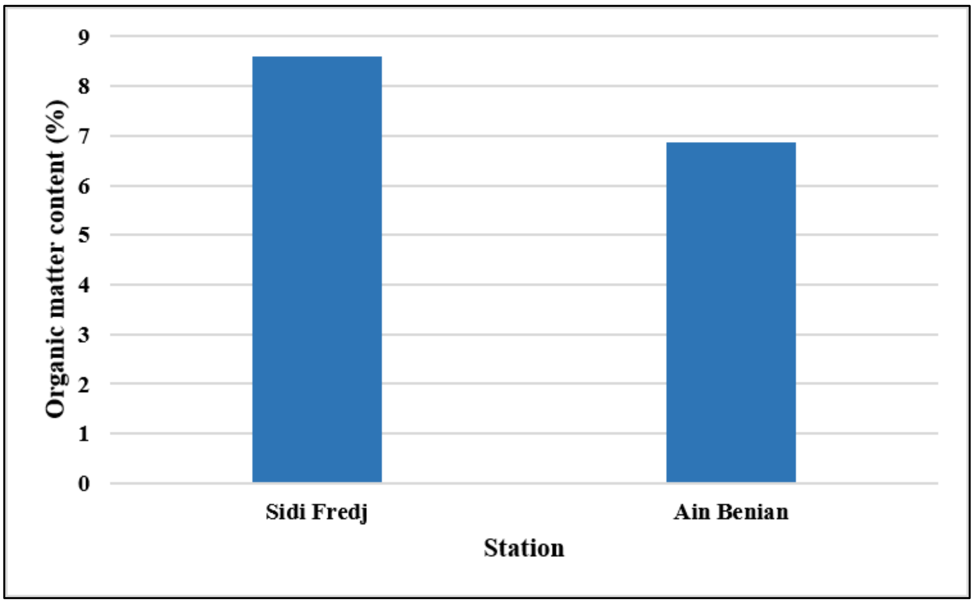


Figure 39: **Organic matter content (%) at each study site**

II. Microplastic characterization & distribution

Microplastics were detected at all sampled stations and matrices, confirming their widespread distribution along the Algiers coastline. Samples also contained biological materials, including algae, seagrass debris, and abundant planktonic organisms, reflecting the complexity of the coastal environment. The identified MPs were classified into five morphological categories: films, fibers, fragments, foams, and pellets, exhibiting a wide range of sizes and colors, including transparent, black, blue, red, and white. This diversity in shape, size, and color reflects diverse pollution sources and different stages of degradation in the marine environment.

The following sections present and discuss microplastic results in terms of abundance, morphology, color, and size distributions, with comparisons across the matrices and study sites to identify accumulation patterns, assess anthropogenic impact, and infer potential pollution sources along the studied coastline.

II.1. MP abundance

The quantitative analysis of MP abundance revealed considerable spatial heterogeneity between the two study sites and across the three environmental matrices (table 10).

At Sidi Fredj, beach sediment exhibited an abundance of 709.81 ± 295.74 MPs/Kg, while marine sediment recorded 773.39 ± 679 MPs/Kg, and seawater 184 MPs/L. At Ain Benian, beach sediment contained 697.98 ± 175.73 MPs/Kg, marine sediment 121.52 ± 46.83 MPs/Kg, and seawater reached 612 MPs/L.

Beach sediment concentrations were comparable among sites, though variability was greater at Sidi Fredj (±295.74), likely reflecting sediment redistribution by strong winds and wave events recorded during sampling, resulting in heterogeneous MP accumulation across the beach surface. The thalassotherapy center near the site may also contribute to the local MP’s input. In contrast, the lower variability observed at Ain Benian suggests a more homogeneous distribution of contamination (±175.73), likely associated with diffuse anthropogenic sources linked to recreational beach activities.

The greatest difference between sites was observed in marine sediments, where MP abundance at Sidi Fredj was more than six times higher than at Ain Benian. Marine sediments are recognized as important sinks for MPs, particularly for dense or biofouled particles that settle from the water column (Kaiser et al., 2017; S. Liu et al., 2022). The elevated concentrations and high variability recorded at Sidi Fredj may therefore reflect enhanced deposition and accumulation processes, combined with hydrodynamic disturbances redistributing MPs across the seafloor.

An opposite trend was observed in seawater, where Ain Benian exhibited concentrations more than three times higher than those measured at Sidi Fredj. This pattern may be associated with local anthropogenic inputs, including urban runoff from the municipality of Ain Benian, particularly during rainfall events, as well as fishing activities, wastewater discharges, and riverine contributions from the adjacent wadi (Beni Messous). Such sources are known to introduce large quantities of MPs directly into the water column, resulting in elevated suspended concentrations (Jambeck et al., 2015; Liffmann & Boogaerts, 1997; Schell et al., 2021; Weiss et al., 2024; Werbowski et al., 2021).

Table 10: **Average MP abundance observed across the study sites**

matrix	Beach sediment		Marine sediment		Seawater
Abundance	**MPs/Kg**	**MPs/m^2**	**MPs/Kg**	**MPs/m^2**	**MPs/L**
Sidi Fredj	709.81±295.74	3529.6±1241.02	773.39±679	5146.67±4274.59	184
Ain Benian	697.98±175.73	2481.6 ± 559.28	121.52±46.83	1173.33 ± 253.25	612

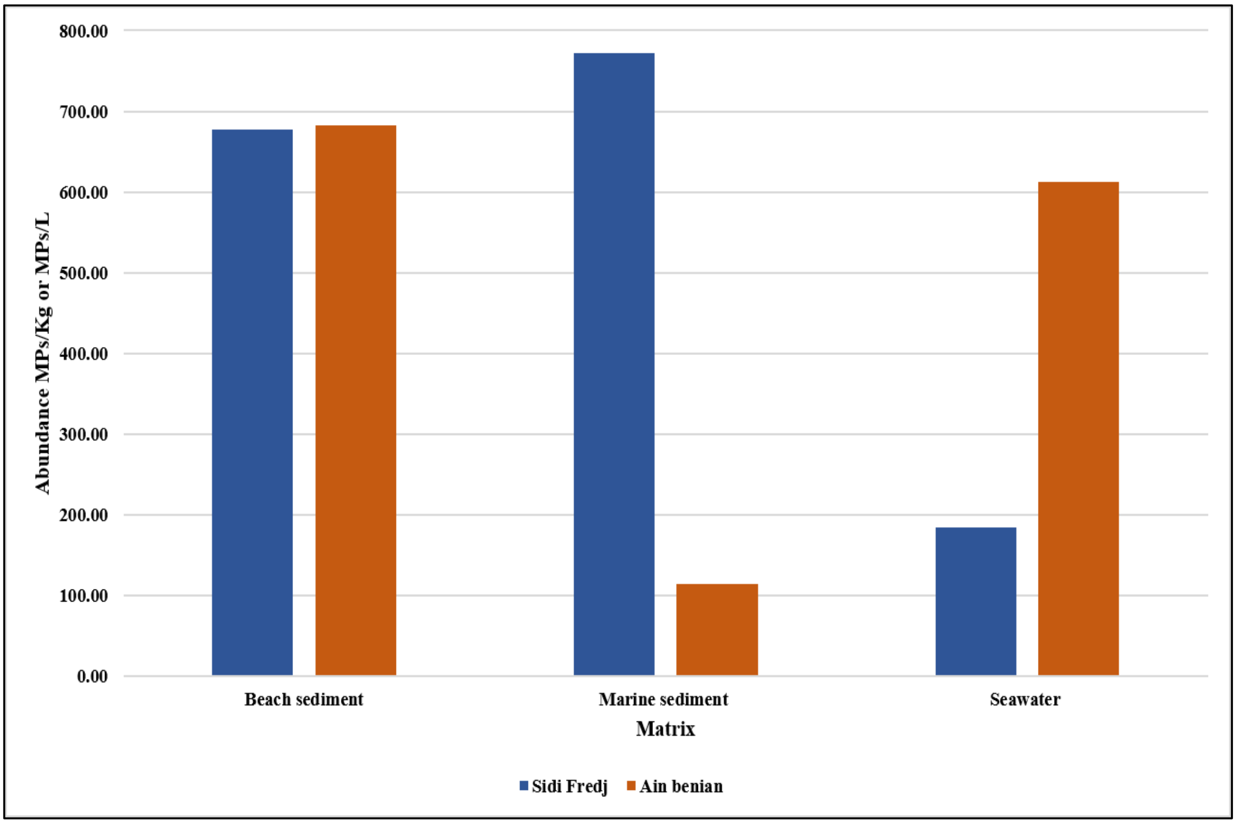


Figure 40: **Microplastic abundance in different coastal environmental matrices at the study sites**

II.2. Morphological characterization

The analysis of MP morphotypes revealed a clear predominance of fibers across all environmental matrices and sampling sites (Figures 41–43). In beach sediments, fibers constituted the most abundant morphotype, 560.92 MPs/Kg at Sidi Fredj and 336.24 MPs/Kg at Ain Benian. Fragments represented the second most abundant category, accounting for 113.38 and 255.7 MPs/Kg, respectively. A similar pattern was observed in marine sediments, where fibers dominated at both sites (approximately 649.01 MPs/Kg at Sidi Fredj and 105.59 MPs/Kg at Ain Benian). Fragments were detected in appreciable quantities only at Sidi Fredj marine sediments (approximately 122 MPs/Kg), while their occurrence at Ain Benian was minimal (0.65 MPs/Kg). In seawater, fiber dominance was even more pronounced. At Sidi Fredj, fibers accounted for nearly all detected MPs (183 out of 184 MPs/L), whereas at Ain Benian they represented 569 out of 612 MPs/L. Foams constituted the only notable secondary morphotype in the Ain Benian water samples, reaching approximately 37 MPs/L.

The predominance of fibers suggests that synthetic textiles, fishing, and related anthropogenic activities constitute the principal source of MP contamination in the study area. Their elongated shape, low density, and small diameter facilitate transport through wastewater treatment processes and subsequent dispersion within marine environments (Acharya et al., 2021; Andrady, 2011; Barrows et al., 2018; Browne et al., 2011). The particularly high abundance of fiber observed in Ain Benian seawater may reflect the combined influence of several local sources. The proximity of the wastewater treatment plant effluent and the adjacent Wadis provides a direct pathway for the introduction of fiber-rich MPs into the coastal environment. In addition, fishing activities occurring within the sampling area contribute to the degradation and abrasion of synthetic ropes and fishing nets, which are recognized sources of microplastic fibers (Napper et al., 2022; Song et al., 2025).

Fragments constituted the second most abundant morphotype, particularly in Ain Benian beach sediments (255.7 MPs/Kg) and Sidi Fredj marine sediments (122 MPs/Kg). These particles result from the progressive degradation of larger plastic items, including bottles, rigid containers, and packaging materials, through processes such as mechanical abrasion and photooxidation (Andrady, 2011; Brožová et al., 2025; Gewert et al., 2015). Their abundance in Ain Benian beach sediments may be associated with the accumulation of litter generated by recreational activities; in contrast, their higher occurrence in Sidi Fredj marine sediments may reflect subsequent transport from coastal sources and deposition within the benthic environment.

Foams were detected mainly at Ain Benian across matrices (55.67 MPs/Kg in beach sediments, 8.47 MPs/Kg in marine sediments, and 37 MPs/L in seawater), most likely derived from expanded polystyrene materials commonly used in packaging, fishing equipment, and floating devices (Song et al., 2020; Turner, 2020). Films were predominantly detected in Ain Benian beach sediments (≈ 35 MPs/Kg), consistent with the degradation of single-use plastic bags and packaging items (Jambeck et al., 2015).

In contrast, pellets were absent from all matrices and sites, suggesting no significant primary MP inputs from industrial plastic resin in the investigated area. However, it may also reflect methodological limitations related to sampling, extraction, and identification procedures, which could reduce the recovery of certain pellet sizes or types.

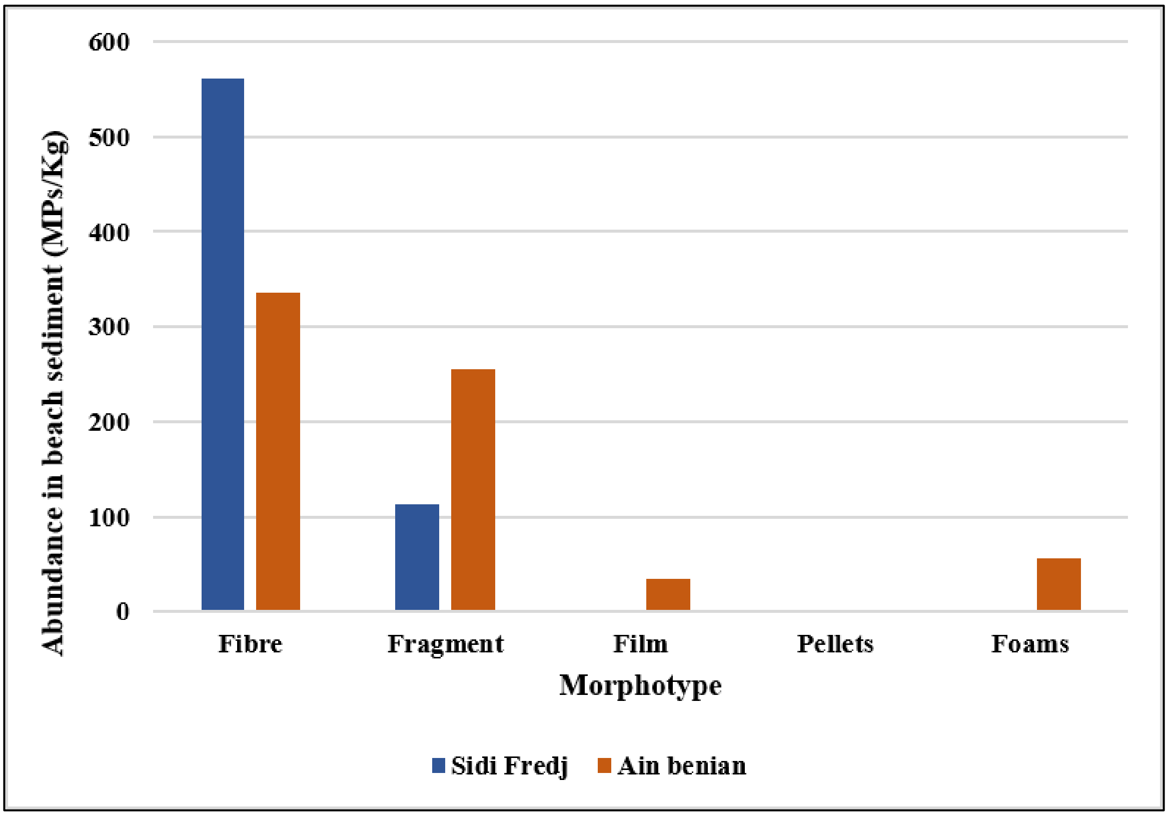


Figure 41: **Distribution of microplastic morphotypes in beach sediment at the study sites**

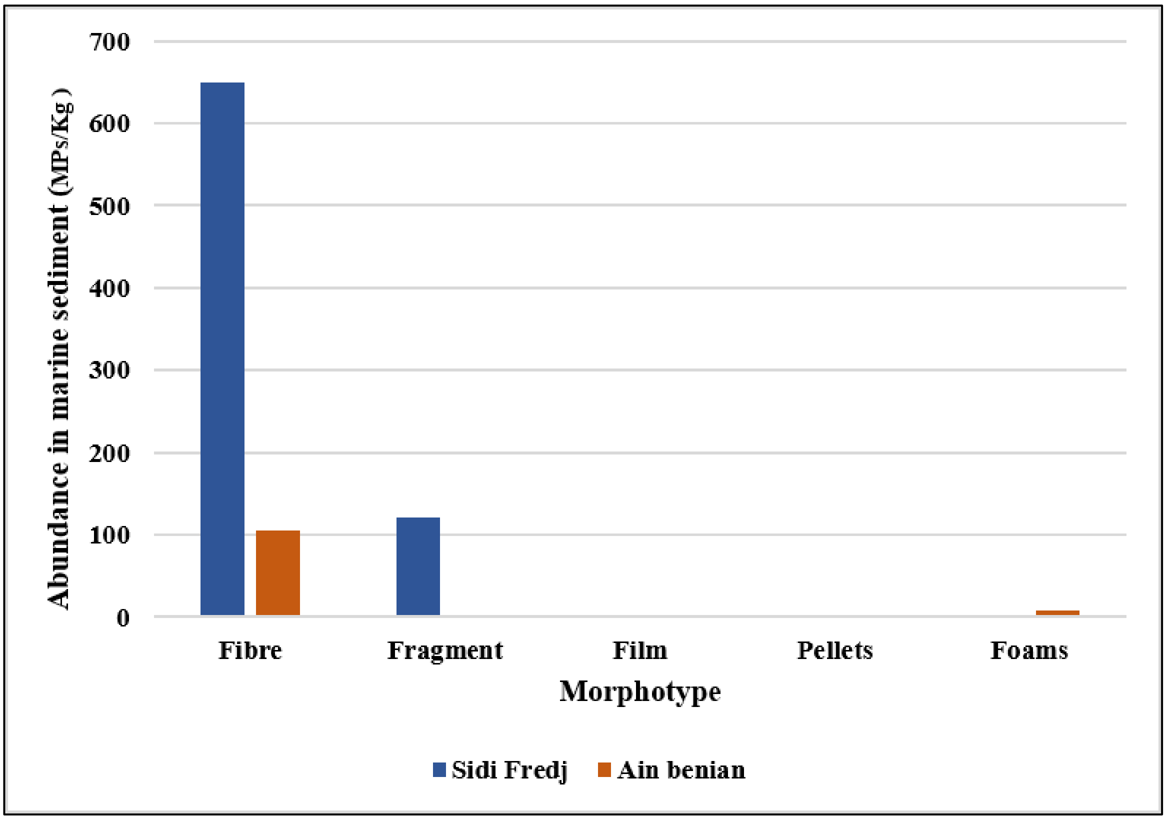


Figure 42: **Distribution of microplastic morphotypes in marine sediment at the study sites**

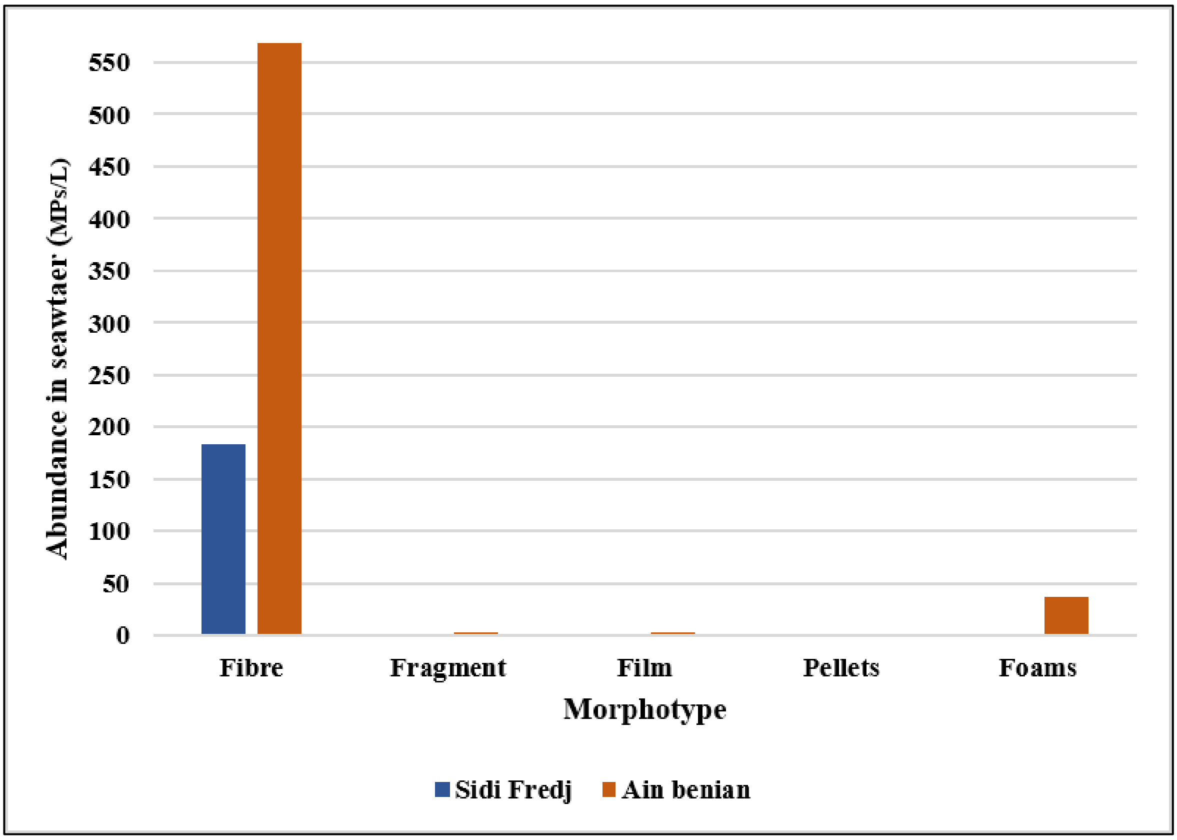


Figure 43: **Distribution of microplastic morphotypes in seawater at the study sites**

II.3. Color distribution

Transparent particles were the most abundant color category across all matrices and both sites. In beach sediment, transparent MPs dominated at Sidi Fredj (3545 particles) and Ain Benian (1287 particles), followed by blue (917 and 551, respectively), while other colors occurred in lower proportions. A similar pattern was observed in marine sediments and seawater, where transparent MPs remained predominant, with blue particles as the consistent secondary color.

The predominance of transparent MPs is likely linked to the widespread use of transparent plastics in single-use packaging, bottles, food containers, and fishing-related materials, which constitute a major source of coastal plastic litter (Andrady, 2011; Geyer et al., 2017). Blue particles are strongly associated with synthetic textiles, fishing ropes, and nets, supporting the morphological evidence of fiber dominance and suggesting fishing gears and domestic wastewater as primary pollution vectors at both sites, particularly at Ain Benian (Acharya et al., 2021; Barrows et al., 2018; Jambeck et al., 2015; L. Lebreton et al., 2022).

Black particles were relatively more abundant in Sidi Fredj marine sediments, potentially reflecting the accumulation of highly weathered plastics, degraded rubber, or bituminous materials. White particles, particularly common in Ain Benian beach sediments, may originate from expanded polystyrene and single-use food packaging, consistent with the occurrence of foam particles at the same site (Song et al., 2020; Turner, 2020).

Overall, the color distribution suggests multiple sources of MP contamination, including domestic plastic waste (transparent and white), synthetic textile and fishing-related inputs (transparent and blue), and weathered debris (black), consistent with the observed patterns of MP morphology and abundance across the study area.

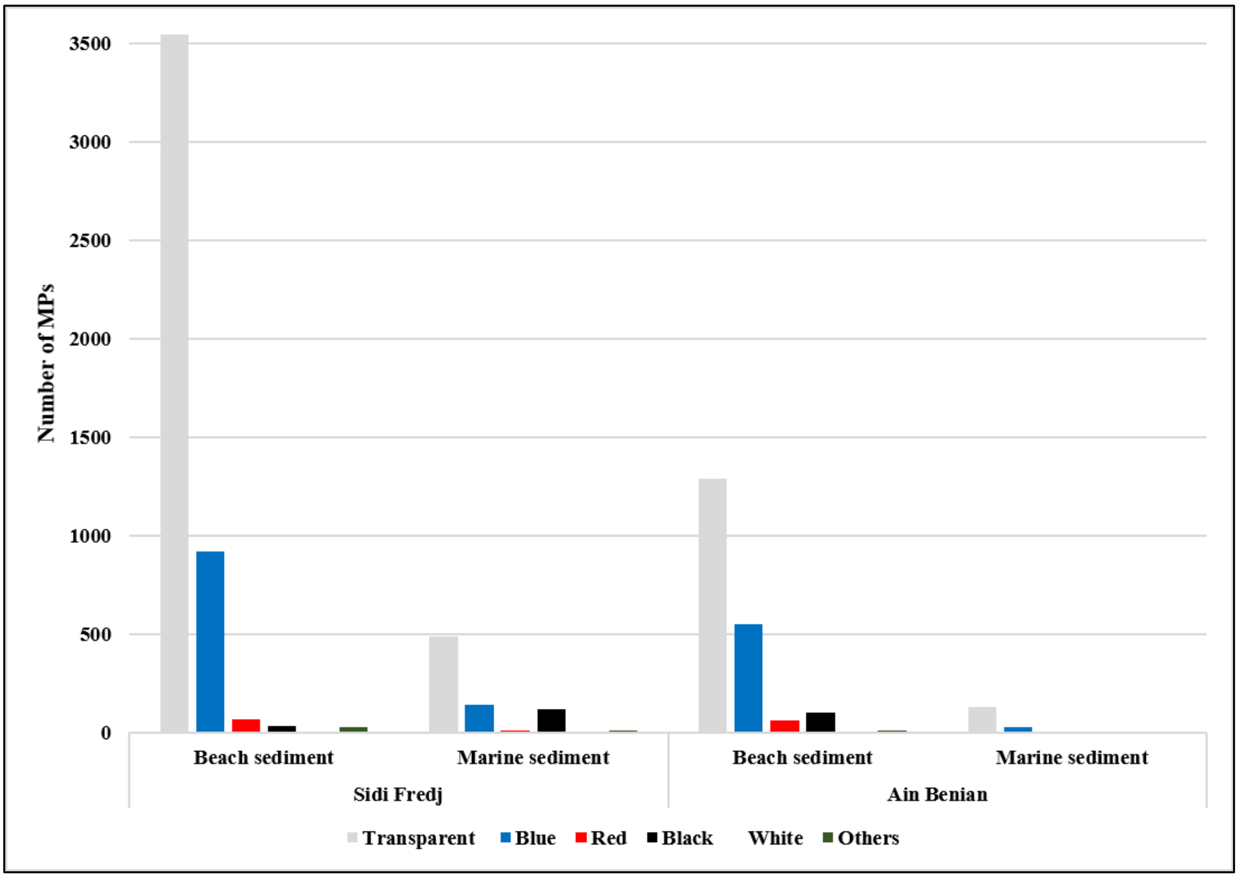


Figure 44: **Distribution of MP color categories in beach and marine sediments at study sites**

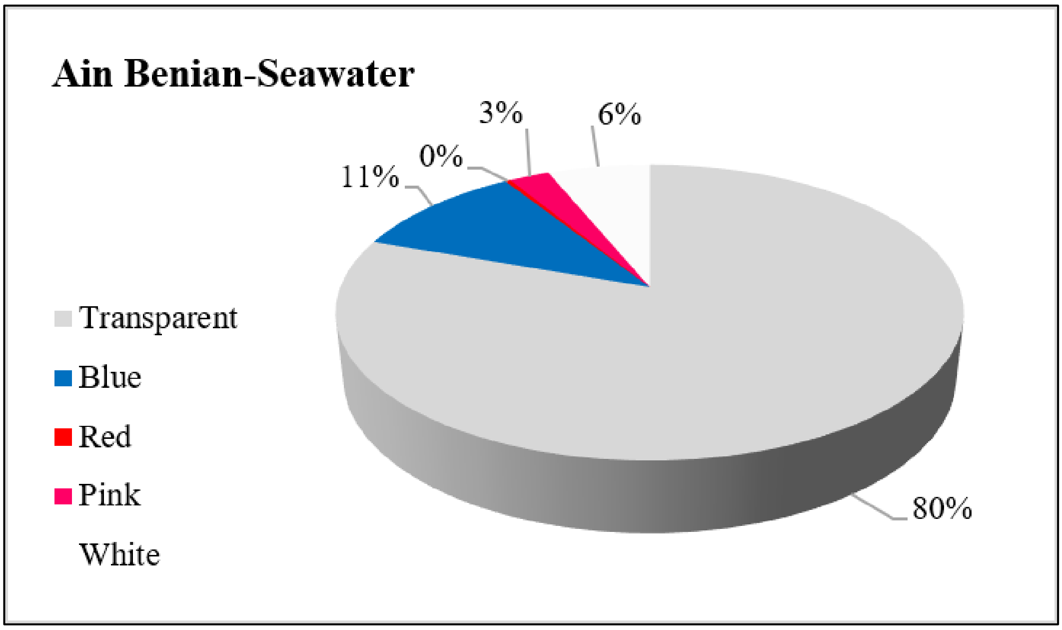


Figure 45: **Relative color distribution of microplastics in seawater at Ain Benian**

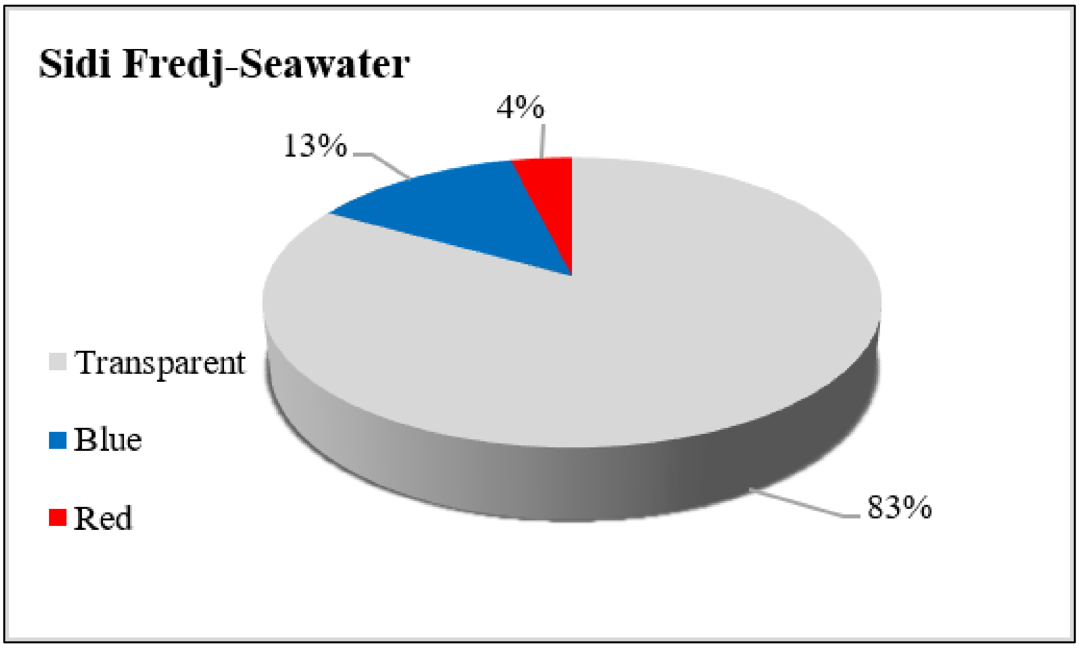


Figure 46: **Relative color distribution of microplastics in seawater at Sidi Fredj**

II.4. Size distribution

The size distribution of MPs across all matrices and both sites (figures 47 & 48) shows a clear dominance of the smallest fraction (1 µm–1 mm), particularly in sediments, indicating advanced fragmentation of plastic debris and long-term accumulation of fine particles in depositional environments (Andrady, 2011; Gewert et al., 2015). In beach sediments, this fraction was most abundant, with 3368 particles at Sidi Fredj and 2902 particles at Ain Benian, compared to 711 and 172 particles in marine sediments, respectively. The 1 mm–5 mm fraction was less frequent but still present, particularly in beach sediments (969 at Sidi Fredj and 162 at Ain Benian), while it was strongly reduced in marine sediments (60 and 4, respectively). The largest fraction (>5 mm) was rare, detected only in beach sediments (75 at Sidi Fredj and 38 at Ain Benian), and completely absent in marine sediments, suggesting limited offshore transport and/or rapid breakdown of larger debris under coastal exposure conditions (Brandon et al., 2016; Chamas et al., 2020). Seawater showed a different profile: at Ain Benian, the 1–5 mm fraction (328 particles) dominates the finest fraction (283 particles), while at Sidi Fredj, both fractions showed similar abundance (92 and 91 particles, respectively). Particles >5 mm were negligible at both sites (1 particle each).

Overall, these results highlight a dynamic coastal system governed by continuous plastic inputs, progressive mechanical and photochemical fragmentation, and the efficient retention of fine particles within sediments (Andrady, 2011; Cole et al., 2011). The predominance of small MPs in sediments reflects their preferential accumulation through biofouling-induced settling, prolonged residence time, and low hydrodynamic energy (Alimi et al., 2018; Kaiser et al., 2017). The scarcity of particles >5 mm, their absence from marine sediments, and the marked decrease in MP abundance from beach to marine sediments together reflect limited offshore transport of larger particles, which are more likely to remain trapped near the shoreline due to their size and lower mobility (L. C.-M. Lebreton et al., 2012). In contrast, the relatively higher proportion of intermediate-sized MPs in seawater, particularly at Ain Benian, likely reflects recent inputs from local sources such as fishing activities, wastewater effluents, and riverine discharges. These sources are recognized as major pathways of MPs into coastal waters, where hydrodynamic processes such as turbulence, currents, and vertical mixing can maintain particles in suspension before undergoing further degradation or eventual sedimentation (Chandra & Walsh, 2024; Leslie et al., 2017; Strokal et al., 2023).

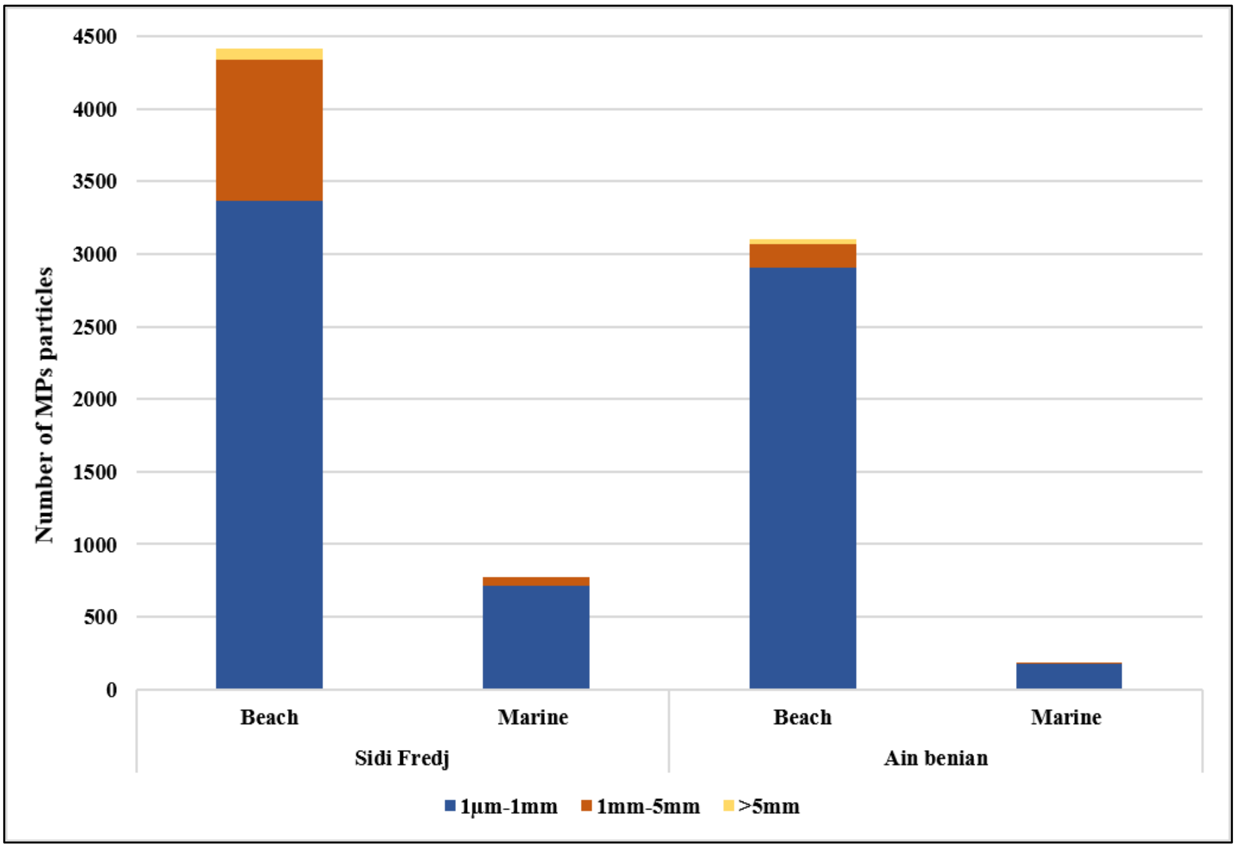


Figure 47: **Size distribution of microplastics in beach and marine sediments at the study sites**

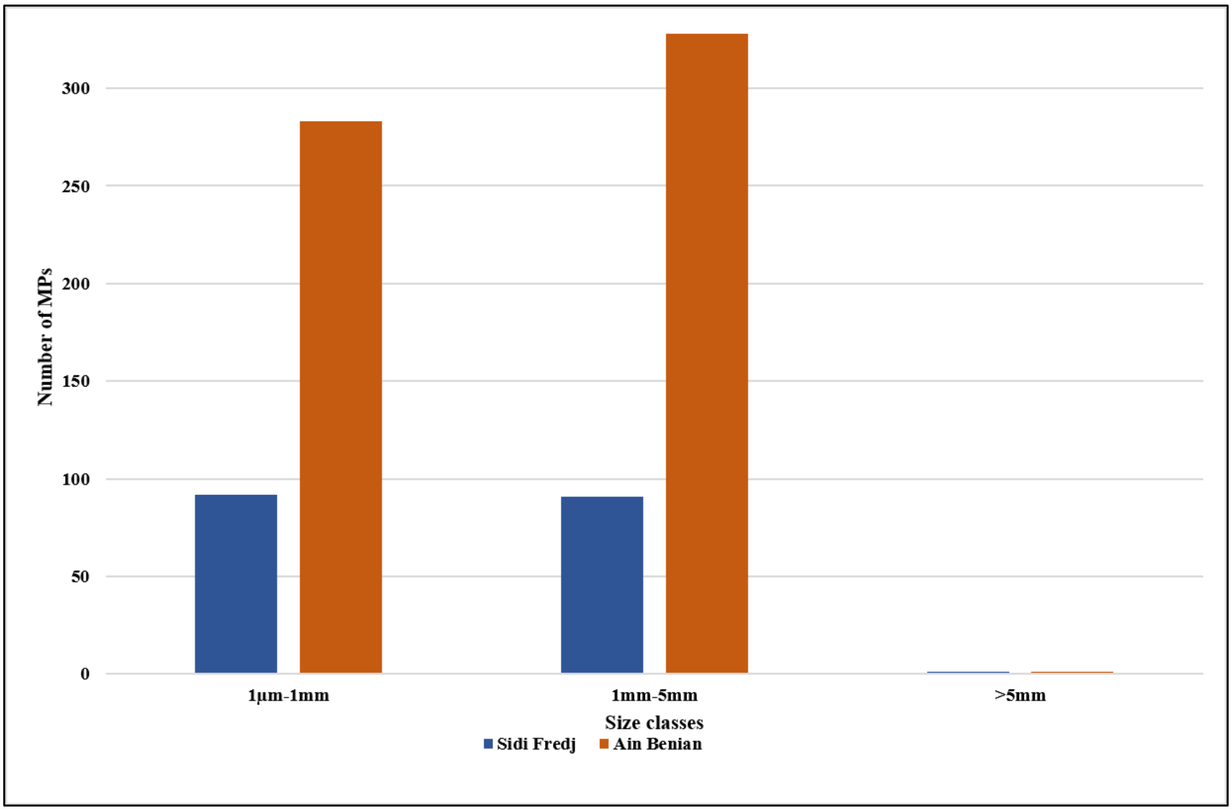


Figure 48: **Size distribution of microplastics in seawater at the study sites**

II.5. Polymer composition

The FTIR analysis shown in the figures has identified seven polymer types: polypropylene (PP) (13.64%), polyethylene (PE) (50%), high-density polyethylene (HDPE) (4.55%), polyethylene terephthalate (PET) (4.55%), polyvinyl chloride (PVC) (9.09%), polyamide (PA/Nylon) (9.09%), and polystyrene (PS) (9.09%), with PE being the most dominant. The prevalence of PE and PP suggests substantial inputs from packaging materials and other consumer plastics (bags, bottles, containers, etc.), as these polymers are among the most produced and frequently detected in marine microplastic pollution (Kowalczyk et al., 2025; Meng et al., 2025). The presence of PET is consistent with the dominance of fibers observed during morphological analysis, indicating a significant contribution from synthetic textile sources (F.-F. Liu et al., 2021). PA (Nylon) reflects fishing-related activities, whereas PVC and PS are commonly associated with urban infrastructure, household materials, and food-packaging products (W. Li et al., 2023; W. C. Li et al., 2016). Overall, the identified polymer assemblage indicates that microplastic contamination at both sites is predominantly derived from secondary plastics stemming from urban, domestic, and maritime sources. These findings are consistent with the morphological, size, and color characteristics of the recovered particles and further support the influence of multiple anthropogenic inputs on the study area(W. Li et al., 2023; Yang et al., 2021).

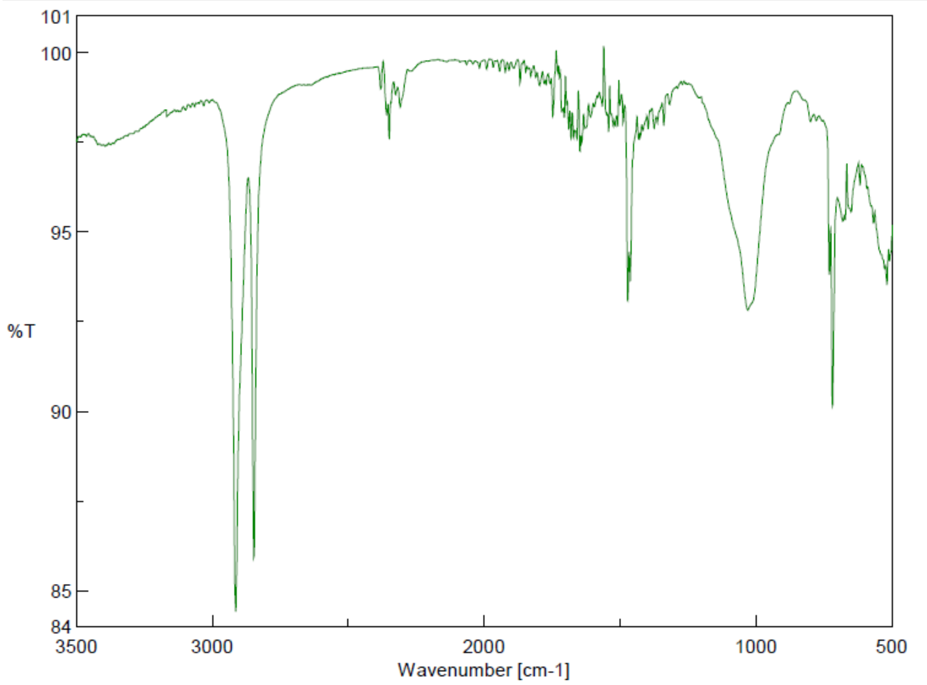


Figure 49: **FTIR Spectra of MPs identified as high-density polyethylene (HDPE)**

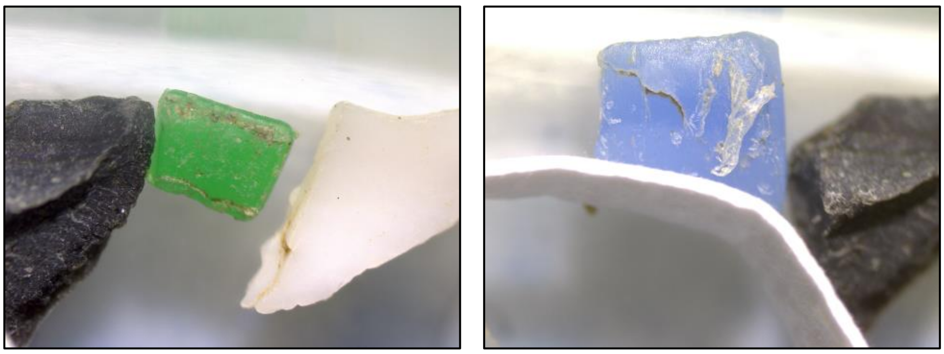


Figure 50: **Representative HDPE microplastic particles**

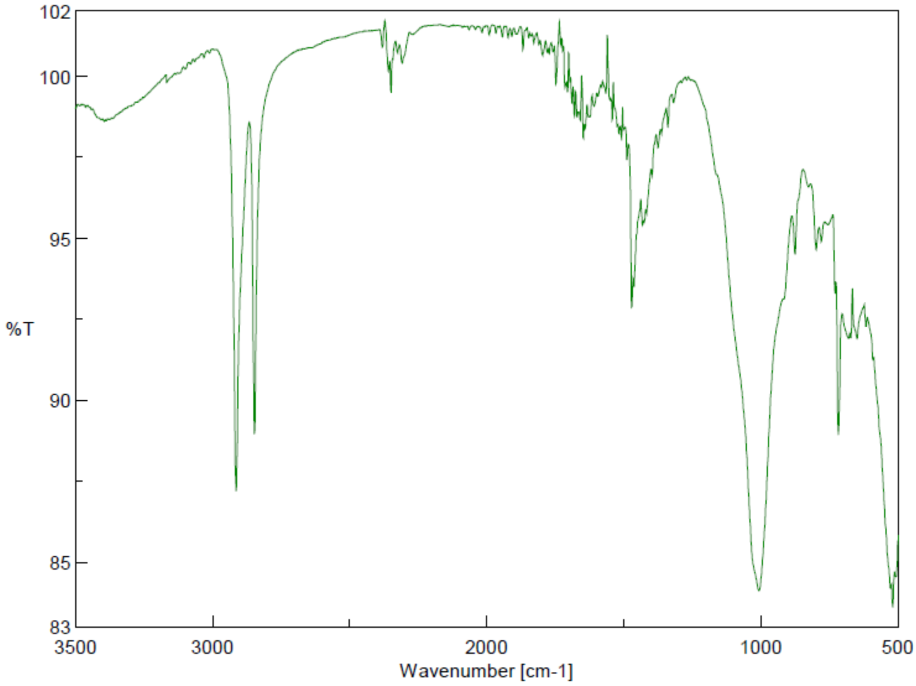


Figure 52: **FTIR Spectra of MPs identified as polyethylene (PE)**

Figure 51:* *Representative PE microplastic particles

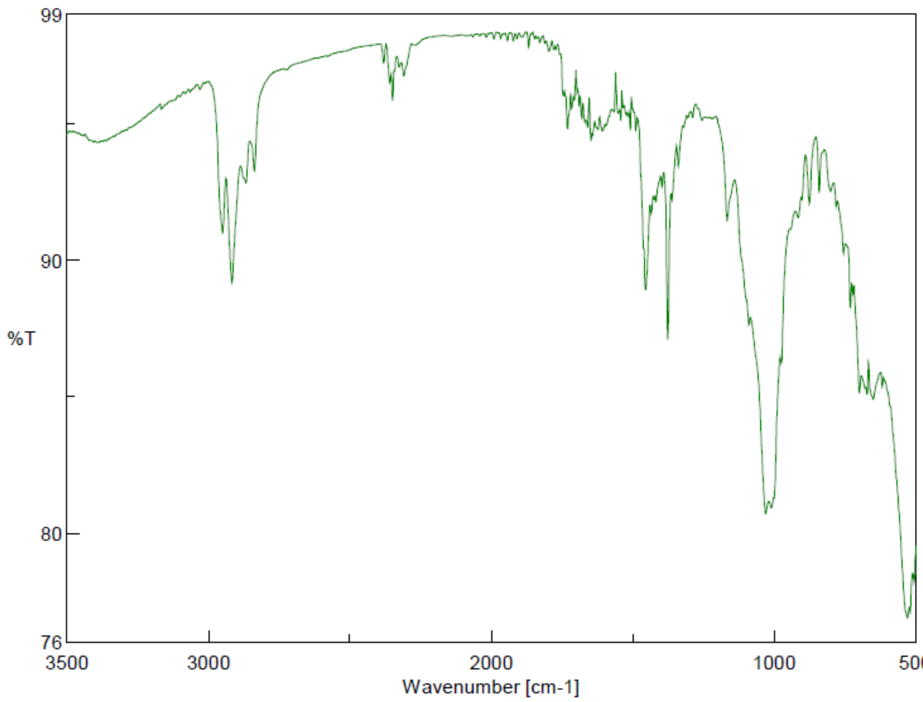


Figure 53: **FTIR Spectra of MPs identified as polyvinyl chloride (PVC)**

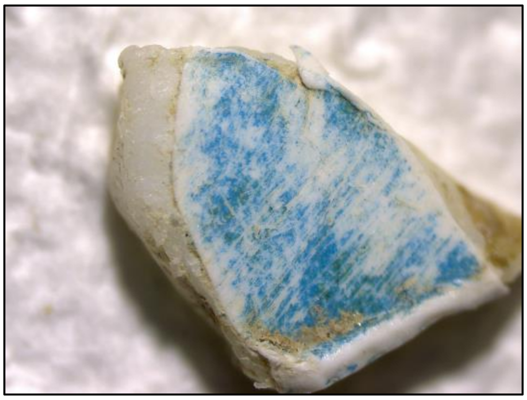


Figure 54: **PVC microplastic particle**

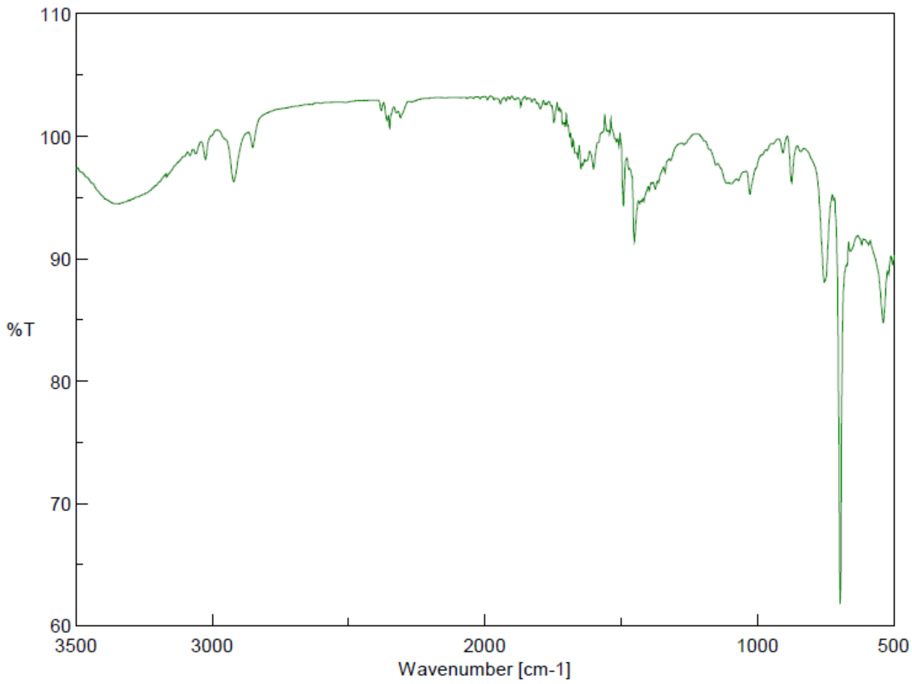


Figure 55: **FTIR Spectra of MPs identified as polystyrene (PS)**

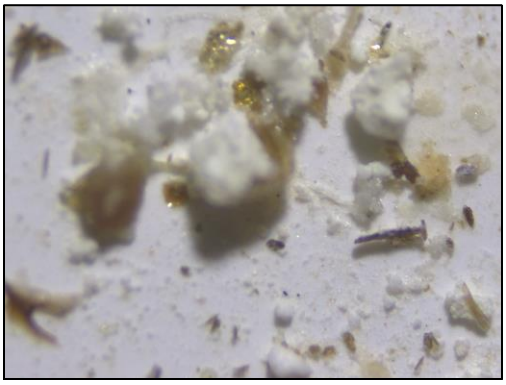


Figure 56: **Representative PS microplastic particles**

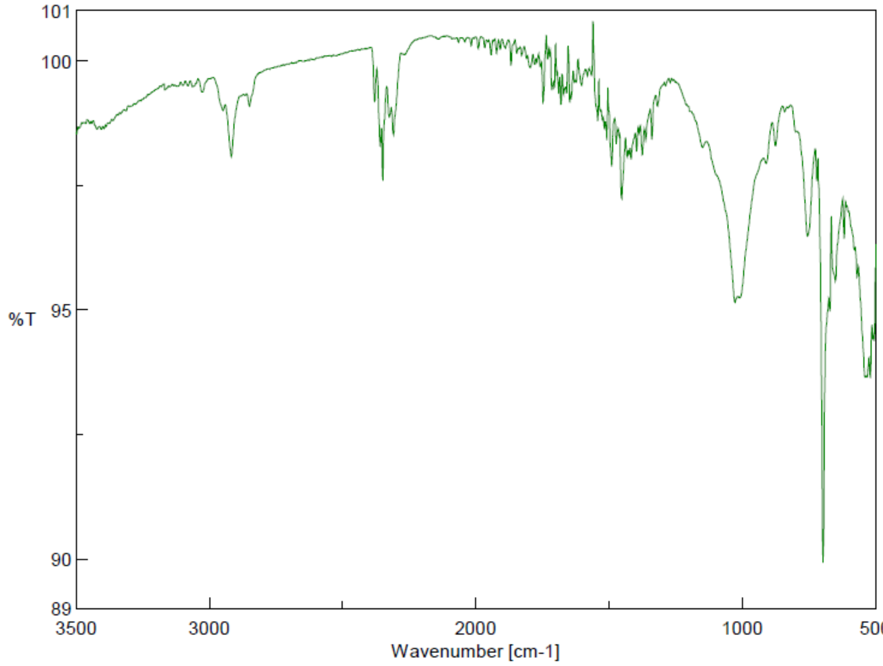


Figure 58: **FTIR Spectra of MPs identified as polyethylene terephthalate (PET)**

Figure 57: **Representative PET microplastic particles**

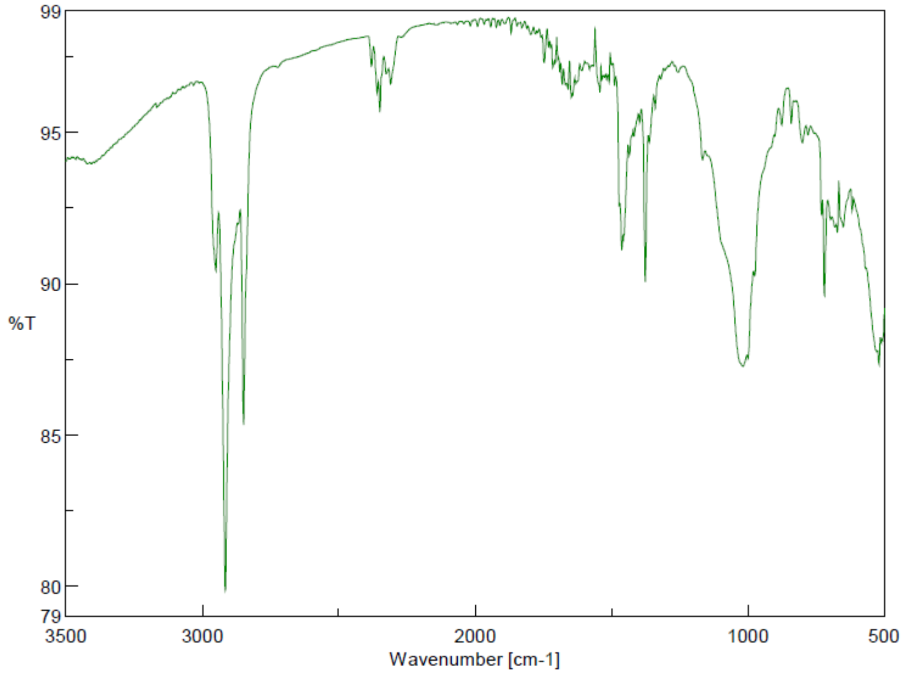


Figure 59: **FTIR Spectra of MPs identified as polypropylene (PP)**

Figure 60: **Representative PP microplastic particle**

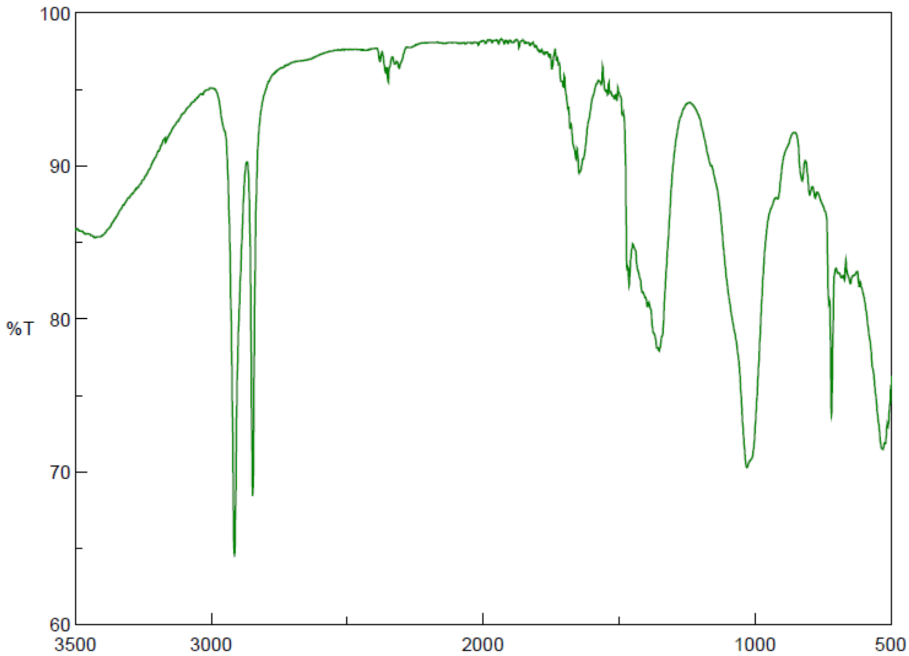


Figure 61: **FTIR Spectra of MPs identified as polyamide (PA)**

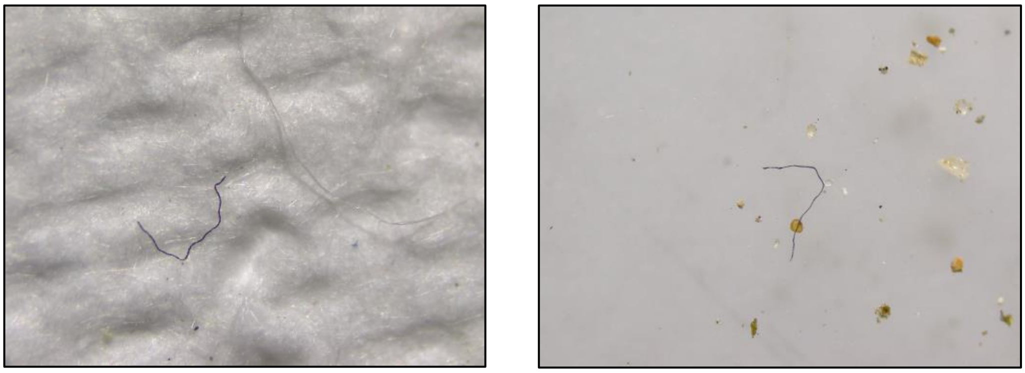


Figure 62: **Representative PA microplastic particles**

II.6. Comparative analysis

To assess the significance of the observed contamination, the concentrations measured at Sidi Fredj and Ain Benian were compared with values reported in previous studies from the Algerian coastline and the Mediterranean basin (table 11), providing a regional context for interpreting the contamination status of the study sites.

Microplastic analysis across all environmental matrices revealed pervasive contamination, reflecting strong anthropogenic pressure on the Algiers coastal environment. Differences in abundance and particle characteristics between sites point to distinct local conditions: Sidi Fredj exhibited greater MP accumulation in sediments, consistent with enhanced deposition and retention under storm-driven hydrodynamic conditions, while Ain Benian showed markedly higher waterborne concentrations, attributed to wastewater discharge, riverine inputs, and fishing activities.

The consistent dominance of fibers, fine particles (<1 mm), and transparent-to-blue coloration across all matrices indicates that contamination is predominantly of secondary MPs. The absence of pellets further suggests a negligible contribution from primary industrial sources. These findings were further supported by FTIR analysis, which identified a similar polymer composition at both sites, dominated by polypropylene (PP), together with PE, HDPE, PET, PVC, PA (Nylon), and PS, indicating inputs from packaging, textiles, urban infrastructure, and fishing-related equipment.

Comparison with published literature indicates that Sidi Fredj (3529.6 MPs/m²) and Ain Benian (2481.6 MPs/m²) rank among the higher microplastic concentrations reported along the Algerian coastline. These values exceed those documented at several western Algerian sites, including 55.47 ± 48.01 MPs/m² (Bentaallah et al., 2024) and 7.6 ± 66 items/m² (Taïbi et al., 2021), while remaining lower than the extreme levels reported at Skikda up to 6183 MPs/m² (Grini et al., 2024). At the Mediterranean scale, the concentrations observed in this study are higher than those reported in many less impacted coastal environments, confirming elevated microplastic pressure along the central Algerian coast. This pattern is mainly attributed to urban proximity, wastewater discharges, and intensive maritime and recreational activities, compounded by inadequate waste management, highlighting the need for long-term monitoring and targeted mitigation strategies to reduce plastic inputs into the marine environment.

Table 11: **Comparison of microplastic abundance in sediments and surface waters with Algerian and nearby regional studies.**

Location	Abundance	Shape	Size	Sample type	Polymer type	Reference
ALGERIAN STUDIES						
Sidi Fredj (this study)	709.81 MPs/kg (beach); 773.39 MPs/kg (marine); 184 MPs/L (SW)	Fibers dominant, fragments, foams	<1 mm dominant	Beach sed., marine sed., seawater	PP, PE, PET, HDPE, PS, PVC	**Current study**
Ain Benian (this study)	697.98 MPs/kg (beach); 121.52 MPs/kg (marine); 612 MPs/L (SW)	Fibers dominant, fragments, films, foams	<1 mm dominant	Beach sed., marine sed., seawater	PP, PE, PET, HDPE, PS, PVC, PA	**Current study**
Macta Marshes, NW Algeria (Ramsar site)	253.54 ± 273.84 MPs/kg (sed.); 3.82 ± 3.14 MPs/L (SW)	Fibers, fragments	SEM-measured	Sediment & surface water	HDPE, PET, PS, PE, ABS, ASA, PAN	**Chaalal et al. (2026)**
Gulf of Bejaia (Sidi Ali Labher, Aokas, Wadisdas)	— (MPPI, PPI, FIPI computed)	Fibers, fragments dominant	—	Beach sediment	—	**Zeghdani et al. (2026)**
Bou-Ismail Bay	0.89 ± 0.5 items/m³ (SW); 0.32 ± 0.13 items/g (sed.)	Fibers (SW), fragments (sed.)	<5 mm	Surface water & seafloor sediment	PE, PP, PS	**Amenouche et al. (2025)**
El-Mellah Lagoon Bizerte Lagoon, Tunisia	—	Fragments dominant, Fibers ~38%	<5 mm	Sediment	PE, PET, PP	**Khazr et al. (2025)**
Western coast (Stidia, Sablettes, Mersat El Hadjaj)	55.47 ± 48.01 MPs/m²	Fragments (79.81%), pellets (10.58%), filaments, foams	—	Beach sediment	Not identified	**Bentaallah et al. (2024)**
Skikda coast (persistence study)	6174 MPs/m² (spring); 6183 MPs/m² (autumn)	Fragments, pellets, foams, Fibers	—	Beach sediment	PE, PP	**Grini et al. (2024)**

Skikda coast (first survey)	106.98 ± 62.39 MPs/kg; 1067.19 ± 625.62 MPs/m²	Fragments, pellets	1–5 mm	Beach sediment	Not identified	**Grini et al. (2022)**
Annaba coast	182.66–649.33 MPs/kg	Fragments, Fibers	<5 mm	Beach sediment	PE, PET, PP	**Tata et al. (2020)**
Bou-Ismail Bay	—	Fibers, fragments, films, foams, granules	<330 µm	Surface water	PE, PP, PS	**Setiti et al. (2021)**
Western coast (Mostaganem–Arzew)	7.6–66 items/m²	Fragments (83.27%), pellets (14.93%)	≥1 mm	Beach sediment	Not identified	**Taïbi et al. (2021)**
MEDITERRANEAN BASIN						
Djerba Island, Tunisia	23–126.67 MPs/kg	Fragments, Fibers	—	Beach sediment	PE, PP	**Chouchene et al. (2025)**
Marmara Sea, Turkey (Istanbul)	13,891–25,156 MPs/kg	Fragments (53%), Fibers	50–150 µm dominant	Beach sediment	—	**Aytan et al. (2025)**
NE Spain, Pletera Beach	1863 MPs/kg (Fibers); 815 MPs/kg (fragments)	Fibers, fragments	—	Beach sediment	—	**Cocozza et al. (2025)**
Al Hoceima Bay, Morocco	—	Fibers, fragments, pellets, films	0.15–5 mm	Surface water	PE, PP, PS, PET	**Bouadil et al. (2024)**
Eastern Moroccan Mediterranean coast	40–230 MPs/kg (avg. 99.79 ± 44.19)	Fibers (74.72%), fragments (20.26%)	1–5 mm dominant	Beach sediment	PE, PS, PP	**Azaaouaj et al. (2024)**
Gulf of Tunis, Tunisia	109–531 MPs/kg (seasonal)	Fragments, filaments, Fibers	—	Beach sediment	PE, PP, PET	**Abidli et al. (2024)**

Three Italian seawaters (Adriatic, W. Med., Ionian)	0.029 MPs/m² (mean)	Fibers dominant	—	Surface seawater	PE, PP, PET	**ISPRA (2023)**
Thermaic Gulf, Greece (N. Aegean)	733.6 ± 136.6 MPs/m²	Fragments, Fibers	<1 mm dominant	Beach sediment	PE, PP	**Simantiris et al. (2023)**
Central Atlantic coast, Morocco	59.33 ± 34.38 MPs/kg	Fibers (77.61%), fragments (15.65%)	<1 mm (54.3%)	Beach sediment	PET, PP, PA	**Abelouah et al. (2022)**
Turkish Mediterranean coast	118–1688 MPs/kg; 0.18–2.21 MPs/m³ (SW)	Fragments dominant (>57.6%)	<1500 µm (>65%)	Beach sed. & surface water	PE (>59%)	**Gündoğdu et al. (2022)**
Spanish Mediterranean continental shelf	45.9–280.3 MPs/kg (avg. 113.2 ± 88.9)	Fragments, Fibers	—	Marine sediment	PE, PP	**Filgueiras et al. (2019)**
North Adriatic Sea, Italy	1512 MPs/kg	Fibers, fragments	—	Beach sediment	PE, PP	**Lots et al. (2017)**

III. Trace metal elements

The determination of trace metals in beach sediments and plastic samples provides important information on the contamination status of the studied coastal area. Sediments act as sinks for metallic pollutants due to their ability to accumulate and retain contaminants through adsorption and sedimentation (Förstner & Wittmann, 1981; Quevauviller, 1998). Plastics, especially MPs may also adsorb trace metals and thereby contribute to their transport and redistribution in the marine environment (Amelia et al., 2021; Brennecke et al., 2016; Hartmann et al., 2017; Holmes et al., 2012; Rafa et al., 2024; Squadrone et al., 2022). Taken together, these data help assess pollution levels and may identify possible anthropogenic and natural sources.

III.1. Beach sediment

Trace metals measured in the El Djamila sediment can be grouped into non-essential toxic elements (Cd, Pb), which have no biological role and are harmful even at trace levels, and essential trace elements or oligo-elements (Cr, Cu, Ni, Zn), which are required in small amounts but become toxic above threshold concentrations due to their persistence and bioaccumulation (Ali et al., 2019) (table 12).

Toxic, non-essential elements (Cd, Pb)

- Lead (Pb) was the only toxic-category metal quantified. Its concentration (5.9 ± 0.037 mg/Kg d.w) is far below the Effects Range–Low (ERL) guideline (Long et al., 1995) of 46.7 mg/Kg, consistent with diffuse, low-level inputs such as urban runoff from nearby roads and residential areas and atmospheric deposition rather than a point-source contamination.
- Cadmium (Cd) was not reliably detected in this analysis. The raw result produced an anomalous negative value, indicating the measurement is invalid rather than evidence of very low Cd. Possible causes include spectral or matrix interference on the Cd 361.051 nm line (a less sensitive, more interference-prone Cd line than the usual 214.4/226.5 nm).

Essential trace elements (Cr, Cu, Ni, Zn)

- Cr recorded the highest concentration (15.89 mg/Kg), attributable predominantly to lithogenic inputs from terrigenous minerals, yet still well below the natural geochemical background for Mediterranean shales (90 mg/Kg; Turekian & Wedepohl (1961)).
- Zn (9.53 mg/Kg) may receive a minor contribution from diffuse anthropogenic sources (road traffic, coastal urbanization), though the level remains non-concerning.
- Ni (4.35 mg/Kg) reflects lithogenic dominance, consistent with naturally metal-bearing geological formations in the region.
- Cu (1.05 mg/Kg) is the lowest, with no identifiable significant anthropogenic source at this site.
- All four elements were quantified well below their respective ERL and Effects Range–Medium (ERM) threshold (table 13), indicating no ecotoxicological concern for this sample.

Table 12: **Concentrations of trace metals in sediment sample (mg/Kg)**

Element	Concentration (mg/Kg)	SD	CV (%)	LOQ(mg/Kg)	LOD (mg/Kg)	Recovery (%)
Pb	5.9	0.037	6.289	2.01	0.63	98.48
Ni	4.35	0.09	2.102	0.46	0.14	98.48
Cd	< LOD	3.19	1.331	14.66	4.4	100.2
Cr	15.89	0.07	0.4476	0.49	0.15	98.78
Cu	1.05	0.04	3.41	0.63	0.19	98.95
Zn	9.53	0.09	0.9033	0.11	0.03	100.73

Table 13: **Assessment of sediment trace metal contamination against ERL and ERM ecological threshold values** (Long et al., 1995)

Element	mg/kg d.w	ERL	ERM
Cr	15.89	81	370
Cu	1.05	34	270
Ni	4.35	20.9	51.6
Pb	5.90	46.7	218
Zn	9.53	150	410

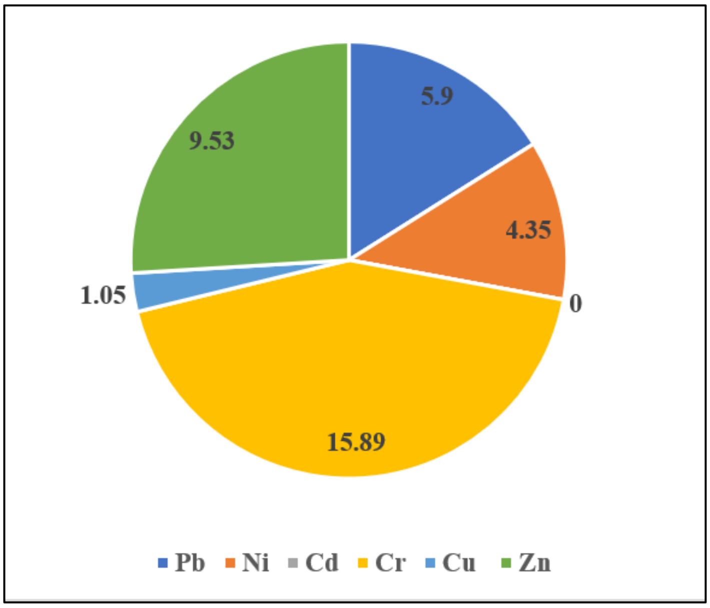


Figure 63: **Concentrations of trace metals in El Djamila beach sediment (mg/Kg)**

To further characterize the degree of metal enrichment, contamination indices: the Geoaccumulation Index (Igeo; Müller, 1969), Contamination Factor (CF; Hakanson, 1980), and Pollution Load Index (PLI; Tomlinson et al., 1980) were calculated using average shale values as geochemical background (Turekian & Wedepohl, 1961).

$$I_{geo} = \log_2\left(\frac{C_n}{1.5 \times B_n}\right) \quad ; \quad CF = \frac{C_{measured}}{C_{background}}$$

Where C_n= measured concentration, B_n= geochemical background value.

Pollution Load Index (PLI) = (CF_Pb × CF_Ni × CF_Cr × CF_Cu × CF_Zn)^(1/5) ≈ **0.057**.

Table 14: **Contamination indices values**

Element	Igeo	Classe	CF
Pb	**−2.35**	**0**	**0.30**
Ni	**−4.89**	**0**	**0.06**
Cr	**−3.09**	**0**	**0.18**
Cu	**−6.00**	**0**	**0.02**
Zn	**−3.91**	**0**	**0.10**

- All Igeo values were below zero, placing the sediment in Class 0, which corresponds to an uncontaminated or practically uncontaminated status.
- CF values were below 1 for all elements, indicating low contamination.
- The calculated PLI of approximately 0.057 is below 1, which suggests no overall deterioration of sediment quality attributable to trace metal loading.

The low metal concentrations and low contamination indices are consistent with the sediment texture. The well-sorted medium-sand matrix, together with a pelitic fraction of only 0.16%, likely reduces the availability of fine particles and organic matter that typically bind trace metals in coastal sediments (Chen et al., 2016; Förstner & Wittmann, 1981; Wang et al., 2024). On this basis, the results are compatible with a predominantly lithogenic geochemical background and limited anthropogenic enrichment.

III.2. Plastic samples

The quantified metals in the El Djamila plastics followed the order Zn (65.59) > Pb (57.67) > Cr (15.78) > Cu (8.70) > Ni (2.59) mg/Kg d.w. (table 15). This pattern may reflect a combination of residual additive-associated metals from polymer manufacture and subsequent environmental sorption onto weathered plastic surfaces. The relatively high Zn and Pb concentrations may be consistent with historical use of these elements in stabilizers, pigments, and other plastic additives, together with uptake from the surrounding environment after release (Ashton et al., 2010; Fajković et al., 2022; Hahladakis et al., 2018; Holmes et al., 2012; Turner, 2020). In contrast, Cr and Cu may also reflect environmental adsorption, and Cu may be influenced by localized maritime activities such as antifouling paints in coastal or harbor-influenced settings (Brennecke et al., 2016; Zou et al., 2020). Ni occurred at the lowest concentration, which may reflect its weaker affinity for plastic surfaces and a stronger association with lithogenic sediment (Ashton et al., 2010). Overall, the data are consistent with microplastics acting as potential carriers of trace metals in the study area (Fajković et al., 2022; Hahladakis et al., 2018).

Table 15: **Concentrations of trace metals in plastic samples (mg/Kg)**

Element	Concentration (mg/Kg)	SD	CV (%)	LOQ(mg/Kg)	LOD (mg/Kg)	Recovery (%)
Pb	**57.67**	**0.47**	**0.8102**	**2.29**	**0.69**	**97**
Ni	**2.59**	**0.13**	**5.142**	**0.45**	**0.13**	**97.6**
Cd	**< LOD**	**1.45**	**6.983**	**14.23**	**4.27**	**101.1**
Cr	**15.78**	**0.09**	**0.5655**	**0.45**	**0.14**	**97.7**
Cu	**8.7**	**0.18**	**2.085**	**0.611**	**0.18**	**100.3**
Zn	**65.59**	**0.25**	**0.375**	**0.15**	**0.04**	**96.3**

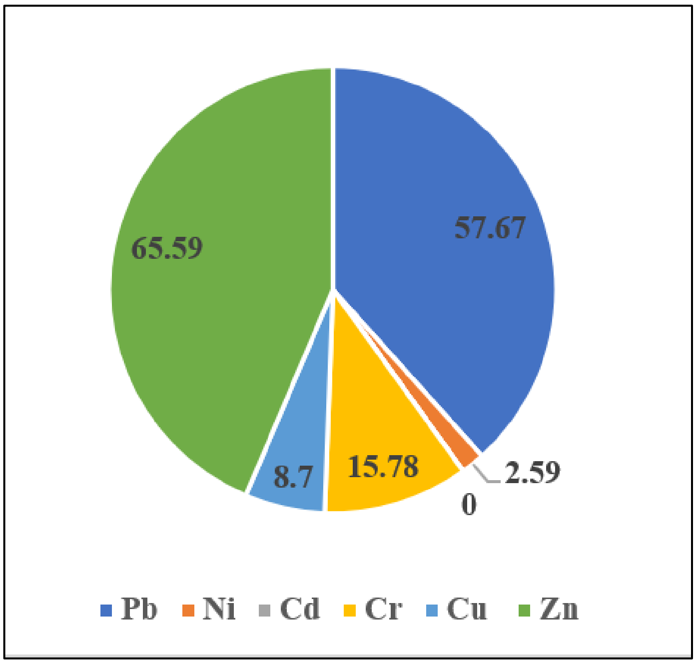


Figure 64: **Concentrations of trace metals in plastics sample (mg/Kg)**

III.3. Comparative metal loads: sediment vs plastic

Comparison between sediment and plastic matrices revealed distinct metal partitioning patterns, reflecting matrix-specific accumulation mechanisms (table 16). Zn, Pb, and Cu showed marked enrichment in plastics relative to sediment (6.88×, 9.77×, and 8.29×, respectively), suggesting combined contributions from residual polymer additives and environmental sorption onto weathered microplastic surfaces (Brennecke et al., 2016; Hahladakis et al., 2018; Turner & Holmes, 2015). Cr displayed nearly identical concentrations in both matrices, indicating limited matrix selectivity and a broadly distributed environmental background. In contrast, Ni was lower in plastics than in sediment, consistent with its predominantly lithogenic origin and weaker sorption affinity (Ashton et al., 2010; Brennecke et al., 2016). Overall, these results support the role of plastic debris as selective carriers and reservoirs of trace metals in Mediterranean and Atlantic coastal environments (Gao et al., 2021; Munier & Bendell, 2018; Squadrone et al., 2022).

Table 16: **Trace metal distribution between sediment and plastics**

Element	Sediment	Plastic	Plastic / Sediment ratio
Pb	5.9	57.67	9.77
Ni	4.35	2.59	0.6
Cr	15.89	15.78	0.99
Cu	1.05	8.7	8.29
Zn	9.53	65.59	6.88

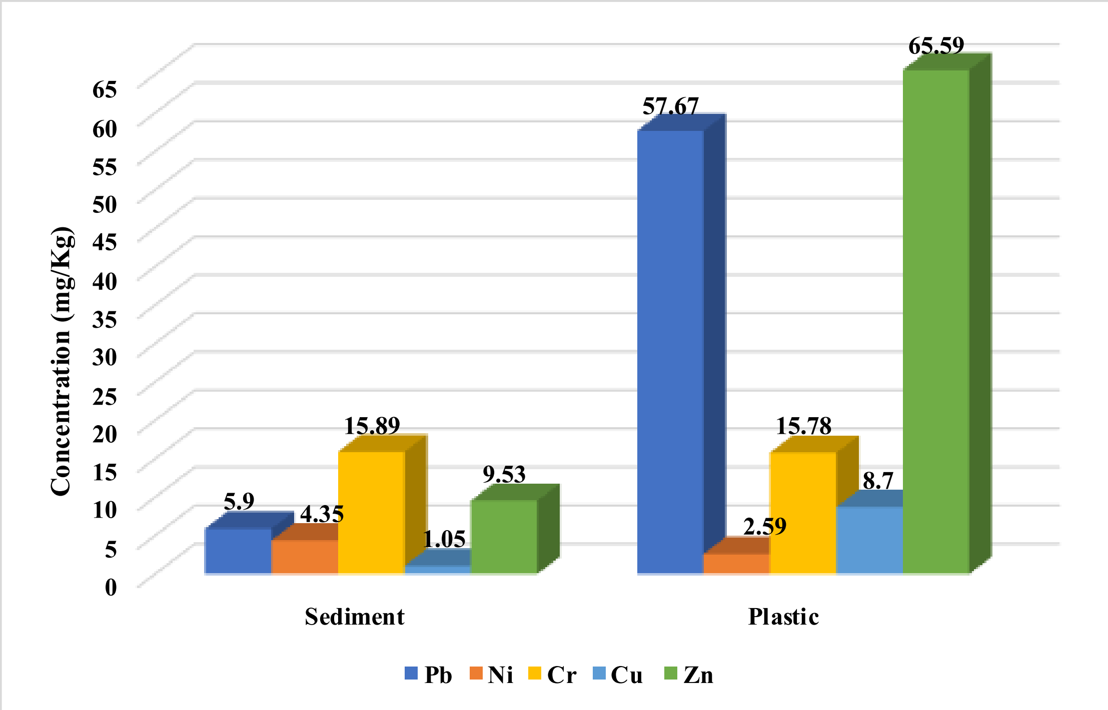


Figure 65: **Trace metal concentrations (mg/kg) in beach sediment and plastics collected at El Djamila Beach, Ain Benian.**

Trace metal concentrations in plastics from El Djamila beach were compared with values reported in coastal studies from southwest England (Holmes et al., 2012), the Persian Gulf (Akhbarizadeh et al., 2018), and Lake Como, Italy (Carnati et al., 2023) (table 17). Pb (57.67 mg/kg) and Zn (65.59 mg/kg) exceeded values reported in all comparable studies (Pb: 0.149–4.59 mg/kg; Zn: 0.299–70 mg/kg), positioning El Djamila among the most metal-enriched coastal plastic samples reported to date. Cu (8.7 mg/kg) was likewise higher than values reported for southwest England (0.064–1.32 mg/kg). In contrast, Cr (15.78 mg/kg) and Ni (2.59 mg/kg) fell within or below the ranges reported elsewhere (Cr: 44.0–751 mg/kg; Ni: 2.03–131 mg/kg), indicating comparatively lower enrichment for these two elements. This pattern is consistent with the elevated Pb concentration exceeding even the global maximum reported by Verla et al. (2019).

Table 17: **Comparative trace metal concentrations measured directly in/on plastics & MPs**

Location	Metal	Concentration (in plastic)	Note	Reference
El Djamila, Ain Benian, Algeria	Pb / Ni / Cr / Cu / Zn	57.67 / 2.59 / 15.78 / 8.7 / 65.59 mg/kg	Plastics act as a vector for trace metals	This study
Saltram, SW England	Cu / Zn / Pb Cr/ Co / Ni/ Cd	0.441 / 2.68 / 1.02 mg/kg; 237, 107, 131, 1.65 ng/g	Plastics act as a vector for trace metal transport in the marine environment	Holmes et al., 2012
Ninney Rock, SW England	Cu / Zn / Pb Cr/ Co / Ni/ Cd	1.11 / 21.0 / 1.34 mg/kg;	Beached pellets adsorb/concentrate metals beyond virgin pellet levels — confirms vector role	Holmes et al., 2012

		751, 17.7, 69.8, 76.7 ng/g		
Sharrow Point, SW England	Cu / Zn / Pb Cr/ Co / Ni/ Cd	1.32 / 23.3 / 1.64 mg/kg; 430, 20.7, 95.1, 74.4 ng/g	Zn levels approached/exceeded contaminated estuarine sediment; supports plastics-as-vector conclusion	Holmes et al., 2012
Watergate Bay, SW England	Cu / Zn / Pb Cr/ Co / Ni/ Cd	0.064 / 0.299 / 0.149 mg/kg; 44.0, 22.5, 40.0, 1.09 ng/g	Lowest baseline site; still shows measurable sorption on plastics	Holmes et al., 2012
Bushehr, Persian Gulf, Iran	Cd / Cr / Ni / Pb / Cu Al/ Fe/ Mn	0.035 / 0.915 / 2.03 / 4.59 / 3.6 115/ 531/ 32.2 mg/kg	Plastics adsorb trace metals from the surrounding environment, confirming plastics as vectors	Akhbarizadeh et al., 2018
Lake Como shoreline plastics, Italy	Zn (most abundant) Ti/ Al/ Mg/ Na/ Fe/ K	Median ~3–70 mg/kg (range across most abundant metals; Zn within this range)	Total metal load in plastics reflects both intrinsic additives and environmental adsorption; not all metal content is sorption-derived, complicating the simple "vector" narrative	Carnati et al., 2023

Taken as a whole, these findings indicate that El Djamila and Sidi Fredj beaches are characterized by well- to moderately-sorted medium sands with very low pelitic fractions and moderate organic matter content, reflecting a coarse-grained, high-energy depositional system with limited fine-particle retention. Despite this low fine-sediment affinity, microplastics were detected at all stations and matrices, alongside marked site-specific contrasts in marine sediment and seawater linked respectively to storm-driven deposition and to wastewater & riverine inputs. Across both sites, MPs were dominated by fibers, fine particles (<1 mm), and transparent-to-blue morphotypes, with FTIR confirming PE, PP, PET, HDPE, PVC, PA, and PS as the principal polymers; a signature of secondary, urban, textile & fishing-derived contamination rather than industrial pellet input, and one that places the Algiers coastline among the more heavily impacted sectors of the Algerian and Mediterranean coasts. Trace metal analysis reinforced this picture of a system governed more by plastic-mediated transport than by sediment-based accumulation: sediment metal concentrations were low, with all Igeo, CF, and PLI values indicating an uncontaminated status consistent with the coarse, low-organic sediment matrix. In stark contrast, the plastic fraction showed strong enrichment in Zn, Pb, and Cu while Cr remained comparable between the two matrices and Ni was depleted in plastics relative to sediment, reflecting its stronger lithogenic affinity. This selective enrichment pattern, pronounced for Zn, Pb, and Cu, negligible for Cr, and absent for Ni, mirrors findings from SW England, the Persian Gulf, and Lake Como, but with El Djamila's Pb and Zn concentrations exceeding all of these reference studies, and Pb surpassing even the global maximum reported by Verla et al. (2019). Taken together, these results indicate that despite low overall sediment contamination, microplastics along the Algiers coast function as active vectors concentrating specific trace metals well beyond ambient sediment levels, underscoring their role as both indicators and carriers of localized anthropogenic metal pollution.

Conclusion & perspectives

This study provides the first multi-matrix assessment of microplastic pollution along the central Algerian coastline, integrating analyses of beach sediment, marine sediment, and surface seawater at two western Algiers sites, alongside sedimentological characterization and trace metal analysis. MPs were detected ubiquitously across all matrices and stations, confirming pervasive and ongoing anthropogenic pressure on the Algiers coastal zone.

Spatial contrasts between sites reflect distinct contamination pathways: higher sedimentary accumulation at Sidi Fredj is driven by storm-induced deposition and recreational pressure, whereas elevated waterborne concentrations at Ain Benian point to wastewater effluent and riverine discharge as the dominant sources. Fibers and fine particles < 1mm prevailed across all matrices, indicating predominantly secondary contamination from synthetic textiles and fragmented plastic debris. Polymer identification by FTIR confirmed inputs consistent with urban, domestic, and fishing-related sources, with PP & PE as the dominant polymers. Comparison with regional literature places both sites among the most contaminated beaches along the Algerian coast, underscoring the cumulative effect of waste mismanagement, inadequate wastewater infrastructure, and intensive coastal use on this urbanized shoreline. Trace metal analysis by ICP-OES indicates that microplastics act as potential vectors of metallic contaminants, with higher concentrations of Zn, Pb, and Cu in plastics than in sediments, likely due to surface adsorption. The association of trace metals with both MPs and surrounding sediments further suggests that plastic debris contributes to the transport and redistribution of metals, increasing the overall environmental impact of plastic pollution beyond its physical presence.

Taken together, these results establish a baseline for MP contamination along the southern Mediterranean coast and highlight the urgent need for coordinated monitoring, governance, and mitigation action. Based on the findings of this study, the following recommendations are proposed:

Monitoring and spatial coverage

- ENSSMAL could play a leading role, in collaboration with CRNA, CNRDPA, AND, and other relevant stakeholders, in establishing a national long-term, seasonal multi-matrix MPs monitoring network covering beach sediment, marine sediment, surface seawater, and biota along the Algerian coastline.
- Expand offshore sampling through multiple transects to assess the spatial dispersion of MP from coastal sources into the marine water column.
- Develop dedicated monitoring stations at major wadi outlets and drainage systems to quantify riverine and stormwater-driven inputs of MPs into coastal environments.
- ONEDD could coordinate the expansion of monitoring coverage across representative beaches, bays, and subtidal zones, to build a more comprehensive national baseline.
- Establish a national georeferenced coastal pollution database integrating MP abundance, polymer composition, trace metal concentrations, and biotic contamination data across spatial and temporal scales, to support evidence-based environmental governance and regional comparisons.

Analytical and methodological

- ENSSMAL and CRNA could lead the implementation of standardized sampling, extraction, and identification protocols, aligned with International Atomic Energy Agency quality assurance frameworks and complementary international guidelines, to ensure analytical reliability and long-term data comparability.
- Integrate advanced analytical techniques, including micro-FTIR and Raman spectroscopy for the identification of fine and ultrafine particles below the 500 µm thresholds, alongside SEM-EDX to assess particle surface weathering, additive composition, and adsorbed metal contaminants.
- Use denser extraction solutions (NaI, $ZnCl_2$, or sodium polytungstate) to improve recovery efficiency for high-density polymers such as PET, PVC, and PA, which may be underestimated by standard NaCl-based flotation.

Ecotoxicology and human health

- CNRDPA, in collaboration with ENSSMAL, could assess ecotoxicological risks in commercially important marine species to evaluate human dietary exposure pathways and inform food safety risk assessments in Algerian coastal communities.
- Investigate interactions between MPs and co-contaminants, particularly trace metals and POPs, to better characterize combined ecotoxicological risk to marine organisms.
- Conduct controlled laboratory microcosm experiments under realistic coastal conditions to quantify the leaching kinetics of plastic additives, as well as the release rates of adsorbed and incorporated heavy metals, to assess their bioavailability and potential toxicity.
- Incorporate smaller size fractions (<250 µm and nanoplastic range) into future studies, given their higher bioavailability, capacity to penetrate biological tissues, and current underrepresentation in Algerian MP studies.

Governance and management

- Upgrade wastewater treatment infrastructure with tertiary filtration systems capable of retaining synthetic fibers and fine plastic particles before coastal discharge.
- The AND, in collaboration with MEER, should strengthen municipal waste management systems, including collection efficiency, selective sorting, and prevention of plastic leakage across coastal wilayas.
- Enforce and strengthen existing coastal legislation with specific provisions explicitly targeting microplastic emissions from urban, industrial, and agricultural sources, and accelerate implementation of the SNGID 2035 strategy through improved infrastructure and extended selective sorting systems.
- Develop hydrodynamic and morphosedimentary dispersion models integrating microplastic transport parameters, wave climate, longshore current dynamics, wadi discharge rates, and runoff intensity to predict coastal accumulation hotspots and spatial dispersion patterns.
- Promote regional scientific cooperation within the Mediterranean framework to harmonize monitoring protocols, share analytical capacity, and develop joint mitigation strategies consistent with the Barcelona Convention and its protocols.
- Implement seasonal beach-cleaning and public awareness campaigns targeting peak tourist periods, engaging local communities, fishers, beachgoers, and municipal authorities in source reduction efforts.

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Certificate Of Participation

This certificate is awarded to

Kateb Rania, Grimes Samir

In recognition of the participation in the 2nd edition of the conference entitled

"Anthropogenic Interactions with the Sea and the Coastal Zone:

Observation And Ecosystem-Based Management"

Held on June 8th 2026 with a poster entitled **"Spatial Distribution and Characterization of Microplastics along the Central Algerian Coast".**

The Director of LCVRM

Laboratoire CVRM ENSSMAL

The Director of ENSSMAL