

Cranfield University

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**Sources, Characterisation and Exposure Risk of Airborne
Microplastic Emissions from Municipal Solid Waste Dumping Site
in Nigeria**

School of Water, Energy and Environment PhD
Academic Year 2020-2024

Supervisor: Dr Gill Drew
Associate Supervisor: Dr Chris Walton

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This thesis is submitted in partial fulfilment of the requirements for the
degree of PhD.

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Abstract

Airborne microplastics (AMPs) represent an emerging environmental and public health challenge, with their sources, transport mechanisms, and impacts still poorly understood, particularly in developing regions with inadequate waste management systems. This research addresses three key gaps: the need for cost-effective and efficient AMP sampling tools, the AMP flux estimations under different environmental conditions, and the modelling of AMP dispersion to understand their transport and potential exposure risks downwind.

This research tackles these challenges by developing a low-cost sampler for AMP collection. The low-cost sampler was validated against the commercial sampler (SKC Deployable Sampler equipped with a Total Suspended Particulate (TSP) head), with a focus on fibres, fragments, and films across diverse environmental conditions. The emission of AMPs was quantified using a modified Fick's law, which incorporates site-specific parameters such as wind speed, temperature, and particle properties. Seasonal variation in AMP emissions was analysed by collecting and processing 226 environmental samples (42 soil and 184 air) from the municipal solid waste disposal site and its environment during dry and wet seasons. Dispersion modelling was conducted using SCREEN3 to simulate the downwind transport of AMPs.

A low-cost sampler (LCS) was developed and evaluated against a commercial sampler, demonstrating a strong correlation ($\rho = 0.976$) and high accuracy (94.12%) compared to a reference sampler. The LCS effectively captured seasonal variations in AMP abundance. Polymer analysis identified five predominant polymers, with nylon (fibres), PVC (fragments), and PE (films) accounting for the majority of microplastics. The cost analysis revealed that the LCS offers 61% savings over second-hand and 98% over new commercial samplers, making it a reliable and affordable tool for AMP research in resource-limited settings.

The airborne microplastics measured on-site reveal seasonal variations in concentrations. Notably, the dry season reveals higher concentrations (mean: 14.37 ± 3.87 MP/m³) comparable to the wet season (mean: 11.31 ± 3.00 MP/m³). Upwind concentrations were considerably lower, averaging 4.25 ± 1.17 MP/m³ during the dry season and 2.75 ± 1.43 MP/m³ during the wet season, reflecting contributions from distant fibre-rich sources, likely indoor emissions. On-site, films exhibited the lowest emissions but retained moderate mobility during the wet season. Fibres showed the highest diffusion coefficients, indicating potential for long-range transport. Fragments were the most abundant microplastic type (55% dry, 53% wet), with high emission factors (188 µg/day dry, 170 µg/day wet). Rising velocities were higher during the dry season due to favourable wind conditions, with values of 0.1056 m/s for nylon fibres, 0.0835 m/s for PVC fragments, and 0.0742 m/s for PE films. The rising velocities and

flux measurements highlighted the influence of soil porosity and wind speed on resuspension and transport of microplastics.

The SCREEN3 dispersion model reveals distinct seasonal variations in the transport of AMP from MSW sites. Peak AMP concentrations occurred at 100–107 m downwind, with wet season levels (fibres: $2.28 \times 10^{-2} \mu\text{g}/\text{m}^3$, fragments: $6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$, films: $2.41 \times 10^{-3} \mu\text{g}/\text{m}^3$) exceeding dry season concentrations by 2.1–2.2 times. Fragments posed the highest health risks (Level III), particularly during short-term exposures, while fibres and films showed lower risks. SCREEN3 agreed well with ground measurements ($R^2 = 0.98$ to 0.96) and identified key drivers such as stability classes and precipitation, affirming its utility for AMP transport modelling and risk assessment.

This study highlights the significant environmental and health implications of airborne microplastic (AMP) emissions from municipal solid waste (MSW) sites. Fragments pose the greatest risks, particularly during the wet season. The development of a low-cost sampler and advanced dispersion modelling provides essential tools for AMP monitoring. To mitigate AMP impacts, improved waste management practices, such as minimising open burning, are necessary. Integrating AMP data into air quality monitoring frameworks and prioritising seasonal mitigation measures are also recommended. Future studies should investigate long-range transport mechanisms, refine emission factor models, and chronic exposure risks to develop comprehensive strategies for mitigating AMP impacts globally.

Keywords: Low-cost Sampler, Airborne Microplastic, Flux Emission, Rising Velocity, Diffusion Coefficient, SCREEN3 and Dispersion Modelling.

Acknowledgements

I am profoundly grateful to my supervisors, Dr Gill and Dr Chris Walton, for their steadfast guidance, encouragement, and support throughout this research. Their dedication and mentorship have been instrumental in the progress and completion of this work. I would also like to acknowledge my former PhD supervisors, Dr Ying Jiang, Dr Muhammed Mead and my reviewer, Dr Francis Hassard, whose extensive expertise in air pollution studies enriched this research.

I sincerely appreciate the Petroleum Technology Development Fund (PTDF), Nigeria, for providing full sponsorship for this study. My gratitude also extends to the Hussaini Adamu Federal Polytechnic, Kazaure, Jigawa, Nigeria, for their institutional support, as well as the Department of Civil Engineering, HAPF, for their cooperation during my academic journey.

I am thankful to my colleagues and friends from various research groups and my workplace for the meaningful discussions, collaborative efforts, and shared moments of growth. I am also indebted to the Nigerian community in Cranfield, United Kingdom, for their camaraderie and encouragement throughout my research period.

To my cherished wife, Mrs Surayya, my children, and my parents, Salma and Abdullahi, your enduring love, understanding, and unwavering support have been my greatest source of strength throughout this endeavour.

Above all, I give thanks to Almighty Allah for the blessings, opportunities, and sustenance that have enabled me to complete this work successfully.

Table of Contents

Abstract	i
Acknowledgements	iii
List of Figures	ix
List of Tables	xi
ABBREVIATION	xiii
Chapter 1.0: Airborne Microplastics in Waste Disposal Sites	1
1.1 Introduction	1
1.1.1 Waste Management Practices in Nigeria	4
1.1.2 Measurement of Microplastics and Airborne Microplastics	6
1.2 Research Aim and Objectives	9
1.3 Thesis Structure	11
Chapter 2.0: LITERATURE REVIEW	13
2.1 Introduction	13
2.2 Microplastic exposure and human health	16
2.2.1 Inhalation Route	17
2.2.2 Ingestion Route	18
2.2.3 Dermal Route	19
2.2.4 Potential Health Impact	20
2.3 Influence of Waste Treatment Methods on Microplastic Emissions	20
2.3.1 Nigeria's Waste Treatment System and Microplastic Emissions	24
2.3.2 Microplastic and Airborne Microplastic Studies in Nigeria	26
2.3.3 Air Quality Standards in Nigeria and the Overlooked Role of Microplastics	30
2.4. Plastic Legislation in Nigeria	31
2.4.1 Federal Legislation on Plastic Pollution in Nigeria	32
2.4.2 Proposed Legislation	32
2.4.3 State-Level Legislation	33
2.4.4 International Agreements	33
2.4.5 Policy Initiatives, Challenges and Opportunities	34
2.5 Airborne Microplastic Sampling Methods	34
2.5.1 Passive Sampling Methods for Airborne Microplastics	34
2.5.2 Active Sampling Methods for Airborne Microplastics	36
2.6 The Role of Low-Cost Samplers in Microplastic Research	39

2.6.1 Low-Cost Samplers for Microplastic Monitoring in Low-Income Countries	39
2.6.2 Low-Cost Samplers for Microplastic Monitoring in Nigeria.....	40
2.6.3 Future Directions for Low-Cost Samplers in Airborne Microplastic Research	41
2.7 Identification of Microplastics	43
2.7.1 Qualitative Sampling in Microplastic Research	43
2.7.2 Quantitative Microplastic Sampling and Analysis.....	48
2.7.3 Qualitative/Quantitative Complementary Roles and Importance.....	50
2.8 Transport and Fate of Airborne Microplastics: Dispersion Model	51
2.8.1 Adaptation of Dispersion Models for Airborne Microplastics	52
2.8.2 Adoptability of Screening Models for Airborne Microplastics	56
2.8.3 Integration of Dispersion Models with Health Impact Assessments.....	60
2.9 Conclusion	61
Chapter 3.0: Design and Development of a Low-Cost Air Sampler for Airborne Microplastic Collection.....	63
3.1 Introduction	63
3.2. Materials and Methods.....	65
3.2.1 Study Area	66
3.2.2 Weather Conditions	67
3.3 Design and Choice of Air Sampler	69
3.3.1 Reference Sampler	69
3.3.2 Low-Cost Sampler Design	69
3.3.3 Sample Storage and Transportation	71
3.4 Sampling Processing	71
3.4.1 Isolation of Airborne Microplastics	72
3.4.2 Microplastic Particles under Microscopy	72
3.4.3 Confirmation of Microplastic Particles via ATR-FTIR Spectroscopy	73
3.5 Quality Assurance and Quality Control (QA/QC)	73
3.6 Data Analysis	74
3.6.1 Air Sampling Volume and Microplastic Calculation.....	74
3.6.2 Performance indicators	74
3.7 Results and Discussion.....	75
3.7.1 Airborne MP Abundance and Shapes.....	77
3.7.2: Agreement between Low-Cost and Reference Samplers: Bland-Altman Analysis	79
3.7.3 Performance Evaluation across Microplastic Size Classes	80

3.7.4 Airborne MP Polymer-Specific Performance Assessment	82
3.8 Cost Analysis and Cost Comparison.....	83
3.8.1 Development of a Low-Cost and Reference Sampler: Components and Costs	83
3.8.2 Cost Comparative Analysis of Low-Cost and Reference Sampler	85
3.8.3 Reproducibility of Low-Cost Sampler	86
3.9 Limitations and Recommendations	88
3.9.1 Limitations.....	88
3.9.2 Recommendations for Future Research and Development	89
3.10 Conclusion	90
Chapter 4.0: Estimation of Airborne Microplastics Flux from Soil to Air.....	92
4.1 Introduction	92
4.2 Method.....	94
4.2.1 Study Area.....	94
4.2.2 Field Sampling	96
4.2.3 Meteorological Data.....	97
4.3 Microplastic Analysis.....	99
4.3.1 Air Sample Preparation.....	99
4.3.2 Soil Sample Preparation	99
4.3.3 Microplastic Visualisation.....	100
4.3.4 Polymer Composition Analysis	101
4.4 Quality Control	101
4.5 Statistical Analysis	102
4.6 Model Development	103
4.6.1 Microplastic Flux Estimation (MPF).....	103
4.6.2 Modifications to Fick's Law	104
4.6.3 Diffusion Coefficient (D_o)	105
4.6.4 Concentration Gradient ($\partial c \partial x$)	106
4.6.5 Estimation of Microplastic Emission Factor and Mass Loading from Waste Disposal Sites	107
4.7 Results and Discussion.....	109
4.7.1 Airborne Microplastic Concentration	109
4.7.2 Microplastic Concentrations in Soil Samples	114
4.7.3 Aerodynamic Properties and Rising Velocity of Microplastics.....	116
4.7.4 Diffusion Coefficient, Concentration Gradient, Flux and Emission Factors	120

4.8 Implications and Limitations	123
4.8.1 Environmental and Health.....	123
4.8.2 Wider Implications	124
4.8.3 Limitations and Future Studies	125
4.9 Conclusion	126
Chapter 5.0: Airborne Microplastic Transport and Exposure (Model-Based Assessment)	127
5.1 Introduction	127
5.2 Field sampling.....	128
5.3 Laboratory Analysis	132
5.4 Framework for Dispersion Modelling.....	133
5.4.1 Theoretical Framework	133
5.4.2 Dispersion Modelling in Microplastic Research.....	134
5.5 SCREEN3 Model Interface	135
5.5.1 SCREEN3 Model Input and Parameterisation	136
5.5.2 Options Menu	137
5.5.4 Receptors Menu	140
5.5.5 SCREEN3 Model Input Data.....	141
5.5.6 Evaluation of Model Performance	142
5.6 Microplastic Exposure Risks Framework for Municipal Solid Waste	142
5.6.1 Risk Assessment Framework	143
5.7 Results and Discussion.....	145
5.7.1 Microplastics Downwind Concentration Profile	146
5.7.2 Microplastic Time-dependent Concentrations.....	148
5.7.3 Comparison of Airborne Microplastic Concentrations to PM Standards ..	149
5.7.4 Exposure Risk Assessment	150
5.7.5 Dispersion Patterns	152
5.7.6 Performance Comparison of Dry and Wet Season Dispersion Results ..	153
5.7.7 Comparison with other studies.....	155
5.8 Conclusions	157
5.8.1 Key Findings and Implications	157
Chapter 6.0: Conclusion, Implications, Contributions, and Future Directions.....	159
6.1 Key Findings and Conclusions.....	159
6.2 Airborne Microplastic (AMP) Sampling Using Low-Cost Sampler	162
6.3 Microplastic Flux Estimation Using Modified Fick's Law	163

6.4 Modelling and Risk Assessment of Airborne Microplastic Dispersion	164
6.5 Key Limitations.....	165
6.6 Recommendations for Further Studies	167
6.7 Contribution to Knowledge	170
7.0 References.....	173
Appendix A.....	217
Appendix B.....	231
Appendix C.....	241

List of Figures

Figure 1. 1: Schematic diagram of the thesis structure	11
Figure 2. 1: Microplastic Exposure in Humans (Sun & Wang 2023).....	17
Figure 2. 2: Distribution of Microplastics Studies Across Six Southern Nigerian States	27
Figure 3. 1: Map of the Study Area.....	68
Figure 3. 2: Airborne-MPs Samplers (a) Low-cost Sampler (b) Low-cost and SKC sampler.	70
Figure 3. 3: Comparative assessment of airborne microplastics collected by Ref and LCS.	77
Figure 3. 4: Examples of micrographs of microplastics identified in air samples. From top left to right (a-c): collected from the reference sampler. From bottom left to airborne microplastic collected from the LCS sampler (d-f). Scale 1:100 μm	78
Figure 3. 5: Bland Altman Agreement of Low-cost vs Reference Sampler.....	79
Figure 4. 1: The map illustrates the sampling locations, with purple points (1-7) indicating soil sample collection sites and a black point designating the air sampler location.....	95
Figure 4. 2: Airborne microplastic concentrations from the disposal site during dry (purple) and wet (green) seasons from on-site and upwind (a) Airborne MP concentrations on-site (b) Airborne MP concentrations Upwind.....	110
Figure 4. 3: Soil Microplastic Particle Sizes Distributed in Dry and Wet Seasons (a) Soil Concentration MP in Dry Season (b) MP Sizes Distribution in Dry Season (c) Soil MP Concentration in Wet Season (d) MP Sizes Distribution in Wet Season.	114
Figure 5. 1A: Distribution of Downwind Sampling Points During Dry Season	130
Figure 5. 2: SCREEN3 Option Menu.....	137
Figure 5. 3: SCREEN3 Source Menu	139
Figure 5. 4: Receptors Menu	140
Figure 5. 5: Windrose Analysis during Dry and Wet Seasons (a) Dry Season (b) Wet Season.....	145
Figure 5. 6: Dry Season Microplastic Downwind Concentration Profiles	146
Figure 5. 7: Wet Season Microplastic Downwind Concentration Profiles	147

Figure 6. 1: Framework for Using the SCREEN3 Air Dispersion Model in Airborne Microplastic (AMP) Studies	160
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List of Tables

Table 2. 1: Classification of Plastic Debris by Size, Composition, and Environmental Occurrence	14
Table 2. 2: Landfills in Nigeria.	25
Table 2. 3: Passive Sampling Methods for Airborne Microplastics	35
Table 2. 4 : Active Sampling Methods for Airborne Microplastics.....	37
Table 2. 5 : Active Sampling Filter Type and Pore Sizes.....	38
Table 2. 6: Future Research Directions for Airborne Microplastic Low-Cost Samplers in Nigeria	42
Table 3. 1: Seasonal Meteorological Conditions During Sampler Validation.....	76
Table 3. 2: Airborne MP concentration difference between Ref and LCS	78
Table 3. 3: LCS Performance Across Particle Sizes	81
Table 3. 4: Performance Indices of LCS.....	82
Table 3. 5: Polymer Composition in Low-Cost and Reference Sampler.....	82
Table 3. 6: Materials and Costs for Low-Cost Sampler Development	84
Table 3. 7: Cost of Imported Air Sampler	85
Table 4. 1: Average Weather Conditions for Dry and Wet Seasons.....	98
Table 4. 2: On-site Comparison of Microplastic Particles < 100 µm (Dry vs. Wet Season).....	110
Table 4. 3: Upwind Comparison of Microplastic Particles < 100 µm (Dry vs. Wet Season).....	112
Table 4. 4: Polymer Composition of Airborne Microplastic (Dry and Wet Season).	113
Table 4. 5: Polymer Composition of Microplastic in Soil (Dry and Wet Season)	116
Table 4. 6: Microplastic Dimensions: Dry vs. Wet Season	117
Table 4. 7: Microplastic Aerodynamic Properties	119
Table 4. 8: Microplastic Volume and Mass Estimation	120
Table 4. 9: Microplastic Diffusion Coefficient and Concentration Gradient.....	121
Table 4. 10: Seasonal Mass Loading of Airborne Microplastics (µg/m ³)	123
Table 5. 1: Model Input Data for Dry and Wet Seasons	141
Table 5. 2: Polymer Hazard Score (Sn).....	143

Table 5. 3: MP Exposure Risk.....	144
Table 5. 4: Microplastic Time Dependent Concentrations.....	149
Table 5. 5: Comparison of Modelled AMPs with NESREA PM Standards	150
Table 5. 6: Risk Assessment Levels and Exposure Duration of AMPs	151
Table 5. 7: Dispersion Characteristics.....	152
Table 5. 8: Model Performance in Dry and Wet Season	154
Table 5. 9: SCREEN3 Performance Metrics	155

ABBREVIATION

AMPs	Airborne Microplastics
ATR FTIR	Attenuated Total Reflectance Fourier
Transform Infrared	
COVID-19	Coronavirus Diseases 2019
DSC	Differential Scanning Calorimetry
EFs	Emission Factors
E/NE	East-Northeast
EPA	Environmental Protection Agency
EPR	Extended Producer Responsibility
FTIR	Fourier-Transform Infrared Spectroscopy
FPA	Focal Plane Array
GC-MS	Gas Chromatography-Mass
Spectrometry	
GUI	Graphic User Interface
HDPE	High-Density Polyethylene
KOH	Particulate Matter
LDPE	Low-Density Polyethylene
LCS	Low-Cost Sampler
NEM	Northeast Monsoon
NEMA	Nigerian Environmental Protection
Agency	
MBE	Mean Bias Error
MSE	Mean Square Error
MPs	Micro Plastics
MPF	Microplastic Flux
MSW	Municipal Solid Waste
MSWDS	Municipal Solid Waste Disposal Site
PA	Polyamide
PAA	Peroxyacetic Acid
PC	Polycarbonate
PAN	Peroxyacetyl nitrate
PE	Polyethylene

PES	Photo Electron Microscope
PET	Polyethene terephthalate
PM	Particulate Matter
POPs	Persistent Organic Pollutants
PP	Polypropylene
PS	Polystyrene
PSD	Particle Size Distribution
PTFE	Polytetrafluoroethylene
PUR	Polyurethane
PVC	Polyvinyl chloride
PYR	Pyrolysis
QA	Quality Assurance
QC	Quality Control
REF	Reference Sampler
RMSE	Root Mean Square Error
(R ²)	Correlation Coefficients
RO	Reverse Osmosis
SEM	Scanning Electron Microscope
SEM-EDS	Scanning Electron Microscopy with
Energy-Dispersive X-Ray Spectroscopy	
SI	System of Units
SKC	Air Sampler
TDS	Thermal Desorption-Gas
TDS-GCMS	Desorption Gas Chromatography-Mass
Spectrometry	
TGR	Thermos Gravimetric Analysis
UNEP	United Nations Environment Programme
VOCs	Volatile Organic Compounds
VS	Volume of Sampled Air
W	Weight
WHO	World Health Organisation
WSW	West-Southwest
WX	Rising Velocity of Microplastic
(D_{mod})	Diffusion Coefficient

$$\left(\frac{\partial c}{\partial x}\right).$$

μFT-IR

Concentration Gradient

Micro-FTIR

Chapter 1.0: Airborne Microplastics in Waste Disposal Sites

1.1 Introduction

Plastics are versatile materials derived from carbon and fossil fuels like crude oil and natural gas (Bîrcă et al., 2019). There are two main types: thermoplastics and thermosets (Prasittisopin et al., 2023). Thermoplastics are flexible and recyclable, ideal for everyday items like plastic bags, bottles, and pipes. Thermosets are durable but hard to recycle, so they are used for construction, adhesives, and food packaging (Scheutz et al., 2019).

Despite the benefits of plastics, their widespread use has caused significant environmental pollution (Koelmans et al., 2022a). Mismanaged plastic waste can persist in the environment for centuries, harming ecosystems. Plastic debris comes in different shapes and sizes due to its various sources; these include Macroplastic (> 25 mm), Mesoplastic (>5 mm to 25mm), Microplastic (1 μ m to 5 mm), and Nanoplastic (1nm-1 μ m) (Caldwell et al., 2022; Galloway, 2015; Gigault et al., 2018; X. Zhao et al., 2023a) based on size (Jeyasanta et al., 2020; Pegado et al., 2024). The growing concern over plastic pollution and the depletion of fossil fuels has led to the development of bioplastics (K. Kumar et al., 2023). Bioplastics are made from renewable resources like plant starch, sugarcane, or microorganisms, offering a more sustainable alternative to traditional plastics (K. Kumar et al., 2023). However, not all bioplastics are biodegradable, so proper disposal methods are still important.

Microplastics (MPs) are the focus of this research and have garnered significant media attention. Microplastics (MPs) possess distinct physical and chemical properties that may influence their environmental behaviour.

Physically, microplastics exhibit considerable variability in size ($\leq$ 5 mm), morphology (fibres, fragments, films, pellets), and polymer density, all of which govern their transport, deposition, and settling behaviours across environmental media. In aquatic systems, polyethylene (PE) (low-density polymers) typically remain buoyant and accumulate near the water surface. Polyvinyl chloride (PVC) (higher-density polymers) tends to sink and deposit in benthic sediments. In the atmosphere, particles with lower

density and smaller aerodynamic diameters remain airborne for extended periods, facilitating long-range transport. Denser or larger particles exhibit higher settling velocities and are more likely to deposit proximally to their source. Understanding these density- and size-driven processes is necessary in predicting the fate and environmental distribution of microplastics in both terrestrial and aquatic compartments.

Microplastics can be further subdivided into two main categories, primary and secondary microplastics. Primary MPs are manufactured from small-sized plastic raw materials, such as feedstock for plastic products, and plastic particles utilised in various abrasion processes, including skincare and personal care items (Hernandez et al., 2017). Secondary MPs, in contrast, are particles that emerge from the mechanical weathering, photo-oxidative degradation, and biodegradation of plastic materials (Colzi et al., 2022; H. Wang et al., 2023). Other secondary gateways include wastewater, runoff, or direct disposal (X. Lin et al., 2023; Naderi Beni et al., 2023; Petrović et al., 2023; Ziajahromi, Neale, et al., 2017).

Research has consistently shown that microplastics (MPs) are ubiquitous in marine and terrestrial environments (Andrady, 2011; J. J. Guo et al., 2020; Radford et al., 2021a). Studies have revealed that small-sized, low-buoyancy MPs tend to become airborne, contributing to atmospheric aerosols (M. A. Bhat, 2024; Thacharodi et al., 2024a). Airborne MPs can be dispersed through various means, including industrial processes and the wear and tear of plastic materials, leading to their widespread distribution (Goßmann et al., 2023; Jan Kole et al., 2017). Airborne microplastics (MPs) can have far-reaching impacts on the environment, including altering the energy balance and climate (Gardon et al., 2018; Sunil et al., 2024).

Airborne microplastics (MPs) can alter the Earth's energy balance by absorbing solar radiation (Revell et al., 2021). MPs can alter surface albedo by changing surface reflectivity and influence climate formation through interactions with cloud droplets and atmospheric circulation patterns (Evangelidou et al., 2020a; X.-F. Wei et al., 2024). The colour diversity of the microplastics can also expedite the melting of snow and ice by absorbing more heat on their surfaces (Y. Zhang et al., 2022).

The atmospheric transport of microplastic is faster than the ocean, moving them from their source to distant locations in days or weeks (Materić et al., 2021). This rapid transport allows plastic pollutants to travel long distances and reach even the most remote and Polar Regions through atmospheric processes (Evangelidou et al., 2020b).

Plastic pollution can infiltrate and damage even the most remote and pristine ecosystems globally (D. Allen et al., 2022a).

Airborne microplastics (MPs) can settle on farmland, water bodies, and food items, leading to direct human exposure through inhalation or indirect exposure through the food chain, and ultimately bioaccumulate in the environment (Sobhani et al., 2020). Human exposure to microplastics can occur through ingestion and inhalation pathways (Ageel et al., 2022a; A. Sun & Wang, 2023a).

The exposure to airborne microplastics has raised concerns about potential health impacts (Revel et al., 2018). These include the risk of cardiovascular disease and bronchitis (Liang et al., 2024a). The chemicals contained in the microplastics may also affect brain development and function, leading to neurological effects (Yong et al., 2020). Recent studies have uncovered the alarming presence of microplastics in different parts of the human body, such as blood, semen, and the placenta (Ragusa et al., 2021; Q. Zhao et al., 2023). The discovery of microplastics in the placenta and blood samples has raised concerns about embryo development and health (Ragusa et al., 2021).

Moreover, the presence of microplastics in human semen has been associated with adverse reproductive health outcomes (Hong et al., 2023). This burgeoning concern underscores the far-reaching implications of microplastic pollution for human health, necessitating a comprehensive investigation into their sources, pathways, and fate. Notably, microplastics are generated at multiple stages of the plastic lifecycle, including during the production of plastics, wherein primary microplastics are formed, and during the disposal of plastic waste, wherein secondary microplastics are created through degradation and fragmentation processes (Ghosh et al., 2023; L. Zhang et al., 2024).

A critical area of focus is the significant roles that landfills, and disposal sites play in microplastic pollution, as highlighted by recent studies (Ghosh et al., 2023; L. Zhang et al., 2024). Specifically, research by Zhang et al. (2024) demonstrates that municipal solid waste (MSW) management sites are substantial reservoirs of microplastics, necessitating further in-depth examination and scrutiny (L. Zhang et al., 2024).

The disposal of plastic waste in landfills and dumpsites can lead to the formation of microplastics through various mechanisms, including mechanical fragmentation, chemical degradation, and biological processes (A. C. Rocha-Santos, 2017). For

instance, the breakdown of plastic bags, bottles, and packaging materials can result in microplastic formation, dispersing into the environment via various pathways, including leachate and runoff (Mangla et al., 2025). Furthermore, the disposal method used can influence the amount of microplastic generated, with open dumping and burning of plastic waste potentially causing higher levels of microplastic formation compared to more controlled disposal methods (A. L. P. Silva et al., 2021).

Recent studies have provided clear evidence to support this assertion, demonstrating that different waste treatment methods can result in significantly varying concentrations of microplastics. For example, a study by Hu et al. (2022) on airborne microplastic emissions from municipal solid waste transfer stations in downtown Shanghai found an average concentration of 2.5 airborne microplastics per cubic meter. The study revealed the presence of nine polymers, with rayon polymer dominating the landscape, and noted seasonal variations with higher abundance in winter than summer. Moreover, the study suggested that an odour treatment facility reduced airborne microplastics, highlighting the potential for targeted interventions to mitigate microplastic pollution (T. Hu et al., 2022a). In contrast, a study by Zapata et al. (2024) at an industrial composting facility in England found airborne microplastic concentrations ranging from 32-49 MP/m³ on-site, 1 MP/m³ upwind, and 3 MP/m³ downwind (Zapata et al., 2024a).

These studies demonstrate that plastic waste disposal methods can impact microplastic generation and emissions, emphasising the need for sustainable and environmentally responsible waste management practices to reduce microplastic pollution. While this research has provided valuable insights into airborne microplastic emissions, significant knowledge gaps remain regarding their environmental fate and the impacts of microplastics, underscoring the necessity for a comprehensive assessment of microplastic pollution and its implications for the environment and human health.

1.1.1 Waste Management Practices in Nigeria

A complex mix of informal practices, infrastructural inadequacies, and regulatory gaps characterises waste management in Nigeria (D. Xu, 2024). In Nigeria, over 32 million tonnes of solid waste are generated annually, however only a limited portion is formally collected and disposed of through structured systems (Isaac Anyaogu, 2022). Consequently, informal methods such as open dumping, burning, and unregulated

recycling have become widespread. One of the most visible and troubling practices is the employment of vacant plots, often undeveloped or abandoned land, as unofficial waste disposal sites, particularly in Kano (Butu & Mshelia, 2014). These plots were located in urban and peri-urban areas and served as convenient dumping grounds for household and commercial waste, including plastic waste (Mathias Agbo Jr, 2025). Over time, waste accumulates without any sorting or management, exposing plastics to environmental stressors such as sunlight, wind, rainfall, and microbial activity (Nayanathara Thathsarani Pilapitiya & Ratnayake, 2024). This prolonged exposure facilitates the fragmentation of plastics into microplastics, which can be dispersed into surrounding soils, waterways, and even the air.

The widespread use of open dumping in vacant plots and along roadsides reflects a lack of access to formal waste collection systems, particularly in densely populated or economically disadvantaged communities (Kitole et al., 2024). These uncontrolled dumpsites are rarely monitored or rehabilitated, allowing plastic waste to degrade naturally over months or years (Ncube et al., 2021). Photo-degradation and mechanical weathering are key processes that convert these plastics into microplastics (Duan et al., 2021). Moreover, when rainfall interacts with waste on vacant plots, it carries fragmented plastic particles into nearby drains, rivers, and agricultural lands, contributing to broader environmental contamination. In some cases, these plots are later developed into residential or commercial areas without thorough remediation, posing long-term ecological and health risks due to persistent microplastic residues (Yoon et al., 2024).

In addition to dumping, open burning is a commonly employed method for waste reduction, especially on vacant plots where large amounts of waste are left unmanaged. Residents and informal waste handlers set fire to accumulated rubbish to clear space or reduce volume (Pathak et al., 2023). While seemingly efficient, this combustion process leads to the emission of toxic pollutants and the formation of combustion-derived microplastics that degrade over time and become suspended in the air (J.-J. Guo et al., 2020). These airborne microplastics can be inhaled or deposited on surfaces, contributing to indoor and outdoor air pollution. This presents a significant public health concern in areas with frequent burning activity, especially near homes and markets (Yoon et al., 2024).

Another contributor to microplastic pollution is the informal recycling sector, which operates widely across Nigeria without adequate regulation or infrastructure (Solaja et al., 2024). Informal recyclers often work in makeshift conditions, manually sorting plastics and shredding them into smaller pieces using basic machinery. These operations, frequently conducted in open spaces or on vacant plots, lack proper containment systems, allowing microplastics and plastic dust to disperse into the environment. Furthermore, without waste segregation at the source, biodegradable and non-biodegradable materials are mixed, reducing the quality of recyclable plastic and increasing the volume of plastics that end up discarded or burnt (Kibria et al., 2023).

Landfills and official dump sites in Nigeria also present significant environmental challenges, particularly where they are constructed without protective liners or leachate collection systems (Gunarathne et al., 2024). Plastics buried in such landfills are subject to degradation processes that slowly release microplastic particles into the surrounding soil and groundwater (Wojnowska-Baryła et al., 2022). Combined with the informal use of nearby land for dumping and burning, the cumulative impact of these practices results in a continuous release of microplastics across terrestrial and aquatic ecosystems.

Overall, Nigeria's prevailing waste management practices, especially the informal use of vacant plots, open dumping, burning, and unregulated recycling, are central to the formation and dissemination of microplastics. These practices, driven by infrastructural limitations and insufficient regulatory enforcement, accelerate the breakdown of plastic waste and contribute significantly to microplastic contamination in soil, water, and air.

1.1.2 Measurement of Microplastics and Airborne Microplastics

Microplastic research in Nigeria has predominantly focused on aquatic and terrestrial environments, with most studies measuring microplastic concentrations in water bodies and soil (Aragaw, 2021a). This focus stems from the significant pollution of rivers, lagoons, and coastal areas due to poor waste management, industrial activities, and the prevalence of plastic usage. Additionally, the methodologies for microplastic analysis in water and soil are well-established, allowing researchers to conduct studies with relatively accessible equipment and procedures (Enyoh et al., 2023). However, the measurement of airborne microplastics (AMPs) remains significantly understudied.

This domain lacks research partly due to limited awareness of airborne microplastic pollution and its potential health and environmental consequences (Rednikin et al., 2024). Furthermore, the technical challenges associated with sampling and analysing airborne microplastics, including the need for specialised air filtration systems and spectroscopic identification methods, contribute to the research gap (Campanale et al., 2020). Unlike water and soil, where microplastic contamination is visually evident and can be collected through sedimentation or filtration techniques, airborne microplastics require advanced sampling strategies to capture minute particles suspended in the atmosphere (Enyoh et al., 2019; A. B. Silva et al., 2018).

Microplastic measurement in water bodies is typically conducted using filtration and sieving, where water samples are collected and passed through filters of varying mesh sizes to retain microplastic particles (Seghers et al., 2022a). Density separation techniques using sodium chloride or zinc chloride to separate microplastics from organic matter, while spectroscopic analysis, including Fourier-transform infrared (FTIR) and Raman spectroscopy, helps in plastic polymer identification (Quinn et al., 2017). Similarly, soil microplastics are measured using extraction with density separation, where microplastics are isolated using high-density salt solutions (K et al., 2021a).

On the other hand, airborne microplastics pose unique challenges in measurement due to their small size and dispersion in the atmosphere. Active sampling methods include high-volume air samplers that draw higher volumes of air through filters (e.g., quartz or PTFE filters) to trap microplastic particles, and low-volume samplers used in areas with lower airborne particle concentrations (Choudhury et al., 2023). Passive sampling methods involve deposition collectors, such as open containers or adhesive surfaces that collect microplastics settling from the air over time, and sticky tape samplers placed at different heights to capture airborne microplastics (Azari et al., 2023a; Gwinnett et al., 2021a). Once collected, airborne microplastics are analysed using microscopy, including optical and electron microscopy for morphological analysis, spectroscopy techniques like FTIR and Raman spectroscopy for polymer composition confirmation, and thermogravimetric analysis (TGA) to determine the thermal degradation profiles of microplastics (Circelli et al., 2024; Mansa & Zou, 2021; Richter et al., 2023).

The measurement of airborne microplastics is crucial for understanding their role in atmospheric pollution, human exposure, and environmental transport. In Nigeria, where waste mismanagement and open burning are prevalent, airborne microplastics may pose significant health risks, including respiratory issues and systemic toxicity. Research in this area is essential for developing regulatory policies and mitigation strategies to reduce microplastic emissions from various sources. Despite the increasing recognition of microplastic pollution, researchers in Nigeria have focused on water and soil, leaving airborne microplastic measurements relatively unexplored. Several factors limit research on airborne microplastics in Nigeria, including:

- **High cost of commercial air sampling equipment:** The specialised equipment required for airborne microplastic sampling, such as high-volume air samplers and electron microscopy, is often expensive and inaccessible to many researchers in Nigeria.
- **Limited availability of affordable and efficient air samplers:** Some studies have developed low-cost air samplers through customisation and innovation (Gaston et al., 2020). The efficiency of these air samplers is yet to be elevated to match the standards of commercial equipment.
- **Limited expertise and resources for sampling and analysis**
- **Limited funding for research initiatives**

Due to potential health and environmental implications, expanding research efforts to include airborne microplastics is crucial. Implementing systematic sampling and advanced analytical techniques will yield essential data for evaluating exposure risks and guiding sustainable waste management practices. Given the resource limitations in Nigeria, this research sought to establish a cost-effective yet reliable method for measuring AMP emissions. By generating accurate and validated emission data, the study contributed to a refined understanding of microplastic flux from MSW which can potentially inform policies for managing AMP pollution. Additionally, it addressed critical gaps in understanding the fate and transport of microplastics, as well as the exposure risks associated with AMP flux in regions with inadequate waste management and environmental monitoring infrastructure.

To comprehensively understand the scope of microplastic pollution, it is imperative to examine the current state of scientific knowledge on the subject. The next chapter

reviews the existing literature on microplastic contamination, focusing on contemporary methodologies for detection and quantification, established exposure pathways, transport mechanisms, and the environmental fate of microplastics. Special attention is given to studies conducted within the Nigerian context, where empirical data remain limited but increasingly vital

1.2 Research Aim and Objectives

This study aims to identify and characterise the release of airborne microplastics from a typical municipal solid waste facility in Nigeria and develop a framework for using a dispersion model to assess exposure risk.

This research aimed to enhance understanding of airborne microplastic emissions from a typical municipal solid waste facility in Nigeria, focusing on emissions and dispersion in the surrounding environment. By developing a dispersion model to assess exposure risk, the study sought to predict exposure concentrations of airborne microplastics downwind of the disposal site. Ultimately, the findings were intended to inform the public and environmental agencies about the need for regulations and enforcement in plastic waste management. To achieve this aim, the following three objectives were established:

The first objective is to develop a low-cost air sampler for collecting airborne microplastics (AMP) from dumpsites and validate its efficiency against a reference sampler.

This thesis commenced by constructing a low-cost sampler to collect airborne MPs from the municipal solid waste facility. The low-cost sampler was then tested against the commercial sampler to assess its suitability and efficiency for collecting airborne MPs at different seasons. The result of the two samplers was compared with the view:

- 1) To design and test the low-cost air sampler for airborne microplastic collection from a typical municipal solid waste dumping site.
- 2) To assess the performance of the low-cost sampler from a typical municipal solid waste dumping site against the commercial sampler. These include the

seasonal performance and variations in the shapes, sizes, and polymer composition of the airborne MPs collected from the two air samplers.

- 3) To compare the cost of the low-cost air sampler with the commercial air sampler, including material, construction, and operational costs, and assess its economic viability and potential for widespread adoption in research and monitoring of airborne microplastics.

The second objective is to model the resuspension flux of airborne microplastics (AMP) from soil to air at dumpsites by utilising site-specific environmental data and providing input data for dispersion modelling.

The flux model was developed using site-specific environmental data and particle properties to estimate the resuspension of airborne microplastics (AMP) from soil to air at dumpsites. The output from this model was then used to provide input data for the dispersion model. The objectives of these were to:

1. Model the resuspension flux of airborne microplastics from soil to air at municipal solid waste dumpsites.
2. Derive emission factors from the model to provide input data for atmospheric dispersion modelling.

The third objective is to develop a framework for using a dispersion model to predict the risk of exposure to airborne microplastics.

The emission factor obtained from objective two of this thesis work was then used as input data to run the dispersion model. However, additional ground measurements were conducted at the sensitive receptor. These measurements were taken with a view to:

- 1) Validate the model by comparing its results with observed microplastic concentrations in the surrounding environment.
- 2) To determine the airborne MP polymer composition in each zone
- 3) To determine the risk associated with the exposure zones using an exposure assessment model.

1.3 Thesis Structure

This study aims to develop and validate low-cost equipment for sampling airborne microplastics in poor resource locations (Nigeria). The development of this equipment will help bridge critical knowledge gaps in designing affordable tools for airborne microplastic (MP). Such low-cost, adaptable sampling methodologies have the potential to amplify research contributions from resource-constrained regions, fostering a more inclusive understanding of global microplastic pollution. The study also investigates the emissions, characterisation, dispersal, and transport of airborne MPs from municipal solid waste to the receptor's location. Additionally, the research examines the exposure zone and associated risks with the emission of airborne MPs

Figure 1.1 Schematic structure of the thesis work.

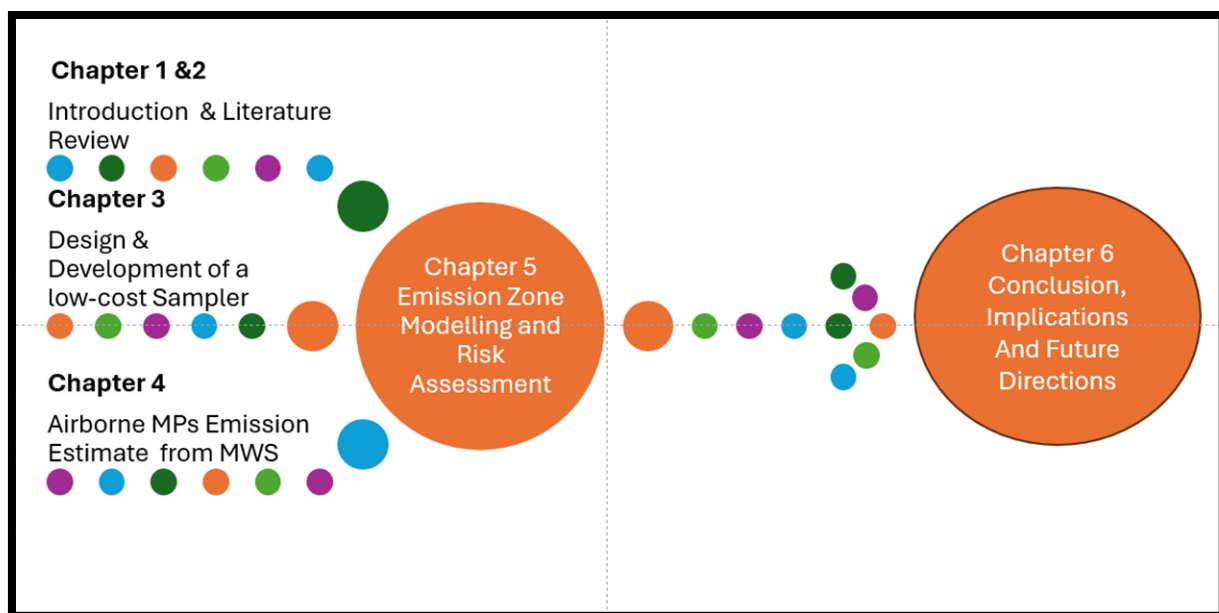


Figure 1. 1: Schematic Diagram of the Thesis Structure

Chapter Two: Literature Review

This chapter provides a comprehensive overview of the current state of airborne microplastic (MP) pollution from municipal solid waste (MSW) facilities, examining the associated exposure risks. The review lays the groundwork for understanding the complex issues surrounding airborne MP pollution and its potential impact on human health and the environment.

Chapter Three: Design and Development of a Low-Cost Air Sampler for Airborne Microplastic Collection

This chapter addresses Objective 1. It outlines the design and construction techniques for developing a low-cost air sampler for collecting airborne microplastics (MPs). The chapter also presents the results of a validation test, where the designed sampler was compared to a commercial sampler to determine its effectiveness in collecting airborne MPs.

Chapter Four: Analysis of Airborne Microplastics from Municipal Solid Waste

This chapter addresses Objective 2 by analysing airborne microplastics (MPs) collected from three key locations. These are upwind, on-site, and downwind of the municipal solid waste facility. The analysis explores the distribution and characteristics of airborne MPs in these areas. This study aims to shed light on their potential environmental impact. Specifically, it investigates whether emissions of airborne MPs lead to increased levels of MPs in nearby soil and air samples. The results will help clarify the influence of the facility on local pollution levels.

Chapter Five: Emission Zone Modelling and Exposure Risk Assessment

Based on the ground measurement in Chapter 4, on-site data from Chapter 4 is used as a preliminary input in Chapter 5 to inform dispersion modelling. This dispersion model is applied to map the airborne microplastic emission zones (MP). Air samples are then collected from each emission zone and analysed to determine the exposure risk associated with airborne MPs.

Chapter Six: Conclusion, Implications, Contributions, and Future Directions

This final chapter summarises the main findings from Chapters 3, 4, and 5, highlighting the aspects of microplastic emissions, exposure risks, and emission zone modelling. The research shows that airborne microplastics are a significant environmental concern, affecting policy, public health, and waste management. Despite some limitations, the study provides valuable insights that can inform policy decisions, protect public health, and improve waste management. Future research should focus on understanding how microplastics move and change, and their potential impacts on health and the environment

Chapter 2.0: LITERATURE REVIEW

This chapter provides an in-depth review of the current understanding of airborne microplastics (AMPs), encompassing their collection, detection, and analysis. With an emphasis on the Nigerian context, this review explores the unique challenges and implications of plastic pollution in the country. The chapter synthesises the existing research and highlights the methods, techniques, and findings on AMPs while also discussing the environmental and health impacts of plastic pollution in Nigeria.

2.1 Introduction

The presence of airborne microplastics in urban, suburban, and remote areas is evidence that microplastics can travel long distances through the atmosphere (Liu, Li, et al., 2019a). The occurrence, fate, transport, and impact of atmospheric microplastics raise concerns due to the limited physical analysis and understanding of this type of pollution, connected with the absence of standardised methods for sampling and identification (Jenkins et al., 2022). These questions arise due to inadequate atmospheric microplastic pollution studies to compare with, coupled with the lack of standardised sampling and identification methods.

Plastic debris can be classified by various names and size ranges (Table 2.1). This classification provides a framework for understanding their composition, sources, and environmental occurrence across different ecosystems. Microplastics are small plastic debris of less than 5 millimetres in size (Thompson et al., 2004). Although there is no universally agreed-upon minimum size for microplastics, the International System of Units (SI) has adopted a lower limit of 1 μ m (Hartmann et al., 2019). Recent research on plastic particles has raised concerns regarding the prevalence of Nanoplastics, which are expected to be as ubiquitous as their larger counterparts (Alimi et al., 2018a). Nanoplastics have a particle size range of 1 nm to 1 μ m (Gigault et al., 2018b). Owing to their small size, nanoplastics can be ingested by marine organisms, posing a risk to the environment and humans (Tirkey & Upadhyay, 2021a). However, environmental nanoplastics have not received much attention; therefore, microplastics will be the focus of this review.

Table 2. 1: Classification of Plastic Debris by Size, Composition, and Environmental Occurrence

Plastic Type	Size Range	Common Composition	Typical Environmental Occurrence	References
Macroplastics	> 25 mm	PET, HDPE, PP, PS	Roadsides, beaches, open dumpsites, rivers	Caldwell et al., 2022; Galloway, 2015
Mesoplastics	> 5 mm – 25 mm	PET, PS, PVC	Urban runoff, riverbanks, and landfills	Jeyasanta et al., 2020
Microplastics	1 μ m – 5 mm	Nylon, PE, PVC, PS, PP	Soil, air, freshwater, and marine environments	Gigault et al., 2018; Zhao et al., 2023a
Nanoplastics	1 nm – 1 μ m	Degraded PE, PP, polystyrene	Airborne particles, surface water, and sediments	Pegado et al., 2024; Gigault et al., 2018

Microplastics are categorised as primary and secondary microplastics (Hanik et al., 2019), with primary microplastics intentionally produced for specific purposes, such as microbeads in personal care products like facial cleansers, toothpaste, and cosmetics (Khan et al., 2020). Secondary microplastics are smaller plastic particles that result from the degradation of plastic items, such as water bottles, straws, and plastic bags, including fibres and synthetic materials (Khan et al., 2022). The distinction helps scientists understand how far and how quickly different types of microplastics can travel from their sources, affecting deposition patterns in remote areas like the Arctic and high mountains (Cole et al., 2011).

Microplastics are also classified based on morphological and geometrical properties (Silva et al., 2020). They are also classified based on their chemical compositions (Burnley, 2007). Microplastics come in various shapes, each with distinct characteristics and potential environmental impacts. Fragments are irregularly shaped microplastics that result from the breakdown of larger plastic items, such as plastic bags, bottles, or other debris (Na et al., 2024). On the other hand, fibres are linear, thread-like microplastics often released from clothing, textiles, or fishing nets

(Gliaudelytė et al., 2024). Beads are spherical microplastics commonly found in cosmetics and industrial applications, while films are sheet-like microplastics originating from plastic bags, wrappers, or flexible packaging materials. Indoor environments typically contain microplastics in the form of fibres, whereas outdoor environments are more likely to contain fragments (K. Liu, Wang, et al., 2019). However, the shape and aerodynamics of microplastic particles can also significantly influence the transport of plastics into the atmosphere (Preston et al., 2023).

In terms of polymer type, microplastics are made from various materials, each with unique properties. The most produced polymers are polypropylene (PP), low-density polyethylene (LDPE), high-density polyethylene (HDPE), polyvinyl chloride (PVC), polyurethane (PUR), polyethylene terephthalate (PET), and polystyrene (PS) (A. L. Lusher et al., 2021). Studies on polymer composition in seawater show that PE is the predominant polymer, followed by PP and PS. PE, PP, PS, and PES are the main types found on beaches and in subtidal sediments (S. Allen et al., 2020).

The chemical composition of airborne microplastics differs across various regions. In Shanghai, synthetic compounds accounted for 54% of the observed particles, with PET, PE, PES, PAN, PAA, and rayon making up 91% of the microplastics (K. Liu, Wu, et al., 2019a). In Dongguan City, microplastics consisting of three different polymers (PE, PP, and PS) were identified (Cai et al., 2017a). In another study, the predominant microplastics were found in dust deposition with polymers of PET and PC (C. Liu et al., 2019b).

In Hamburg, Germany, atmospheric microplastic samples are of PE and ethyl vinyl acetate (EVA) copolymers, constituting 48.8% and 22.0% (Klein & Fischer, 2019a). In Shanghai, China, the predominant polymers found in municipal solid waste dumpsites were rayon (69%), followed by nylon (7%), with the winter season having the highest prevalence (Hu et al., 2022b). Conversely, the most abundant polymers at a composting facility in England consisted of 50% polyvinyl alcohol (PVA), 17% polypropylene (PP), 17% polyvinyl chloride (PVC), 13% polystyrene (PS), and 4% polyethylene (PE) (Zapata et al., 2024b).

Currently, there is no definitive correlation or explanation for the variability in polymer types found in atmospheric samples. Further research is necessary to determine whether a predominant group of polymers exists within atmospheric microplastic

pollution and if the composition of these polymers varies with the location of the sample and the distance the particles have travelled.

The atmosphere remains an important route through which pollutants are transported locally or globally, due to the constant movement of pollutants driven by wind and pressure systems (D. Allen et al., 2022b; S. Allen et al., 2021). These dynamics enable the local redistribution and long-range dispersion of contaminants, contributing to their widespread environmental presence and exposure. Recent studies have suggested that airborne MP particles can travel from terrestrial to marine environments (Geyer et al., 2017). The direction, speed, and horizontal and vertical drafts of the wind are among the dynamics that influence the movement of microplastics in the atmosphere (Liu, Wu, Wang, et al., 2019a). Turbulence and convection are believed to be significant vectors that affect the transmission of airborne MPs (Liu, Wang, Fang et al., 2019). Most studies published to date have focused on atmospheric deposits that involve the passive collection of microplastics at a specific location (Kernchen et al., 2022a). Numerous studies have investigated the occurrence of microplastics in terrestrial atmospheric environments (A. Lusher, 2015; Peter Sundt, Per-Erik Schultze, 2014). However, the investigation into the presence of microplastics in the atmosphere surrounding municipal solid waste sites over prolonged duration's remains limited (Bergmann et al., 2015).

To gain a deeper understanding of airborne microplastics, it is essential to collect and compare data from various environments on the occurrence of microplastics in the air. This literature review provides a concise overview of the latest research on airborne microplastic pollution. It highlights recent advancements, identifies ongoing gaps in our understanding, and proposes avenues for future studies to promote uniformity and comparability in research efforts.

2.2 Microplastic exposure and human health

Microplastic exposure is a growing health concern, with humans potentially being exposed through multiple routes, including inhalation of airborne particles, ingestion of contaminated food and water, and skin contact, as highlighted in Figure 2.1. These exposure pathways may lead to various adverse health effects, including physical harm from particle accumulation or blockages, chemical transfer where microplastics act as vectors for toxic substances like PCBs, DDT, and phthalates, and biological impacts such as inflammation, toxicity, or immune responses (A. Sun & Wang, 2023b).

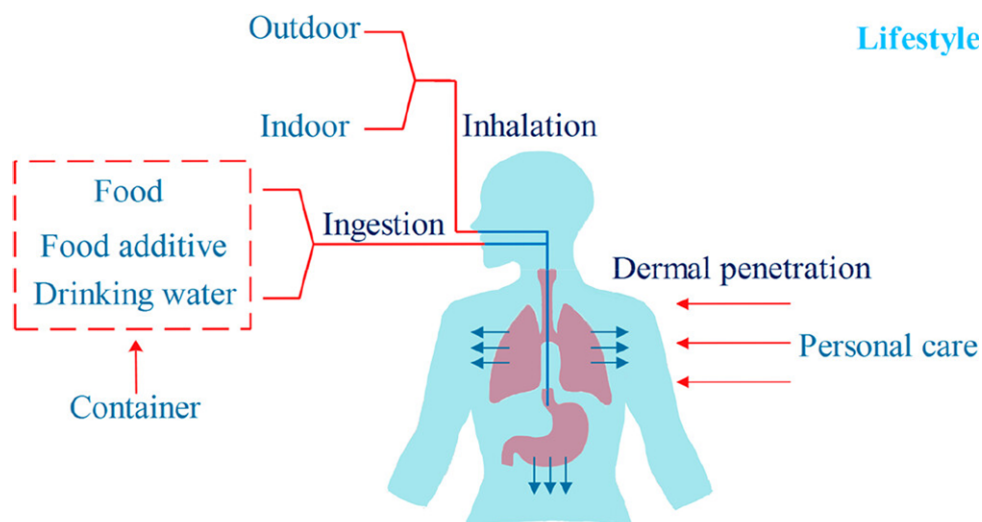


Figure 2. 1: Microplastic Exposure in Humans (Sun & Wang 2023).

2.2.1 Inhalation Route

Inhalation is a significant exposure route for microplastics, particularly in areas with high levels of airborne microplastics. These areas include urban centres, industrial zones, and waste management sites. Microplastic particles with smaller diameters, especially those smaller than 10 μm , can stay in the air and be inhaled, reaching deep into the lungs (N. Kumar et al., 2024).

The disintegration of materials like clothing, furniture, and building products releases microplastics into the air, contributing to secondary microplastic exposure via inhalation (L. Rahman et al., 2021). Synthetic textiles and urban dust are primary sources of microplastics (Correia Prata, 2017). Inhalation can pose significant health risks due to microplastic contamination in the air (Lai et al., 2022).

- **Indoor and Outdoor Inhalation Risks:** Indoor environments typically have higher concentrations of suspended microplastics (1583 MPs/m³), comparable to outdoor settings (Dris et al., 2017a). Research suggests humans inhale up to 130 airborne microplastics daily, posing direct lung health risks (Correia Prata, 2017). In contrast, the rural microplastic is smaller in size and concentration of 101 MPs/m³ (Vianello et al., 2019a).
- **Occupational Exposure:** Certain occupations, particularly those involved in plastic manufacturing, recycling, or waste management, have a higher risk of microplastic inhalation (Shahsavaripour et al., 2023). For example, workers in

textile industries are at risk of being exposed to synthetic fibres and microplastics (Murashov et al., 2021). The constant handling and processing of plastics in these workplaces can lead to elevated concentrations of airborne particles, increasing the risk of inhalation and subsequent respiratory issues (Xie et al., 2022).

2.2.2 Ingestion Route

The ingestion of microplastics can occur mainly through food, additives, drinking water, and packaging containers:

- **Food Processing, Additives, and Packaging:** Microplastics have been detected in various food and water sources, including bottled and tap water, seafood, fruits, vegetables, groundwater, and treated drinking water. This widespread contamination highlights the need for concern about the potential impact on human health and the environment. (Pham et al., 2023). Marine pollution contaminates seafood, making it a substantial source of microplastic intake. Studies reveal that marine environments can contain alarming levels of microplastics up to 102,000 particles per cubic meter, which are then ingested through seafood consumption (Hossain et al., 2022). Although the World Health Organisation (WHO) has found no immediate health risks from microplastics in tap water, the long-term effects of microplastic consumption remain unknown (Shao et al., 2022a; Q. Zhao et al., 2023). Further research is essential to understand the potential health implications.
- Microplastics can enter food through packaging and processing when plastic containers or wraps are employed (Sobhani et al., 2020). Household plastics, like bottles and wrappers, shed microplastics into food, especially under heat (Pham et al., 2023). Urban settings with widespread plastic packaging increase the risk of ingesting microplastics. Human dietary patterns significantly impact microplastic consumption. For example, Europeans ingest 37 microplastics (numbers) annually through table salt, while Chinese individuals consume 101 particles (Karami et al., 2017; B. Wei & Yang, 2010). This disparity highlights how regional food packaging influences microplastic intake. Research has found microplastics in popular beverages, including soft drinks, cold teas, and

energy drinks. These microplastics range in size from 0.1 to 3 millimetres, likely resulting from the complex manufacturing processes involved (Shruti et al., 2020). Food packaging can leach chemicals into food, posing a potential health risk. This transfer of chemicals from packaging to food increases human exposure to harmful additives (Pham et al., 2023). Microplastics can leach from plastic containers into food and drinks. Drinking bottled water, for instance, adds 90,000 microplastic particles annually (Makhdoumi et al., 2021).

2.2.3 Dermal Route

The dermal pathway facilitates the absorption of deleterious chemicals associated with plastics, including phthalates and bisphenols, identified as endocrine disruptors. Microplastics have been shown to act as vectors for these toxic additives, which can subsequently leach into the bloodstream via cutaneous exposure (Burgos-Aceves et al., 2021). Given the extensive surface area of human skin, approximately 1.5-2 m², this organ provides a substantial interface for exposure to environmental microplastics.

Recent studies have highlighted the ability of certain chemicals to penetrate the skin barrier, raising questions about their impact on human health (Alijagic et al., 2024; Jayavel et al., 2024a). Research has shown that the skin is a significant route of exposure for various chemicals, including those found in plastics (Burgos-Aceves et al., 2021). The dermal absorption of these chemicals through skin can lead to systemic toxicity, highlighting the need for further investigation into the potential health consequences.

Further research is needed to investigate the types of chemicals that can penetrate the skin barrier, the rates and mechanisms of dermal absorption, and the impact of skin conditions on absorption rates. Additionally, studying the potential health effects of long-term exposure to these chemicals through dermal absorption is crucial. By gaining a deeper understanding of these factors, researchers can inform the development of effective strategies for reducing exposure and mitigating potential health effects, ultimately contributing to a better understanding of the risks associated with dermal absorption of toxic chemicals from plastics.

2.2.4 Potential Health Impact

The health risks associated with microplastic exposure are still under investigation, but some concerns have emerged:

- **Particle Size:** The size of microplastic particles plays a crucial role in their potential health effects (Y. Li et al., 2024). The smaller particles, particularly those in the nano-plastic range (below 1 μm), may be more dangerous as they are more likely to cross biological barriers, such as the respiratory and gastrointestinal lining, and enter the bloodstream (Lear et al., 2021). Once in the bloodstream, microplastics can potentially reach other organs, leading to a range of health outcomes, though the full extent of their impact is understudied.
- **Chemical Additives:** Microplastics often contain chemical additives, such as plasticisers, flame retardants, and stabilisers (Yu et al., 2024). These additives can leach out of the microplastic particles and have toxic effects on human health, including endocrine disruption, reproductive issues, and carcinogenicity (Jayavel et al., 2024b). Furthermore, microplastics can act as vectors for environmental pollutants, such as heavy metals and persistent organic pollutants (POPs).
- **Inflammatory Responses:** Studies have suggested that inhaling or ingesting microplastic particles may trigger inflammatory responses in the body (A. Sun & Wang, 2023c). Inhaled microplastics may lead to respiratory issues, such as asthma or chronic inflammation of the lung tissue (Pimentel et al., 1975). Ingested microplastics may cause gut irritation and disrupt normal gut microbiota, potentially affecting digestive health (Fang, et al., 2019).

2.3 Influence of Waste Treatment Methods on Microplastic Emissions

The environmental impact of microplastics originating from disposed waste is significantly influenced by the type of waste treatment method employed. The waste treatment method can impact the quantity of microplastics released into the environment. Here are some different ways waste treatment methods can influence microplastic pollution.

Open dumping is an uncontrolled deposition of municipal solid waste directly onto land without engineering or environmental safeguards (Akintola et al., 2021). This practice presents serious public health hazards, including groundwater pollution, the release of greenhouse gases, and the spread of disease-carrying organisms (Tun et al., 2022). Open dump sites are increasingly recognised as significant sources of airborne pollutants, including emerging contaminants like bioaerosols and microplastics, posing environmental and health risks (Shammi et al., 2021; Stapleton & Hai, 2023). The combination of unregulated waste burning, wind-driven particle resuspension, and the degradation of plastic waste in these environments contributes to the release of microplastics into the atmosphere (Pathak et al., 2023). Despite their widespread presence, the role of open dump sites in airborne microplastic emissions remains understudied, underscoring the need for targeted research in this area.

Landfilling, for instance, facilitates the long-term degradation of plastic materials, often leading to the formation of leachate that can carry microplastics into surrounding soils and water systems (Petrović et al., 2023). This leachate can contaminate groundwater, surface water, and soil, posing significant environmental risks. Microplastics in soil can alter its structure, fertility, and biodiversity, while in water systems, they can harm aquatic life and potentially work their way up the food chain (Bodor et al., 2024).

The environmental impact of microplastic pollution from landfills is further compounded by the fact that these particles can persist in the environment for extended periods (Ihenetu et al., 2024). To mitigate these impacts, effective landfill management strategies are crucial. This includes implementing efficient leachate collection and treatment systems, designing landfills with microplastic mitigation in mind, and reducing the plastic waste sent to landfills through recycling and waste reduction efforts.

Incineration reduces waste volume but releases airborne pollutants, including microplastics and toxic residues in the form of ash (Shen et al., 2021; Z. Yang et al., 2021a). Incineration reduces waste volume but poses significant environmental risks due to the emission of airborne pollutants and toxic residues (Z. Yang et al., 2021a). When plastic waste is incinerated, microplastics can be released into the atmosphere, contributing to air pollution and potentially being deposited in surrounding

environments (Z. Yang et al., 2021b). This can lead to widespread contamination, as microplastics can be transported long distances by wind and water.

Advanced incineration technologies with robust air pollution control systems are crucial to mitigate these impacts. These systems can capture particulate matter, including microplastics, remove acidic gases, and adsorb volatile organic compounds. However, advanced incineration is not completely clean, and the residual ash still poses environmental risks.

Recycling is generally considered a sustainable alternative; however, it may inadvertently generate microplastics through mechanical processing and material breakdown (Suzuki et al., 2022). As plastics are fragmented and processed during recycling, they can break down into smaller particles, including microplastics. This can occur through various mechanisms, such as shredding, grinding, and melting, which can cause physical degradation of the plastic materials (Bekwapu et al., 2024; Dumbili & Henderson, 2020). Furthermore, the quality of the recyclable materials can also impact microplastic generation. Contaminated or mixed plastics can be more prone to breakdown during recycling, leading to increased microplastic formation. To mitigate these impacts, improve recycling technologies, enhance sorting and separation processes, and design products with recyclability in mind.

Composting and biological treatment: Composting and biological treatment are effective methods for managing biodegradable plastics, but they require careful implementation to avoid contamination (Lara-Topete et al., 2024). When improperly mixed with organic waste, conventional plastics can disrupt the composting process and form microplastics. This contamination can have significant environmental impacts, including microplastic pollution into soil and water systems, potentially harming ecosystems and human health (Ebrahimi et al., 2022; Rodrigues et al., 2019). To mitigate these risks, establish robust waste management systems that ensure proper sorting of biodegradable and non-biodegradable plastics. This can be achieved through education and awareness campaigns, effective waste collection infrastructure, and clear labelling and certification of biodegradable products.

Advanced treatment technologies, such as chemical depolymerisation and high-pressure mechanical processes, offer varying efficiency in managing plastic waste (Puteri et al., 2025). These technologies can break down plastics into their chemical building blocks or raw materials, enabling closed-loop recycling and reducing the volume of waste sent to landfills (Liguori et al., 2021). Chemical depolymerisation, for

instance, uses chemical reactions to break down plastics into monomers, which can then be reused to produce high-quality plastic (Clark & Shaver, 2024). The efficiency of these technologies can vary depending on factors such as the type of plastic being processed, the energy consumption, and the quality of the output materials (Fernandes, 2021). Nevertheless, these advanced treatment technologies have the potential to significantly reduce plastic waste, conserve resources, and promote a more circular economy.

The environmental implications and ability to contain microplastic emissions remain under evaluation. Collectively, these treatment pathways illustrate the complex relationship between waste management strategies and the release of microplastics into the environment. This study reviews and summarises findings from existing literature, highlighting the influence of treatment methods:

- Hu et al. (2022) conducted the first study on airborne MP emissions from the Shanghai municipal solid waste transfer station. They suggested that emissions from waste transfer stations may contribute to the accumulation of microplastics in terrestrial ecosystems. The concentration found in the transfer station exceeded $2.5 \pm 1.3 \text{ MP/m}^3$. Furthermore, the study estimated that the emissions of airborne MP from the transfer station vary between 41,297 and 82,593 particles per hour, fluctuating with seasonal changes (Hu et al., 2022). In another study, Zapata et al. (2024) conducted a study at a municipal solid waste composting facility in England and suggested that the on-site concentration of airborne microplastic was 32 to 49 MP m^{-3} , which is significantly higher than the upwind (1 MP m^{-3}) and downwind (3 MP m^{-3}) concentrations (Zapata et al., 2024b).
- Kannankai and Devipriya (2024) investigate the influence of microplastic emissions from landfill sources on monsoon winds, deposition and characteristics of atmospheric microplastics in Kochi, Kerala, India. The research focuses on seasonal variations and the impact of wind patterns originating from a municipal landfill site on microplastic distribution. Below are some key points and insights:

- The study measured a deposition flux averaging 21.27 particles/m²/day, with peak levels during the Northeast Monsoon (NEM), reaching 37.29 particles/m²/day.
- Backwards trajectory analysis using HYSPLIT indicated that NEM winds pass over the Brahmapuram landfill, suggesting the landfill is a primary source, with the potential to transport particles to the Arabian Sea. (Kannankai & Devipriya, 2024a)

The above studies highlight how different waste treatment methods directly influence the concentration and dispersion of airborne microplastics. Transfer stations (Hu et al., 2022), composting facilities (Zapata et al., 2024), and landfills (Kannankai & Devipriya, 2024) were all identified as significant emission sources, with microplastic levels varying widely based on treatment type and seasonal conditions. These findings underscore the role of waste management practices in shaping microplastic pollution levels in the surrounding environment. However, further studies are needed to establish and optimise treatment methods that emit lower concentrations of microplastics and minimise their environmental impact.

2.3.1 Nigeria's Waste Treatment System and Microplastic Emissions

Nigeria's waste treatment system, like those in developing nations, is characterised by widespread reliance on open dump sites, limited recycling infrastructure, and inadequate segregation of plastic waste (Sobhani et al., 2020; Velis & Cook, 2021). These practices create favourable conditions for microplastic emissions through wind-driven dispersal, open burning, and improper waste handling. With little to no environmental control at disposal sites, plastics undergo physical fragmentation and chemical degradation, generating airborne microplastic particles that pose risks to nearby communities and ecosystems.

Nigeria presents a compelling case study of microplastic emissions, driven by harsh environmental conditions, demographic factors, and policy-related influences. Nigeria's population has surpassed 220 million and is experiencing rapid urbanisation (Pona et al., 2021). The situation is further complicated by factors such as inadequate waste collection and disposal infrastructure, resulting in widespread littering and the

open burning of plastics. Nigeria's extensive coastlines and waterways also facilitate the transport and accumulation of microplastics in the environment. These challenges underscore the need for targeted research and policy interventions to mitigate microplastic pollution in Nigeria.

The country generates 27.6 million tonnes of municipal solid waste annually, of which 13% (roughly 3.6 million tonnes) is plastic waste (Babayemi et al., 2018). This growing waste burden, placed on an already under-resourced management system, increases the likelihood of uncontrolled microplastic release into the environment. The major disposal sites in Nigeria (Table 2.2) include the Lagos (established 1992; 2.1 million t/yr; 43 ha; 18 m depth), Solous 2 also in Lagos Olusosun (2006; 820 000 t/yr; 8 ha), and Epe (2010; 12 000 t/yr; 80 ha), as well as Awotan (Apete) in Ibadan (1998; 36 000 t/yr; 14 ha), Lapite in Ibadan (1998; 9 000 t/yr; 20 ha), and Eneka in Port Harcourt (45 600 t/yr; 5 ha), all of which operate as open dump sites and contribute to the uncontrolled release of microplastics into the surrounding environment (Arogundade, 2022a).

Table 2. 2: Landfills in Nigeria.

Item	Olusosun	Solous 2	Epe	Awotan (Apete)	Lapite	Eneka
Year of establishment	1992	2006	2010	1998	1998	-
State	Lagos	Lagos	Lagos	Ibadan	Ibadan	Port Harcourt
Tonnes/year	2.1 million	820,000	12,000	36,000	9,000	45,600
Area in hectares	43	8	80	14	20	5
depth	18 meters	-	-	-	-	-

The majority of waste in Nigeria is disposed of through open dumping or informal burning, practices that are largely unregulated and lack environmental safeguards and protections (Daffi et al., 2020). These conditions create multiple pathways for microplastics to enter the atmosphere, soil, and aquatic systems. Compounding this issue is the limited empirical data available on airborne microplastics within the Nigerian context. While studies have documented microplastic contamination in water bodies (AKINBILE, 2012) and soils, airborne microplastics remain severely

understudied. This represents a critical research gap, particularly considering the proximity of communities to waste sites and the potential health risks associated with inhalation.

Furthermore, Nigeria's distinct climatic patterns, characterised by alternating dry and wet seasons, offer a unique opportunity to assess how seasonal variations influence microplastics transport, deposition, and resuspension. This environmental variability can provide valuable insights into the behaviour of microplastics under different atmospheric conditions.

The absence of effective regulation and the informal nature of waste disposal heighten exposure risks, particularly among vulnerable populations living near open dump sites. Research in this area can inform targeted interventions, guide sustainable waste management strategies, and inform regional policies for reducing microplastic pollution and its associated impacts in Nigeria.

2.3.2 Microplastic and Airborne Microplastic Studies in Nigeria

Microplastic pollution has become a pressing environmental concern globally, and Nigeria is not an exception (Enyoh et al., 2023). Scientific investigation into microplastics within Nigeria is a relatively recent endeavour, with the earliest documented studies dating back to 2019 (Ebere et al., 2019). An initial phase of research activity occurred between 2019 and May 2021, yielding twelve distinct studies focused on detecting and quantifying microplastics across various environmental settings. These foundational studies geographically concentrated on six states in the southern part of the country (Figure 2.2): **Bayelsa, Imo, Ogun, Osun, Oyo, and Rivers** (Enyoh et al., 2023).

These initial studies on microplastic contamination warrant further exploration, particularly in understanding the underlying factors driving this research. Potential drivers include local ecological disruptions, the impact of global research and international collaborations, or specific human activities common in southern states, such as industrial operations, agricultural practices, or tourism.

The researchers examined several critical environmental matrices known to accumulate or transport microplastics, including:

- **Surface water:** Assessing contamination levels in rivers and lakes.

- **Sediment:** Investigating accumulation zones where microplastics often settle.
- **Biota:** Analysing presence in aquatic organisms, indicating potential food web transfer.
- **Food:** Exploring direct contamination pathways to humans.
- **Land/Soil:** Examining terrestrial deposition.

A crucial finding from this initial research period was that **all twelve studies reported detectable levels of microplastics**, underscoring the ubiquity of this pollutant within the environments studied. Notably high concentrations were documented in Ox-Bow Lake, located in Bayelsa State. During the rainy season, levels reached 8369 items per cubic meter (m³) in water and 7593 items per kilogram (kg) in sediment (Oni et al., 2020).

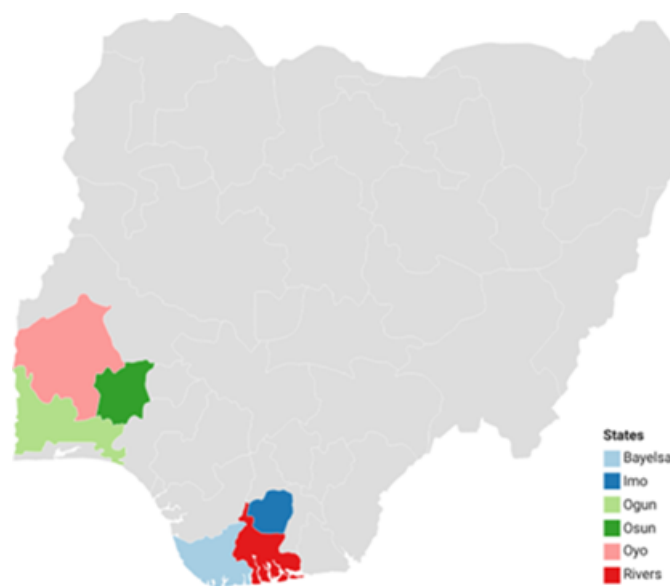


Figure 2. 2: Distribution of Microplastics Studies Across Six Southern Nigerian States

These figures are exceptionally high, placing them among the highest recorded concentrations for surface water bodies across the African continent at the time (Aragaw, 2021b), potentially indicating significant local sources or environmental factors favouring concentration.

However, despite these initial findings, the momentum in primary microplastic research within Nigeria appears to have significantly slowed since 2021. Literature indicates that fewer than five new studies were conducted between late 2021 and late 2023/early 2024 (Ololade et al., 2023). This apparent stagnation hinders the development of a more comprehensive national understanding of the issue. Recent scholarly work has consequently shifted towards synthesising the existing data. Review articles published up to early 2024 have focused on collating and analysing the findings from the original twelve studies conducted between 2019 and 2021 (Nyaga et al., 2024). While these reviews confirm the presence of microplastics in the studied regions, they highlight the significant gaps in the current knowledge base, emphasise the need for methodological harmonisation, and reiterate the urgency for renewed and expanded investigation into microplastic pollution across Nigeria. The lack of new primary data collection underscores the need for revitalisation in this research area, including expanding studies to unrepresented geographical regions.

A significant gap in Nigerian microplastic research is the absence of comprehensive studies focusing on Municipal Solid Waste (MSW) environments. Critical locations like landfills, dumpsites, and their leachate systems act as potential major sources and routes for microplastic migration into soil, water, and possibly the atmosphere, yet remain largely unstudied. Furthermore, while the section heading mentions Airborne Microplastics, the provided text indicates a critical gap: there appear to be no dedicated studies investigating airborne microplastic pollution in Nigeria reported in the referenced literature (Yalwaji et al., 2022). The potential sources (e.g., vehicle tire wear, synthetic textile fibres, agricultural plastics, fragmentation of waste), atmospheric concentrations, transport mechanisms, deposition patterns, and associated human health risks (particularly via inhalation) of airborne microplastics within the Nigerian context remain largely unexplored (Gan et al., 2023; Kabir et al., 2023; Ratnasari et al., 2024; X. Zhao et al., 2023a), representing a critical area for future research. Addressing these knowledge deficits is crucial for understanding the full scope of microplastic exposure and developing effective mitigation policies in Nigeria.

In many cases, open burning occurs due to inadequate waste management infrastructure (Daffi et al., 2020; Noiki et al., 2021; Okedere et al., 2019). Open burning becomes attractive when waste inflow exceeds the system's capacity; shipment and

landfill management become challenging, albeit harmful, solutions. However, this approach exacerbates the issue, releasing MPs into the environment and contaminating soil, water, and air (Velis & Cook, 2021).

In Nigeria, open burning of waste and poor waste management systems are significant contributors to airborne microplastics (MPs). Open burning of waste can lead to MPs' release into the atmosphere, where they can be transported over long distances (Noiki et al., 2021). High temperatures from burning waste break the plastics into smaller MP particles, increasing their airborne concentration. Furthermore, open burning generates ultrafine MPs ($<10\ \mu\text{m}$), which are more easily inhalable and potentially more harmful (Subramanian & Khatri, 2019). Moreover, inadequate waste processing and disposal can lead to MP formations through mechanical breakdown.

The consequences of airborne microplastics (MPs) in Nigeria are alarming. Inhaling them can lead to physical and chemical harm, potentially causing respiratory issues and other health problems. Airborne MPs can also settle on surfaces, contaminating soil, water, and vegetation in Nigeria's ecosystems. Additionally, they can be transported over long distances, spreading pollution to pristine areas.

Notably, most dump sites in Nigeria are nested within urban centres, exacerbating microplastic pollution (Arogundade, 2022b). This proximity poses significant risks to human health, as urban residents are more likely to inhale airborne MPs and experience related health issues.

Effective mitigation of microplastic pollution in Nigeria requires a nuanced understanding of its sources, pathways, and environmental and health impacts. To achieve this, experts from various disciplines must work together to assess the issue, evaluate its consequences, and develop sustainable solutions and policies tailored to Nigeria's urban context.

By addressing these knowledge gaps, researchers can inform evidence-based policies to mitigate microplastic pollution's impacts on human health and the environment. Potential research questions include investigating microplastic concentrations and characteristics in MSW environments, understanding microplastic transport mechanisms, and assessing human exposure routes and health impacts.

2.3.3 Air Quality Standards in Nigeria and the Overlooked Role of Microplastics

Globally, air quality standards have traditionally focused on pollutants such as particulate matter (PM_{2.5} and PM₁₀), sulphur dioxide (SO₂), nitrogen oxides (NO_x), carbon monoxide (CO), and ozone (O₃) all of which have well-established thresholds under frameworks like the World Health Organization (WHO) Air Quality Guidelines and national environmental protection agencies (Ravindra et al., 2024). These standards aim to mitigate respiratory and cardiovascular risks associated with long-term exposure to air pollutants (Adeoye et al., 2024; Manisalidis et al., 2020). Despite mounting evidence suggesting potential health risks, current environmental standards haven't recognised airborne microplastics (AMPs) as a separate pollutant.

Airborne microplastics can fall within the size range of PM₁₀ and PM_{2.5} (Subramanian & Khatri, 2019) they are not explicitly regulated or monitored. Scientifically, AMPs can be considered a subset of PM because of their size and airborne behaviour. However, AMPs are synthetic plastics with irregular shapes, such as fibres, films, and fragments, and contain additives like plasticisers and flame retardants (Y. Li et al., 2020). The airborne microplastics' unique shapes distinguish them from traditional particulate pollutants such as dust or soot. These characteristics raise distinct toxicological concerns, including respiratory stress, inflammation, and chemical exposure (Donaldson & Tran, 2002; J. J. Guo et al., 2020). These make them more than “ordinary” particulate matter. Studies have shown that AMPs can be inhaled, deposited in lung tissues, and potentially trigger inflammatory responses, similar to conventional particulate matter (Abad López et al., 2023; Zurub et al., 2024). Nevertheless, a globally accepted threshold for airborne microplastics does not exist, due to limited long-term toxicological data and the challenge of standardising sampling and analysis techniques.

In Nigeria, air quality is regulated by the National Environmental Standards and Regulations Enforcement Agency (NESREA), which sets permissible limits for conventional pollutants (FGN, 2021). For example:

- PM_{2.5}: 40 µg/m³ (24-hour), 20 µg/m³ (Annual).
- PM₁₀: 150 µg/m³ (24-hour), 60 µg/m³ (Annual).

These standards mirror international norms but do not account for microplastic particles, even though airborne MPs fall within the same size range as PM₁₀ and can exhibit similar airborne behaviour.

Given the growing global evidence on the health and environmental impacts of airborne microplastics (AMPs), there is an urgent need to broaden air quality standards to include this emerging pollutant class. Internationally, air quality regulations continue to focus on traditional pollutants such as PM_{2.5} and PM₁₀, yet microplastics, though often falling within these size ranges, remain unaccounted for in most monitoring and policy frameworks. This regulatory gap persists despite increasing documentation of AMPs in urban air, indoor environments, and especially near waste treatment facilities (Lin et al., 2023; Petrović et al., 2023; Shi et al., 2023). This omission is significant, especially considering Nigeria's heavy dependence on open dumping for waste disposal (Daffi et al., 2020), where exposed and degraded plastic waste fragments contribute to AMP release. These conditions create ideal scenarios for airborne microplastic emissions into surrounding communities and ecosystems. While Nigeria's current air quality regulations, set by NESREA, offer standards for particulate matter and gaseous pollutants, they do not explicitly address microplastics as a measurable pollutant or a health risk.

2.4. Plastic Legislation in Nigeria

Currently, there are no specific regulations targeting microplastics in Nigeria. However, the government has initiated plastic regulations to caution against the effects of microplastics. While these regulations do not focus exclusively on microplastics, they signify an important step towards tackling the increasing issue of microplastic pollution in the country.

The Nigerian, existing laws and regulations indirectly address the secondary sources of microplastics, thereby mitigating their detrimental environmental impacts (OBIDIMMA & AZUBUIKE, 2023). This legislative oversight necessitates an examination of the efficacy of current waste management frameworks in curbing microplastic pollution. This study aims to review the laws by investigating Nigeria's legal framework and its implications for microplastic pollution management.

2.4.1 Federal Legislation on Plastic Pollution in Nigeria

The federal government of Nigeria has enacted several laws to address plastic pollution. The Nigerian Environmental Protection Agency (NEPA) Act (1993) empowers NEPA to regulate and enforce environmental standards, including monitoring and controlling pollution (Eneh & Agbazue, 2011).

The National Environmental (Waste Management) Regulations (2009) establish guidelines for waste management, including plastic waste (FGN Gazette, 2009). This regulation mandates waste generators:

- To sort and segregate it before its disposal.
- The waste should be disposed of properly.
- The generators should provide waste management facilities.

The Harmful Waste (Special Criminal Provisions) Act (2004) prohibits dumping harmful waste, including plastics and prescribes penalties for offenders, comprising fines and imprisonment. These laws demonstrate the federal government's commitment to addressing plastic pollution.

2.4.2 Proposed Legislation

The government has announced an ambitious plan to ban single-use plastic materials by 2024 (Essen, 2019). This forward-thinking initiative is part of a broader project to enhance waste management and address environmental concerns related to plastic pollution.

Recent studies have raised concerns about the alarming levels of microplastics in the Osun River, which is one of the water sources for neighbouring communities (Idowu et al., 2024). These developments indicate a growing awareness and response to the challenges posed by plastic waste in Nigeria.

Additionally, in 2019, the Nigerian government (May 2019) passed the Plastic Bags Prohibition Bill (PBPB 2019). The purpose of the bill act is to prohibit the manufacture and import of plastic bags for commercial and household packaging, including those for food packaging (Nwafor & Walker, 2020). This bill comprises two clauses.

“Plastic bag Prohibition act (PBPB 2019)

1. Clause 1
2. The use, manufacturing, importation, or sale of plastic bags (standard industrial classification [sic]). Prohibited.

3. A retailer shall not offer plastic bags to a customer at the point of sale.
4. Retailer [sic] who provides customer [sic] with the plastic bag at a point of sale is guilty of an offence [sic].
5. A person [sic] who manufactures [sic] plastic bags to sell [sic] is guilty of an offence.
6. A person [sic] who imports [sic] plastic bag, whether as a carryout bag or sale, is guilty of an offence.”

“Plastic bag Penalties act (PBP 2019)

1. Clause 2
2. Any person found guilty of the offence under clause one shall be liable to a conviction to a fine not exceeding Five Hundred Thousand Naira, ~~N500,000-~~ pound equivalent =£237.98 (XE currency converter 2024) or
3. Imprisonment term 3 years or more, or both.

2.4.3 State-Level Legislation

Some states have taken proactive steps to address plastic pollution through legislation in Nigeria. For instance, Lagos State's Environmental Management Protection Law (2017) explicitly prohibits single-use plastics and imposes fines on offenders, demonstrating a commitment to reducing plastic waste. Similarly, Kaduna State's Plastic Bag Ban (2020) prohibits single-use plastic bags, further highlighting state-level initiatives to mitigate plastic pollution.

These state-level efforts complement federal initiatives, showcasing a multi-tiered approach to addressing plastic pollution in Nigeria (Duru et al., 2019). Such legislative actions can help reduce plastic waste, promote sustainable practices, and raise awareness about the importance of environmental protection.

2.4.4 International Agreements

Nigeria is a signatory to international agreements addressing plastic pollution. The Basel Convention (1989) regulates the transboundary movement of hazardous waste, including plastics, to prevent illegal dumping. Nigeria has also signed the United Nations Environment Programme (UNEP) Global Plastic Treaty, to reduce plastic pollution globally and encourage countries to develop national action plans (Ogunkan, 2022).

2.4.5 Policy Initiatives, Challenges and Opportunities

Nigeria has developed policies addressing plastic waste. The National Policy on Plastic Waste Management (2020) outlines strategies for reducing plastic waste, promoting recycling and waste-to-energy initiatives. The Extended Producer Responsibility (EPR) Policy (2022) requires manufacturers to take responsibility for waste generated by their products. Despite progress, challenges persist. The Inadequate enforcement of existing laws limited public awareness and insufficient funding for waste management infrastructure hinders effective plastic pollution management. Opportunities exist to harmonise state-level initiatives with federal legislation, increase public awareness and encourage businesses to adopt sustainable practices.

2.5 Airborne Microplastic Sampling Methods

The sampling of airborne microplastics (MPs) exhibits significant variability across different environments, necessitating tailored approaches to quantify MP concentrations accurately. The sampling of airborne MP is usually conducted in indoor and outdoor settings (Dris et al., 2017a). Each environment requires distinct sampling protocols due to differences in particle size distribution, air circulation patterns, human activity levels, and environmental conditions such as temperature and humidity (Ainali et al., 2021).

2.5.1 Passive Sampling Methods for Airborne Microplastics

Passive sampling has emerged as a widely employed technique in airborne microplastics (AMPs) detection and quantification, particularly in studies seeking to assess atmospheric fallout without relying on active air pumping systems (Alimi et al., 2018b). This method capitalises on the natural deposition of microplastics from the atmosphere (Enyoh et al., 2019), offering a cost-effective and low-maintenance alternative to active sampling, especially in remote or resource-limited environments. Several studies have demonstrated the applicability of passive samplers in capturing airborne microplastics. For instance, Dris et al. (2016) utilised glass collection jars placed on rooftops in urban areas to estimate microplastic fallout, reporting deposition rates between 2 and 355 particles/m²/day (Dris et al., 2016).

Similarly, Allen et al. (2019) conducted a passive sampling campaign in a remote mountain region, using stainless steel funnels and pre-weighed filters to capture

settled particles over extended exposure periods, revealing that atmospheric transport of microplastics could occur over considerable distances (S. Allen et al., 2019).

The design of passive samplers varies across studies but generally involves the placement of collection surfaces, such as glass plates, metal funnels, or quartz filters, at a standard height (typically 1–2 metres) to approximate human exposure levels (Enyoh et al., 2019). These surfaces are exposed for defined durations, ranging from daily to weekly intervals, and subsequently analysed for microplastic content (Dong et al., 2023). The deposition is usually recovered through rinsing with filtered deionised water, followed by vacuum filtration, and particles are identified using different polymer characterisation techniques such as Fourier-transform infrared (FTIR) or Raman spectroscopy (Tirkey & Upadhyay, 2021b).

Despite its simplicity, passive sampling has its limitations (Table 2.3). The method is sensitive to meteorological factors such as wind speed, precipitation, and turbulence, which can influence the deposition rate and particle retention. Moreover, passive samplers do not allow direct calculation of airborne microplastic concentrations in volumetric units (e.g., $\mu\text{g}/\text{m}^3$ or MP/m^3) unless coupled with air exchange or deposition velocity. Nonetheless, passive sampling remains a tool that can provide baseline data on AMP contamination, particularly in long-term monitoring and comparative studies between urban, suburban, and rural or remote locations (Y. Huang et al., 2021; K. Liu, Wu, et al., 2019b).

Table 2. 3: Passive Sampling Methods for Airborne Microplastics

Sampler Type	Description	Advantages	Limitations	Reference
Deposition Plates	Collects MPs by gravitational settling	Low cost, easy deployment	Uncontrolled airflow	(Kernchen et al., 2024)
Self-adhesive Tape	Uses a sticky surface to capture MPs	High collection efficiency	Potential contamination	(Gwinnett et al., 2021b)
Glass Bottles	Collects MPs through deposition	Low maintenance	Limited spatial representation	(Knobloch et al., 2021)

By deploying passive samplers in these diverse settings, researchers can identify spatial patterns and gradients of microplastic pollution, informing our understanding of the sources, transport, and fate of microplastics in the environment. This information can, in turn, support the development of targeted strategies for mitigating microplastic pollution and promoting environmental sustainability.

Overall, passive methods are advancing the understanding of microplastic deposition dynamics and their spatial variability across different environments. Their integration with complementary active sampling and modelling approaches is increasingly recommended to overcome current methodological constraints and improve the resolution of airborne microplastic flux assessments.

2.5.2 Active Sampling Methods for Airborne Microplastics

Active sampling methods use mechanical devices drawn to the air to collect airborne microplastics (Sharaf Din et al., 2024). These methods provide controlled airflow and precise volume measurement, allowing for accurate quantification of microplastics. Active sampling devices typically employ filters with specific pore diameters to capture microplastics. The filters are analysed to determine the microplastic counts, size, and shape (Cai et al., 2017b). The active sampling approach offers several advantages, including higher efficiency, precision control and real-time data (Table 2.4). The advantages of active sampling methods make them particularly useful in various environments, such as urban areas, industrial settings, municipal solid waste disposal sites and indoor environments.

In urban areas, active sampling can help researchers understand the extent of microplastic pollution in densely populated cities. In industrial settings, it can monitor microplastic emissions from specific sources, like manufacturing facilities. By applying active sampling methods in different contexts, researchers can better understand microplastic pollution, its sources, and its potential impacts on human health and the environment. In industrial settings, active sampling can monitor microplastic emissions from specific sources like manufacturing facilities, enabling researchers to pinpoint the types and amounts of microplastics being released. By applying active sampling methods in different contexts, researchers can gain a comprehensive understanding of microplastic pollution, its sources, and potential impacts on human health and the environment.

Table 2. 4 : Active Sampling Methods for Airborne Microplastics

Sampler Type	Description	Advantages	Limitations	Reference
High-Volume Air Samplers	High-volume air sampling	High efficiency, precise control	High cost, power consumption	(P. Sharma et al., 2024a)
Portable Aerosol Monitors	Real-time monitoring	Real-time data, High sensitivity	Limited spatial representation, calibration	(Childers et al., 2023)
Cascade Impactors	Size-fractionated sampling	Precise size fractionation	Complex operation, maintenance	(Aldekheel et al., 2023; M. A. Bhat & Gaga, 2024)

Indoor Active Sampling Methods: The indoor active sampling methods include standalone dust collection pumps and vacuums, or modified household vacuum cleaners equipped with specialised filters (Enyoh et al., 2019). The active method uses pumping devices to suction air through sequential filters and allows the estimation of microplastic particles per volume of air (S. Yang et al., 2024). This method was used for testing in Paris, France, to measure indoor and outdoor deposition of microplastic particles (Dris, 2017).

Outdoor Air Sampling Protocol: Active sampling protocols have been successfully implemented in different outdoor environments, including urban centres, industrial zones, coastal regions, and near waste disposal sites. For example, Zhang et al. (2020) employed high-volume air samplers in a metropolitan setting, reporting AMP concentrations between 0.3 and 1.5 particles/m³, while Catarino et al. (2021) used a low-volume pump system in suburban areas, highlighting significant temporal variation linked to local human activities and meteorological conditions. The air sampler is delivered to the site and positioned at a specified height (Table 2.5), typically from 1.5 to 3 meters above ground level, and it's secured to prevent movement or tampering and configured according to the manufacturer's guidelines (Prata, Castro, et al., 2020).

Sampling duration and frequency are set, and appropriate filters or collection media are chosen (Y. Zhang et al., 2020a). Once configured, the sampler is started, and air sampling begins. The sampling start time, date, and environmental conditions (temperature, humidity, wind speed) are recorded. After the specified sampling

duration, the sampler is stopped, and the sampling end time and environmental conditions are recorded (Enyoh et al., 2019).

Following sampling completion, the sampler and collected samples are retrieved, handled carefully to prevent contamination, and stored in a secure, climate-controlled environment (Leslie et al., 2017). Samples are labelled and documented for further analysis (Brander et al., 2020a) .

Key considerations for outdoor air sampling include sampler height and orientation, sampling duration and frequency, environmental conditions (weather, temperature, humidity), and potential sources of interference or contamination. Sampler height can vary depending on the study's objectives, with a breathing zone (1.5-2 meters) representing human exposure and a rooftop or tower (~10-20 meters) assessing regional air quality (Cai et al., 2017b; Choi et al., 2022).

Table 2. 5 : Active Sampling Filter Type and Pore Sizes

Reference	Filter Type	Pore Diameter	Filter Size	Air Volume/Flow Rate
Liu et al., 2019c	Glass Microfiber	1.6 µm	90 mm	1-144 m ³
O'Brien et al., 2020	Glass Microfiber	1.6 µm	-	55 m ³ /h
Wang et al., 2020	Glass Microfiber	1.6 µm	90 mm	100 ± 0.1 L/min
Liu et al., 2019a	Glass Microfiber	1.6 µm	90 mm	100 ± 0.1 L/min
Vianello et al., 2019	Silver Membrane	0.8 µm	20 mm	0.82 L/min
González-Pleiter et al., 2021b	Stainless Steel	25 µm	-	-
Peñalver et al., 2021	Glass Fiber	-	150 mm	720 m ³ /day
Wright et al., 2019	Quartz Microfiber	2.2 µm	-	16.71 L/min
	Polytetrafluoroethylene	2.0 µm	-	
	Mixed Cellulose Ester Membrane	0.8 µm	-	
	Alumina-Based Membrane	0.2 µm	-	
	Silver Membrane	1.2 µm	-	
Amato-Lourenço et al., 2022	Glass Microfiber	<1 µm	110 mm	3 L/min

Sampling durations can also be tailored to capture peak concentrations (short-term, minutes-hours) or provide average exposure levels (long-term, hours-days) (Kim et

al., 2023). Researchers may collect reliable and accurate data by carefully planning and executing outdoor air sampling protocols.

2.6 The Role of Low-Cost Samplers in Microplastic Research

The absence of airborne microplastic data in developing countries is a stark reminder of the expensive nature of traditional active samplers, limiting research and policy development in these regions (X. Zhang et al., 2020). In contrast, developed countries have utilised active sampling to study airborne microplastics, but at a prohibitively excessive cost (Zhu et al., 2021). However, a low-cost active sampler offers an inexpensive, efficient alternative for researchers and communities, bridging the gap in airborne MP research.

Currently, low-cost sampling applications in microplastic research are limited due to higher uncertainty in data reliability (Gaston et al., 2020). This stems from a lack of validation against commercial samplers, a limited understanding of sampling efficiencies, and uncertainty in particle size and shape detection (Torres-Agullo et al., 2021). Microplastic sampler designs typically consist of filter holders connected to laboratory air pumps for airborne microplastic collection, low-cost fans or pumps, DIY filter holders, affordable filters, sampling inlets, and power sources (Truong-Phan & Rasmus, 2020).

This low-cost solution boasts several advantages, including affordability (less than £100-2000), portability, lightweight design, ease of assembly and maintenance, customisable design, and suitability for indoor/outdoor sampling (Chew et al., 2022). However, limitations include lower collection efficiency comparable to commercial samplers, potential contamination, and limited flow rate control (Dybwad et al., 2014).

2.6.1 Low-Cost Samplers for Microplastic Monitoring in Low-Income Countries

The advancement of low-cost active samplers can be a game-changer for monitoring microplastic pollution in low-income countries (Alimov et al., 2021). These devices

provide an affordable solution for researchers and communities to collect reliable data on airborne microplastics, which can inform policy decisions, support community-led initiatives, and raise environmental awareness (Kang et al., 2022). By making microplastic monitoring more accessible, low-income countries can enhance their research capacity and drive sustainable development, ultimately mitigating the impacts of microplastic pollution on human health and the environment. The benefits of low-cost active samplers in microplastic research are numerous. They can increase accessibility, allowing for deployment in more locations and providing a more comprehensive understanding of microplastic pollution. This technology may empower communities to take ownership of microplastic monitoring and advocacy, fostering a sense of responsibility and engagement. Furthermore, reliable data from these devices can inform policy decisions and support evidence-based interventions, leading to more effective microplastic pollution management.

However, potential challenges and limitations may arise. Ensuring the accuracy and reliability of data collected from low-cost devices can be a concern, and regular maintenance and calibration may be necessary to ensure optimal performance. Additionally, local researchers and community members may require training to effectively use and maintain the devices, which can be an issue in resource-constrained settings. Despite these challenges, low-cost active samplers have the potential to democratise microplastic pollution monitoring and promote sustainable development in low-income countries.

2.6.2 Low-Cost Samplers for Microplastic Monitoring in Nigeria

Introducing affordable samplers in Nigeria may alter the country's approach to microplastic research. These may provide a cost-effective solution for monitoring and analysing microplastic pollution. They can help bridge the knowledge gap and inform evidence-based policies. This, in turn, can lead to more effective strategies for mitigating the impacts of microplastic pollution on the environment, human health, and the economy.

The benefits of adopting low-cost samplers in Nigeria are multifaceted. They can enable researchers to collect and analyse microplastic samples more efficiently, generating valuable data that can inform policy decisions and drive sustainable

development. Additionally, affordable samplers can facilitate community engagement and participation in microplastic research, raising awareness about the issue and promoting collective action. This can be particularly important in Nigeria, where community-led initiatives can drive environmental change. By promoting the adoption of low-cost samplers, Nigeria can take a significant step towards addressing the pressing issue of microplastic pollution. This can contribute to broader environmental benefits, including improved waste management practices, reduced plastic pollution, and a healthier environment for future generations. Furthermore, the insights gained from microplastic research can inform policy decisions and support the development of effective strategies for mitigating the impacts of microplastic pollution.

Moreover, the adoption of low-cost samplers may also contribute to a more robust and sustainable research infrastructure in Nigeria. Researchers with access to affordable and reliable technology can help build research capacity, foster innovation, and drive scientific discovery. This, in turn, can have far-reaching benefits for the environment, human health, and the economy.

2.6.3 Future Directions for Low-Cost Samplers in Airborne Microplastic Research

Building on the LCS foundations, Nigeria can effectively pursue the research directions outlined in Table 2.6 for advancing and monitoring airborne microplastics. These directions can help address key knowledge gaps and technological limitations, ultimately enhancing the country's capacity to monitor and mitigate microplastic pollution more efficiently.

Validation and calibration activities are necessary to establish the reliability of measurements and to ensure comparability with conventional sampling methods.

Optimising the sampler's design is another critical step. This involves refining its physical structure and internal mechanisms to make it more efficient. For instance, design improvements could enhance the sampler's ability to capture a wider range of microplastic sizes, from large fragments to tiny fibres. Furthermore, optimization can adapt the sampler to perform better under diverse environmental conditions, such as different wind speeds, humidity levels, and temperatures, ensuring its effectiveness in various regions and seasons across the country.

Table 2. 6: Future Research Directions for Airborne Microplastic Low-Cost Samplers in Nigeria

Low-cost samplers' durability, maintenance needs, and reliability	Description
Validation and Calibration	Conduct studies comparing low-cost samplers with standard high-precision instruments to ensure measurement accuracy and establish robust calibration protocols.
Optimisation for Microplastic Capture	Explore design modifications to improve sampler efficiency for microplastic collection, considering particle size, airflow dynamics, and material compatibility.
Long-Term Field Deployment	Assess low-cost samplers' durability, maintenance needs, and reliability during prolonged field use across different Nigerian climatic and environmental conditions.
Community-Based Monitoring Programmes	Initiate pilot projects involving local communities in sample collection and analysis, evaluating training approaches and resulting data quality.
Cost-Benefit Analysis	Perform economic evaluations to determine the viability of widespread adoption and integration of low-cost samplers into national environmental monitoring systems.
Integration of Human Health Impact Studies	Utilise data from low-cost samplers in dispersion modelling to assess potential human health impacts of airborne microplastic exposure.

Long-term studies using LCS will address issues relating to the durability and operational sustainability of these devices. Furthermore, community-based monitoring programmes offer opportunities to expand data coverage and to enhance public engagement with environmental research. Cost-benefit analyses will be instrumental in assessing the scalability and economic feasibility of LCS for widespread implementation.

To maximise the benefits of low-cost samplers, it is essential to ensure that they are used in conjunction with other research approaches and policy interventions. This can include implementing policies to reduce plastic use, promoting sustainable waste management practices, and supporting environmental education and awareness programs. Furthermore, integrating human health impact studies and utilising data

from low-cost samplers to inform dispersion modelling is a significant advancement in understanding the broader implications of airborne microplastic pollution. In summary, these research initiatives will establish a solid foundation for a scientifically robust, accessible, and sustainable microplastic monitoring framework in Nigeria.

Overall, the introduction of affordable samplers has the potential to revolutionise microplastic research in Nigeria, driving meaningful change and supporting a more sustainable future for the country. By harnessing the power of technology and innovation, Nigeria can take a significant step towards addressing the pressing issue of microplastic pollution and create a better future for its citizens

2.7 Identification of Microplastics

Microplastics can be identified and characterised using quantitative and qualitative methods, each providing distinct but complementary insights into their presence and behaviour in environmental samples (Y. Shi et al., 2024). The methods are essential for a comprehensive understanding of microplastic pollution. While quantitative analysis provides data on the extent of contamination, qualitative assessment offers critical insight into the sources, environmental behaviour, and potential risks associated with different microplastic types.

2.7.1 Qualitative Sampling in Microplastic Research

Qualitative sampling focuses on detecting and characterising microplastics, determining whether they are present at a site and what types they are, without primarily aiming to measure total amounts (Z. Huang et al., 2023). This means collecting samples (e.g. grab samples of water, sediments, air filters, or biota) and identifying the microplastic particles by their shape, size, colour, and chemical composition (Riaz et al., 2024). Methods typically include visual or microscopic inspection (fragments, fibres, beads, films, etc.) and spectroscopic analysis to determine polymer type.

MICROSCOPIC INSPECTION: Visual/microscopic involves the physical identification. These include identifying the shapes of microplastics in the sample matrix. Studies have shown that visual identification can result in false positives and negatives, with

accuracy rates ranging from 20% to 70%, depending on particle type and analyst experience (Hidalgo-Ruz et al., 2012; Lenz et al., 2015a). The common shapes typically found in airborne microplastics include the following.

- **Fibres:** Thin, elongated structures with consistent thickness and uniform edges (O'Brien et al., 2020).
- **Fragments:** Irregular shapes with sharp or jagged edges, often resembling broken pieces (Mateos-Cárdenas et al., 2020).
- **Films:** Flat, flexible sheets that may exhibit folding or curling (Q. Chen et al., 2021a).
- **Pellets:** Spherical or near-spherical particles, often smooth and uniform.

In addition to morphology, surface characteristics such as texture, abrasion, cracks, and pits provide valuable information about the weathering and degradation of MPs in sample matrices (Z. Yuan & Xu, 2023). Furthermore, colour and transparency are important optical properties of the MPs that distinguish them from natural materials (X. Zhao et al., 2022).

SPECTROSCOPY IDENTIFICATION: The spectroscopy involves the chemical identification of microplastic polymer composition in the sample matrix. For example, Fourier-transform infrared (FTIR) and Raman micro-spectroscopy are widely used to identify the chemical “fingerprint” of particles above ~20 µm in size (De Frond et al., 2023). These techniques can differentiate plastics from natural debris and classify particles by polymer (e.g. polyethylene, polypropylene, PET) (M. Huang et al., 2024). The result is a qualitative profile of what microplastics are present, information on polymer types and morphologies that can be linked to likely sources (e.g. fibres from textiles vs. fragments from packaging) (Chia et al., 2024). This compositional information is crucial for source verification: knowing the polymer and shape of particles helps trace them to origins or pollution pathways (for instance, the prevalence of polyester fibres might indicate laundry effluent, whereas polystyrene beads suggest industrial feedstocks) (Riaz et al., 2024). FTIR and Raman spectroscopy offer high chemical identification accuracy for microplastics, typically above 90% for particles larger than 20 µm, but their reliability declines with particle size, contamination, or fluorescence interference (Cerutti et al., 2019). In complex atmospheric samples,

careful pre-treatment and robust spectral libraries are crucial to maintaining analytical precision. Details of these methods are described below.

Fourier Transformed Infra-red (FTIR): Many analytical methods are employed to estimate the polymer compositions of airborne microplastics (MPs). Fourier transform infrared (FTIR) microscopy is the most prevalent technique in airborne-MP research. The FTIR spectra of an unidentified microplastic sample were compared with FTIR spectra from polymer IR libraries until a match was found. Matching spectra provides comprehensive insights into the chemical bonds and polymer composition in the microplastics. Additionally, FTIR instruments are often combined with a microscope, facilitating the visualisation and measurement of the plastic particles. This method is called micro-FTIR (μ FT-IR) or focal plane array (FPA) and has been used in previous research studies (Cai et al., 2017a; Crawford & Quinn, 2017a; K et al., 2021b). FTIR remains the optimal technology for identifying airborne microplastic polymers (Tirkey & Upadhyay, 2021c). Micro-FTIR with attenuated total reflectance has been effective for small-sized plastic polymers. This technique is especially suitable for analysing the chemical composition of microplastic particles with diameters under 20 μ m (J. Li et al., 2018). Microplastic samples are analysed directly on the filters where they are collected, including various types such as glass and microfibre filters (Ding et al., 2021; Liao et al., 2021), Nitrocellulose membrane filters (Zhang et al., 2020c), Silver membrane filters (Wright et al., 2020), and quartz filters (Dris et al., 2017b).

Several studies have adopted a methodology for transferring particles onto a substrate amenable to FTIR analysis. Vianello et al. (2019) described a process in which microplastic particles were removed from silver membrane filters onto a zinc selenide (ZnSe) window for such analysis (Vianello et al., 2019b). Alternative substrates used in FTIR analysis include calcium fluoride slides (CaF_2) (Kernchen et al., 2022b). Other studies have employed gold-coated slides (Soltani et al., 2021, 2022), KBr slides (González-Pleiter et al., 2021), and PTEF filters (Liao et al., 2021).

Raman Spectroscopy: Raman spectroscopy involves a laser beam on an unknown sample, producing a spectrum comparable to the known polymer libraries, like in the FTIR method (Collard et al., 2015; Zhang et al., 2020c). Raman can identify microplastics up to nano-dimensions (Imhof et al., 2016). Although FTIR offers greater image precision, Raman spectroscopy coupled with microscopy can yield high-resolution spectra comparable to FTIR (Hengstmann & Fischer, 2019a). Raman

spectroscopy provides a contactless profile of polymers during sampling (Mukhanov et al., 2019). Raman is a non-destructive, rapid technique for identifying microplastic polymers (Baruah et al., 2022).

For high-quality and reliable Raman analysis, selecting a suitable substrate is crucial (A. Rahman et al., 2021; Rocha-Santos et al., 2022). Previous studies have identified various filter papers as appropriate substrates, including S&S filter papers (Abbasi, Turner, et al., 2022), cellulose filters, aluminium oxide (Kernchen et al., 2022a), membrane filters (Welsh et al., 2022), and glass microfibre filters (Wright et al., 2019). In addition, numerous studies have employed substrates such as quartz, Teflon filters, silver membrane filters, polytetrafluoroethylene (PTFE), polycarbonate membranes, mixed cellulose ester membranes, double-sided copper adhesives, alumina-based membranes, and CaF₂ for Raman spectroscopy analysis. These substrates have been documented in several research works (Abbasi, Rezaei, et al., 2022; A. Rahman et al., 2021; Trainic et al., 2020; Wright et al., 2019). Additionally, the use of Klarite substrate was demonstrated in the previous study (M. Xu et al., 2019).

Scanning Electron Microscope with Energy Dispersive (SEM-EDS): The Scanning Electron Microscope (SEM) is a powerful optical technique that produces high-resolution images of airborne microplastics (Baruah et al., 2022). However, this method allows clear differentiation between biomaterials, other impurities, and the distinct morphology of microplastics within a sample mix (Crawford & Quinn, 2017b). Visual and stereoscopic methods often require the removal of organic matter from the sample before the examination to avoid confusion between organic substances and polymer particles (Lenz et al., 2015b; Monteiro et al., 2022). Furthermore, SEM capabilities can be enhanced by coupled with energy-dispersive X-ray spectroscopy (SEM-EDS) (Z.-M. Wang et al., 2017). This combination facilitates the direct identification of polymers and elemental analysis of airborne MPs (Crawford & Quinn, 2017b; Fries et al., 2013). SEM-EDS offers the advantage of providing both physical and chemical characteristics of airborne MPs. Despite the demonstrated ability of SEM-EDS to analyse the physical and chemical properties of microplastics simultaneously, it is a time-consuming method and can only handle a small number of samples at a time (W. Wang & Wang, 2018).

Thermal Analysis: Thermal analysis is performed through a synergistic application of multiple techniques, including differential scanning calorimetry, thermal desorption, and pyrolysis (Sgier et al., 2016). This analytical technique is used to examine the physical and chemical properties of plastic polymers by evaluating the thermal stability of the polymers (Castañeda et al., 2014). To date, the use of thermal methods has not been employed in the analysis of airborne microplastics, but they have been successfully used in the identification of microplastics within various environmental samples (Nuelle et al., 2014). Thermal analysis is a destructive process, making reanalysis of microplastic samples after the initial attempt impossible (Shim et al., 2017). Although thermal techniques can substitute for spectroscopy in identifying some polymers, however, some polymer species cannot be identified. For example, thermogravimetry (TGA) combined with differential scanning calorimetry (DSC) can detect polyethylene (PE) and polypropylene (PP), but not polymers like polyethylene terephthalate (PET), polyvinyl chloride (PVC), polyamide (PA), polyester (PES), among others (Majewsky et al., 2016). Nevertheless, the limitations of this method are addressed by combining thermogravimetric analysis (TGA) with solid-phase extraction and thermal desorption gas chromatography-mass spectrometry (TDS-GC-MS) (Majewsky et al., 2016).

The Raman spectroscopy method can be combined with either thermogravimetric analysis (TGA) with solid-phase extraction or thermal desorption gas chromatography-mass spectrometry (TDS-GC-MS), this integrated approach allows the processing of large sample sizes at once and yields high-resolution images (Dümichen et al., 2015).

RECENT ADVANCES IN MICROPLASTIC IDENTIFICATION: Recent advances in analytical technologies have significantly improved the ability to detect, characterise, and quantify airborne microplastics (AMPs), moving beyond traditional methods such as FTIR, Raman spectroscopy, and SEM-EDS. While these conventional techniques remain foundational for identifying microplastic particles based on chemical composition and morphology, emerging methods offer enhanced sensitivity, higher throughput, and the ability to detect smaller particle sizes, including Nanoplastics, which are of particular concern in atmospheric studies.

Techniques such as Pyrolysis–Gas Chromatography–Mass Spectrometry (Py-GC/MS) and Thermal Extraction Desorption–Gas Chromatography–Mass Spectrometry (TED-GC/MS) are increasingly applied to airborne microplastic analysis (C. Li et al., 2021; Pimpke et al., 2020). These methods enable the thermal decomposition and chemical fingerprinting of microplastics collected on air filters, providing direct quantification even in complex atmospheric matrices. TED-GC/MS, in particular, has proven useful for microplastics screening in ambient air (Velimirovic et al., 2021). However, these thermal methods are destructive and cannot provide particle morphology information. For finer particle resolution, especially for detecting airborne Nanoplastics, techniques such as Atomic Force Microscopy-Infrared Spectroscopy (AFM-IR) and Nano-FTIR are emerging as powerful tools (Nene et al., 2025). These allow chemical identification and spatial mapping of microplastics down to the nanoscale, enabling researchers to study the fate of ultrafine plastic particles suspended in the atmosphere. Confocal Raman Imaging further complements these approaches by offering three-dimensional profiling of microplastics embedded in atmospheric dust or captured on filters, providing valuable insights into particle aggregation and deposition behaviour (Aramendia et al., 2024).

Automation and machine learning are also transforming airborne microplastic analysis. Automated imaging and AI-based particle recognition systems enable high-throughput classification and quantification of microplastic particles based on size, shape, and colour, reducing human bias and analysis time (B. Zhao et al., 2024). Similarly, Laser Direct Infrared (LDIR) spectroscopy offers rapid, automated chemical mapping of microplastic particles across large air filter surfaces, significantly accelerating environmental monitoring efforts (Cheng et al., 2022).

2.7.2 Quantitative Microplastic Sampling and Analysis

Quantitative sampling aims to measure the amount or concentration of microplastics in environmental samples using a standardised method or reproducible way. This requires collecting representative samples with known volumes or masses of media so that results can be expressed per unit volume or area. Standardised techniques have been developed for water, sediment/soil, and air to ensure comparability between studies:

Water Sampling: For surface waters (lakes, rivers, oceans), *net trawls* (e.g. manta or neuston nets) with known flow rates to capture suspended microplastics (Pasquier et al., 2022). Alternatively, *bulk sampling* pumps (e.g. submersible pump filtering a known volume through a sieve or filter) can quantify smaller particles. The U.S. ASTM D8332-20 standard provides a protocol for collecting water samples with high, medium, or low suspended solids for microplastics identification and quantification (Water Board, 2022). This method uses open-sieve sampling; a known drawback is potential air contamination, so it mandates a field blank to account for atmospheric and handling contamination (Water Board, 2022).

Sediment/Soil Sampling: Use core or grab samplers to collect a known mass or volume of sediment/soil (Adomat et al., 2022). The sample is then processed (e.g. density separation, digestion of organic matter) to extract microplastics, which are counted or weighed (Tan et al., 2022). Knowing the weight or volume of sediment processed, concentrations are reported (e.g. particles per kg dry weight). Standard lab protocols (based on environmental guidelines) are applied to minimise loss and contamination (Shruti & Kuttralam-Muniasamy, 2023a).

Air Sampling: There are two common approaches to collecting microplastics in the air, as mentioned earlier: passive deposition and active pumping. Passive samplers (e.g., open funnels or sticky surfaces) collect particles that settle out of the air by gravity (dry or wet deposition) (Brander et al, 2020). As the sampled air volume cannot be determined, results are expressed as particles per square meter per day (particles/m²/day), not per cubic meter. Active samplers draw known volumes of air through a filter or impactor over time (e.g. using high-volume air pumps with glass fibre or quartz filters). The filter pore size is chosen for the target size range. After sampling, particles are counted on the filter. The volume of air sampled must be recorded (e.g. using flow meters) so that concentrations can be expressed as particles per cubic meter. For example, Brander et al. (2020) note that analysts should record the volume of air filtered and report microplastic counts per unit volume (litre or m³).

In all cases, quality control and blank corrections are critical (Brander et al., 2020b). Because microplastics are ubiquitous (e.g. fibres in air), both field and laboratory blanks are processed alongside samples (Gao et al., 2023a). Field blanks (e.g. clean

containers or filters exposed during sampling) capture any contamination introduced in the field, while lab blanks (pure water or clean media run through all lab procedures) reveal contamination from lab air, reagents, or equipment (Shruti & Kutralam-Muniasamy, 2023b). Blanks are recommended for every batch of samples (e.g. one lab blank per 10–20 samples) (Munno et al., 2023a). Any particles found in blanks are reported and often subtracted from sample counts if they match in colour/shape. For instance, if a certain number of blue fibres appear on a lab blank, the same number of blue fibres can be subtracted from each sample in that batch (Dawson et al., 2023). This blank correction ensures that quantified concentrations exclude background contamination.

After collection and processing, microplastics are quantified. Common concentration units include particle counts or mass per volume (or area) (Klein & Fischer, 2019b). In air studies, concentrations are often reported as particles per cubic meter (particles/m³) or mass per cubic meter (µg/m³) of air in the active sampling method (Trainic et al., 2020). In the passive sampling method, the concentrations are reported as particles/m²/day. In water, one might report particles per cubic meter of water. In soil/sediment, one can report particles per kilogram of dry soil. As the National Institute for Public Health and the Environment notes (RIVM2021-0059). Microplastic abundance is “most commonly reported as the number (e.g. of microplastic) or mass (e.g. kg) of particles per volume (e.g. m³) of air”, typically obtained by active filtration of known air volumes. Microplastic samples are examined under a microscope or spectrometry to count and characterise particles (Michel et al., 2020). These counts (after blank subtraction) are then divided by the sampled volume to give, e.g. particles/m³. Sometimes both count and mass concentrations are reported; mass concentration (e.g. µg/m³) can be measured by weighing the residue or using thermogravimetric methods and is often considered a more stable indicator of environmental load (Y. Xu et al., 2023).

2.7.3 Qualitative/Quantitative Complementary Roles and Importance

Both qualitative and quantitative approaches are important and complementary. Qualitative analysis led to the identification of polymer types, shapes, colours and associated source signatures (M. Huang et al., 2024). These are essential for understanding origins and transport pathways. For example, the presence of

polyamide in microplastic samples is likely coming from synthetic textiles, providing clues about how it entered the environment (Periyasamy & Tehrani-Bagha, 2022). Quantitative data indicate the concentration or burden, which is essential for exposure assessment, comparison across sites or times, and validating fate/transport models (Adomat et al., 2022; Riaz et al., 2024). Standardised quantitative measurements (particles/m³) enable researchers to compare pollution levels between locations and detect trends (e.g., increasing microplastics following a pollution event).

Together, these approaches feed into broader environmental assessments. Identification (qualitative) and quantification (quantitative) of microplastics are “essential information” for environmental protection and policy development (Kaydi et al., 2024). For instance, regulators may set concentration targets (requiring quantitative monitoring) but also need to know major polymer sources to design effective controls. Quantitative data provide the quantity to calibrate environmental fate models and estimate human/animal exposures, while qualitative data refine models by specifying particle properties (density, degradation rate, toxic additives). The recent review emphasises that accurate quantitative analysis is “an important prerequisite to understand microplastics pollution in the environment” (Ashrafy et al., 2023a). At the same time, polymer characterisation is critical to link measurements to real-world pollution pathways (Galafton et al., 2025). For comparative studies across regions or years, consistent sampling (quantitative) and consistent analytical identification (qualitative) are needed to ensure valid comparisons. In sum, qualitative methods characterise *types* and origins of microplastics, whereas quantitative methods measure the *extent* of contamination; using both provides a complete picture of microplastic pollution

2.8 Transport and Fate of Airborne Microplastics:

Dispersion Model

Dispersion models have proven effective in predicting the transport and fate of airborne microplastics (MPs) from distant mountain areas (Allen, et al., 2019), marine atmospheres (Allen et al., 2022), and the planetary boundary layer (González-Pleiter et al., 2021). However, their success in estimating emissions from municipal solid waste facilities is not well demonstrated. Theoretically, dispersion models should

accurately predict airborne MPs from waste facilities, given their track record in simulating dispersion and impact zones for different types of pollutants.

2.8.1 Adaptation of Dispersion Models for Airborne Microplastics

Although no model has been specifically designed for simulating the transport and fate of microplastics (MPs), existing models can be adapted and utilised for airborne microplastics. Models such as COMPLEX-I, HYSPLIT, and CALPUFF were parameterised and used for non-conventional pollutants, such as odour and mercury, with modifications to accommodate their unique characteristics. Below is an example of model parameterisation conducted using different pollution scenarios.

1. The **COMPLEX-I dispersion** model is fundamentally based on Gaussian plume theory, which mathematically describes how a pollutant plume emitted from a source disperses in the atmosphere under specific meteorological conditions (Jawad, 2018a). It assumes that the pollutant concentration within the plume has a Gaussian (bell-shaped) distribution horizontally and vertically (Sarkar et al., 2003). Originally developed for conventional pollutants, researchers have demonstrated the adaptability of this model for other pollutants. For instance, Sarkar et al. (2003) successfully applied COMPLEX-I to simulate the dispersion of odour emissions, a non-conventional pollutant, from a landfill, showcasing its potential beyond its initial design. Notably, the landfill was considered an area source in this simulation. Although COMPLEX-I was originally designed for conventional simulation, the study demonstrated the model's ability to predict pollutant dispersion and generate exposure contour maps for a non-conventional pollutant such as odour. Sensitive receptors were used to link model results with odour annoyance, showing how dispersion outputs can be related to human impacts. Similarly, dispersion modelling of airborne microplastics from disposal sites could adopt this approach to map exposure zones and assess risks to nearby populations.
2. **CALPUFF** is an advanced air quality dispersion model that utilises a puff-based Lagrangian approach to simulate the transport and transformation of pollutants in the atmosphere (Rzeszutek, 2019). Unlike simpler Gaussian plume models that assume steady-state conditions and uniform meteorology, CALPUFF

pollutant releases are simulated to be a series of discrete puffs advected by a three-dimensional, time-varying wind field (Leelössy et al., 2014). The "model base" of CALPUFF lies in its ability to handle complex meteorological conditions and terrain. It is often coupled with its diagnostic meteorological model, CALMET, which can incorporate observed data and/or prognostic model output to generate a detailed representation of the wind, temperature, and stability fields over the modelling domain (Rzeszutek, 2019). This allows CALPUFF to account for phenomena like complex terrain flows, coastal effects, and changes in atmospheric conditions over time and space, making it suitable for a wider range of applications than traditional Gaussian plume models (Tagliaferri et al., 2025). Its ability to handle unconventional pollutants was demonstrated in the Meat Packaging and Rendering Plant. Elbir et al. (2007) employed the CALPUFF dispersion model to simulate odour concentrations from two meat-rendering plants. In this study, the plants were modelled as area sources (Elbir et al., 2007). Although CALPUFF was not specifically developed for odour predictions, the model accurately predicted odour dispersion, with strong agreement between model results and ground-measured concentrations. This validation process highlights how dispersion models, even those not initially intended for particulate simulation, can be adapted for airborne microplastic studies by following similar parameterisation and validation methodologies.

3. **The Hysplit model** is a hybrid approach combining Lagrangian and Eulerian methodologies. In the Lagrangian framework, the model calculates the movement of individual air parcels or "particles" as they are advected by the mean wind field, with dispersion simulated by a turbulent component. Alternatively, a puff model is employed as pollutants expand in three-dimensional volumes. The Eulerian aspect is applied when calculating concentrations on a fixed grid. HYSPLIT relies on gridded meteorological data. (e.g., from weather models or reanalysis) for its wind fields and other atmospheric variables, which are interpolated spatially and temporally to determine the movement and dispersion of the pollutants.

HYSPLIT is primarily designed for the transport and dispersion of conventional pollutants such as particulate matter (PM), nitrogen oxides (NO_x), sulfur dioxide (SO₂), and dust. Its capabilities are well-established for tracking these

substances released from various sources. However, researchers have successfully demonstrated its adaptability for other pollutants. For example, Karaca et al. (2021) effectively used HYSPLIT to model the dispersion of mercury emissions, a non-conventional pollutant, from a coal-fired power plant, showcasing the model's potential beyond its traditional applications. This highlights that with appropriate parameterisation and validation, HYSPLIT can be a valuable tool for studying the fate of a wider range of atmospheric substances. However, the ability of HYSPLIT to predict the transport and fate of airborne microplastics is widely acknowledged within the scientific community. For instance, the adaptability demonstrated by Allen et al. (2020) for airborne microplastics in a remote mountain area.

4. **AERMOD View**, a user-friendly interface for the widely recognised AERMOD atmospheric dispersion model, presents a valuable tool for simulating the potential transport of MPs from an MSW site into the broader ambient environment (Y. Hu et al., 2024). While traditionally applied to conventional pollutants, AERMOD's framework can be adapted to model MP dispersion by treating the MSW storage area as an area or volume source of emissions. Although MP emission rate estimation from such a diffuse source remains a significant challenge requiring further research, this parameter would serve as a crucial input. AERMOD's requirement for meteorological data, including wind speed, direction, temperature, and atmospheric stability, is facilitated by AERMOD View's data input and processing capabilities (dos Santos Cerqueira et al., 2019). Furthermore, the model's ability to account for terrain complexities and building downwash effects enhances its applicability to real-world MSW site scenarios. By defining the source characteristics and relevant environmental data, AERMOD View can simulate the downwind dispersion of MPs, predicting ground-level concentrations and deposition patterns at varying distances and directions. To more accurately represent MP behaviour, adaptations such as modelling different particle size fractions with corresponding settling velocities and utilising AERMOD's advanced depositional parameters based on MP physical and surface properties would be necessary as more data becomes available.
5. **FLEXPART**, a sophisticated Lagrangian particle dispersion model widely employed for tracking atmospheric transport (Evangelidou et al., 2022), presents

a valuable alternative for simulating the potential dispersal of MPs from an MSW site into the broader environment. In contrast to traditional Eulerian models, FLEXPART tracks the movement of multiple or individual air parcels or "particles," offering a detailed perspective on dispersion pathways (Bakels et al., 2024). Evangelou et al. (2024) demonstrated that global MP transport from bare soil resuspension can be simulated using a sophisticated framework by employing FLEXPART. These studies highlight FLEXPART's ability to model MP transport under varying particle shape scenarios (spheres and fibres), detailed meteorological data, and even incorporate uncertainty assessments through Monte Carlo techniques (Evangelou et al., 2024). However, to successfully run FLEXPART, detailed meteorological input from global or regional weather models or site-specific measurements is required. Furthermore, for AMP, the characteristic shape and size distribution of MPs emitted from disposal sites through local sampling and analysis is crucial for accurate model parameterisation. These may account for particle-specific properties like settling velocity. FLEXPART can be utilised for short-range dispersion of AMP through its Lagrangian method and detailed weather data; it does have limitations (Herzke et al., 2025). High-resolution simulations are computationally expensive and require fine-scale meteorological data. Turbulence modelling at very short ranges and building downwash treatment may be less specialised than near-field models. Therefore, using FLEXPART for local studies to simulate the dispersion of AMP at the disposal site presents a challenge.

The successful adaptation of dispersion models like HYSPLIT, FLEXPART, COMPLEX-I, and AERMOD for simulating both airborne microplastics (AMPs) and other non-conventional pollutants strongly indicates their potential for investigating the environmental transport and fate of AMPs. However, their effective application demands substantial data, detailed meteorological information, and expert handling, potentially limiting their practical use where resources and specialised knowledge are constrained. Therefore, simpler screening tools are also necessary to complement these complex modelling approaches, particularly for scenarios where the waste is poorly disposed of, data and expertise might be insufficient to ensure a more thorough understanding of AMP dispersion. Therefore, simpler screening tools are also

necessary to complement these complex modelling approaches (IAQM 2020), particularly in scenarios with poor waste disposal, where data and expertise may be insufficient for a thorough understanding of AMP dispersion.

2.8.2 Adoptability of Screening Models for Airborne Microplastics

Screening modelling encompasses several conservative simulation techniques for estimating extreme upper-bound concentrations (EPA, 1992). These "worst-case" estimates are based on simplified assumptions/representations of source-receptor geometries (Schulman & Scire, 1993). The primary purpose of screening modelling is to quickly and easily eliminate sources whose impacts are low enough to influence ambient air quality standards or ambient increment standards, from further analysis. Additionally, the screen model pinpoints the "worst-case" operational scenario or emission load for a specific source or group of sources, to confirm adherence to air quality regulations. Functionally, a screening model is an abridged atmospheric dispersion instrument designed to estimate the maximum ground-level concentration of pollutants, including emerging contaminants, such as bioaerosols, at varying downwind distances from an emission source (TAHA et al., 2007). There are many screening tools available to air quality professionals. Screening Models include AERSCREEN, SCREEN3, and the Industrial Source Complex Short Term model (ISCST3) (IAQM, 2020).

In contrast to more sophisticated, computationally intensive models such as COMPLEX-I or CALPUFF, screening models exhibit key differentiating characteristics (AQD-ODEQ, 2017). Notably, they obviate the requirement for extensive hourly meteorological datasets, relying instead on inherent assumptions of steady-state and adverse atmospheric conditions (Singh, 2024). This simplification renders screening models computationally efficient and relatively straightforward, particularly in regions with poor waste management, data scarcity or limited computational resources. This section will provide a critical review of selected screening models, with a specific focus on AERSCREEN and SCREEN3, and their demonstrated or potential application in simulating the atmospheric dispersion of non-conventional pollutants, thereby evaluating their adoptability for assessing the environmental fate of AMPs.

SCREEN3: is an older regulatory screening model based on the steady-state Gaussian plume formulation. EPA describes SCREEN3 as “*a single source Gaussian plume model*” that computes conservative (upper bound) concentrations for point, area, flare and volume emissions. SCREEN3 is the screening version of the ISC3 model, using traditional Pasquill–Gifford plume formulas with simple rural/urban stability assumptions. Earlier studies have demonstrated the applications of SCREEN3 for handling emerging pollutants; for instance, Khaniabadi et al (2018) demonstrated the atmospheric dispersion of fine particulate matter from a cement plant stack in Doroud, Iran, utilising the SCREEN3 model and a Gaussian plume approach. The findings revealed maximum particle concentrations within a 1600-1800 m radius surrounding the source, exhibiting a strong correlation with ground measurements ($r^2 = 0.98$). The simulated and measured PM₁₀ concentrations exceeded the World Health Organisation's daily and annual limit values. Consequently, the study recommends maintaining a distance of at least 7500 m between the plant and residential areas to mitigate the adverse impacts of hazardous emissions (Khaniabadi et al., 2018).

AERSCREEN: the screening version of the AERMOD regulatory model (including Monin–Obukhov similarity scaling and mixing-height parameterisations), is designed to estimate maximum ground-level concentrations under worst-case atmospheric conditions (A. S. Bhat & Kumar, 2011). It achieves this by integrating core modules such as MAKEMET, which generates a site-specific matrix of conservative meteorological conditions, AERMAP for terrain processing, and BPIPPRM to account for building downwash effects (VSTGD, 2014). AERSCREEN does not require detailed meteorological input, making it highly suitable for limited data (EPA, 2021). For instance, the efficiency of the model under limited data was showcase in assessing emissions from an Asphalt Batching Plant at Tornoe Quarry, Ardarhan, Co. Galway the analysis reveals that the projected emissions of key pollutants, including nitrogen dioxide (NO₂), sulfur dioxide (SO₂), carbon monoxide (CO), and particulate matter (PM₁₀ and PM_{2.5}), are anticipated to remain within the permissible limits set by applicable short-term and long-term ambient air quality standards at all off-site locations considered (Nolan & Amiaqm, 2020). This provides a conservative yet regulatory-acceptable estimate of pollutant dispersion that aids in permitting and impact evaluation.

Both AERSCREEN and SCREEN3 have the potential to offer practical advantages depending on the context and data availability. AERSCREEN, based on the more advanced AERMOD dispersion framework, provides refined handling of site-specific meteorological conditions through its MAKEMET module and accounts for terrain and building downwash effects via AERMAP and BPIPPRM (Schulman & Scire, 1993). These features make it particularly valuable in studies where environmental complexities may significantly influence AMP behaviour, especially around urban waste sites or heterogeneous terrains.

However, SCREEN3, based on the traditional Gaussian plume model (ISCST3), remains advantageous in specific scenarios, particularly in the early stages of environmental assessments (EPA, 1995). Its key strengths lie in its simplicity, rapid setup, and minimal input requirements, making it highly suitable for preliminary evaluations or situations where time, technical capacity, or detailed environmental data are limited. This characteristic is particularly valuable in developing countries or emerging pollutant studies, such as airborne microplastics (AMPs), where comprehensive emission inventories, real-time meteorological data, or access to advanced dispersion platforms (e.g., AERMOD, CALPUFF) may not be readily available.

SCREEN3 can generate conservative concentration estimates that serve as a baseline for regulatory planning, siting assessments, or prioritising further investigation. Its ability to simulate point, area, and volume sources with simple terrain adjustments enables its application in diverse environmental settings, including municipal solid waste disposal sites, where emissions are often diffuse and intermittent (EPA, 2024). Additionally, the graphical interface versions of SCREEN3 further enhance its accessibility for early-career researchers and regulatory bodies by reducing the technical barrier typically associated with air quality modelling. While SCREEN3 lacks the temporal resolution and chemical transformation modelling of more complex systems, it compensates by offering quick-turnaround insights for risk screening, scenario testing, and community health planning. Ultimately, SCREEN3's adaptability and low resource demands underscore its value as an entry point for emerging environmental issues such as microplastics, particularly in resource-

constrained regions or for pilot studies that require interpretable results to inform policy or research direction.

It is important to recognise the limitations of these simple screening models in real-world applications. These limitations can be especially significant when modelling microplastics because of their specific and unique properties. Recognising these shortcomings will be valuable for guiding future modelling efforts. Here are some of the limitations:

- **Simplified Assumptions:** AERSCREEN and SCREEN3 assume steady-state conditions, flat terrain, uniform wind flow, and a continuous, ground-level source (Lakes, 2024; Yellow Tree, 2025). These assumptions may not hold in complex terrain or where the emissions are fluctuating.
- **Uncertainty in Emission Rates:** Estimating the emission rate of microplastics from a waste site is highly uncertain.
- **Particle Size and Shape Effects:** The model does not explicitly account for the influence of particle size and shape on deposition and atmospheric behaviour.
- **Dry and Wet Deposition:** This basic model does not include removal mechanisms like dry or wet deposition, which can significantly affect long-range transport and ground-level concentrations.

Despite these limitations, Screening models have the potential to offer a valuable entry point for understanding the complex issue of airborne microplastic dispersion. Their relative simplicity and lower data requirements make them highly adoptable for initial assessments, spatial extent estimation, and guiding early monitoring efforts. While their limitations must be acknowledged, their ability to provide rapid insights and facilitate communication makes them a crucial tool in the early stages of AMP research, paving the way for more detailed investigations and ultimately contributing to effective mitigation strategies. Additionally, screen models may offer several advantages and benefits. These include:

- Assisting the regulatory authorities when designing airborne MP emission exposure limits for existing and proposed municipal solid waste dumping sites.

- Assist researchers in understanding the exposure levels of airborne MPs on spatial and temporal scales.

2.8.3 Integration of Dispersion Models with Health Impact Assessments

The potential health impacts of airborne microplastics (MPs) are of significant concern, particularly for populations living near waste disposal sites. These areas are often sources of high microplastic emissions due to open waste burning, poor waste management, and the degradation of plastic materials (Pathak et al., 2023; K. Zhang et al., 2021).

Using dispersion tools based on screening models like ARESCREEN and SCREEN3, it is possible to simulate the movement and concentration of microplastics downwind from a given emission site (e.g., the waste disposal site). This is achieved by inputting relevant meteorological data such as wind speed, temperature, humidity, and site-specific emission rates. The model's output generates **concentration maps** that indicate the **spatial distribution** of airborne microplastics at various distances from the emission source (Cristiane Thé & Michael Johnson, 2024). These predictions (model output) are subsequently integrated with established exposure pathways, such as inhalation, ingestion of deposited particles, or dermal contact, to assess potential health risks. Quantitative risk assessment frameworks: the hazard quotient (HQ) for non-carcinogenic effects or the lifetime cancer risk (LCR) for carcinogenic potential are used to link the dispersion model with health risk assessment.

Although direct research linking microplastic dispersion with health assessments remains limited, the approach of using dispersion modelling to inform health risk assessments has been effectively applied to other airborne pollutants, such as heavy metals and conventional air pollutants. This represents a significant research gap, especially in regions where communities live near poorly managed waste sites, such as Nigeria. Integrating microplastic dispersion models with health risk frameworks could provide crucial insights into exposure dynamics and public health implications, supporting evidence-based policy and mitigation strategies.

To address this gap, future research should establish robust methodologies that link dispersion modelling outputs with health risk assessment frameworks. This includes the development of standardised exposure metrics for microplastics, the development of polymer-specific toxicity reference values, and applying models such as hazard quotient (HQ) and lifetime cancer risk (LCR). Additionally, spatial mapping of health risk gradients downwind of disposal sites can help identify vulnerable populations and inform public health planning. Advancing this area of research is essential not only for understanding the health implications of airborne microplastics but also for guiding regulatory actions, such as establishing protective buffer zones or improving waste handling practices to reduce atmospheric emissions.

2.9 Conclusion

The presence of microplastics in atmospheric environments, including urban, rural, and remote areas, has gained significant attention. Research has confirmed that municipal solid waste dumpsites are potential sources of microplastic emissions into the air. Airborne microplastics exhibit diverse characteristics and quantities across regions, with fibres and fragments being the most common shapes identified.

However, significant research gaps exist, particularly in developing countries like Nigeria and the African continent. Waste management practices are often inadequate, and dumpsites are poorly regulated in these regions. Limited research capacity, funding, and expertise hinder comprehensive studies on airborne microplastics in these regions. As a result, there is insufficient data on microplastic emissions from dumpsites and a limited understanding of their impacts on local ecosystems and human health.

Resource constraints further exacerbate the research gap. Limited financial resources, high-cost sampling equipment, and analytical techniques restrict research capacity. Lack of skilled researchers in the field also contributes to the challenge. Furthermore, low-cost sampling methods, which have been proposed as an alternative to expensive equipment, have not been:

- Applied in dumpsite settings to assess microplastic emissions.

- Validated for airborne microplastic research. These raise concerns about data accuracy and comparability.

To address these constraints, low-cost homemade samplers can provide an affordable alternative if properly designed and validated against the commercial sampler. Open-source designs and community-driven initiatives can facilitate widespread adoption.

Collaborative research partnerships between developed and developing countries can enhance research capacity, share knowledge and expertise, and develop context-specific solutions. Establishing standardised protocols for sampling and collecting airborne microplastics, particularly at dumpsites, is crucial. Expanding global research efforts on airborne microplastics to map the spatial and temporal distribution of microplastics originating from dumpsites is also necessary.

Investigating interactions between airborne microplastics, chemicals, ecosystems, and human exposure is vital. Prioritising research in developing countries, especially Africa and Nigeria, will help assess dumpsite-related microplastic emissions, develop context-specific mitigation strategies, and build local research capacity. Developing and validating low-cost sampling methods for airborne microplastics will democratise research and facilitate global understanding. The next chapter, therefore, focuses on the design, construction, and field deployment of a low-cost air sampler for collecting airborne microplastics. This includes the selection of affordable materials, system design rationale, and the experimental setup used for sample collection. This foundational step is critical in bridging the empirical data gap and validating the modelling framework discussed in this chapter.

Chapter 3.0: Design and Development of a Low-Cost Air Sampler for Airborne Microplastic Collection

This chapter presents the design, assembly, and performance evaluation of a novel, low-cost sampling apparatus specifically tailored for airborne microplastics in resource-constrained settings. The sampler's design and operational parameters were optimised to ensure affordability, ease of use, and robustness, facilitating microplastic research participation in low-income countries. A comparative cost analysis between the custom-built air sampler and a commercially available sampler is also presented, demonstrating the substantial financial benefits of the proposed design. The datasets supporting the findings presented in this chapter are available in Appendix A, facilitating transparency, reproducibility, and future research endeavours.

3.1 Introduction

Airborne microplastics studies are a rapidly evolving field, yet it is constrained by a significant gap in standardised sampling protocols, particularly in developing countries (Torres-Agullo et al., 2021). This lack of methodological consistency can lead to inaccuracies and inconsistencies in research findings.

A major challenge in advancing airborne microplastic research is the absence of harmonised sampling techniques, which hinders the ability to compare and integrate data across studies (Bhat, 2023). The varied sampling methods employed in existing studies include active, passive, and precipitation (Dong et al., 2023; Edo et al., 2021; Markovic et al., 2015). Precipitation sampling for airborne microplastics involves collecting rain or snow samples and filtering them to capture microplastic particles (Chapman et al., 2024). These highlight the need for standardised protocols to ensure data reliability and comparability. However, inconsistency complicates the cross-comparison of results. Additionally, studies report varying sampling durations from a few hours to several days or weeks, potentially influencing the number of microplastics collected (D Allen, S Allen, J Sonke, 2019; Klein & Fischer, 2019b).

The spatial variability of sampling locations is a significant consideration in airborne microplastic research, with studies conducted in diverse environments, including urban, rural, coastal, and mountainous areas (Dehghani et al., 2017; Izquierdo & Avila, 2012). This heterogeneity in sampling locations substantially influences transport mechanisms and deposition patterns of airborne microplastics, thereby complicating inter-study comparisons.

The analysis of airborne microplastics is complicated by methodological diversity in identification, quantification, and characterisation techniques. Different methods, such as microscopy, spectroscopy, and chromatography (Song et al., 2015), can lead to inconsistencies in detection, enumeration, and classification, hindering a comprehensive understanding of the issue. Furthermore, monitoring airborne microplastics poses additional challenges, including the high cost of air samplers (£4,200 to £12,600) (Giordano et al., 2021) and the requirement for expertise in sampling and analytical methods (Kurniawan, 2024). This financial burden can be particularly significant for developing countries, limiting their ability to conduct research in this area.

In Nigeria, although there is established expertise in basic analytical chemistry techniques that can support microplastic quantification, the country lacks specialised capacity in air pollution assessment monitoring. This gap has significantly limited the advancement of research on airborne microplastics (AMPs). To address this limitation and enhance local research capabilities, the present study proposes the development of a locally adaptable, low-cost sampling solution.

Existing literature highlights several constraints in AMP research, particularly in resource-constrained regions. These include the high cost of commercial air samplers, the lack of standardised protocols, and the limited validation of existing low-cost designs (Gaston et al., 2020; Perera et al., 2022). In response, this study focuses on the design, development, and evaluation of a cost-effective airborne microplastic sampler tailored to the Nigerian context. The custom-built Low-Cost Sampler (LCS) is specifically engineered for deployment around municipal solid waste facilities, where AMPs are likely to be concentrated due to poor waste handling practices.

To determine its effectiveness, the performance of the LCS was rigorously tested and benchmarked against a commercial reference sampler, the SKC Deployable Sampler fitted with a Total Suspended Particulate (TSP) head. This comparison enables the

assessment of the LCS's reliability, accuracy, and suitability for monitoring in resource-constrained settings.

The primary objectives of this study are:

1. Design and evaluate a low-cost air sampler for airborne microplastic collection in Nigeria.
2. Compare the performance of the low-cost sampler with a commercial sampler across seasons.
3. Assess the cost-effectiveness and economic viability of the low-cost air sampler.

Nigeria is home to one of the largest dumping sites in the world, with a staggering number of these sites failing to meet the United Nations standards for waste disposal operations (Ferreira et al., 2023). The poor waste management practices in these areas can exacerbate the presence of microplastics, which can become airborne and pose significant environmental and health risks. Despite these concerns, no efforts have been made to assess the impact of these disposal sites due to a lack of access to the necessary technology and equipment. However, introducing cost-effective monitoring tools can pave the way for this crucial work. These innovative tools can enable researchers and policymakers to track and analyse microplastic pollution, identify areas of high risk, and develop targeted strategies for mitigation and remediation.

3.2. Materials and Methods

As part of the fieldwork, airborne microplastic (MP) sampling was conducted from May 10, 2022, through December 11, 2022. 80 samples collected across various seasons were used to assess the efficiency and suitability of a novel low-cost sampler relative to a reference sampler.

The SKC Deployable Sampler was selected as the reference sampler for this study due to its well-established use in air sampling applications, including monitoring non-conventional pollutants such as bioaerosols and particulate matter in occupational health settings (Patterson et al., 2010). Notably, it has been recently deployed to monitor microplastic particles from compost sites, demonstrating its suitability for emerging research areas (Zapata et al., 2024a). The SKC Deployable Sampler was configured with a suitable sample head and filter assembly. Specifically, a total

suspended particulate (TSP) sampler head was chosen to capture airborne microplastic particles without size fractionation, allowing for a wide range of particle sizes (EPA, 1999). This approach enables a comprehensive assessment of microplastic concentrations in the air, without biases introduced by size-selective sampling.

The sampling method entailed running two collocated samplers concurrently for eight hours at a flow rate of 5 litres per minute. Notwithstanding the prevailing recommendations in existing literature, including those proffered by the United States Environmental Protection Agency (EPA), which suggest a collocation distance of 1-20 meters between samplers, this study employed a more proximal spacing of 1 meter (US EPA 2024). This deliberate methodological choice was motivated by recognising that microplastics have a high degree of ubiquity in natural environments, necessitating a more precise monitoring approach to capture their spatial variability. This simultaneous sampling protocol allowed for direct comparison of results between the two samplers, enabling validation of the low-cost sampler's performance against the reference sampler. In each season, 20 samples were collected (10 per sampler). The sampling schedule was as follows: spring (May 17-26, 2022), summer (June 10-19, 2022), autumn (October 14 - 24, 2022), and winter (December 2 - 11, 2022). The samplers were strategically placed 1 meter apart and raised to the same height of 1.6 meters above the ground. This setup ensured a direct comparison of results between the two samplers, enabling a comprehensive assessment of the low-cost sampler's performance against the reference sampler.

3.2.1 Study Area

The Gyadi-Gyadi municipal solid waste disposal site is strategically located in the heart of Kano City, Northwestern Nigeria, making it a central receptacle for waste generated from both commercial and residential areas. Situated within the urban core, its proximity to vibrant economic and residential centres leads to a continuous influx of waste, encompassing commercial and household refuse (Figure 3.1).

The facility has an 11°58.274766'—11°58.43868' N latitude and 08°33.576774' — 08°34.773196' E longitude (Figure 3.1). It is permitted to process up to 36 tonnes of waste per day. However, these practices are unsustainable and may pose environmental and public health risks (UNEP, 2024).

3.2.2 Weather Conditions

Weather conditions at the sampling site were monitored using a Kestrel™ 2000 environmental meter at three-hour intervals throughout the air sampling process to ensure that any changes in weather conditions, such as temperature, humidity, or wind speed, were captured and recorded. The study excluded rainy days from the sampling campaign, except when rain occurred while sampling was underway. As rain falls, the sampling campaign ceases while precipitation data is measured. This prevents the sampling equipment from damage and the risk of electric shock (Abbasi et al., 2019).

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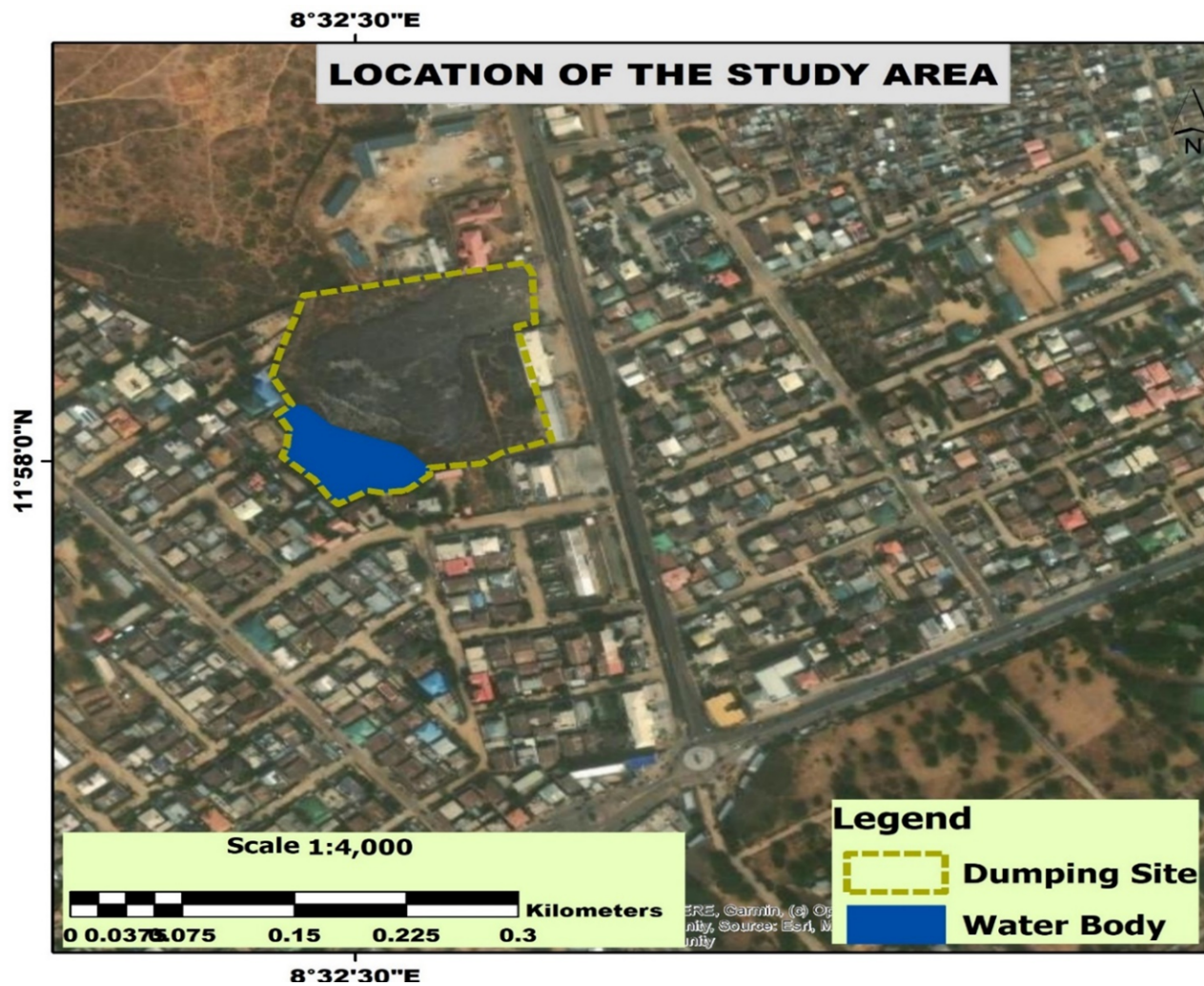
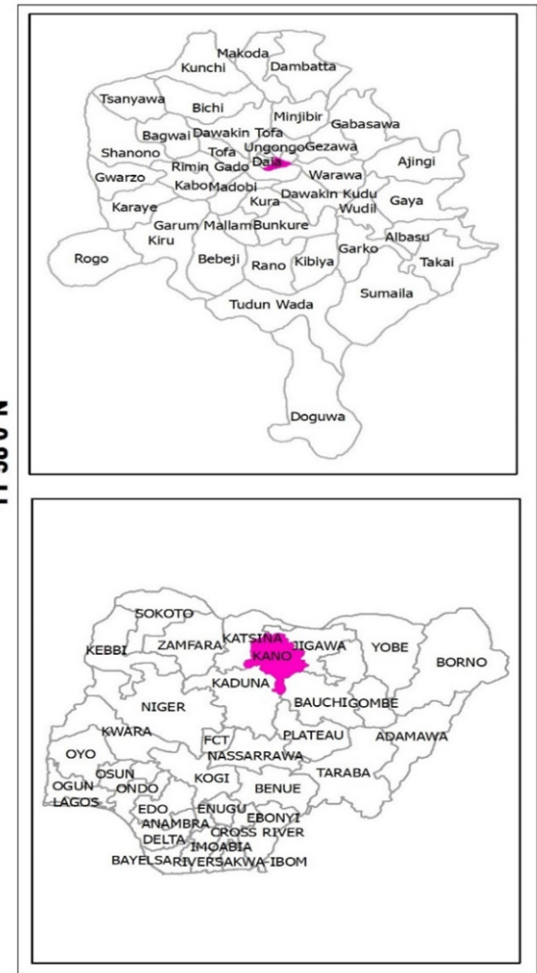


Figure 3. 1: Map of the Study Area.

3.3 Design and Choice of Air Sampler

The sampler's body and fittings are from repurposed materials such as an old lab pump, wooden components, and glassware sourced from laboratory and local markets. The selection of filter paper (polytetrafluoroethylene (PTFE), pore 2.0µm, 47 mm in diameter, ThermoFisher Scientific for this research is due to its proven effectiveness in collecting microplastics in previous studies (Abbasi et al., 2019) and its availability. This approach reduces costs and promotes sustainability.

On the other hand, the choice of the SKC air sampler as the reference sampler in this study is justified by its proven history of collecting airborne microplastics (MP) in similar environments, such as composting chambers, in previous studies (Zapata et al., 2024c). Furthermore, the SKC air sampler is readily available and accessible for this experiment, making it a convenient and reliable choice for comparison. Employing the SKC air sampler as a reference, this study can effectively assess the performance of the low-cost airborne microplastic sampler and ensure accurate results in monitoring airborne microplastics in various environments.

3.3.1 Reference Sampler

To ensure the accuracy and reliability of the results, the SKC sampler was carefully calibrated according to the manufacturer's guidelines (Jawad, 2018a). The sampler's flow rate was adjusted to 5 litres per minute, consistent with established protocols (Rahman et al., 2021). Calibration was performed to guarantee accurate airflow measurement, and the sampler was operated using a DC battery as specified by the manufacturer. This rigorous calibration process helps ensure the accuracy and reliability of the air sampling data collected.

The SKC sampler has been widely used in various air sampling studies, including those focused on occupational health, environmental monitoring, and aerosol research (e.g., Jawad, 2018a). Its application in diverse settings demonstrates its versatility and reliability as a sampling device.

3.3.2 Low-Cost Sampler Design

This study employed an innovative design and cost-effective air sampling equipment to collect airborne microplastics (MPs). The sampler was formed by joining inverted glass funnels to trap airborne pollutants. The funnels were meticulously sealed with

tape (lab-grade Fisher-brand) to ensure a secure and airtight seal, preventing leaks or contamination. The inverted sealed funnels are then connected to a Charles Austin pump (DYMAX-8) via a tube (6mm diameter by 2m length), allowing air to be drawn (Figure 3.2). The LCS was calibrated by installing a volume counter between the collection funnels and the DYMAX-8 pump to verify a steady airflow rate of 5 L/min. Calibration was performed each time before sampling commenced under laboratory conditions to ensure an accurate air volume was sampled for a specific duration. If deviations in flow rate were detected, either above or below the target value, adjustments were made via an external flow control valve of the pump. The power required for the Charles Austin pump to draw air from the atmosphere is 0.45 kW. However, this study supported the pump with a 0.5 kW generator to ensure an adequate power supply and mitigate potential electric surges.

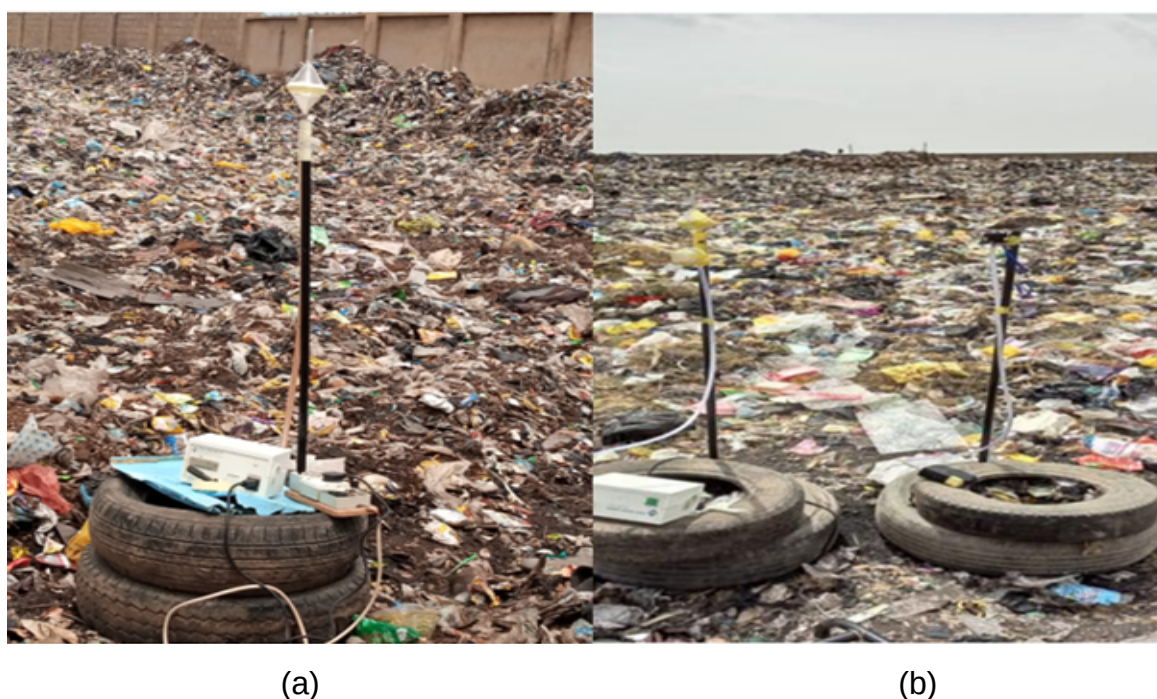


Figure 3. 2: Airborne-MPs Samplers (a) Low-cost Sampler (b) Low-cost and SKC sampler.

The sampler's design is straightforward, with the uppermost funnel serving as the air intake point, drawing in airborne particles at a flow rate of 5 L/min (L. Rahman et al., 2021). The lower funnel inlet acts as the air exhaust outlet, allowing air to escape while trapping the MPs on filter paper. The design features polytetrafluoroethylene (PTFE) filter paper with a pore size of 2.0 μm (47 mm in diameter; ThermoFisher Scientific). The filter was positioned in the lower funnel and served as the collection point for

airborne MPs. Following the laboratory design, the samplers and all the equipment needed for the field sampling campaign were transported to the municipal solid waste disposal site, including the control experiments (refer to section 3.5). At the chosen location (Figure 3- 2a), the low-cost sampler (LCS) was installed 1.6 meters above the ground and 1m apart from the neighbouring LCS (Figure 3-2b) (US EPA, 2024).

The samplers were programmed to operate continuously for 8 hours daily, commencing at 08:00 hours and concluding at 18:00 hours, contingent upon favourable weather conditions, with a consistent nightly shutdown protocol to conserve energy and maintain operational integrity. Access to the site was only possible during the daytime. The daily cycle allowed for simultaneous sampling and data comparison without hindrance (S. Allen et al., 2021).

3.3.3 Sample Storage and Transportation

The field samples were carefully transported from the sampling site to the laboratory in sealed glass containers. The distance between the disposal site and the laboratory is approximately 6 to 8km, depending on the route choices due to prevailing traffic conditions, and the transportation time varies between 20 and 30 minutes. Upon arrival, the samples were immediately stored in a walk-in dark refrigerator maintained at a controlled temperature of 4°C (S. Allen, Simonneau, et al., 2019). This controlled environment ensured the samples' integrity and stability were preserved during storage. Typically, analysis of the samples was conducted within 3 to 5 days after storage. The short transportation time and the controlled storage conditions have the potential to minimise degradation or contamination, ensuring accurate analysis and reliable results.

3.4 Sampling Processing

The laboratory procedure for microplastic analysis begins with processing samples from the refrigerator within the laminar flow bench to minimise contamination (Gao et al., 2023b). Microplastic samples from disposal sites may contain organic materials (K et al., 2021b). Therefore, it is necessary to digest the microplastic samples to reduce interference during microplastic identification (Nava & Leoni, 2021).

3.4.1 Isolation of Airborne Microplastics

Airborne microplastic samples are isolated from organic matter by rinsing/soaking the filter with Fenton's Reagent. (Tagg et al., 2017). To prepare the Fenton's reagent, 25 ml of hydrogen peroxide (H_2O_2) is mixed with 25 ml of iron catalyst solution ($\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$, 1g/L) (Radford et al., 2021b). Concentrated sulphuric acid (95% v/v) was added to the Fenton's reagent while stirring to lower the pH to 3 (B. Liu et al., 2022). This step was crucial to prevent the formation of iron hydroxide, which can inhibit the catalytic activity of the Fenton reaction. The filter papers (including the cassette and funnels) from the low-cost and reference samplers were soaked/rinsed in the RO water to dislodge the particles (B. Liu et al., 2022). Fenton's (30ml) was added to the beakers containing the microplastics and covered with aluminium foil to allow digestion for 24 hours (Abbasi et al., 2019).

After digestion, the beaker solution was filtered through a filter paper (0.45 μm , 47mm ThermoFisher Scientific) to trap the microplastic particles (Zhu et al., 2021). Subsequently, the particles collected on the PTFE filter were stained with three drops of Nile red and left for 30 minutes at room temperature (20 to 40°C, depending on the seasons) (Zhu et al., 2021). The staining solution was prepared by dissolving Nile Red 10 μg in an acetone concentration of 1 ML (Meyers et al., 2022)

3.4.2 Microplastic Particles under Microscopy

A fluorescence stereomicroscope (Olympus SZX9) was connected to a computer monitor to photograph the stained particles (Maes et al., 2017). The resulting images were analysed using ImageJ software, which enables visualisation of the MP particle dimensions and shape estimates (S. Chen et al., 2021). The common MP shapes found at the disposal site are:

Fragments: Microplastic fragments are small, irregularly shaped pieces of plastic that result from the breakdown of plastic items (Tanaka & Takada, 2016).

Fibre: Microplastic fibres are long and released from synthetic clothing, such as polyester, nylon, and acrylic. Fibres can also come from other sources, like plastic carpets or upholstery. They are typically longer than 3:1 in length-to-diameter ratio, as defined by the World Health Organisation (Y. Li et al., 2020).

Films: Microplastic films are thinner than fragments, flexible sheets of plastic often used in packaging materials, such as cling film, food packaging, and plastic bags.

Plastic with thinner material breaks down into smaller pieces and becomes microplastic films (Verla et al., 2019).

3.4.3 Confirmation of Microplastic Particles via ATR-FTIR Spectroscopy

The fluorescing Nile red and non-fluorescing particles on the filters were validated using ATR-FTIR spectroscopy (Agilent Cary 630 FTIR spectrometer). A representative sample of 10 approximate particles (fluorescing and non-fluorescing) per filter, encompassing all size and shape classes, was randomly selected for rigorous analysis. The detector was operated within the 650 4000 cm^{-1} wave number range, with a collection time of 3s and integration of 32 scans at a resolution of 8 cm^{-1} , ensuring optimal spectral resolution. The subsequent spectral analysis facilitated by Open-Spey to determine the chemical composition of the particles through comparison with the polymer spectra library, requiring a minimum of 70% similarity to confirm a specific polymer (Cowger et al., 2021).

3.5 Quality Assurance and Quality Control (QA/QC)

Contamination controls were crucial in this experiment. Initially, filter papers are preheated (105 °C) before sealing them in aluminium foil (Aminah & Ikejima, 2023). After each sampling collection, air sampler filter cassettes and glassware were thoroughly cleaned with alcohol and left to dry before use for the subsequent experiments (C. Yuan et al., 2022). Laboratory benches and other work areas were cleaned regularly. The colours of clothing and shoes were recorded during each sampling event as a measure of cross-checking for contamination from the researchers (Cunningham et al., 2022).

These three controls were employed: a field blank, a laboratory blank, and a procedural blank.

The field blank is a control sample collected in the field using the same setup as the actual experiment, but without drawing air. This control sample is exposed to the same environmental conditions as the actual samples but does not draw any air from the pump (Gaston et al., 2020). In contrast, the laboratory blank is a control sample consisting of beakers placed within the laminar flow hood to collect any fallout of microplastics from the hood's atmosphere (Al-Azzawi et al., 2020). The procedural

blank is a control sample using distilled water as a sample (Brander et al., 2020c). The water is assumed to be the sample and processed as the original samples. These include all steps, such as filtration, digestion, and analysis, to account for any potential contamination introduced during the procedural steps. Following three blanks, visual microscopy identified 13 particles in the blank samples. The results of these blank samples were removed from the original counts before the presentation.

3.6 Data Analysis

Microplastic data was analysed using Microsoft Excel 2016 software. A Spearman and Pearson test was employed to examine correlations between variables and investigate the seasonal variation of microplastics (MPs). The microplastic (MP) data were represented following the protocol outlined in the earlier research. (T. Hu et al., 2022a; K. Liu, Wang, et al., 2019).

3.6.1 Air Sampling Volume and Microplastic Calculation

The sampled air volume (VS) was obtained by multiplying the flow rate (5L/min) and the sampling duration (8 hours):

$$VS = 0.3 \text{ m}^3/\text{hour} \times 8 \text{ hours} = 2.4 \text{ m}^3$$

The microplastic numeric concentration (herein referred to as MP abundance). Using the following formula (D Allen, S Allen, J Sonke, 2019):

$$\text{MP abundance} = (\text{Number of MP}) / (\text{Sampling volume (VS)}) \text{ -----eqn. 3.1}$$

Where: VS = 2.4 m³. The resulting unit of measurement for MP abundance was MP per cubic meter (MP m⁻³).

3.6.2 Performance indicators

The low-cost Sampler (LCS) was assessed against the reference sampler. Statistical measures were employed for the performance evaluation of the low-cost sampler against the reference sampler. These include:

Bland Altman: The Bland Altman graph was employed to analyse the concordance of the measured data. Microsoft Excel 2019 software was utilised to generate a Bland-Altman plot, which is used to assess the agreement between the reference (Ref) and low-cost sampler (LCS) measurements. This method regards a zero difference between measurements as the ideal agreement. Consequently, in this context, the

mean difference and its limits can also be found close to zero. The Bland-Altman plot was created by calculating the difference between the reference and LCS measurements and plotting these differences against the average of the two measurements. The plot was then analysed to determine the mean difference, standard deviation, and 95% limits of agreement

Accuracy: Accuracy is the degree of closeness between the measured values obtained by a low-cost sampler and those obtained by a reference sampler, expressed as a percentage. It represents how close the measured values are to the actual values (KARA, 2020).

$$\% \text{ Accuracy} = (1 - | \text{LCS} - \text{Ref} | / \text{Ref}) \times 100 \text{-----eqn. 3.2}$$

Bias: Bias refers to the systematic error or deviation of measurements obtained by a low-cost sampler (LCS) from the values measured by a reference sampler (Ref) (Sousan et al., 2021).

$$\text{Bias (B)} = (1/N) \sum [(\text{LCS} - \text{Ref}) / \text{Ref}] \text{-----eqn. 3.3}$$

Where:

- N = number of paired measurements in this case, N = 40
- Ref = Values measured by Reference Sampler (MP/m³)
- LCS = Values measured by Low-Cost Sampler (MP/m³)

Precision: Precision evaluates how well the low-cost method reproduces identical measurements under the same conditions as the reference sampler (appendix A, Text 3.2). Precision measures include the Mean Bias Error (MBE), Root Mean Square Error (RMSE), and Coefficients of determination (R²) (Molina Rueda et al., 2023).

3.7 Results and Discussion

The meteorological conditions observed during the co-location of the Low-Cost Sampler (LCS) with the SKC Deployable Sampler provide essential insight into seasonal influences on sampler performance and accuracy. The wind speed and direction affect the resuspension and airborne transport of microplastics across the seasons. For instance (Table 3.1), higher wind speeds in winter (up to 7.2 m/s) likely enhanced the entrainment of particles, enabling both samplers to collect larger quantities (Table 3.2). Under these favourable wind conditions, the LCS showed high agreement with the reference sampler, indicating reliable performance when particle

mobility was elevated. Conversely, in the spring, average wind speeds were lowest (2.8 m/s) and atmospheric stability was likely greater; the LCS tended to slightly underestimate AMP concentrations compared to the SKC reference, possibly due to reduced particle lift and lower flow consistency under calm conditions (Section 3.7.1, Table 3.2).

Temperature extremes also had differential impacts (Table 3.1). For example, high daytime temperatures in spring (up to 42 °C) may have affected the plastic housing and flow stability of the LCS more than the industrial-grade SKC system, potentially introducing measurement bias. Similarly, higher humidity in autumn (60.8%) may have influenced filter performance, with the LCS more prone to minor moisture-related blockages due to its simpler design. These conditions may have slightly reduced its filtration efficiency relative to the SKC sampler, though the deviation remained within acceptable ranges.

Importantly, the absence of rainfall events (>2 mm) during spring provided an ideal baseline for comparing both devices under stable, dry conditions.

Table 3. 1: Seasonal Meteorological Conditions during Sampler Validation

Season	Average. Wind	Min Wind (m/s)	Max Wind	Predominant Wind	Avg. Temp	Min Temp	Max Temp	Avg. Humidity	Rainfall >2 mm	Rainfall >5 mm
Spring	2.8	2.0	3.4	SSE	28.0	23.1	42.0	9.0	0	0
Summer	3.7	2.7	6.3	SSW	26.4	19.3	38.1	50.6	–	–
Autumn	3.3	1.4	5.6	SSE	30.2	15.1	37.2	60.8	–	–
Winter	4.2	2.9	7.2	NEE	17.2	9.1	32.2	–	–	–

Under these conditions, the LCS demonstrated the highest accuracy, with a correlation coefficient of $\rho = 0.976$ and 94.12% agreement with the reference (Table 3.4), underscoring its potential for reliable AMP monitoring when environmental interference is minimal.

Overall, the performance of the LCS varied modestly with meteorological conditions, but it maintained strong accuracy relative to the commercial sampler across seasons

(Table 3.2). These findings emphasise the need for seasonally informed calibration and deployment strategies when using low-cost samplers in diverse climatic settings.

3.7.1 Airborne MP Abundance and Shapes

A comparative analysis of microplastic concentrations revealed similarities between the reference and the low-cost sampler (Figure 3.3).

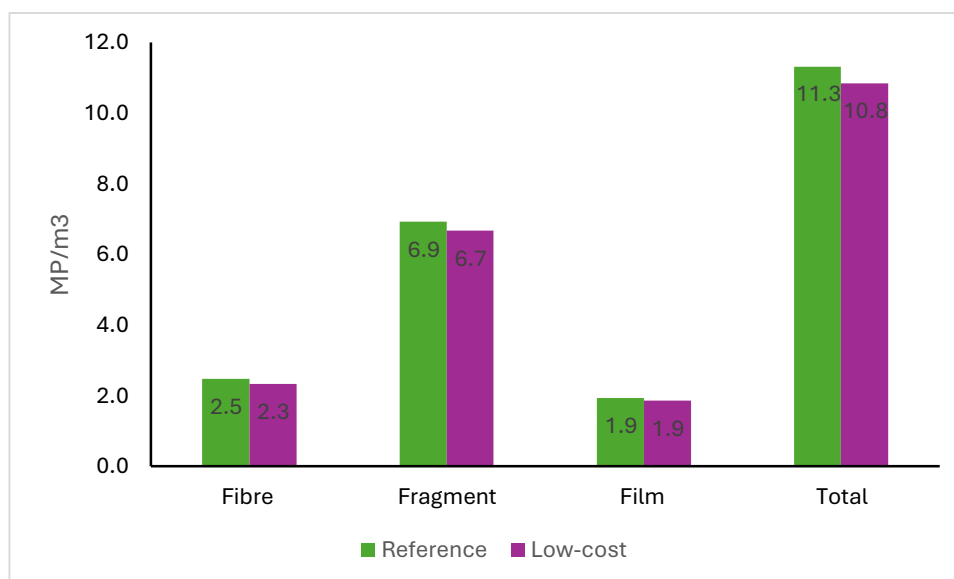


Figure 3. 3: Comparative assessment of airborne microplastics collected by Ref and LCS.

The average concentration of microplastics captured by the reference across the seasons was 11.3 ± 6.1 MP/m³, while the low-cost sampler yielded a comparable average concentration of 10.8 ± 6.4 MP/m³. This small difference of 0.5 MP/m³ between the two samplers indicates that the low-cost sampler could be a feasible alternative for monitoring microplastic pollution, potentially providing a cost-effective solution without sacrificing data precision.

Furthermore, a seasonal analysis of microplastic concentrations revealed distinct fluctuations, with elevated concentrations detected during the autumn (12.7 MP/m³ for the reference sampler and 11.9 MP/m³ for the low-cost sampler) and Winter (14.5 MP/m³ for the reference sampler and 14.4 MP/m³ for the low-cost sampler) seasons (Table 3.2).

Conversely, the spring and summer seasons showed relatively lower microplastic concentrations compared to other times of the year. Specifically, in the spring, the reference sampler collected 8.4 microplastic particles per cubic meter (MP/m³), while

the low-cost sampler collected 7.8 MP/m³. Similarly, during the summer, microplastic concentrations were slightly higher than in spring but still relatively lower, with 9.7 MP/m³ recorded by the reference sampler and 9.3 MP/m³ by the low-cost sampler.

Table 3. 2: Airborne MP concentration difference between Ref and LCS

Season	Reference Sampler (MP/m ³)	Low-Cost Sampler (MP/m ³)	Difference (MP/m ³)
Spring	8.4	7.8	0.6
Summer	9.7	9.3	0.4
Autumn	12.7	11.9	0.8
Winter	14.5	14.4	0.12

The observed seasonal trends were consistent across both the reference sampler and the low-cost sampler (LCS), underscoring the LCS's reliability and accuracy in capturing microplastic pollution patterns. Notably, both samplers observed a similar shape distribution of microplastics, with fragments being the most predominant form, followed by fibres, and films being the least prevalent (Figure 3.4).

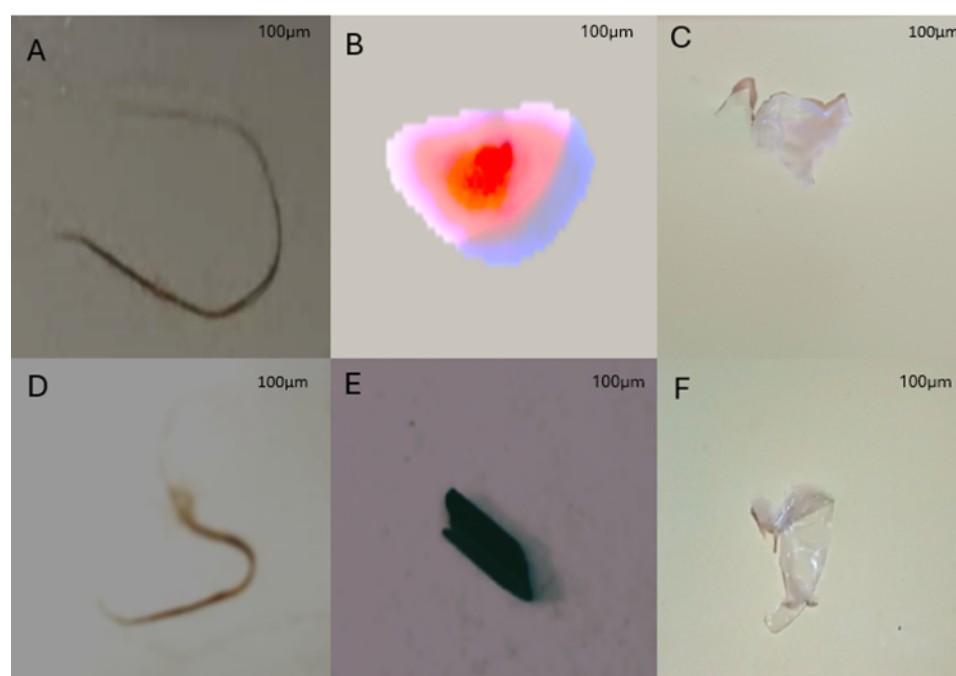


Figure 3. 4: Examples of micrographs of microplastics identified in air samples. From top left to right (a-c): collected from the reference sampler. From bottom left to airborne microplastic collected from the LCS sampler (d-f). Scale 1:100 µm.

This consistency in shape distribution, alongside the seasonal trends, suggests that the LCS can be a valuable tool for monitoring microplastic pollution, particularly in resource-constrained environments. The agreement between the two samplers lends credibility to the findings, indicating that the LCS can effectively track microplastic pollution patterns, including the distribution of shapes and seasonal fluctuations in microplastic concentrations. This capability is crucial for understanding and mitigating microplastic pollution.

3.7.2: Agreement between Low-Cost and Reference Samplers: Bland-Altman Analysis

This study employed a Bland-Altman analysis to assess the agreement between the low-cost sampler (LCS) and the reference sampler (Ref) in measuring microplastic concentrations. The Bland-Altman analysis measured the agreement between the low-cost sampler (LCS) and the reference sampler (Ref). The mean bias was calculated as 0.12, indicating a minimal systematic difference between the two sampling methods (Figure 3.5).

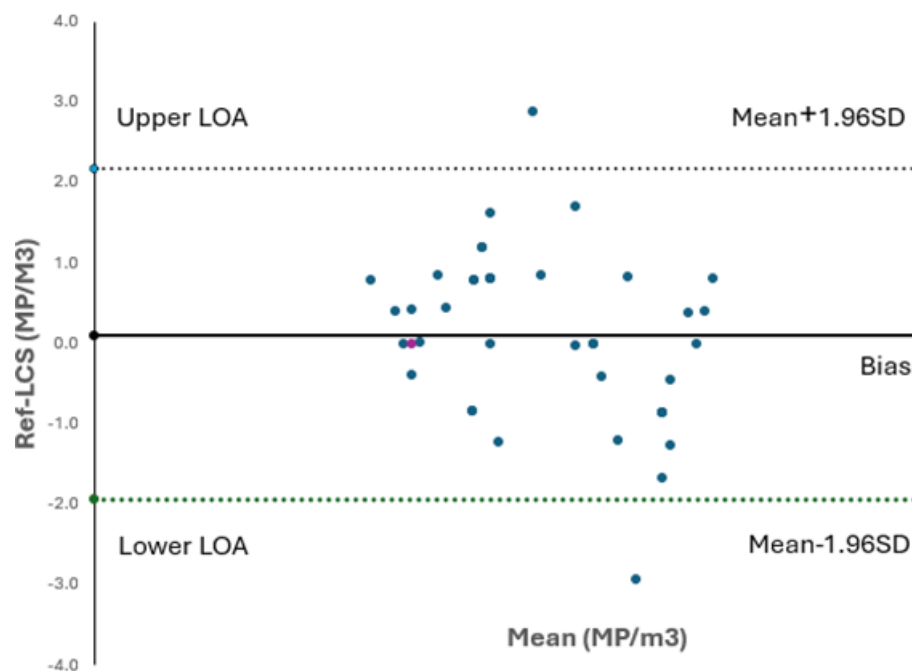


Figure 3. 5: Bland Altman Agreement of Low-cost vs Reference Sampler

The standard deviation of the differences was 1.05, and the limits of agreement (LOA) ranged from -1.93 (lower LOA) to 2.17 (upper LOA). These relatively narrow limits confirm the strong agreement between the samplers. The average bias is within the acceptable limit of EPA $\pm 10\%$ (Camalier et al., 2007).

The distribution of points around the zero line was random, with no apparent trend, suggesting consistent performance of the LCS across varying concentration levels. The narrow limits between the upper and lower limits of agreement further confirmed the reliability of the LCS in capturing airborne microplastic concentrations. While a few outliers were observed, these deviations were minimal and are likely attributable to environmental or operational variability, rather than fundamental deficiencies in the LCS design.

These findings highlighted that the low-cost sampler for MP sampling applications can provide an alternative sampling solution for resource-limited settings. The sampler can provide robust performance comparable to the reference sampler. However, further refinement or calibration under specific conditions may enhance its accuracy and ensure optimal reliability.

3.7.3 Performance Evaluation across Microplastic Size Classes

The performance of the low-cost sampler (LCS) was evaluated against the reference sampler (Ref) across various microplastic size classes (Malyan et al., 2023). The accuracy and bias of the LCS were calculated based on Equations 3.2 and 3.3, respectively. The accuracy indicates how well the LCS measurements relate to the Reference Sampler measurements, representing the degree of closeness between the two. Meanwhile, the bias is the systematic difference between the LCS and Reference Sampler measurements, highlighting the magnitude and direction of the deviation. The results revealed consistent performance with minimal bias and high accuracy (Table 3.3). Specifically, the LCS showed a bias of 11.83% for the 0–20 μm size class, achieving an accuracy of 88% relative to the reference sampler. In the 20–50 μm size class, the bias was reduced to 7.92%, with an improved accuracy of 92%. Similarly, the 50–75 μm range recorded a bias of 6.12%, and the accuracy further increased to 94%.

In the 75–100 μm size class, the bias decreased to 3.70% (96% accuracy), and indicating robust agreement. For larger particles in the 100–150 μm and 150–200 μm

classes, the LCS maintained low biases of 2.06% and 2.27%, achieving high accuracies of 98% in both classes.

In the 200–1010 μm size class, the LCS showed a bias of 2.55%, with an accuracy of 97%, further emphasising its reliability for capturing larger microplastic particles.

Table 3. 3: LCS Performance across Particle Sizes

Class	Ref	LCS	Bias	Accuracy
0-20 μm	93	82	11.83%	88%
20-50 μm	101	93	7.92%	92%
50-75 μm	98	92	6.12%	94%
75-100 μm	135	130	3.70%	96%
100-150 μm	243	238	2.06%	98%
150-200 μm	220	215	2.27%	98%
200-1010 μm	196	191	2.55%	97%
Total	1086	1041	4.14%	96%

The total bias across all size classes was 4.14%, with an impressive average accuracy of 96%. These results confirm the LCS as a reliable and cost-effective alternative to the reference sampler for measuring airborne microplastics across a wide range of particle sizes. The performance trend of the LCS demonstrated increasing accuracy at the higher-class intervals (from 100 to 1010 μm), suggesting its suitability for environments like municipal solid waste (MSW) disposal sites, where particle concentrations are expected to be higher. Consequently, the LCS can be a viable alternative to more expensive equipment, offering reasonable accuracy in such settings. However, it may be beneficial to categorise performance based on size-class ranges to refine its application and further validate its deployment in varied scenarios. Additionally, the coefficient of determination (R^2) was calculated to assess the strength of the relationship between the two samplers (Table 3.4).

The results showed that the LCS agree well with Ref, with R^2 values ranging from 0.94 to 0.97 across different seasons.

The root mean square error (RMSE) and mean absolute error (MAE) were also calculated to quantify the differences between the LCS and Ref measurements. The RMSE values ranged from 1.93 to 2.51, while the MAE values ranged from 1.53 to 2.04.

The strong agreement and low error values demonstrate the reliability and accuracy of the LCS in measuring microplastic concentrations. This suggests that the LCS can be a valuable tool for monitoring microplastic pollution, particularly in situations where traditional reference samplers may be unavailable or excessively costly.

Table 3. 4: Performance Indices of LCS

Season	R ²	RMSE	MAE
Spring	0.94	2.35	1.88
Summer	0.96	2.17	1.73
Autumn	0.97	1.93	1.53
Winter	0.95	2.51	2.04
Overall	0.96	2.24	1.79

3.7.4 Airborne MP Polymer-Specific Performance Assessment

This section focuses on a comparative analysis of low-cost and reference samplers for plastic particle identification. Due to the analytical limitation of ATR-FTIR spectroscopy, assessing particles below 20µm may not be possible. Nevertheless, ATR-FTIR spectroscopy remains a preferred option for budget-constrained studies, as it offers a fast and cost-effective method for polymer identification. As a preliminary test, 30 particles larger than 20µm from each sampler underwent ATR-FTIR analysis. Due to constraints related to time, laboratory access, and COVID-19 restrictions, the sample analysis was limited to 30 samples.

The comparative analysis of low-cost and reference samplers for plastic particle identification revealed a similar distribution of polymers (Table 3.5).

Table 3. 5: Polymer Composition in Low-Cost and Reference Sampler

Polymer	Ref. (%)	LCS (%)
Polyvinyl Chloride (PVC)	28.53	26.92
Polyamide (PA)	24.83	23.08
PE (Polyethylene)	17.42	19.23
Polyethene terephthalate (PET)	13.91	15.38
PP (Polypropylene)	10.11	11.54
PS (Polystyrene)	5.20	3.85

Specifically, the polymer composition identified by the Reference Sampler consisted of Polyvinyl Chloride (PVC) (28.53%), Polyamide (PA) (24.83%), Polyethylene (PE) (17.42%), Polyethylene terephthalate (PET) (13.91%), Polypropylene (PP) (10.11%), and Polystyrene (PS) (5.20%). In comparison, the Low-Cost Sampler (LCS) identified a similar polymer composition, with PVC (26.92%) and PA (23.08%) being the most abundant, followed by PE (19.23%), PET (15.38%), PP (11.54%), and PS (3.85%). However, this analysis reflects the ability to detect polymer particles from the filter surface, which may be linked to bias during particle selection. As such, these performance trends only indicate the types of polymers the LCS can detect, rather than a definitive measure of its accuracy in polymer identification. This highlights the importance of addressing potential biases in future studies. These include further validating the LCS's effectiveness in polymer-specific assessments.

3.8 Cost Analysis and Cost Comparison

Airborne microplastics are a growing concern, and the lack of research in developing countries highlights a significant knowledge gap. The challenges of accessing air samplers and technology for detecting airborne microplastics are barriers to advancing research in this area. Exploring low-cost alternatives for air sampling and microplastic detection, such as low-cost samplers or adapting existing technologies, could be beneficial. While designing such samplers is undoubtedly essential, it is important to consider the cost of the sampler to gain insight into its cost and adaptability

3.8.1 Development of a Low-Cost and Reference Sampler: Components and Costs

The design and development of a low-cost sampler required procuring various materials, focusing on optimising cost-effectiveness while maintaining functionality (Table 3.6). The sampler was supported with an old tyre (R15 x 205mm), sourced from a municipal solid waste site for ₦4,000 (£ 1.79). Ancillary materials, including a conductive pipe (£0.22) and a metal stand (£0.45), were chosen based on a comprehensive market survey to ensure the procurement of high-quality materials at reasonable prices.

The Charles Austin Pump (DYMAX-8), priced at ₦60,000 (£26.91) by the Kano Pollution Control Lab, was integrated into the sampler design to facilitate efficient sample collection. Additionally, the fuel expenses for operating the sampler, totalling

₦140,000 for 40 days (£ 62.78), were carefully budgeted to ensure sustainable operation.

The total cost of the materials amounted to ₦333,700.00, equivalent to approximately £149.64 (based on an exchange rate of £1 = ₦2,230). This cost represents a significant achievement in designing an affordable and effective sampler, demonstrating the viability of cost-effective solutions for environmental monitoring. The strategic sourcing of materials, combined with a comprehensive market survey, ensured that costs were optimised and justifiable within the project's objectives

Table 3. 6: Materials and Costs for Low-Cost Sampler Development

Material	Description	Quantity	Cost (₦)	Total Cost (₦)	Cost (£)
					Nov 2024 £1= ₦2230
Conductive Pipe	6mm x 1500mm	1	500.00	500.00	0.22
Metal Stand	10mm x 1600mm	1	1,000.00	1,000.00	0.45
Old Tyre	R15 x 205mm (sourced from Municipal Solid Waste site)	2	2,000.00	4,000.00	1.79
Charles Austin Pump	DYMAX-8 (priced by Kano Pollution Control Lab)	1	60,000.00	60,000.00	26.91
Power Source	SUMEC 0.5 kW	1	120,000.00	120,000.00	53.81
Fuel	Gasoline @ 5liter/day	40 days	3,500/day	140,000.00	62.78
Wire Cables	6mm cable (10 yards)	10	700.00	7,000.00	3.14
Glass Wears	Funnel (2 sets)	2	600.00	1,200.00	0.54
Total				333,700.00	149.64

On the other hand, the costs of air samplers pose a significant challenge for researchers in developing countries seeking to investigate airborne microplastics. An examination of online sources reveals a mean price of £382.33 for used air samplers, with a range of £135.97 to £495.37 (Table 3.7).

In contrast, new air samplers exhibit a higher mean cost of £8,346.53, with a range of £7,816.62 to £8,877.45 (Table 3.7). This cost disparity was particularly problematic for researchers in developing countries, where limited funding and resources hinder the

ability to conduct comprehensive airborne microplastic studies. Transportation costs, including shipping and handling fees, contribute to the overall cost.

Furthermore, customs duties, taxes, and other logistical fees imposed by local authorities add to the financial burden. These extra costs can increase the initial price of the air sampler by 50% to 200%, making it even more challenging for researchers to access this essential equipment. As a result, the cost of air samplers becomes prohibitively expensive, hindering the progress of airborne microplastic research in developing countries.

Table 3. 7: Cost of Imported Air Sampler

Brand	Price (GBP)	Origin	Advertiser	Condition	Shipping (GBP)	Tax/Custom Duty (GBP)	Total Cost (GBP)
GILIAN BDX-II	135.97	USA	E-bay	Used	37.3	51.98	225.25
SKC-223	382.28	USA	E-bay	Used	58.23	132.15	572.66
SKC-MBA47	495.37	USA	E-bay	Used	Free	148.61	643.98
SKC-224	310.02	USA	Express Lab Werks	Used	59	110.71	479.73
PUMP, ESCORT ELF	7816.62	USA	Environmental Express Zefon	New	80	2369	10266
RCS High Flow Touch	8877.45	USA	Weber Scientific	New	82	2687.8	11647

3.8.2 Cost Comparative Analysis of Low-Cost and Reference Sampler

The Low-Cost Sampler (LCS), priced at £149.64, presents a viable solution for researchers in poor-resource countries, enabling them to collect vital data on airborne microplastic pollution. In contrast, the Use Air Sampler costs about £382.33, and the new Air Sampler costs £8346.53. However, these samplers are less accessible due to their higher costs, hindering comprehensive research (Pinder et al., 2019).

The LCS offers a 61% cost savings compared to the Used Air Sampler (£382.33 - £149.64 = £323.69) and a substantial 98% cost savings compared to the New Air Sampler (£8,346.53 - £149.64 = £8,196.89). These cost savings are crucial in poor resource countries, where limited funding and resources restrict the ability to conduct environmental research (Acharya & Pathak, 2019). Notably, the LCS are free from importation barriers, ensuring effortless sourcing and transportation (Temitope & Rotimi, 2018; Vose & Cervellini, 1983). Furthermore, the LCS are easy to use and maintain (Ródenas García et al., 2022). This will guarantee functional equipment for researchers. By adopting the LCS, researchers can overcome the challenges of accessing affordable and functional air samplers (Nalakurthi et al., 2024). Facilitating the advancement of airborne microplastic research in poor resource countries.

3.8.3 Reproducibility of Low-Cost Sampler

Reproducing the Low-Cost Sampler (LCS) is highly feasible, owing to the widespread availability of its constituent components. Sourcing the materials for building the LCS proved relatively straightforward, with most components readily procurable from online electronics suppliers, local hardware stores, and municipal solid waste sites. This accessibility ensures that the LCS can be easily replicated and deployed in diverse environmental settings.

The air pump is a crucial component of the Low-Cost Sampler (LCS), but sourcing one that meets the specific air pollution standards for detecting airborne microplastics can present challenges. Although these pumps are not as abundant as other components, they can be found from local institutions, environmental monitoring suppliers, or online retailers specialising in atmospheric or air quality monitoring equipment. Small diaphragm pumps or suction pumps, commonly used for environmental sampling, are well-suited for this purpose. These pumps are typically available from suppliers who cater to scientific, industrial, or environmental monitoring sectors.

While finding the right pump may take some effort, they are accessible within the broader market, with several options suitable for low-cost, small-scale applications like the LCS. Furthermore, there are opportunities to calibrate local pumps to verify their effectiveness for airborne microplastic sampling, which could help reduce costs.

A critical consideration is to ensure that the LCS remains operational, especially in off-grid or remote environments. In this study, the pump was powered by a 0.5 kW

electrical source, which provided sufficient power for consistent operation during sampling.

For future deployments of LCS, pumps may be powered by rechargeable batteries or standard power adapters (AC or DC), depending on the specific application. Rechargeable lithium-ion batteries are commonly used in field deployments due to their portability and relatively long operational life. However, energy efficiency is key; selecting pumps with low power consumption will help extend battery life, allowing for prolonged sampling without frequent recharging. This is especially important for remote or off-grid locations where access to power may be limited.

In regions characterised by resource constraints and unreliable power supply, exemplified by countries such as Nigeria. Solar-powered pumps may provide an alternative for air sampling applications. Although necessitating supplementary setup and configuration, these systems can furnish a sustainable and uninterrupted power supply, facilitating prolonged sampling campaigns, particularly in environmental settings distinguished by elevated sunlight exposure.

Some components were specialised, requiring careful selection to ensure optimal performance. Building LCS was moderately complex; it involved assembling the funnels and filter components, wiring, and calibration. Clear instructions and some technical knowledge were necessary, but the overall process could be managed with basic skills.

To ensure the reproducibility of the Low-Cost Sampler (LCS), consider the following factors:

Component consistency: Sourcing the same components is crucial for maintaining consistency.

Assembly and sizing precision: Precise instructions for assembly and sizes of the funnels and filters must be followed to avoid variations in air intake methods that could affect results.

Power Source: This research employed a 0.5 kW power supply. For Future research, applications may consider using rechargeable lithium-ion batteries or solar-powered solutions based on the specific deployment needs.

Energy Efficiency: Researchers should prioritise low-energy consumption pumps when utilising rechargeable batteries or solar-powered systems to maximise operational time.

Software and tool accessibility: Access to the appropriate software and tools to process and analyse data results should be provided

Environmental influences: Potential environmental conditions, such as temperature and humidity, which may affect sampler performance, must be accounted for.

Attention to detail: The LCS design is not overly complex; ensuring accuracy and reliability requires attention to detail. Anyone using the setup must consider these aspects cautiously.

3.9 Limitations and Recommendations

Although the protocol employed in this study may not be perfect, LCS nonetheless yielded promising results, exhibiting a strong correlation with the reference sampler. The LCS demonstrated satisfactory performance before and after calibrations, indicating its potential as a viable alternative. However, the study has its limitations. This section identifies these limitations encountered and recommends solutions for future research.

3.9.1 Limitations

The Low-Cost Sampler (LCS) has its limitations. One of the primary concerns is the potential impact of environmental influences on the sampler's performance. Temperature and humidity, in particular, can introduce variability in the results, which may compromise the accuracy of the data obtained.

From a technical standpoint, the LCS also presents several challenges. Ensuring compatibility between components, such as filter papers and air pumps, is crucial; this can be tough to achieve, particularly with air pumps that meet air pollution standards. Furthermore, the manual calibration requirements for pumps and other components introduce an additional step that may be prone to error without proper technical skills. Additionally, the replicability of the LCS can be complex due to the assembly requirements, which demand attention to detail, moderate technical knowledge, and precision.

In terms of performance, the LCS has demonstrated slight biases in particle size classes and marginal deviations compared to reference samplers, particularly in high-concentration scenarios. These limitations underscore the need for further refinement and optimisation of the LCS to enhance its accuracy and reliability.

This study faced significant logistical challenges, particularly with the procurement of filter papers. Specifically, the study notes that filter papers with the required specifications, especially those with small pore sizes, are not readily available in the local market.

Another significant limitation of this study is the omission of a comprehensive cost analysis, except for the Low-Cost Sampler (LCS), due to the lockdown imposed during the COVID-19 pandemic. Future research endeavours must include detailed financial considerations to enhance the replicability and scalability of findings.

A thorough cost analysis should encompass various critical components, including the costs associated with equipment procurement or rental, reagents required for sample analysis, fees for analytical techniques such as Fourier Transform Infrared Spectroscopy (FTIR), and expenses related to laboratory space rental or leasing. By integrating these financial parameters, researchers can provide a more nuanced understanding of the economic viability of their methodologies, thereby facilitating informed decision-making and resource allocation in future studies.

This limitation may affect future studies and the replicability and scalability, potentially affecting the accuracy and reliability of results. Furthermore, the reliance on imported materials may introduce additional challenges, such as supply chain disruptions, customs delays, and increased costs, which could hinder the progress of future research endeavours.

3.9.2 Recommendations for Future Research and Development

To enhance the effectiveness of the Low-Cost Sampler (LCS) in monitoring airborne microplastics, it is essential to address its current limitations. Future studies should focus on improving the LCS's sensitivity, accuracy, and reliability in various environmental conditions. This will enable the LCS to provide more precise and dependable data, ultimately contributing to a better understanding of airborne microplastic pollution.

The integration of emerging technologies, such as machine learning and sensor technologies, also holds great promise for enhancing the performance and accuracy of the LCS. By leveraging these innovations, researchers can optimise the LCS's design and functionality, allowing for more efficient and effective monitoring of airborne microplastics.

Design improvements are also necessary to facilitate the deployment of LCS in large-scale research projects. Such improvements are crucial, particularly for deployment in regions where the waste is directly exposed to the environment and waste management conditions are less controlled.

Sampling in these challenging settings is not only justified but necessary. In regions characterised by inadequate waste management infrastructure, including open dumping, exposure of municipal solid waste (MSW) to a harsh environment, and routine open burning, the emission of airborne microplastics (AMPs) is likely to be elevated. Furthermore, ambient temperatures in these areas can reach between 38°C and 40°C, accelerating the degradation of exposed plastic waste and promoting the emission of fine microplastic particles into the atmosphere.

Enhancing the LCS to operate under high temperatures, variable humidity, and wet conditions is essential. Specifically, future iterations of the sampler should consider:

- Waterproofing and sealing of critical components to ensure functionality during rainy seasons or in humid climates.
- Thermal resilience, using heat-tolerant materials to maintain operational stability under high ambient temperatures.
- Structural durability, particularly in areas where the device may be exposed for extended periods without supervision.

Incorporating these improvements is anticipated to improve the reliability of AMP sampling across different climatic zones and enable more representative data collection in high-risk environments where AMPs are most likely to be emitted in substantial quantities. This would significantly advance the understanding of airborne microplastic dynamics and contribute to more inclusive, geographically diverse environmental assessments.

3.10 Conclusion

The Low-Cost Sampler (LCS) demonstrated in this chapter offers an innovative solution, more comprehensive monitoring of airborne microplastics, addressing the financial constraint and its usability across various environmental conditions. By providing a cost-effective and efficient tool for capturing microplastics at waste disposal sites, the LCS contributes to a more holistic approach to airborne microplastic research. The chapter also highlighted the potential of the LCS to support overlooked contributions of AMPs from disposal sites, which are vital for the estimation of fluxes

and emission factors (objective 2). The results show that LCS can enhance the understanding of microplastic flux and distribution with an accuracy of 96 to 98%, comparable to the reference sampler. Future advancements in LCS technology could lead to improved sampling techniques, offering even more precise data on microplastic pollution.

Moreover, the LCS provides significant cost savings, potentially reducing monitoring expenses by as much as 61 to 98%, making it an economically viable option for researchers and policymakers. Additionally, future research should explore the financial costs of analysing airborne microplastics (AMP), including the long-term operational costs of monitoring and the economic feasibility of widespread LCS deployment. Integrating these financial considerations will be crucial for scaling up monitoring efforts and ensuring that the LCS can be implemented effectively in different environmental settings. Research is needed to incorporate broader LCS data monitoring systems to fully assess the scope and impact of airborne microplastics, locally and globally.

Chapter 4.0: Estimation of Airborne Microplastics Flux from Soil to Air

This chapter presents a comprehensive analysis of AMP emissions from a Nigerian waste disposal site, shedding light on the factors contributing to their resuspension from soil to air. The study focuses on particle sizes ranging from less than 5 μm to 1,010 μm . This discussion highlights the seasonal dynamics of microplastic fluxes, their correlation with soil porosity, and the variability of emission factors (EFs) for different microplastics. Through a synthesis of findings, this chapter demonstrates the mechanisms governing microplastic transport at the soil–air interface, offering insights into the environmental and health implications of AMP pollution in Nigeria. Furthermore, it underscores the significance of site-specific data in refining emission models and informing targeted mitigation strategies.

4.1 Introduction

Microplastics (MP) are tiny synthetic polymer particles, measuring from 1 μm to 5 mm, that can be either from the primary source, created in small sizes, or a secondary source, formed due to plastic degradation (Thacharodi et al., 2024). MP particles come in various sizes, shapes, and colours, appearing as fibres, fragments, and films (Falakdin et al., 2024). The microplastic ubiquity raises ecological and health concerns (Bharath et al., 2023). Disposal sites, including landfills and wastewater treatment facilities, contribute significantly to microplastic pollution (Petrović et al., 2023). Inadequate waste management practices facilitate environmental entry points (Upadhyay & Bajpai, 2021).

Microplastics from discarded waste infiltrate soil layers through vertical and horizontal leaching, contaminating the immediate environment around disposal sites (Shen et al., 2022). As wind moves across the soil and waste surfaces, it mechanically abrades plastics in the waste, reducing their mass and diameter. This process generates a greater abundance of smaller microplastic particles within the soil (Bullard et al., 2023). The low density of microplastics within the waste piles and soil, often ranging from 0.9 to 2.2 g/cm^3 (S. Yang et al., 2024), makes them highly susceptible to atmospheric suspension. Once airborne, they can remain suspended for extended periods and be carried by wind currents over short and long distances (Emenike et al., 2023; Xie et

al., 2022). This atmospheric suspension allows microplastics to travel far beyond their original emission source, potentially spreading contamination to distant ecosystems (Q. Chen et al., 2023; X. Shi et al., 2023). The settled microplastics in the new areas may continue to disperse into soils, bodies of water, and the air, gradually enlarging their environmental impact across different regions (C. Liu, Li, et al., 2019a). The combined effects of wind erosion and atmospheric suspension may ensure that microplastics from waste disposal sites impact the local environments and contribute to widespread contamination in distant ecosystems (Kabir et al., 2023).

The transfer of microplastics (MPs) from waste disposal site soils to the atmosphere is a complex process, primarily driven by wind shear stress, facilitating MP detachment from contaminated surfaces. However, the complex mechanisms and influencing factors remain inadequately understood, especially regarding site-specific conditions and broader environmental variability (Ratnasari et al., 2024, Evangelou et al., 2024). Previous studies have employed backwards trajectory models to explain how microplastics travel through the air from disposal sites (X. Shi et al., 2023; Ward et al., 2024). These approaches may oversimplify the complex interactions between wind dynamics and the unique surface characteristics of these sites, leading to an inaccurate representation of microplastic dispersion. Furthermore, current models don't fully capture the intricate relationships among soil properties, wind patterns, and the resulting resuspension of microplastics.

Microplastic flux estimation from waste disposal sites posed significant challenges (Zhou et al., 2017), making it difficult to assess their environmental impact. Reliable flux estimation may require site-specific measurements, including local conditions and emission dynamics (S. Yang et al., 2024). Developing advanced models incorporating factors such as wind erosion, particle fragmentation and their shapes may significantly enhance prediction accuracy and address existing knowledge gaps.

On the other hand, emission factors (EFs) are crucial for understanding how microplastics move through the environment (Ren et al., 2020). EFs assist in measuring the strength and weakness of the emission and inform atmospheric modelling (Q. Wang et al., 2024). Accurate EFs lead to good modelling practices. The reliability of EFs depends on the method used in flux estimation (Newton et al., 2024). Inaccurate flux measurements can make EFs unreliable, leading to flawed modelling predictions and potentially misguided policies (Mahrt, 2010). Therefore, precise flux

measurements are crucial. This study aims to measure microplastic emissions, estimate the flux, and develop specific emission factors by integrating environmental processes to enhance flux accuracy and provide an emission factor for future use in the predictive dispersion model.

4.2 Method

4.2.1 Study Area

The Gyadi-Gyadi disposal site in the northern region of Nigeria, geographic footprint spanning $11^{\circ}58.274766'$ - $11^{\circ}58.43868'$ N latitude and longitude $08^{\circ}33.576774'$ - $08^{\circ}34.773196'$ E was used for this study (Figure 4.1). This facility is authorised to process a maximum of 36 tonnes of waste per day, making it a significant hub for waste management in the area.

The Gyadi-Gyadi disposal site's practices have been deemed unsustainable by the United Nations Environment Programme (UNEP, 2024). The site's operations pose potential environmental and public health risks, including uncontrolled odours from waste disposal, microplastic emissions, and water pollution (Dauda & Ali, 2024). The lack of proper waste management infrastructure, such as leachate collection systems and runoff management, exacerbates these concerns (Abubakar et al., 2022). The site's unsustainable practices cause severe implications for the environment and public health (A. Kumar, Thakur, et al., 2023). Uncontrolled waste disposal contaminates soil, water, and air, while microplastic emissions contribute to environmental pollution and potential human exposure (Mangla et al., 2025). Furthermore, leachate and runoff from the site can pollute nearby water sources, posing a significant risk to human health and the ecosystem. Understanding the microplastic flux from these sites under different environmental conditions assists in defining the strength of the release of microplastics.

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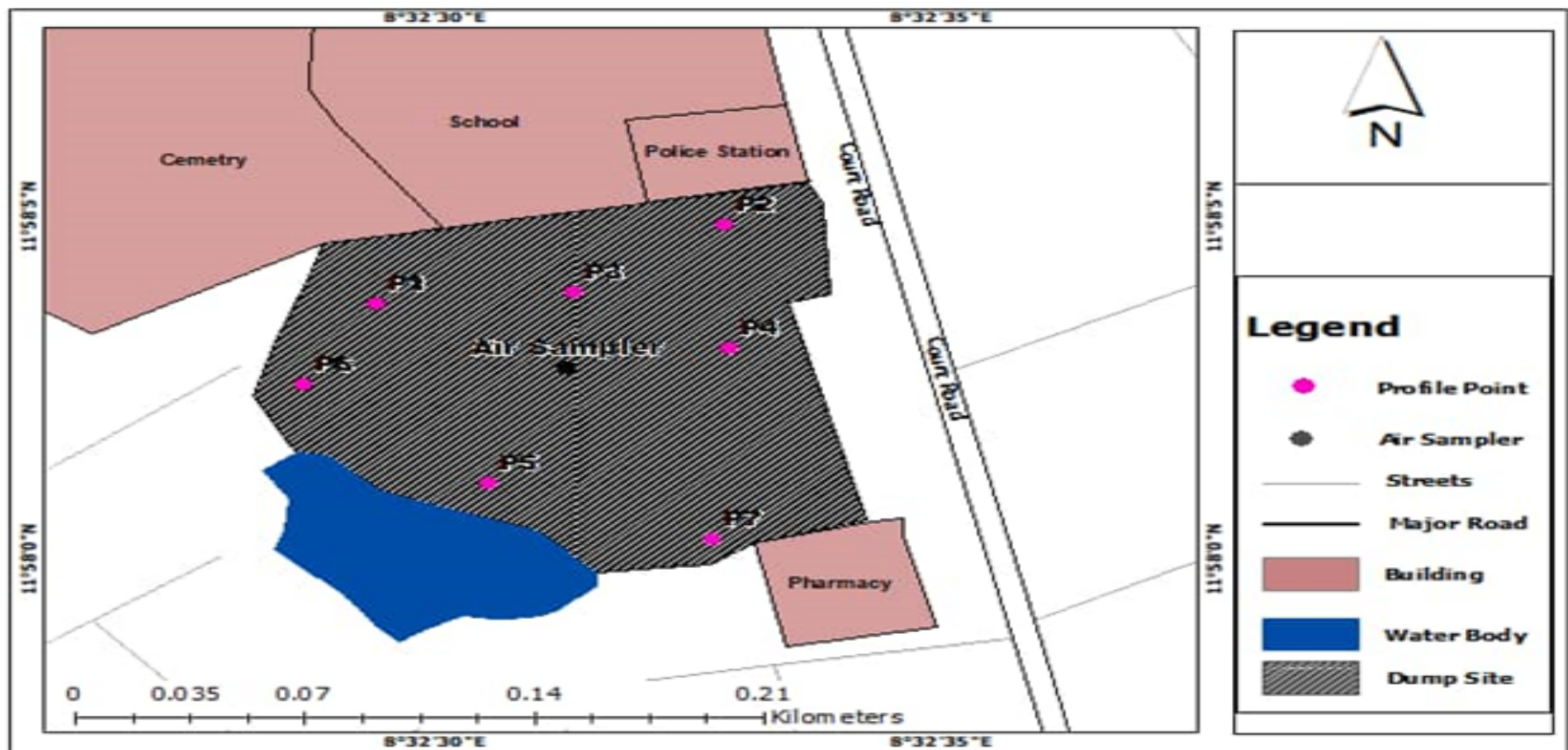


Figure 4. 1: The map illustrates the sampling locations, with purple points (1-7) indicating soil sample collection sites and a black point designating the air sampler location

4.2.2 Field Sampling

Before the commencement of field sampling, initial measurements were taken in the laboratory to record the weights of the filter papers and glass impactors used during the experiment. These initial weight measurements provided a critical reference point, enabling reliable estimation of particle mass concentration and other relevant parameters during data analysis. Air samples were collected 5000 meters upwind of the disposal site before initiating the main sampling program (Appendix B, Table 4.1a and 4.1b). This preliminary sampling enabled the establishment of baseline values for subsequent comparison with samples collected near the disposal site.

Due to uncertainties in airborne microplastic (AMP) transport, such as wind shifts and turbulent dispersion, a shorter distance could risk an overlap with site-related emissions (X. Zhao et al., 2023b). The 5000m buffer may ensure the establishment of a true background concentration. This study selects the upwind station within the same local government area, sharing features with the disposal site, such as nearby roads and small-scale industries, which allowed for environmental comparability while ensuring the background data reflected local, non-waste-related conditions. The approach aligns with site selection methods used in municipal solid waste landfill-related health risk assessments and baseline assessments, such as the case study of a solid waste landfill in southern Italy (Davoli et al., 2010).

A comprehensive 84-day air and soil sampling program was conducted at a municipal solid waste disposal site (MSWDS) to assess the microplastic emission from soils to the air. The study consisted of two 42-day phases: a dry season phase (February 14, 2021, to March 26, 2021) (Appendix B, Table 4.2a) and a wet season phase (July 16, 2021, to August 25, 2021) (Appendix B, Table 4.2b). The downwind samples are presented in Chapter 5 for model validations.

During each phase, air samples were collected from on-site and downwind locations on alternating days, resulting in 21 on-site and 21 downwind samples per phase. The on-site and downwind sampling locations were alternated every other day, providing a good spatial representation of pollutant distribution between the on-site and downwind sampling and temporal resolution of pollutant variability. These methodologies were adopted due to COVID-19 restrictions and access limitations on facilities. The adaptations made were:

- Restricted access: Sampling was conducted under limited access to facilities, requiring careful planning and coordination.
- Enhanced safety protocols: this research followed strict COVID-19 guidelines, including social distancing, mask-wearing, and sanitation procedures

Additionally, soil samples were collected for three days during the dry season (February 14, March 6, and March 26 (Appendix B, Table 4.3A)) and the wet season (June 16, July 5, and July 25 (Appendix B, Table 4.3B)). Soil samples (n=21) were collected at 7 points, coinciding with air samples 1, 14, and 21. This synchronisation ensured equal sample numbers for soil and air, enabling direct comparison and robust statistical analysis of pollutant levels. Soil sampling locations were carefully selected to represent different areas of the MSWDS and potential contamination pathways. Seven strategic locations were established using Garmin E-Trex 10 GPS devices, including the upwind edge, centre, downwind edge, northeast corner, southwest corner, and near the water body suited in the MSWDS (Figure 4.1). These locations were sampled during both phases. At each point, the waste was removed to expose the soil surface. About 20 grams of the soil samples were collected using an Auger at 0-5 cm depth and stored in a glass container and transported to the laboratory.

The air sampling was conducted according to the protocol established in the earlier studies (S. Allen et al., 2021; Klein & Fischer, 2019b; Xiao et al., 2023). Briefly, the sampler was set at a rate of 5 L/min at 1.6 meters above ground level, and a target analyte of airborne microplastics (size range: 2- 1000 μm , filter pore 2 μm (PTFE 47mm). The equipment used was a low-cost air sampler, as specified in Chapter 3. Sampling typically occurred over 8 hours (08:00-16:00) with nighttime shutdown (16:00-08:00) to conserve energy and minimise security risks. However, adjustments (9:00 to 17:00) were made during rainfall to prevent equipment damage and the risk of electrical shocks (Azari et al., 2023; Q. Chen et al., 2023). The soil and air samples were sealed in sterile glass containers and transported to the laboratory for subsequent processing.

4.2.3 Meteorological Data

Site meteorology was measured during each sampling campaign using a Kestrel™ 2000 environmental meter at three-hour intervals throughout the air sampling process (Table 4.1). As rain falls, the sampling campaign ceases while precipitation data is

measured. The method adopted in this work was due to resource constraints, limited availability of air sampling equipment, and access restrictions due to the pandemic. To accurately model microplastic flux and estimate the emission factor, it is essential to collect detailed meteorological data for the wet and dry seasons. The following variables were considered:

- **Wind Speed:** Wind speed influences the suspension and transport of microplastics in the air. Higher wind speeds during the dry season may increase microplastic release, while lower wind speeds in the wet season may reduce it (Rezaei et al., 2019).
- **Wind Direction:** The wind direction is crucial in the dispersion of airborne microplastics. The data on predominant wind directions are usually used for estimating the downwind distribution of microplastics from the source (Haque & Fan, 2023).

Table 4. 1: Average Weather Conditions for Dry and Wet Seasons.

Parameter	Dry	Wet	Average
Wind Speed Range (m/s)	4.2-6.8	2.8-4	4.05-5.3
Average Wind Speed (m/s)	6.0	3.5	4.9
Prevailing Wind Direction	E/NE (45%)	WSW (55%)	ENE/WSW
Degree (°)	45.0	225.0	135.0
Temperature Range (°C)	19.1-32.0	25-38	22.55-38.8
Average Temperature (°C)	21.0	34.0	31.3
Humidity Range (%)	10.0-19.0	75.0-82.0	42.5-50.5
Average Humidity (%)	14.5	80.5	47.5

- **Humidity:** Humidity levels differ significantly between the wet and dry seasons. The wet season has higher humidity, thus increasing soil moisture. This may reduce microplastic resuspension. Conversely, low humidity in the dry season can enhance the suspension of microplastics.
- **Temperature:** Temperature affects microplastics by speeding up their breakdown (Conti et al., 2024). Meteorological variables allow for precise modelling of seasonal impacts on microplastic emissions and transport. These inputs are usually incorporated into the simulation models to account for the varying environmental conditions between seasons.

4.3 Microplastic Analysis

Field samples were transported to the laboratory in sealed glass containers over a short distance of 6-8 km, with different routes selected depending on traffic conditions (approximately 20-30 minutes), which helped safeguard sample quality. Upon arrival, the samples were stored in a walk-in dark refrigerator at 4°C to maintain sample stability (Allen et al., 2019). This controlled storage environment preserved sample integrity, minimising potential degradation or contamination (Prata, da Costa, et al., 2020). The samples were typically analysed within 3-5 days after storage, ensuring reliable results and enabling accurate analysis and comparison of results.

4.3.1 Air Sample Preparation

Air samples collected on filter paper were carefully removed from the sampler glass funnels (impactor) and immersed in a beaker containing reverse osmosis (RO) water to facilitate the dislodgment of microplastics (Zapata et al., 2024a). To ensure comprehensive recovery, the funnel walls of the impactor were subsequently rinsed with RO water to capture any microplastics adhering to the surfaces. Fenton's reagent was prepared and added to the solution, and allowed to decompose the organic matter until there were no visible reactions (K. Hu et al., 2022).

After digestion, zinc chloride (ZnCl_2) salt solution was added to the treated air sample to achieve a density separation (J. Li et al., 2023). The mixture was allowed to settle for an hour, causing microplastics to float to the surface while other particles settled at the bottom (Crutchett & Bornt, 2024). The floating microplastics were then filtered through 0.45 μm pore-sized polytetrafluoroethylene (PTFE) filter papers (47mm, Thermo-Fisher Scientific) to concentrate and isolate microplastics (Blair et al., 2019).

4.3.2 Soil Sample Preparation

The conventional 2 mm sieving threshold in soil microplastic particle size distribution analysis is inconsistent with microplastic classification, which categorises MPs as smaller (<1 mm) particles in soil samples for resuspension analysis (Dorau et al., 2023; Müller et al., 2022; Rolf et al., 2022). This study employed a <1.2 mm sieving protocol to collect smaller microplastics with aerodynamic diameters conducive to airborne transport, exhibiting similar characteristics to those observed in air samples (section 4.3.1). Firstly, the soil samples were sieved through a 1.2 mm mesh, and the larger

materials were discarded (Chia et al., 2024). Out of the 20 grams of the soil sample, 10 g of the sieved sample was then transferred to glass jars, and the sample was mixed with 100 ml of 10% potassium hydroxide (KOH) solution and heated at 60°C while stirring for 1 hour (Radford et al., 2021c). The glass jar was covered with aluminium foil and left for 24 hours. This step was taken to break down the organic matter, preparing the sample for microplastic isolation (Mbachu et al., 2021).

Subsequently, zinc chloride solution was added to the sample to float the microplastics for an hour. This process exploited the density differential between microplastics and other soil components, causing microplastics to float to the surface while organic compounds and residual soil particles settled at the bottom (Rani et al., 2023).

The floating microplastics were then filtered through 0.45 µm pore-sized polytetrafluoroethylene (PTFE) filter papers (47mm, Thermo-Fisher Scientific) to concentrate and isolate microplastics (Blair et al., 2017).

Additionally, 14 soil samples were selected for bulk density analysis from dry and wet seasons. Following sieving (1.2 mm), 10 grams of sieved soil were used for bulk density analysis. The soil samples were filled into a volumetric cylinder (5-6 cm diameter × 5-6 cm height), and the initial volume (V) was recorded. The weight (W) of the filled cylinder was measured. The soil samples were dried in an oven at 105°C ± 5°C until constant weight was achieved (AL-SHAMMARY et al., 2018). The dry weight is then recorded. The densities were estimated as the ratio of dried soil to the volume of soil recorded from the cylinder. Soil porosity was calculated using the method described in the previous studies (F. Wang et al., 2022). The bulk density and porosity estimates are presented in Appendix B, specifically in **Tables 4.8a** and **4.8b**.

4.3.3 Microplastic Visualisation

Microplastics are visualised and counted by staining the filter papers with Nile Red for 30 minutes (Hengstmann & Fischer, 2019b; Shruti et al., 2022). The microplastics were visually examined using a camera-connected stereo microscope (Olympus, SZX9) (Ziajahromi, Kumar, et al., 2017). The resulting images were analysed using the ImageJ software's particle counter, following established protocols (Azari et al., 2023a). This approach enabled the accurate identification and quantification of microplastic particles.

4.3.4 Polymer Composition Analysis

Attenuated Total Reflectance Fourier Transform Infrared (ATR FTIR) spectroscopy was employed to identify the polymer composition of the selected microplastics (Käppler et al., 2018). The microplastics selected included 21 air samples (21 each for dry and wet seasons) and 21 soil samples. The samples are selected based on the sampling criteria established in the earlier studies (Samson, 2024). This study employed Standardised methodologies (ASTM E1252-98) to guide detailed spectroscopic analysis and polymer identification protocols, ensuring reliable microplastic characterisation (Sarathana & Winijkul, 2022). The detector was set to operate within a 650-4000 cm^{-1} wave range, ensuring optimal spectral resolution. Key settings included a 3-second collection time, 32 scan integrations and 8 cm^{-1} resolution (Circelli et al., 2024).

This methodology provided rapid, non-destructive, and precise chemical fingerprinting for polymer classification (Michel et al., 2020). Optimised ATR FTIR parameters yielded comprehensive spectral capture, enhanced signal-to-noise ratio, and high-resolution spectroscopy.

4.4 Quality Control

To ensure the integrity of the experiments, rigorous contamination controls were implemented. Filter papers used were sealed in aluminium foil to prevent airborne contamination (W. Zhong & Li, 2020). After each sampling collection, air sampler filter cassettes and glassware were thoroughly cleaned with alcohol and dried before reuse. Laboratory benches and work areas were also regularly cleaned. The researchers' clothing and shoe colours were documented during each sampling event to monitor potential cross-contamination (D.Li et al., 2022). Three types of control samples were utilised: field blanks, laboratory blanks, and procedural blanks (Munno et al., 2023b). Field blanks involved collecting control samples from the field using the same experimental setup but without drawing air through the pump, exposing them to the same environmental conditions as actual samples (Gao et al., 2023b).

Laboratory blanks consisted of beakers placed in a laminar flow hood to capture any microplastics falling from the hood's atmosphere. Procedural blanks involved using distilled water as a sample and undergoing all procedural steps, including filtration,

digestion, and analysis, to account for potential contamination introduced during these steps (Brander et al., 2020c).

Microscopic analysis of blank samples revealed the presence of 14 particles. These blanks were subtracted from the original counts before the result presentations (Bakir et al., 2023). Detailed results are presented in Appendix B (Tables 4.5a and 4.5b).

4.5 Statistical Analysis

Microplastic data analysis was performed using Microsoft Excel 2016 and Python (3.12.3). The data analysis investigated correlations between variables and seasonal patterns in microplastic concentrations. The sample analysis encompassed soil and airborne microplastic data, comparing concentrations during dry and wet seasons. The data presentation adhered to the conventional guidelines of microplastics established in the prior research in the field. This is to enable straightforward comparisons (P. Sharma et al., 2024a).

The shapes and sizes of microplastic particles were measured using ImageJ to account for their complex geometries (Mark Willett, 2019). The particles were then classified into specific size ranges to facilitate a comprehensive understanding of their size distribution and formation mechanisms.

The mass concentrations of microplastics were estimated using methods outlined by Simon et al. (2018) and Corradini et al. (2019). To calculate the volume of each plastic particle type, fibres, fragments, and film-specific shapes were assumed: cylindrical for fibres, and ellipsoidal for fragments (Koelmans et al., 2022b). Additionally, the z-axis was estimated by 67% of the minor axis for ellipsoids (Simon et al., 2018). The airborne microplastic concentrations are expressed as the number of microplastics counted per volume of sampled air (MP/m³).

The volume of sampled air = sample duration (8 hours) × flow rate of the air sampler (5L/min) or (0.3m³/hour) = 2.4m³. On the other hand, the concentration of microplastics in soil samples was estimated by the number of microplastics per gram of soil sample. This approach provided a direct measure of microplastic abundance in the soil, allowing for a standardised comparison of microplastic contamination across different samples and locations.

4.6 Model Development

This study developed a novel method for estimating microplastic flux (MPF), built on the fundamental principles of Fick's law of diffusion, to quantify the rate of microplastic emission from soil to air compartments at the disposal site.

Fick's law has been widely used in microplastic research across different media, such as water bodies (Cook et al., 2020), and leaching processes (Henkel et al., 2022), where diffusion across concentration boundaries drives the movement. This well-established use makes Fick's law an ideal framework for extending microplastic flux studies to the air-soil interface, providing a consistent approach to model microplastic transport across different environmental media (Chang et al., 2024).

For airborne microplastics, this choice was especially relevant for several reasons:

1. **Concentration Gradients:** Fick's law captures the movement of particles driven by concentration gradients (Esders et al., 2024). This is essential when assessing microplastic emissions due to differential concentrations in soil and air.
2. **Environmental Relevance:** Fick's law is commonly applied to environmental studies involving gaseous and particulate matter transport, including microplastics in water bodies and wind tunnels, making it adaptable for airborne microplastics when concentration and diffusion are key variables (Al-Zawaidah et al., 2024a).
3. **Model Simplicity:** Fick's law offers a straightforward approach for calculating flux, allowing for modifications without excessive complexity. This is beneficial for creating a model that is both accessible and effective for simulating airborne microplastic dispersion.

4.6.1 Microplastic Flux Estimation (MPF)

In recent research, Fick's law has undergone significant modifications to expand its application beyond traditional diffusion models. For instance, Fick's law was recently adapted to study the Mechanisms of Nucleation and Growth of Nanoparticles in Solution (Thanh et al., 2014). This study further advances Fick's law by developing a model that integrates site-specific factors, particle characteristics, and soil properties to simulate airborne microplastic flux in waste disposal environments more accurately. These adaptations help address the unique environmental and particle-specific factors

affecting microplastic resuspension from soil to air. To apply Fick's law to the unique context of airborne microplastics, several modifications were necessary to reflect the particular environmental processes and parameters at play:

- **Rising Velocity of Microplastic (wx):** Integrates Reynolds number, drag coefficient, shape factor, density, and shape into the model. These account for particle motion.
- **Environmental Adaptability:** Incorporates site-specific wind speed and temperature fluctuations to reflect real-world environmental conditions.
- **Soil Heterogeneity Module:** Accounts for soil porosity (ϵ) and concentration gradients between soil (C_{soil}) and air (C_{air}) to better represent flux
- **Thermal Sensitivity Analysis:** Uses the Boltzmann constant (k) to quantify temperature-dependent effects on microplastic transport.

These modifications allowed Fick's law to be adapted from a basic diffusion model into a more comprehensive framework that captures the complex behaviour of airborne microplastics under environmental conditions specific to waste disposal sites. This approach enhances the model's predictive power and aligns it with site-specific, seasonally varying factors critical to this research goal.

4.6.2 Modifications to Fick's Law

Fick's Law of diffusion describes the MPF as proportional to the product of the diffusion coefficient (D_{mod}) and concentration gradient ($\frac{\partial C}{\partial x}$). The study made specific modifications to reflect the behaviour of particles in soil during transfer to air. The equation for microplastic flux is represented and expressed as:

$$MPF = D_{mod} \times \frac{\partial C}{\partial x} \text{ --- eqn 4.1}$$

Where:

- MPF is the microplastic flux ($\text{g/m}^2/\text{s}$) or ($\text{MP/m}^2/\text{s}$)
- D_{mod} Is the diffusion coefficient (m^2/s) following the modification of the Stokes-Einstein Relation (for small particles in a fluid) to account for the critical velocity for the resuspension of microplastics
- $\frac{\partial C}{\partial x}$ is a concentration gradient

4.6.3 Diffusion Coefficient (D_o)

This study modifies the diffusion coefficient to account for factors influencing the transport dynamics of microplastics. The modification incorporates particle size, shape, and rising velocity. Building upon the existing law of diffusion, the Stokes-Einstein Equation was employed (Kooi & Koelmans, 2019): $D_i = \frac{kT}{(6\pi\eta r)}$, Where: D_i = initial diffusion coefficient (m^2/s), k = Boltzmann constant ($1.3806505 \times 10^{-23}$ J/K), T = temperature (K), r = radius of microplastic particle (m), η = viscosity of surrounding air (Pa · s). The rising velocity is given by: $(w_x) = \frac{2}{9} \times \frac{(\rho_p - \rho_a) \times g \times r^2}{\eta}$, the reciprocal of air. Arrangement of the equation and taking the reciprocal of viscosity (η), the equation was expressed as $\{ \frac{1}{\eta} = \frac{9}{2} \times \frac{w_x}{g \times r^2 (\rho_p - \rho_a)} \}$ (Eso et al., 2017). The expression was then substituted into the diffusion coefficient (eqn 4.1). The new modified expression:

$$D_{mod} = \frac{kT}{(6\pi r)} \times \frac{9 \times w_x}{2 \times (\rho_p - \rho_a) \times g \times r^2}$$

Simplifying yields:

$$D_{mod} = \frac{3}{4} \times \frac{k \times T \times w_x}{\pi \times (\rho_p - \rho_a) \times g \times r^3} \text{----- eqn 4.2}$$

Additionally, previous studies have expressed the settling and rising velocities in the context of microplastic flux as (Waldschläger et al., 2020a):

$$w_x = \sqrt{\frac{3}{4} \times \left(\frac{d_{eqi}}{C_d} \right) \times g \times \left| \frac{\rho_p - \rho_a}{\rho_p} \right|} \text{----- eqn 4.3}$$

Where: w_x = rising velocity (m/s), ρ_p = the microplastic particle density (kg/m^3), ρ_a = air density (kg/m^3), g = acceleration due to gravity (m/s^2), r = radius of microplastic particle (m), and C_d = drag coefficient.

In equation (4.3), the drag coefficient C_d is theoretically determined using Stokes' solution as a function of the Reynolds number $Re = \frac{\rho_a \times U \times d_{eqi}}{\eta}$ where U is the speed in m/s and equivalent aerodynamic diameter $d_{eqi}MP = \sqrt[3]{abc}$ Where: a is the longest length, b is the intermediate, and c is the shortest length, respectively (Akdogan & Guven, 2024).

$$C_d = \begin{cases} \frac{3}{CFS \times \sqrt[3]{Re}}, & \text{non-fibre} \\ \frac{4.7}{Re} + \sqrt{CFS}, & \text{fibre} \end{cases} \quad \text{----- eqn 4.4}$$

Where CSF = Corey Shape Factor and is defined as $CFS = \frac{c}{\sqrt{ab}}$

4.6.4 Concentration Gradient ($\frac{\partial c}{\partial x}$)

The microplastic concentration gradient is modified to account for the soil properties. The modification incorporates additional parameters to account for the complexities of microplastic transport. These modifications include:

- **p-soil (Bulk density of soil, g/m³):** Measures the mass per unit volume of soil, affecting microplastic resuspension (Z. Wang et al., 2024).
- **C-soil (Concentration of microplastics in soil, MP/g):** Represents the initial source strength of microplastic contamination in the soil.
- **e (Porosity, dimensionless):** Describes the ratio of pore space to solid material in the soil, impacting microplastic transport (Z. Wang et al., 2024).
- **C-air (Concentration of microplastics in air, MP/m³):** Influences the net flux of particles from soil to air.
- **H (Height of flux measurement, m):** Affects the microplastic concentration dispersed in the atmosphere, with greater heights leading to longer distances and potential dispersion or settling (Pukos et al., 2023).

These modifications enable the model to account for the physical availability of microplastics in the soil, the source strength of microplastic contamination, and the transport dynamics of air concentration and vertical dispersion. The modified Fick's law grants a more comprehensive understanding of microplastic transport and enables more accurate predictions of airborne microplastic concentrations.

The concentration gradient ($\frac{\partial c}{\partial x}$) Represented and expressed as:

$$\frac{\partial c}{\partial x} = \frac{\{\text{Source Term} - \text{Sink Term}\}}{\text{Height}}$$

$$\frac{\partial c}{\partial x} = \frac{\{(\rho_{\text{soil}} \times E \times C_{\text{soil}}) - C_{\text{air}}\}}{h} \text{ --- eqn 4.5}$$

- $\frac{\partial c}{\partial x}$ = Microplastic concentration gradient (MP/m⁴)
- C-soil = Concentration of microplastic in the soil (MP/g)
- ρ -soil = Bulk density of the soil (g/m³)
- E = porosity (unitless)
- C-air = Concentration of microplastic in the air (MP/m³)
- h = Height at which the flux measurement was observed (m)

4.6.5 Estimation of Microplastic Emission Factor and Mass Loading from Waste Disposal Sites

Measuring microplastic emissions from waste disposal sites is crucial for understanding the environmental implications of improper waste management. This research employed direct flux measurements from the disposal site to estimate the microplastic emission factor (EF) (Kota et al., 2014). This approach facilitates the calculation of total microplastic emissions from a waste disposal site. The accurate estimation of microplastic emission factors (EFs) is crucial for predicting the transport and fate of airborne microplastics. Building on the flux equation 4.5, the EF was calculated as

$$EF = \text{Flux (MPF)} \times \text{Area} \text{ --- eqn 4.6}$$

Where:

- EF = the microplastic emission factor (g/s or MP/s)
- A = the surface area of the waste disposal site (m²).

The framework enables the estimation of microplastic flux from disposal sites, while EF informs modelling software with vital parameters to predict airborne microplastic transport and environmental fate.

Airborne Microplastic Mass Loading (AML): mass loading refers to the weight of pollutants present per unit volume of a given environmental medium (Joo et al., 2015). In the context of air, it describes the burden of particulate contaminants, such as microplastics, suspended in the atmosphere, typically expressed in units of mass per cubic metre (g/m³ or µg/m³). This provides a useful analogue to established particulate matter (PM) metrics. Although microplastics are not currently classified within regulated PM categories, their physical characteristics (particularly size and aerodynamic behaviour) align with fine to coarse particulate fractions. Many studies have reported that airborne microplastic particles (AMPs) are found within the respirable size range (<10µm) (O'Brien et al., 2023), indicating that their atmospheric behaviour and inhalation potential may resemble that of PM₁₀ and PM_{2.5}. Given this overlap, it is important to develop a comparable metric that allows the mass loading of AMPs to be evaluated against established particulate matter (PM) air quality standards, such as those recommended by the National Environmental Standards and Regulations Enforcement Agency (NESREA). In this context, the current study expresses the mass loading of airborne microplastics (AMPs) as the emission factor (EF) ratio to the sampler flow rate. This allows for a preliminary comparison with conventional particulate matter (PM) concentration values (µg/m³).

$$AML \left(\frac{\mu g}{m^3} \right) = \frac{EF (\mu g/s)}{flow\ rate\ (m^3/s)} \text{ --- eqn 4.7}$$

Where:

- *AML* = Airborne Microplastic mass loading (µg/m³)
- *EF* = the microplastic emission factor (µg/s or µg/day) as expressed in equation 4.6
- *Flow rate* = the sampler flow rate (m³/s or m³/day). In this case the Flow rate (m³/s) = 5L/min × 0.001 × 1/60 sec = 8.33 × 10⁻⁵ m³/s or 7.2m³/day

4.7 Results and Discussion

4.7.1 Airborne Microplastic Concentration

The airborne microplastic samples collected from both dry and wet seasons reveal similar shapes, though distinct concentration differences are evident. Notably, airborne microplastic shows higher concentrations during the dry seasons, reaching a maximum concentration of 17.92 MP/m³ and a minimum of 12.08 MP/m³ (Figure 4.2a). Conversely, wet seasons displayed diminished concentrations of 9.60 to 15.4 MP/m³, averaging 11.31 ± 3.00 MP/m³. These findings suggest that dry conditions facilitate atmospheric microplastic suspension and dispersal through wind, contributing to enhanced concentrations (S. Yang et al., 2024). Wet conditions likely mitigate atmospheric microplastic resuspension from the soil. However, this does not prevent the rain mobilisation of microplastics from disposal sites into waterways through runoff and infiltration into groundwater (Y. Wei et al., 2024).

On the other hand, airborne microplastic concentrations upwind during dry and wet seasons exhibited a mean concentration of 4.25 (± 1.17) MP/m³ and 2.75 (± 1.43) MP/m³ mostly fibres, suggesting different sources of microplastic emissions (Figure 4.2b).

These findings indicate that the signature of the upwind location is affected by distant sources that emit microplastic fibres in both seasons. This is likely due to indoor activities such as walking, washing clothes, or agitating fabrics, which release microscopic fibres into the air (Ageel et al., 2022b). These lightweight fibres can travel long distances via prevailing winds, reaching far-flung areas. In contrast, the on-site location's microplastic profile differs significantly, with fragments likely formed through photo-degradation of larger plastic items such as bottles and packaging. This process, triggered by UV radiation, breaks down plastics into smaller, irregular pieces (Gan et al., 2023). Unlike fibres, these fragments don't undergo long-range transport, indicating local emission sources.

The dominance of fibres at the upwind site sources is likely from textile wear (M. A. Bhat, 2023). The on-site location's fragment-rich profile suggests local microplastic degradation (Liro et al., 2023). These findings highlight distinct source origins for microplastics at each location.

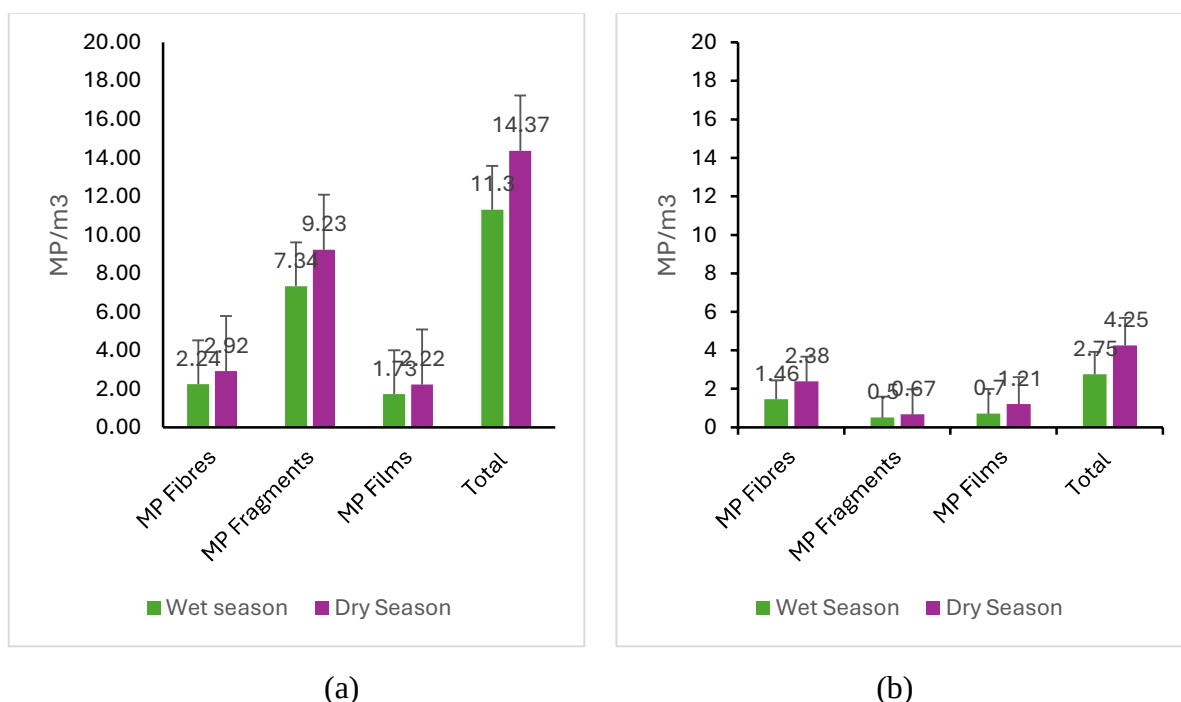


Figure 4. 2: Airborne microplastic concentrations from the disposal site during dry (purple) and wet (green) seasons from on-site and upwind (a) Airborne MP concentrations on-site (b) Airborne MP concentrations Upwind.

The particle size analysis of microplastic particles on-site reveals that more than 73% (527 out of 724 total particles) and 57% (323 out of 570 total particles) were less than 100 µm in size during the dry and wet seasons, respectively (Appendix B Table 4.6a and b). The higher proportion of fragments in both seasons suggests that the disposal site is a significant source of microplastics like fragments. Notably, the wet season recorded a higher proportion of fragments at 61%, while the dry season recorded 57% (Table 4.2).

Table 4. 2: On-site Comparison of Microplastic Particles < 100 µm (Dry vs. Wet Season)

Type	Dry Season	Percentage	Wet Season	Percentage
Fibres	118	22%	64	20%
Fragments	300	57%	197	61%
Films	109	21%	62	19%
Total	527	73%	323	57%

These findings indicate that the disposal site emits a substantial amount of microplastics in fragment shapes, with a slight increase during the wet season. Several factors may contribute to this phenomenon, including mechanical breakdown, weathering, and biodegradation.

The fragmentation of plastic debris into microplastics is a complex process influenced by various mechanical and environmental factors. Mechanical forces, including wind, raindrops, and human activities, can cause the physical breakdown of plastic materials into smaller fragments. Moreover, exposure to environmental stressors such as sunlight, temperature fluctuations, and precipitation can weaken plastic materials, rendering them more susceptible to fragmentation (Matavos-Aramyan, 2024).

Notably, the observed increase in fragment proportion during the wet season suggests that moisture plays a significant role in accelerating the plastic degradation. Higher humidity and precipitation during this season may weaken plastic materials, facilitating their breakdown into smaller fragments (Sutkar et al., 2023). This environmental degradation process can be further exacerbated by other factors such as increased runoff or altered waste transport dynamics.

Changes in waste management practices during the wet season may also contribute to variations in microplastic emissions. For instance, variations in waste collection frequency, disposal methods, or treatment processes may lead to increased exposure of plastic waste to environmental stressors, ultimately influencing the amount and type of microplastics released into the environment (Belioka & Achilias, 2024a). Understanding these seasonal dynamics is crucial for developing effective strategies to mitigate microplastic pollution.

In contrast, the particle size distribution upwind of the disposal site revealed that microplastic particles smaller than 100µm constituted a significant proportion of the particles in the dry and wet seasons. Specifically, 66.7% (68 out of 102 particles) of the total particles in the dry season and 63.6% (42 out of 66 particles) of the total particles in the wet season were found to be smaller than 100µm. This suggests that the bulk of the particles in both seasons are within the smaller size range (Appendix B Table 4.7a and b).

The upwind analysis of microplastic particles revealed a notable presence of fibres in both seasons, their proportion varied slightly between the dry and wet seasons in the size range of 100 µm (Table 4.3). Fibres accounted for 40.2% of the particles during

the dry season and 33.3% during the wet season. In contrast, fragments and films were present in smaller proportions, with fragments constituting 10.8% and 15.2% of the total particles in the dry and wet seasons, respectively.

Table 4. 3: Upwind Comparison of Microplastic Particles < 100 µm (Dry vs. Wet Season)

Type	Dry Season	Percentage	Wet Season	Percentage
Fibres	41	40%	22	33%
Fragments	11	11%	10	15%
Films	16	16%	10	15%
Total	68	67%	42	64%

The dominance of fibre-source microplastics at upwind sites, which are situated within residential areas, suggests that indoor emissions play a significant role in shaping the local microplastic landscape. This finding is consistent with previous research indicating that indoor environments are substantial sources of fibre emissions, which can subsequently contribute to outdoor microplastic pollution (Q. Zhang et al., 2020). The persistence of fibres in the environment highlights the complex interplay between indoor and outdoor microplastic sources, underscoring the need for a comprehensive approach to addressing microplastic pollution. The link between indoor and outdoor microplastic sources implies that efforts to mitigate microplastic pollution should consider both domestic and environmental factors.

AMP Polymer Composition: A comprehensive analysis of airborne microplastic polymers in dry and wet seasons (N=85 particles) revealed consistent polymer-type distributions (Table 4.4). This study's polymer analysis revealed three predominant shapes of microplastics detected, each characterised by a distinct plastic polymer. Notably, six common plastic polymers were analysed, with Polyethylene (PE), Polyamide (PA), and Polyvinyl Chloride (PVC) emerging as the most abundant, collectively accounting for 85-90% of the total microplastics (Table 4.4).

The morphological characteristics of these polymers varied significantly. PVC was predominantly found in fragmented form, whereas PA, specifically nylon, was primarily detected as fibrous microplastics. In contrast, PE was mainly identified as film-like microplastics. These findings underscore the diverse forms of microplastic pollution present in the environment. Fibres accounted for 26.51% and 27.91% of total MP

particles in dry and wet seasons, respectively, with nylon being the predominant polymer (86% and 85%, respectively). A standard density of 1.14 g/cm³, derived from literature (Suaria et al., 2020), was applied to the fibre category for both seasons.

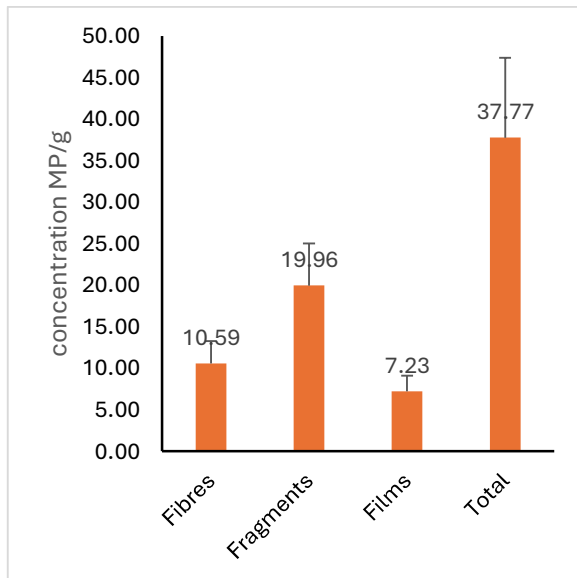
Table 4. 4: Polymer Composition of Airborne Microplastic (Dry and Wet Season)

Shape	Dry Season	Wet Season	Total	Percentage (%)	Polymer (Dry/Wet)	Predominant Polymer (%)	Density (g/cm ³)
Fibre	10.00	12.00	22.00	26.51	Nylon PA (19: 10/9), PET (3: 2/1)	PA (86%)	1.14
Fragment	26.00	20.00	46.00	55.42	PVC (42: 23/19), PE (4: 3/1)	PVC (91%)	1.35
Film	5.00	5.00	10.00	12.05	PE (9: 5/4), PET (1: 1/0)	PE (90%)	0.97
Others	3.00	4.00	7.00	8.43	PS (5: 2/3), PP (2: 1/1), PET (0: 0/0)	PS (71%), PP (29%)	-
Total	44.00	41.00	85.00	100.00	-	-	-

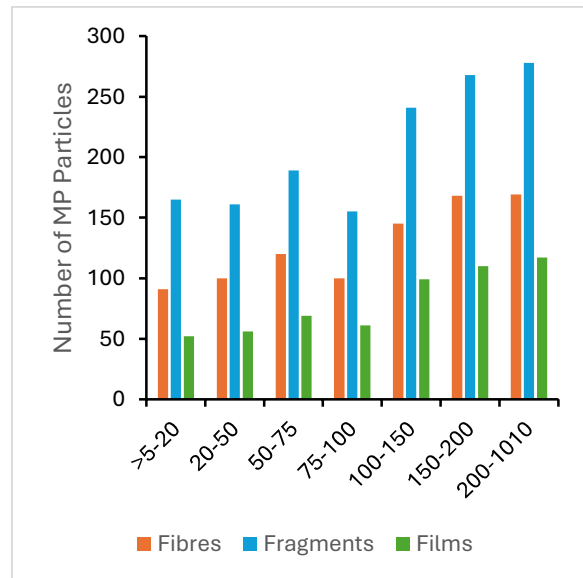
Fragments constituted the majority of microplastics, comprising 55.42% and 53.66% in dry and wet seasons, respectively. Polyvinyl chloride (PVC) dominated this category, making up 91% of fragments in both seasons. A density of 1.35 g/cm³ was applied to align with established PVC density ranges (Borges-Ramírez et al., 2020a). Films represented 12.05% and 11.76% of total particles in dry and wet seasons, respectively, with polyethene being the primary polymer (90% and 80%, respectively). A density of 0.97 g/cm³ was applied to films across both seasons, reflecting literature-reported values (Wei et al., 2021). The "Others" category exhibited diverse polymer compositions, with polystyrene (71%) and polypropylene (29%) present in the dry season, and polystyrene (57%), polypropylene (29%), and polyethene terephthalate (14%) detected in the wet season. Densities for this category were determined based on specific polymer compositions and literature values (Borges-Ramírez et al., 2020b).

4.7.2 Microplastic Concentrations in Soil Samples

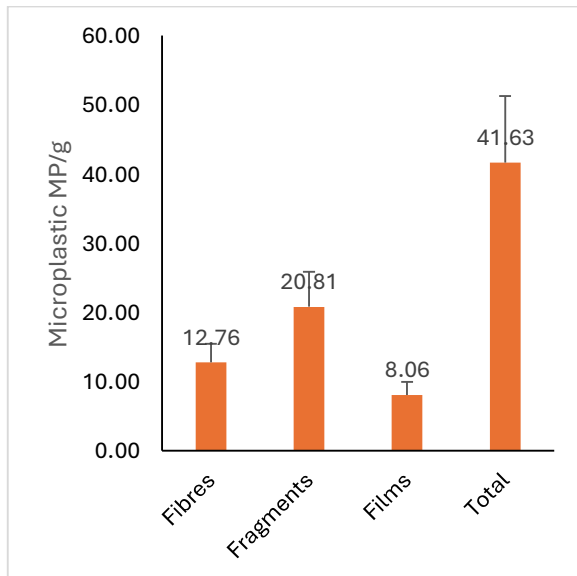
Microplastic concentrations at the disposal site varied seasonally. In the dry season, the average concentration was 37.77 ± 9.6 MP/g, with fragments (53%) dominating, followed by fibres (28%) and films (19%) (Figure 4.3a and b). In contrast, the wet season has the highest concentrations 41.63 ± 10.45 MP/g (Figure 4.3c and d).



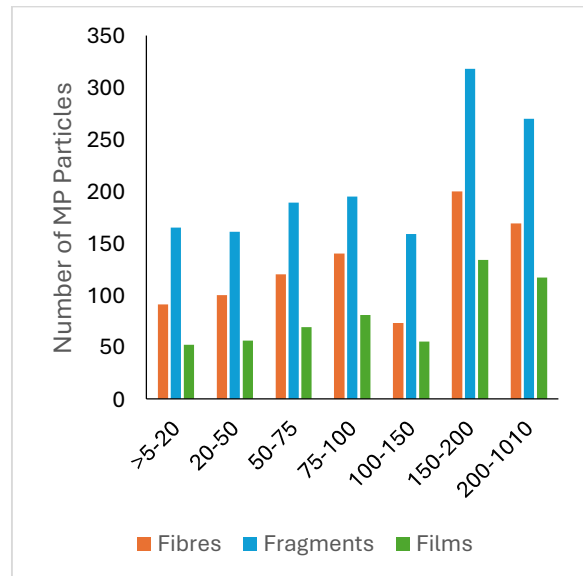
(a)



(b)



(c.)



(d)

Figure 4. 3: Soil Microplastic Particle Sizes Distributed in Dry and Wet Seasons (a) Soil Concentration MP in Dry Season (b) MP Sizes Distribution in Dry Season (c) Soil MP Concentration in Wet Season (d) MP Sizes Distribution in Wet Season.

The microplastics mainly fell within the 100-150 μm to 200-1010 μm size range in both the dry and wet seasons. Notably, the higher wet season MP concentrations can be attributed to increased soil density (1600 kg/m^3) and reduced porosity (0.36 Appendix Table 4.4a and b), thereby retaining microplastics within the soil, whereas the dry season's lower density (1430 kg/m^3) and larger pores (0.46, Table 4.4b) may facilitate aerosolisation (Belioka & Achilias, 2024).

The disposed materials in municipal solid waste facilities primarily included household plastics (e.g., packaging, bags, and bottles), textile waste, agricultural films, and electronic waste (Ma & Singhirunnusorn, 2012). As these MPs are produced at the disposal sites due to various processes, wind may lift and transport these microplastics into the air, enhancing their dispersion and potential human exposure (Rezaei et al., 2019). This process can be influenced by seasonal factors, such as wind speed and direction, humidity, and precipitation, which can either facilitate or limit the aerosolisation and transport of microplastics (Borah et al., 2024).

In this study, the dry season is characterised by relatively high wind speeds of 6.0 m/s (average). The prevailing winds blow from the east/northeast (45%) at 45.0° , creating favourable conditions for microplastic aerosolisation (Al Harraq et al., 2022). The increased wind turbulence facilitates the lift and transport of smaller microplastics, such as fibres, while dispersing larger fragments and films over wider areas (Al-Zawaidah et al., 2024b).

This wind-driven mechanism enhances environmental dispersion, potentially increasing human exposure and health risks associated with airborne microplastics (Abad López et al., 2023; Chukwuka et al., 2024).

In contrast, the wet season is characterised by relatively lower wind speeds, with an average of 3.5 m/s. Prevailing winds shift to the west/southwest (55%) at 225.0° , reducing the effectiveness of microplastic aerosolisation. The decreased wind turbulence may limit microplastics' lift and transport, allowing them to settle and accumulate in the soil. Additionally, the wet season's increased humidity and precipitation further reduce microplastic aerosolisation, trapping particles in the soil and reducing atmospheric transport (D. Huang et al., 2022).

Soil MP Polymer Composition: The polymer analysis of microplastic particles in dry and wet seasons revealed consistent polymer-type distributions (Table 4.5, N=84 particles). Fibres constituted 26.19% and 28.57% of total particles in dry and wet

seasons, respectively, with nylon being the predominant polymer (73% and 75%, respectively), and a density of 1.14 g/cm³ was applied to fibres in both seasons.

Table 4. 5: Polymer Composition of Microplastic in Soil (Dry and Wet Season)

Shape	Dry	Wet	Total	(%)	Polymer Type (Total: Dry/Wet)	Predominant Polymer (%)	Density (g/cm ³)
Fibres	11.00	12.00	23.00	27.38	Nylon (17: 8/9), PET (6: 3/3)	Nylon (74%)	1.14
Fragments	23.00	22.00	45.00	53.57	PVC (39: 20/19), PE (6: 3/3)	PVC (87%)	1.35
Films	5.00	4.00	9.00	10.71	PE (7: 4/3), PET (2: 1/1)	PE (78%)	0.97
Others	3.00	4.00	7.00	8.33	PS (4: 2/2), PP (2: 1/1), PET (1: 0/1)	PS (57%), PP (29%), PET	-
Total	42.00	42.00	84.00	100.0	-	-	-

Fragments represented the largest proportion of microplastics, accounting for 54.76% and 52.38% in dry and wet seasons, respectively. Polyvinyl chloride (PVC) was the primary polymer in this category, comprising 87% of fragments in both seasons. Consequently, a density of 1.35 g/cm³ was applied to fragments.

Films comprised 11.90% and 9.52% of total particles in dry and wet seasons, respectively, with polyethene being the dominant polymer (80% and 75%, respectively). 0.97 g/cm³ was applied to the films for both seasons. The "Others" category exhibited varied polymer compositions across seasons. In the dry season, polystyrene (67%) and polypropylene (33%) were present, whereas in the wet season, polystyrene (50%), polypropylene (25%), and polyethene terephthalate (25%) were detected. Densities for this category were applied based on specific polymer compositions.

4.7.3 Aerodynamic Properties and Rising Velocity of Microplastics

The aerodynamic properties of microplastics play a crucial role in understanding their transport from soil to air (Yang et al., 2024). This study investigated the aerodynamic properties of microplastics, specifically nylon fibres, PVC fragments, and polyethene

(PE) films, and found significant variations in transport properties across materials and environmental conditions. The Image analysis of the microplastic particles reveals that Nylon fibres exhibited the smallest equivalent diameter, ranging from 46.60 to 46.07 μm (Table 4.6).

Table 4. 6: Microplastic Dimensions: Dry vs. Wet Season

Shape	Parameter	Dry	Wet
Fibres	Length (μm)	78.00	78.65
	Width (μm)	24.82	23.60
	Aspect Ratio	3.14	3.13
	Thickness (c) (μm)	52.26	52.69
	Eq. Dia. (dequiv) (μm)	46.60	46.07
	Corey Shape Factor (CSF)	1.22	1.19
Fragments	Length (μm)	102.11	95.83
	Width (μm)	44.40	43.10
	Aspect Ratio (μm)	2.30	2.22
	Thickness (c) (μm)	68.41	64.21
	Eq. Dia. (dequiv) (μm)	67.69	64.25
	CSF	1.00	1.02
Films	Length (μm)	77.59	94.25
	Width (μm)	36.95	44.81
	Aspect Ratio	2.10	2.10
	Thickness (c) (μm)	51.99	63.15
	Eq. Dia. (dequiv) (μm)	53.02	64.37
	CSF	0.97	0.97

In contrast, PVC fragments displayed larger diameters, between 67.69 and 64.25 μm . Meanwhile, PE films showed intermediate diameters, ranging from 53.02 to 64.37 μm . The Reynolds number (Re) plays a pivotal role in determining the flow regime that influences the resuspension of microplastics (MPs) from plastic waste at disposal sites. As a dimensionless ratio of inertial to viscous forces in airflow (Dittmar et al., 2024), Re helps predict the likelihood of MP entrainment into the air. Elevated Re values, indicative of turbulent airflow often generated by wind or site activities such as waste management operations, enhance the resuspension of MPs by overcoming the adhesive forces holding them to the waste surface (Shamskhany & Karimpour, 2022). In turbulent flow regimes, characterised by high Re values, the chaotic motion of air

facilitates the detachment and suspension of MPs, increasing their potential for long-distance transport.

In contrast, lower Re values, indicative of laminar flow, result in a more predictable and streamlined airflow pattern. In this regime, the viscous forces dominate, and MPs tend to remain adhered to the surface, limiting their resuspension. The effect of Re on the flow pattern of MPs is critical in understanding the dynamics of MP resuspension and transport. By characterising the Re at disposal sites, researchers can better predict the likelihood of MP resuspension and transport, ultimately informing strategies to mitigate their impact on nearby communities. The resuspension of MPs from disposal sites can have significant implications for the health and environment of surrounding communities. Airborne MPs can be inhaled, potentially leading to respiratory issues, and can also contaminate soil, water, and air, affecting local ecosystems.

This study investigates the relationships involving the Reynolds number (Re) and several key factors that influence the resuspension of microplastics (MP). Specifically, it focuses on how the Drag Coefficient and Rising Velocity interact with Re , highlighting their significance in understanding the dynamics of microplastic movements in aquatic environments. By examining how these factors interact, the study aims to enhance our understanding of the processes that result in the resuspension of microplastics from plastic waste at disposal sites into the air. The findings will help inform strategies to minimise MP emissions, reduce exposure risks for nearby communities, and promote more effective waste management practices.

The impact of Re was observed across three shapes (Table 4.7): fibres/nylon (PA), fragments/PVC, and films/PE, under dry and wet conditions. Fibres/nylon (PA) exhibited laminar flow in both dry ($Re = 18.327$) and wet ($Re = 10.522$) conditions. However, the drag coefficient increased significantly from 0.872 in dry conditions to 2.539 in wet conditions. This resulted in a notable reduction in rising velocity from 0.020 m/s to 0.015 m/s, indicating higher resistance and reduced transport capacity under moist conditions. Consequently, fibre-shaped microplastic mobility and dispersal are limited in wet environments.

For fragments/PVC, the flow pattern suggests a shift from a slightly turbulent tendency ($Re = 25.556$) towards a more streamlined, laminar flow ($Re = 15.285$) as conditions change from dry to wet.

Table 4. 7: Microplastic Aerodynamic Properties

Description	Fibres/Nylon (PA)		Fragments/(PVC)		Films/(PE)	
Season	Dry	Wet	Dry	Wet	Dry	Wet
Reynolds Number (Re)	18.327	10.522	25.556	15.285	25.605	11.972
Drag Coefficient (Cd)	0.872	2.539	1.0194	1.190	1.047	1.351
Rising Velocity (wx, m/s)	0.020	0.015	0.022	0.021	0.021	0.017

Even though fragments exhibit some turbulence-related properties comparable to fibres, their flow remains within a stable, streamlined regime. The drag coefficients remain relatively stable (dry = 1.0194, wet = 1.190), allowing for consistent rising velocities (dry = 0.022 m/s, wet = 0.021 m/s). Overall, the rising velocities of fragments remained consistent across seasons. These velocities align with findings from the earlier study (Hinata et al., 2023a).

The aerodynamics of films/PE remain consistent across dry and wet conditions, exhibiting laminar flow ($Re = 25.605$ and 11.972 , respectively), in comparison to fragments. Although the drag coefficient increases slightly in wet conditions (from 1.047 to 1.351), Films/PE maintain a relatively high rising velocity in dry conditions (0.021 m/s) comparable to fragments but lower rising velocities (0.017 m/s) in wet conditions. This aerodynamic behaviour of films can be attributed to their thin, flat shape, which facilitates smooth laminar airflow and reduces turbulence (Q. Chen et al., 2021b). Their moderate density and flexibility further contribute to consistently rising velocities across seasons, making them less affected by drag variations.

Studies have documented rising and sinking velocities for microplastics in environmental settings. Specifically, velocities for weathered microplastics collected from natural environments have been reported to vary from 0.002 to 0.199 m/s (Jalón-Rojas et al., 2022; Waldschläger et al., 2020b). Compared with this literature value, this study's rising velocities fall within the reported range. These findings highlight the

importance of considering microplastic size, shape, and surface properties in transport models.

4.7.4 Diffusion Coefficient, Concentration Gradient, Flux and Emission Factors

This section provides a comprehensive analysis of microplastic emissions, focusing on PVC fragments, nylon fibres, and PE films during dry and wet seasons. The emissions were quantified using flux measurements at a height of **1.6 m** above ground from a site covering an extensive area of **23,759.56 m²**. This study offers critical insights into the interplay of material properties, seasonal variability, and soil porosity in influencing microplastic fluxes and emission factors. These findings have significant implications for understanding the environmental transport and potential human health impacts of airborne microplastics.

To facilitate the conversion of microplastic (MP) flux to standard units, the volume and mass of microplastics were first estimated, providing a basis for subsequent calculations of flux concentration gradients and diffusion coefficients. The volume and mass of microplastics were calculated using geometrical approximations based on their shapes and material densities. Fibres were approximated as cylinders, fragments and films as ellipsoids. The dimensions of MP obtained using Image-J software were used for the volume calculations (Table 4.8).

Table 4. 8: Microplastic Volume and Mass Estimation

Season	Shapes	Volume m ³	Density g/m ³	Mass (g)	Mass (µg)
Dry Season	Fibre	3.44E-14	1.14E+06	3.92E-08	3.92E-14
	Fragment	1.39E-13	1.35E+06	1.88E-07	1.88E-13
	Films	1.40E-13	9.70E+04	1.36E-08	1.36E-14
Wet Season	Fibre	3.78E-14	1.14E+06	4.30E-08	4.30E-14
	Fragment	1.62E-13	1.35E+06	2.19E-07	2.19E-13
	Films	7.81E-14	9.70E+04	7.57E-09	7.57E-15

This approach allowed for a more accurate estimation of the volume of microplastics based on their shapes and sizes. The calculated volumes were then multiplied by the respective densities of each microplastic: 1.14 g/cm³ for nylon fibres, 1.35 g/cm³ for

PVC fragments and 0.97 g/cm³ for PE films to estimate the mass of individual particles (Kooi et al., 2021).

By combining image analysis with density values, researchers can gain a better understanding of the mass and volume distributions of MPs in environmental samples. This information can be used to inform models of MP transport, fate, and impact, ultimately contributing to a more comprehensive understanding of the implications of microplastic pollution.

In this study (Table 4.9), the field sampling period lasted only 8 hours; mass fluxes (g/m²/s) correspond to 8-hour sampling averages, whereas the emission factors (µg/day) were extrapolated to a 24-hour interval by applying a time-scaling factor (Schissel et al., 2024).

Table 4. 9: Microplastic Diffusion Coefficient and Concentration Gradient

Material	Fragment		Fibre		Films	
Season	Dry	Wet	Dry	Wet	Dry	Wet
Flux (g/m ² /s)	9.15×10 ⁻¹⁴	8.26×10 ⁻¹⁴	2.98×10 ⁻¹⁴	2.77×10 ⁻¹⁴	3.27×10 ⁻¹⁵	2.71×10 ⁻¹⁵
Emission Factor (µg/day)	187.79	169.54	61.26	56.83	6.72	5.57
Dmod (m ² /s)	5.94 ×10 ⁻¹⁵	4.83×10 ⁻¹⁵	1.75×10 ⁻¹⁴	1.32×10 ⁻¹⁴	8.12 ×10 ⁻¹⁵	1.16 ×10 ⁻¹⁴
Gradient (MP/m ⁴)	8.21×10 ⁷	7.80 ×10 ⁷	4.35 ×10 ⁷	4.88 ×10 ⁷	2.97 ×10 ⁷	3.08 ×10 ⁷
Flux (MP/m ² /s)	4.88×10 ⁻⁷	3.77×10 ⁻⁷	7.61×10 ⁻⁷	6.43×10 ⁻⁷	2.41×10 ⁻⁷	3.58×10 ⁻⁷
conversion Mass (µg)	1.88×10 ⁻¹³	2.19 ×10 ⁻¹³	3.92×10 ⁻¹⁴	4.30 ×10 ⁻¹⁴	1.36 ×10 ⁻¹⁴	7.57×10 ⁻¹⁵

In practice, the site-wide emission factor (µg/day) was calculated by first converting the 8-hour average flux to a daily mass release by multiplying the flux by the dump-site area (m²), the total number of seconds in 24 hours (86,400 s), and the unit conversion from grams to micrograms (×10⁶), thereby yielding an emission rate expressed over a standard 24-hour period interval.

Fragment (PVC): PVC fragments exhibit the highest flux in both seasons, with 9.15 × 10⁻¹⁴ g/m²/s and 8.26 × 10⁻¹⁴ g/m²/s in dry and wet seasons, respectively (Table 4.9). These translate to approximately 188 µg/day and 170 µg/day emission factors in

the dry and wet seasons. Despite their relatively low diffusion coefficients ($5.94 \times 10^{-15} \text{ m}^2/\text{s}$ in dry and $4.83 \times 10^{-15} \text{ m}^2/\text{s}$ in wet), their high concentration gradient ($8.21 \times 10^{-7} \text{ MP/m}^4$ dry, and $7.80 \times 10^{-7} \text{ MP/m}^4$ wet) drives significant fluxes of $4.88 \times 10^{-7} \text{ MP/m}^2/\text{s}$ ($9.15 \times 10^{-14} \text{ g/m}^2/\text{s}$) dry and $3.77 \times 10^{-7} \text{ MP/m}^2/\text{s}$ ($8.26 \times 10^{-14} \text{ g/m}^2/\text{s}$) in wet. The high density of PVC (1.35 g/cm^3) and rigid structure allow them to easily detach and transport, especially in the dry season when soil porosity is higher (0.45) (Ben Stride et al., 2024). This predominance can be attributed to their abundance and moderate rising velocity.

Fibres (Nylon, PA): Nylon fibres display a notable flux of $2.98 \times 10^{-14} \text{ g/m}^2/\text{s}$ in the dry season and $2.77 \times 10^{-14} \text{ g/m}^2/\text{s}$ in the wet season, corresponding to $56.83 \text{ } \mu\text{g/day}$ and $57 \text{ } \mu\text{g/day}$, respectively. The higher diffusion coefficients ($1.75 \times 10^{-14} \text{ m}^2/\text{s}$ in dry; $1.32 \times 10^{-14} \text{ m}^2/\text{s}$ in wet) enhance their mobility, resulting in flux values of $7.61 \times 10^{-7} \text{ MP/m}^2/\text{s}$ (dry) and $6.43 \times 10^{-7} \text{ MP/m}^2/\text{s}$ (wet). Despite this, their tendency to adhere to soil particles due to their high aspect ratio and density (1.14 g/cm^3) limits their total emissions (Roy et al., 2022). The reduced porosity in wet conditions further inhibits their resuspension, but their structural resilience and moderate rising velocities ensure continued contribution to airborne microplastics.

Films (PE): PE films have the lowest emission factors among the three types, with 6.72 g/day in the dry season and 5.57 g/day in the wet season, equivalent to $3.27 \times 10^{-15} \text{ g/m}^2/\text{s}$ (dry) and $2.7 \times 10^{-15} \text{ g/m}^2/\text{s}$ (wet). Although their diffusion coefficient increases in wet conditions (8.12×10^{-15} dry to wet $1.16 \times 10^{-14} \text{ m}^2/\text{s}$). The unique properties of microplastic films, particularly their thin, flexible structure and low density of 0.97 g/cm^3 , significantly influence their environmental behaviour. Due to their low density, microplastic films readily adhere to soil particles, resulting in minimal upward transport via wind or water erosion. Additionally, their flexible nature enables them to conform to soil surfaces, reducing mobility. The film's thin, flexible structure and low density (0.97 g/cm^3) provide adhesion to soil, resulting in minimal upward transport. The flux for PE films in the dry season was $3.58 \times 10^{-7} \text{ MP/m}^2/\text{s}$ in the wet season. The low porosity of wet soil further limits their detachment and transport, resulting in the lowest mass equivalent emissions among the three materials.

The mass contributions of PE films are the lowest, with $1.36 \times 10^{-8} \text{ g}$ in dry conditions and $7.57 \times 10^{-9} \text{ g}$ in wet conditions. However, their diffusion coefficients show slight seasonal variability, increasing from $8.12 \times 10^{-15} \text{ m}^2/\text{s}$ (dry) to $1.16 \times 10^{-14} \text{ m}^2/\text{s}$ (wet).

Despite their low density (0.97 g/cm³), which aids their atmospheric suspension, their flexible structure limits their vertical transport compared to fragments and fibres.

4.8 Implications and Limitations

4.8.1 Environmental and Health

The airborne microplastic mass loading (AML) measured in this study ranged from 0.77 to 26.07 µg/m³, with notable variation across particle types and seasonal conditions. The highest mass loading was recorded for fragments during the dry season (26.07 µg/m³), followed by fibres (8.51 µg/m³) and films (0.93 µg/m³) under the same conditions (Table 4.10).

Table 4. 10: Seasonal Mass Loading of Airborne Microplastics (µg/m³)

Material	Season	EF (µg/day)	AML (µg/m ³)
Fragment	Dry	187.79	26.07
Fragment	Wet	169.54	23.54
Fibre	Dry	61.26	8.51
Fibre	Wet	56.83	7.89
Films	Dry	6.72	0.93
Films	Wet	5.57	0.77

Wet season values were consistently lower, reflecting the dampening effects of increased humidity, rainfall, and reduced wind speed on particle resuspension from the disposal site. This seasonal contrast supports the hypothesis that dry, turbulent conditions facilitate vertical and horizontal transport of microplastics.

Although the AMP loadings fall below the 24-hour ambient air quality standards established by the National Environmental Standards and Regulations Enforcement Agency (NESREA), 40 µg/m³ for PM_{2.5} and 150 µg/m³ for PM₁₀, the significance of these values should not be underestimated. A considerable proportion of AMPs detected in this study fall within the respirable range of <100 µm, indicating that their aerodynamic properties and deposition behaviours may closely mimic those of regulated particulate matter. From a health risk perspective, particles in this size range

are capable of penetrating the upper and lower respiratory tracts, potentially eliciting adverse biological responses such as inflammation, cytotoxicity, or endocrine disruption, particularly given their persistent polymeric composition and capacity to adsorb hazardous chemicals (Correia Prata, 2017; Ebrahimi et al., 2022; Revel et al., 2018).

The current regulatory frameworks for air quality monitoring and control do not include microplastics as a recognised pollutant class (Yuk et al., 2022). As a result, even low-level exposures remain largely unquantified and unregulated despite their potential for chronic environmental and physiological accumulation. To address this gap, the present study introduces a mass-based metric for airborne microplastics expressed in $\mu\text{g}/\text{m}^3$ (Equation 4.7). The mass loading of airborne microplastics provides a meaningful basis for situating AMP exposure within existing particulate matter frameworks.

While the AML values reported in this study remain lower than NESREA's limits for PM, the airborne microplastics' toxicological and ecological uniqueness demands independent assessment standards that account for particle morphology, polymer type, and associated contaminants. Therefore, a microplastic-specific regulatory threshold, analogous to $\text{PM}_{2.5}$ or PM_{10} limits, is timely and necessary to support future air quality policies in microplastic-prone environments such as waste disposal sites.

4.8.2 Wider Implications

This chapter offers a detailed examination of airborne microplastic (AMP) emissions from a waste disposal site in Nigeria, focusing on particles ranging in size from $<5\ \mu\text{m}$ to $1010\ \mu\text{m}$. The study identified that more than 73% (dry) to 57% (wet) of the airborne MP were below $100\ \mu\text{m}$, falling within the respirable range typically associated with PM_{10} . These findings suggest that AMPs may behave similarly to conventional particulate matter in the atmosphere, reinforcing the relevance of comparing their mass loadings with established air quality standards, as discussed in Section 4.8.1.

The investigation further explores how soil porosity, seasonal variations, and polymer type influence the resuspension of microplastics from soil to air. These factors were reflected in the emission factor (EF) values, which were higher during the dry season due to increased soil dryness and porosity that favour particle release. For instance, fragments showed the highest EF ($187.8\ \mu\text{g}/\text{day}$), while fibres and films followed. In the wet season, EFs declined across all types, e.g., fragments reduced to

169.5 µg/day, likely due to increased soil moisture and reduced particle mobility. Polymer characteristics further influenced these trends, with denser fragments more readily mobilised than lighter but more adhesive films. Despite higher fluxes during the dry season, the absolute flux values remain small (order of magnitude 10^{-20} µg/m²/s). Nevertheless, when aggregated across the surface area of the disposal site and scaled temporally, these small fluxes generate non-negligible emission factors (EFs). These EFs quantified for each microplastic type and season demonstrate that even modest fluxes can contribute to moderately high airborne AMP concentrations, with on-site levels found to be 6–8 times higher than upwind concentrations. Importantly, the estimated mass loadings fall below NESREA's 24-hour limits for PM_{2.5} and PM₁₀ but still indicate a moderate contribution to local air quality burdens, particularly due to the size, shape, and persistence of AMPs.

4.8.3 Limitations and Future Studies

This study is not without limitations. While this study thoroughly examines how particle size, soil moisture, and porosity influence microplastic emissions, it did not establish correlations between microplastic fluxes and broader factors such as waste composition, local population density, or specific management practices at the site. The analysis indicates a strong correlation between microplastic emissions and soil porosity, with seasonal changes in porosity significantly affecting flux rates. Concerns about microplastic emissions are rising, especially in Nigeria, where numerous disposal sites and poor waste management practices complicate the issue. This study provides preliminary insight into how microplastics can affect nearby communities around disposal sites, highlighting the need for further research and effective waste management strategies to mitigate these impacts. Moreover, differences in waste handling practices across regions may exacerbate the issue, underscoring the importance of standardised waste management protocols and regional cooperation to address the escalating problem of microplastic pollution.

Future research should prioritise expanding the geographical scope of AMP measurements in Nigeria and other regions with similar waste management challenges. This includes collecting more comprehensive data across various soil types, climatic conditions, and seasonal cycles. Such efforts would enhance the accuracy of flux estimations and emission factors, reducing the current uncertainties

and improving our understanding of the microplastic emissions from waste disposal sites and the contributions to the global transport of airborne microplastics. Addressing these research gaps is crucial for developing targeted mitigation strategies and informing policies for reducing microplastic pollution in the atmosphere.

4.9 Conclusion

This chapter demonstrates and quantifies the seasonal flux of airborne microplastic (AMP) from a municipal waste disposal site in Nigeria. The study characterised AMP particles ranging from $<5\text{ }\mu\text{m}$ to $1010\text{ }\mu\text{m}$, revealing that over 73% (dry season) and 57% (wet season) of particles were below $100\text{ }\mu\text{m}$ within the respirable range typically associated with PM_{10} and $\text{PM}_{2.5}$. Despite relatively small surface flux values (on the order of $10^{-20}\text{ }\mu\text{g}/\text{m}^2/\text{s}$), the calculated emission factors (EFs) when scaled across area and time demonstrated a non-trivial contribution to local airborne microplastic burdens. The AMP loadings fall below NESREA's 24-hour limits for $\text{PM}_{2.5}$ and PM_{10} ; their size and persistence raise important concerns regarding inhalation exposure and accumulation.

Overall, this chapter contributes to the overarching research objective by offering a bottom-up quantification of microplastic fluxes and emissions from terrestrial sources, a domain that remains significantly underexplored. It lays the groundwork for the predictive models, incorporating site-specific emission data to simulate AMP dispersion. The next chapter builds upon these findings by integrating the calculated emission factors into an atmospheric dispersion model. This modelling component will simulate airborne microplastics transport, dilution, and potential deposition under varying environmental conditions. Ultimately, this modelling approach will provide deeper insights into the regional and possibly global movement of AMPs, contributing to improved policy formulation and air quality management strategies.

Chapter 5.0: Airborne Microplastic Transport and Exposure (Model-Based Assessment)

This chapter provides a comprehensive discussion of the findings from the study on airborne microplastic emissions from waste disposal sites, focusing on dispersion, potential environmental and health implications, and transport dynamics. The Chapter develops a framework for using the SCREEN3 model to investigate the downwind concentrations of airborne microplastics. The SCREEN3 model outputs are critically examined, highlighting the spatial variation of microplastic concentrations, their transport and fate in the atmosphere, and associated risks to human health and ecosystems. Additionally, the limitations and assumptions of the modelling approach are discussed, placing the results within the wider context of microplastic research and laying the foundation for future studies on their environmental impact.

5.1 Introduction

Municipal solid waste disposal sites have become critical sources of airborne pollutants, including microplastics (Farid et al., 2024; Y. Hu et al., 2024; Petrović et al., 2023). Once released into the atmosphere through open burning, waste handling, and mechanical abrasion, airborne microplastics (AMPs) can travel considerable distances, affect air quality and potentially cause risks to human health and ecosystems (Pekdogan et al., 2024). Quantifying the dispersion of these pollutants from waste sites is essential for understanding their environmental impacts and formulating effective mitigation strategies.

Several atmospheric dispersion models have been employed to simulate pollutant transport, including HYSPLIT, ADMS, AERMOD, CALPUFF, and FLEXPART (Arciszewska & McClatchey, 2001; Dispersion, 2021; Mohan et al., 2011; S. Yang et al., 2024). These models are widely applied in waste disposal sites for complex and long-range dispersion studies, providing detailed simulations of pollutant behaviour under varying meteorological and topographical conditions (Connan et al., 2013; Leroy et al., 2010). However, these advanced models often require extensive computational resources, large datasets, specialised expertise, and sometimes a fee for accessing the model package, which can limit their accessibility for certain applications (Haq et al., 2019; Snoun et al., 2023a).

In contrast, the SCREEN3 model, a simplified Gaussian plume dispersion model developed by the U.S. Environmental Protection Agency (EPA), offers a more accessible alternative (Goudarzi et al., 2017). SCREEN3 has been widely used for regulatory purposes and assessing the impact of various pollutants, such as particulate matter, volatile organic compounds (VOCs), and heavy metals (S. Sharma et al., 2020; Umurhan, 2013). It is particularly suited for preliminary assessments, providing quick estimates of ground-level pollutant concentrations at specified downwind distances (BAROUTIAN et al., 2006). Its user-friendly interface and minimal data requirements make it an ideal choice for studies that prioritise rapid and straightforward simulations (AQIEG, 2002).

This study represents the first application of the SCREEN3 model for simulating the dispersion of airborne microplastics from a municipal solid waste disposal site. The choice of SCREEN3 is driven by its capability to deliver reliable results under the constraints of limited input data and its proven track record in modelling point and area source emissions. By integrating site-specific emission data and meteorological conditions, the model can simulate microplastics' downwind concentrations and evaluate their dispersion patterns across dry and wet seasons.

This study aims to demonstrate the potential of SCREEN3 as a practical tool for airborne microplastic research. The outputs provide insights into the environmental distribution of microplastics and establish a foundation for future studies on pollutant dispersion from waste disposal activities.

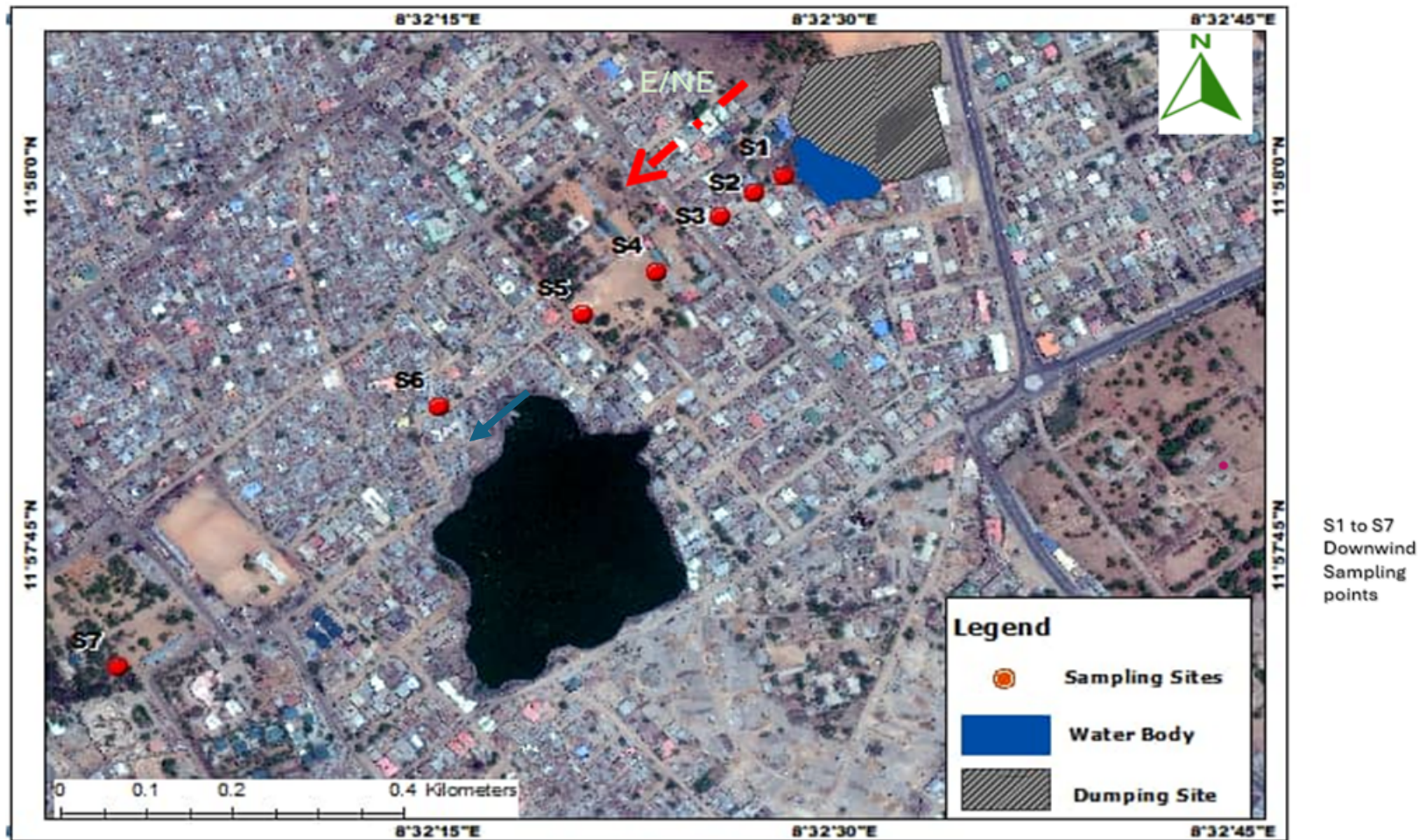
5.2 Field sampling

The study applied a systematic approach to investigate the emission of airborne microplastic (AMP) concentrations and dispersion patterns at a typical waste disposal site in Nigeria.

The sampling campaigns were conducted in two phases. Firstly, 20 upwind samples were collected at a fixed point upwind of the disposal site for dry and wet seasons, serving as a reference point for AMP emissions. The decision not to collect immediate upwind samples, in conjunction with on-site and downwind samples, was informed by the prior establishment of the site's preliminary assessment, as detailed in Chapter 4. On-site and downwind sampling were conducted in series. Notably, the fieldwork campaigns were conducted alternately between on-site and downwind locations, necessitated by logistical constraints and limited availability of air sampling equipment.

This methodology ensured temporal balance and representation across the 84-day study period, encompassing dry and wet seasons. Out of the 84 samples collected, 42 on-site samples were utilised in Chapter 4 for flux estimation purposes. Conversely, the remaining 42 downwind samples were employed in the current chapter to determine the model capacities in simulating airborne MP (Table 5.1A and 5.1B, Appendix C). This guarantees that data from both on-site and downwind sites are collected under comparable meteorological conditions (C. Lin et al., 2011). Downwind sampling was conducted at seven distinct locations along the direction of the prevailing wind. At each location, triplicate samples were taken over three days. These samples were then averaged to produce a representative value. This approach helped to reduce variability and increase the reliability of the data. These data were used to evaluate the model's performance and determine its capability to accurately represent real-world conditions. Once sampling is complete at one site, it moves to the next point, and this process continues until all seven points have been sampled. Details about the number of samples and distances are presented in Figure 5.1A and B.

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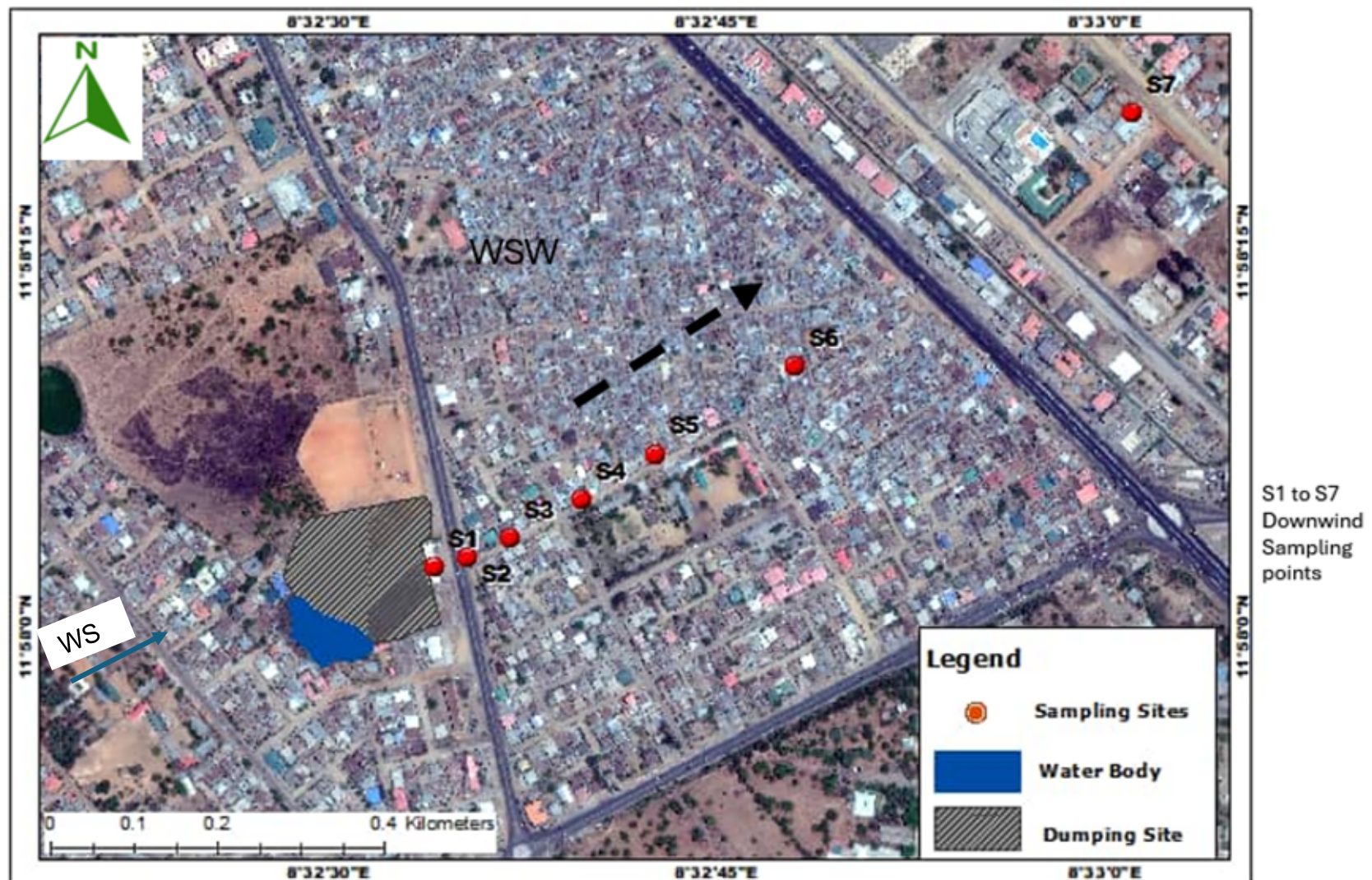


Figure 5.1B: Distribution of Downwind Sampling Points during Wet Season

The primary benefit of adopting this method is its ability to overcome limitations by offering a strong and efficient framework for investigating airborne microplastic (AMP) concentrations and dispersion patterns at a waste disposal site. Notably, the method effectively defies the constraints faced during the study, including the limited resources, time, and uncertainties related to the collection process. The study faced a lack of air samplers; however, the methodology adopted in this work defies the deployment of multiple samplers and may provide analogous results to other locations facing similar challenges.

The low-cost air samplers described in previous Chapters (3 and 4) were set to operate at a flow rate of 5L/min (Kjærgaard et al., 2024). Each sampling event lasts for a fixed duration of 8 hours per day. The sampler was positioned at a height of 1.6m above ground to simulate human exposure levels (Maurizi et al., 2024).

The air sampling protocol targeted airborne microplastics with a size range of 2-1000 μm using a 47mm PTFE filter paper with 2 μm pore size (Azari et al., 2023b). The air sampling typically occurred over 8 hours (08:00-16:00) with night-time shutdown to conserve energy and minimise security risks (Azari et al., 2023c). However, in the event of rainfall, the sampling time was usually adjusted to prevent equipment damage. The collected air samples are sealed in sterile glass containers to avoid contamination. The air samples, including the blanks, were sealed and transported to the laboratory for further processing and analysis. Meteorological data was collected using a Kestrel 2000 weather meter.

Ultimately, this methodology aims to assess spatial variations in AMP concentrations, validate a dispersion model, provide robust data collection, and improve the reliability of results. Following this structure, the study can effectively investigate AMP dispersion patterns and concentrations, contributing valuable insights to environmental monitoring and pollution research.

5.3 Laboratory Analysis

Microplastics collected from air samples were processed and analysed following the established protocol detailed in Chapter 4. This involved the decomposition of organic matter using Fenton's reagent, density separation with a zinc chloride solution, and subsequent filtration to isolate microplastic particles. The collected microplastics were then stained with Nile Red and visualised using fluorescence stereomicroscopy, with images analysed using ImageJ software for quantification. Additionally, contamination

control outlined in Chapter 4 was implemented for these samples throughout the experimental process. These include the analysis of field and procedural blanks to ensure data reliability. Microscopic analysis of these blank samples revealed minimal contamination, with only 11 particles subtracted from the final results (Bakir et al., 2023).

5.4 Framework for Dispersion Modelling

Microplastic (MP) concentrations in air are often reported as MP/m³, representing the numerical abundance of particles per cubic meter of air (Shao et al., 2022b; X. Zhao et al., 2023a). While this unit is suitable for initial quantification, dispersion models require mass-based units (e.g., µg/m³) to accurately simulate transport and deposition dynamics (Sabdo Yuwono & Arif, 2020). To bridge this gap, a robust conversion framework is essential.

Mass-based units are crucial for validating dispersion models, which predict the transport and deposition of microplastics under varying environmental conditions (Dispersion, 2021). Converting downwind measurements to µg/m³ ensures consistency between observed and modelled concentrations, enabling more accurate comparison of these models. Without this conversion, the models' ability to predict microplastic dispersion and environmental impact remains limited.

5.4.1 Theoretical Framework

The mass of microplastic particles was estimated using a multi-step process. The mass was calculated following the framework established in the earlier studies (Oladejo, 2017). First, the filter paper was weighed to determine its initial weight. After experimentation, the filter paper with the extracted microplastic particles was weighed again. The initial weight of the filter paper was then subtracted from the combined weight of the filter paper and microplastics to determine the accurate mass of the microplastics (Seghers et al., 2022). The number of particles was counted using the ImageJ software (S. Chen et al., 2021). Following this, the mass fraction of each particle was calculated as:

$$\text{Mass per Shape (m)} = \frac{\text{Number of particles in a shape}}{\text{Total Number of particles}} \times \text{total mass of microplastics} -$$

—eqn 5.1

The total mass of microplastics = microplastic mass measured at the downwind location – mass of microplastic background sample.

The average mass flow rate of microplastic particles was then calculated using the formula:

$$\text{Average Mass } (\mu\text{g}/\text{m}^3) = \frac{\text{Mass per shape (m)}}{\text{Sampler Flow rate (Q)}} \text{ --- eqn 5.2}$$

Where m represents the mass of microplastics per shape, Q represents the volume flow rate of air. Additionally, the volume flow rate $Q \text{ (m}^3\text{)} = \text{sample duration (8 hours)} \times \text{sampler flow rate (0.3m}^3\text{/hour)} = 2.4\text{m}^3$.

5.4.2 Dispersion Modelling in Microplastic Research

Dispersion modelling is a critical tool for understanding the transport and distribution of airborne microplastics (AMP) in the atmosphere (Martina & Trini Castelli, 2023). It provides insights into how microplastic particles emitted from sources such as waste disposal sites are carried by the wind, dispersed across distances, and eventually deposited on various surfaces (Kannankai & Devipriya, 2024). Modelling these processes is essential for assessing the impacts of microplastics on the environment and human health, especially in urban and industrial areas where emissions are significant (Uzun et al., 2022). The predictive capabilities of dispersion models help identify microplastic hotspots, provide insights into how seasonal variations affect transport, and evaluate the effectiveness of mitigation strategies (Pilechi et al., 2022). These models also help quantify the downwind concentrations of microplastics, which is crucial for validating empirical measurements.

For this study, **SCREEN3** was selected as a foundational dispersion modelling tool. SCREEN3 is a steady-state Gaussian plume model developed by the U.S. Environmental Protection Agency (EPA), designed to estimate pollutant concentrations from point, area, and volume sources (Zhong et al., 2011). Its strength lies in its simplicity and efficiency, making it particularly suitable for studies conducted in resource-limited settings or where detailed emission inventories and meteorological datasets are unavailable. The model's ease of use and low computational demand align with the current study objective of generating first-order estimates of microplastic transport from an open dumpsite in Nigeria.

Although SCREEN3 was originally developed for conventional air pollutants such as sulphur dioxide or particulate matter, it can be adapted to simulate airborne

microplastics by converting measured concentrations into mass-based units ($\mu\text{g}/\text{m}^3$), as demonstrated in Section 5.1.4. This enables airborne microplastics to be treated similarly to coarse particulates (PM_{10}), allowing basic simulations of their atmospheric dispersion. Furthermore, SCREEN3 can be customised to incorporate site-specific parameters such as emission height, wind speed, and atmospheric stability classes. These features make it a suitable choice for modelling emissions from a designated source area, such as a dumpsite (Taha et al., 2007).

While SCREEN3 offers a useful starting point for exploring AMP dispersion, it should be viewed within its limitations. The model assumes flat terrain and steady-state conditions, and it does not account for complex factors such as dry or wet deposition, chemical transformation, or long-range transport influenced by regional meteorological dynamics. More advanced models, such as ADMS, AERMOD, or CALPUFF, offer higher-resolution simulations and are widely utilised in regulatory and research contexts (Haq et al., 2019; Mohan et al., 2011). However, these models demand extensive input data and modelling expertise that may not yet be readily available for microplastic emissions, especially in developing regions.

On the other hand, Lagrangian models such as HYSPLIT and FLEXPART have recently been applied to microplastic research (Q. Chen et al., 2023; Evangelou et al., 2024; Martina & Trini Castelli, 2023; Xiao et al., 2023). However, these models are designed for long-range transport and require extensive datasets and detailed meteorological data, making them less suitable for this study. The lack of access to such extensive meteorological datasets in many regions, particularly Nigeria, hinders their precise application (Dinku, 2019; Lawal et al., 2021). Nevertheless, due to the localised nature of this research study area, SCREEN3 was selected as the most suitable option for this investigation, leveraging its capabilities for simulating near-field dispersion. As such, SCREEN3 provides a practical starting point for microplastic dispersion modelling, with the potential for integrating more sophisticated models as data availability and research capabilities grow.

5.5 SCREEN3 Model Interface

Microplastic pollution has become a pressing environmental concern, and simple modelling tools like SCREEN3 could play a crucial role in understanding and addressing this issue in local studies. SCREEN3 has not been specifically applied in microplastic research, but it may have potential applications in microplastic studies.

These include modelling the transport and distribution of microplastics in different environments, assessing the environmental and health impacts of microplastic pollution, and providing evidence-based support.

The SCREEN3 application software comes in two packages (Cristiane et al., 2024). These include the DOS prompt and Windows applications software. The EPA provided the DOS prompt as the model base (EPA, 1995). In contrast, several graphic user interface (GUI) providers, including Providence, Lakes, and Breezes, offer enhanced interfaces for SCREEN3, each with distinct features and benefits. The SCREEN3 Graphic User Interface (GUI) could greatly boost microplastic research. These include increased efficiency through simplified data input processes that save time and reduce errors, and intuitive visualisations that facilitate understanding complex model outputs. The SCREEN3 model can be accessed through various paid and free services, including the EPA and the Providence websites. This research employed the Providence GUI version, which offers an intuitive interface compared to the EPA's DOS prompt (EPA, 1995). The Providence GUI was chosen because it is free software, making it an attractive option for researchers with limited funding. Particularly in regions with limited resources, such as Africa, the accessibility of free software packages like Providence's GUI could facilitate critical research. By leveraging this improved GUI, researchers can explore new opportunities to understand microplastic pollution and contribute to a better grasp of this complex environmental challenge. This research examines the adaptation and application of SCREEN3 for microplastic studies, utilising the enhanced capabilities of the GUI. It aims to incorporate microplastic-specific processes and validate the model using ground data.

5.5.1 SCREEN3 Model Input and Parameterisation

The SCREEN3 model, developed by the United States Environmental Protection Agency (USEPA), provides a simplified approach for estimating maximum pollutant concentrations from point, area, or volume sources. The model interface is structured into three core menus that guide users through input data entry and parameter selection. These menus are:

5.5.2 Options Menu

The Options Menu enables users to establish the basic modelling configuration. This includes six different sub-menus. These include the title bar in which the project name is inserted (Figure 5.2)

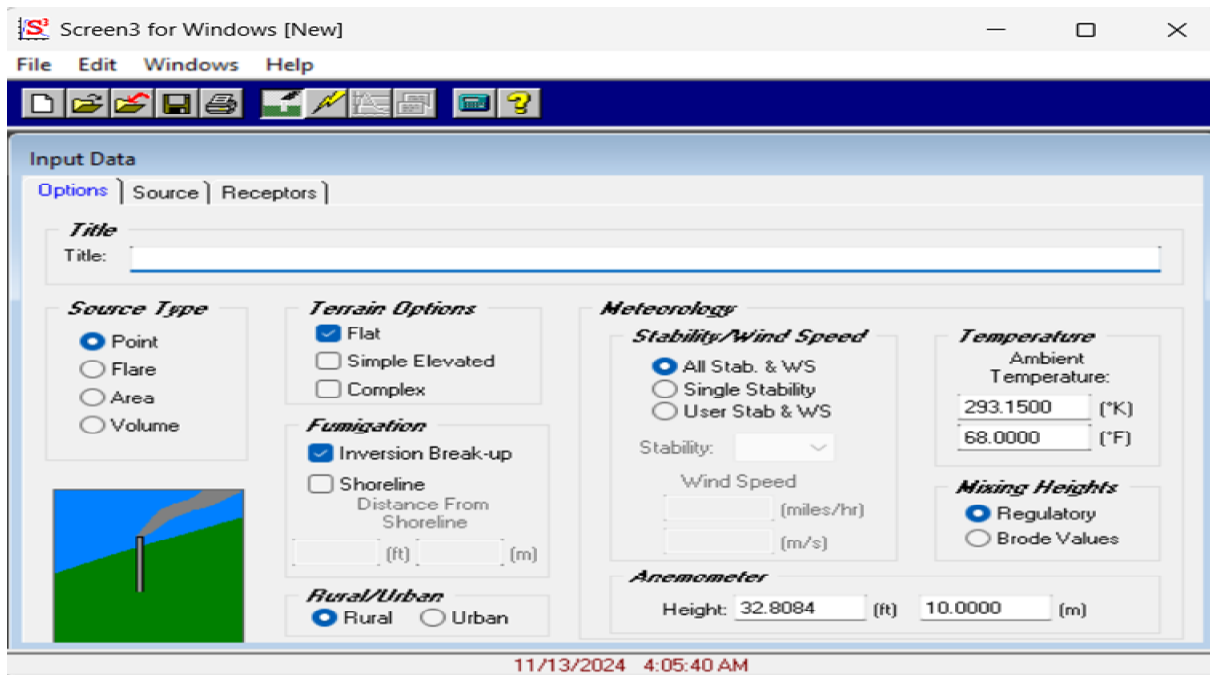


Figure 5. 2: SCREEN3 Option Menu

Source Type: The municipal solid waste dumping site was modelled as an "Area" source, accounting for the spatial distribution of emissions.

Terrain: The study area's flat topography was represented by selecting the "Flat" terrain option, consistent with previous research (Ado Umar et al., 2014). This assumption simplifies the modelling of atmospheric transport and dispersion.

Fumigation: Fumigation in the SCREEN3 model refers to fumigant movement and behaviour in agricultural settings. SCREEN3 is an environmental fate and transport model used to predict pesticide behaviour. The model's fumigation option was left unselected, as the focus was on airborne microplastics transport via atmospheric dispersion, rather than volatile organic compounds or fumigants.

Rural/Urban: The study area, Kano, was classified as an "Urban" environment, reflecting its densely populated city characteristics and urbanised landscape. This classification is a critical initial step, as it defines the environmental context for the entire study. By designating Kano as urban, the research acknowledges its key features: a high population density, extensive built-up areas, commercial and

residential structures, and a complex network of infrastructure like roads, electricity, and water systems.

Meteorology/Stability Class: In this study, two distinct seasons were considered: the dry season and the wet season. This approach was essential for testing the model's ability to accurately predict the transport and fate of airborne MP under different climatic conditions.

For the **dry season (February–March 2021)** of this study, the experiments conducted from morning to afternoon experienced moderate to high wind speeds (4.2–6.8 m/s) and low humidity (10–19%). During this period, strong solar radiation and elevated temperatures support well-mixed atmospheric conditions, aligning with **Stability Class C** (slightly unstable).

In the **wet season (July–August 2021)**, lower wind speeds (2.8–4.0 m/s) and high humidity (75–82%) prevail. Although these conditions typically favour stable atmospheric layers, daytime solar heating from morning to afternoon promotes neutral conditions, making **Stability Class D** appropriate for the experimental timeframe.

Wind Direction: During the dry season, the prevailing wind direction was E/NE (east-northeast), with an average of 45°, contributing to pollutant dispersion in the north-eastern direction. In the wet season, winds predominantly came from the WSW (west-southwest), with an average angle of 225°, shifting dispersion patterns towards the southeast.

Wind Speed: The wind speed in the study area exhibits seasonal variations. During the dry season, wind speeds range from 4.2 to 6.8 m/s, with an average of 6.0 m/s. In contrast, the wet season experiences lower wind speeds, ranging from 2.8 to 4 m/s, with an average of 3.5 m/s. The overall average wind speed is 4.9 m/s, with prevailing wind directions shifting between E/NE (45%) during the dry season and WSW (55%) during the wet season.

Anemometer Height: The anemometer height was set to 1.6 meters, similar to the sampler height, to provide representative wind conditions for dispersion modelling. This alignment is crucial because it allows for the direct correlation of two distinct variables: wind speed and direction, with the concentration and distribution of airborne microplastics. By collecting both sets of data at a similar height, the study can more accurately model how wind influences the transport and fate of these particles in the immediate environment.

Temperature: During the dry season, temperatures range from 19.1°C to 32.0°C, with an average of 21.0°C. Consequently, 21.0°C represents the dry season. In contrast, the wet season experiences warmer temperatures, spanning from 25°C to 38°C, with an average of 34.0°C. The overall average temperature is 31.3°C. The temperature variations may affect dispersion, degradation, and atmospheric stability, underscoring the need to account for temperature fluctuations in modelling efforts.

The source menu consists of the Area Source, Cavity and Building options. The cavity and building preferences are inactive due to the selection made on the Options Menu tab of the SCREEN3 model (Figure 5.3). Selecting "Area Source" on the Options Menu tab of the SCREEN3 model automatically makes the Cavity and Building options inactive on the Source Menu.

SCREEN3 for Windows [New]

File Edit Windows Help

Input Data

Options | **Source** | Receptors

Area Source

English Metric

Emission Rate: (lb/hr) (g/s)

Release Height: (ft) (m)

Longer Side Length: (ft) (m)

Shorter Side Length: (ft) (m)

Note: Emission Rate (g/s) will be divided by the area (Longer Side Length times Shorter Side Length) and written to the model as (g/s/m2).

Building

Height: (ft) (m)

Length: (ft) (m)

Width: (ft) (m)

Cavity

☒ Regulatory Default

☐ Schulman-Scire -- Stack not on structure

☐ Schulman-Scire -- Stack on structure

☐ Print values for all wind speeds

Ratio of distance of stack from the center of structure to side length

D/L for long side:

D/L for short side:

Example: $D/L \text{ (long)} = 35/100 = 0.35$
 $D/L \text{ (short)} = 15/50 = 0.30$

Diagram: A rectangular area source with dimensions 100m by 50m. A point source is located at a height of 15m from the center of the rectangle. The distance from the point source to the long side is 35m, and the distance to the short side is 15m.

11/13/2024 6:40:21 AM

Figure 5. 3: SCREEN3 Source Menu

Area: The disposal site covers a total area of 23,759.56 square meters. For modelling purposes, the site was approximated as a rectangle, with a characteristic length of 165.15 meters and a width of 143.87 meters. While the actual shape of the dump site is likely irregular, this rectangular simplification was applied to meet the input requirements of the SCREEN3 model, which assumes simplified geometric source configurations. This compromise allows for consistent calculations and dispersion simulations, providing a practical and reproducible basis for further analysis while

recognising that more advanced models may accommodate complex source geometries.

Release Height: The release height of 1.6 meters is selected to simulate emissions at breathing level, reflecting the height at which humans inhale pollutants (Cristiane Thé & Michael Johnson, 2024; EPA, 2024). This enables an accurate assessment of potential health risks associated with the disposal site.

Although microplastic emissions from open dump sites primarily originate at ground level, SCREEN3 was configured to use the sampling height (1.6 m) as the effective release height, aligning with the height at which airborne concentrations were measured. This assumption was adopted to maintain consistency between observed field data and model inputs, recognising that SCREEN3 requires a defined release height. However, given that microplastic resuspension is strongly influenced by near-surface turbulence, future modelling could incorporate a ground-level area source configuration or sensitivity analysis to evaluate the influence of different release heights on dispersion outcomes

5.5.4 Receptors Menu

The Receptors Menu provides options for configuring downwind measurement distances. Users can choose between two primary options (Figure 5.4).

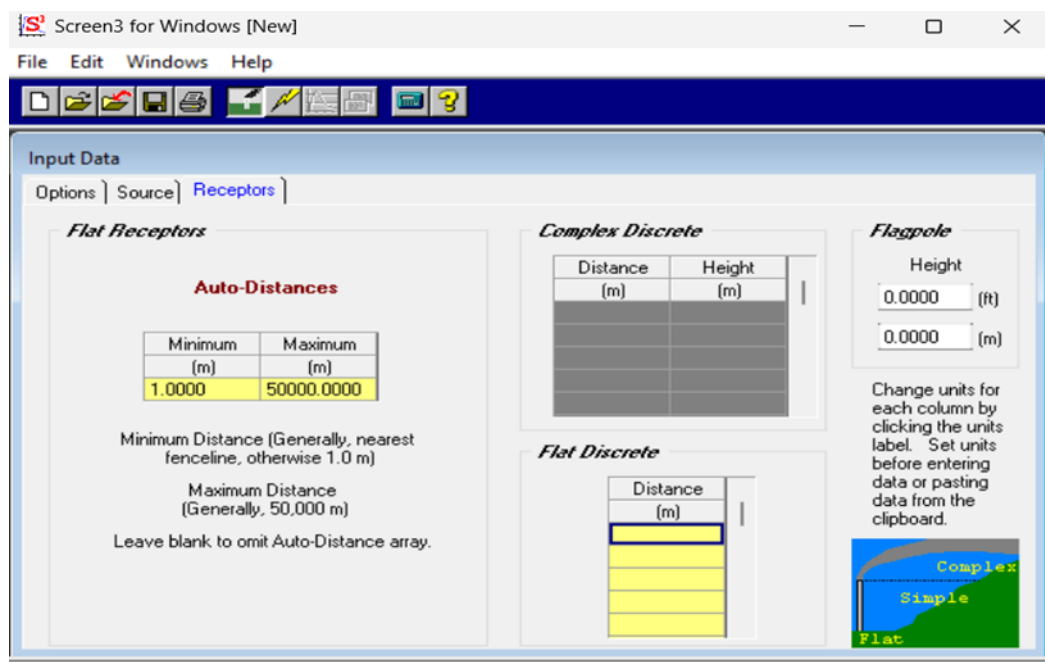


Figure 5. 4: Receptors Menu

Auto-Distance, which automatically calculates receptor distances based on model settings, or User-Defined Distance, allowing customised receptor placement. Additional settings include Complex Discrete receptor placement for nuanced simulations and Flagpole Height adjustment. These configurations enable users to define receptor locations for capturing downwind concentrations, studying factors such as downwind direction, distance intervals, and receptor density.

5.5.5 SCREEN3 Model Input Data

This study investigates the dispersion of airborne microplastics (AMPs) under two distinct seasonal conditions, the dry and wet seasons, which are characterised by contrasting meteorological dynamics. Accurately representing these conditions in the dispersion model requires season-specific input parameters, such as wind speed, direction, ambient temperature, and atmospheric stability (Table 5.1).

Table 5. 1: Model Input Data for Dry and Wet Seasons

Parameter	Dry Season	Wet Season	Units	Notes
Source Height	1.60	1.60	M	Height of the emission source.
Emission Factor (Fibres)	6.13×10^{-5}	5.68×10^{-5}	g/s	Emission factor for fibres.
Emission Factor (Fragments)	1.88×10^{-4}	1.70×10^{-4}	g/s	Emission factor for fragments.
Emission Factor (Films)	6.72×10^{-6}	5.57×10^{-6}	g/s	Emission factor for films.
Average Wind Speed	6.00	3.50	m/s	From site-specific data.
Wind Direction	E/NE (45°)	WSW (225°)	°	Predominant wind direction.
Temperature	21.00	34.00	°C	Average temperature.
Humidity	14.50	80.50	%	Seasonal average humidity.
Stability Class	C	D	N/A	Based on atmospheric conditions.
Downwind Distances	10, 50, 100, 200, 300, 500, 1000	10, 50, 100, 200, 300, 500, 1000	M	Sampling points for model comparison.
Terrain Type	Flat	Flat	N/A	Flat, urban terrain.
Measurement Duration	8.00	8.00	Hours	Duration for averaging concentration.

These variables significantly influence the behaviour and transport of microplastics in the atmosphere. The spatial variability of AMP dispersion and downwind points is strategically selected across three zones based on distances. The near-field zone (10 m, 50 m) represents immediate surroundings and potential hotspots. The mid-field zone (100 m, 200 m) captures intermediate-scale dispersion patterns. The far-field zone (300 m, 500 m, and 1000 m) assesses long-range transport and deposition (Nolan & Amiaqm, 2020).

These downwind points facilitate the evaluation of AMP concentrations and deposition patterns at varying distances from the source. This analysis provides valuable insights into spatial distribution, seasonal influences, and potential exposure risks for nearby populations.

Utilising the emission rates from Chapter 4 and carefully selected downwind locations, this study provides a thorough understanding of AMP dispersion during dry and wet season conditions.

5.5.6 Evaluation of Model Performance

The performance of SCREEN3 was evaluated by comparing its simulated microplastic concentrations ($\mu\text{g}/\text{m}^3$) with 8-hour in-situ downwind measurements taken during the dry and wet seasons. Data were collected at multiple downwind sites from the waste disposal site.

The accuracy of the model was evaluated using a simple statistical analysis and correlation methods. The correlation results provided insights into the model's ability to replicate real-world dispersion patterns, with discrepancies analysed to refine model inputs.

5.6 Microplastic Exposure Risks Framework for Municipal Solid Waste

In municipal solid waste (MSW) environments, where improper disposal and open burning are common, assessing the risk levels of microplastics (MPs) is essential to mitigate pollution and its adverse effects. Exposure to microplastics has been linked to severe health consequences, including cancer and genetic problems. The **Polymer Hazard Index** is a tool used to assess the ecological risk of microplastic pollution. It's a numerical index that combines two key factors: the **toxicity of the polymer type** and its **relative abundance** in a given environment (Esmailbeigi et al., 2023).

The index works on the principle that not all microplastics pose the same level of risk. The core of the index is based on a **hazard score** assigned to different types of polymers. This score reflects the inherent chemical toxicity of the polymer and its additives, which can cause harm to ecosystems

According to recent research, Polyvinyl Chloride (PVC), Polyethene (PE), and Polystyrene (PS) have been identified as high-priority pollutants (Wan et al., 2024). Furthermore, based on the Polymer Hazard Index, PVC is classified as having the highest risk score and Polypropylene (PP) is considered to have the lowest risk score among the assessed polymers (Table 5.2).

Table 5. 2: Polymer Hazard Score (S_n)

Polymer	Hazard Score (S_n)
Polyvinyl Chloride (PVC)	5001
Polyethylene (PE)	11
Polystyrene (PS)	30
Nylon (PA)	47
Polypropylene (PP)	1
Polyethene-Terephthalate (PET)	4

The polymer hazard index (S_n) is a semi-quantitative score derived from published toxicological studies that assess the potential harm associated with individual polymer types (Tajwar et al., 2023). These factors include persistence, bioaccumulation, toxicity, and the use of chemical additives. For example, polymers like PVC (polyvinyl chloride) are typically assigned higher hazard scores due to the presence of plasticisers and heavy metals, as polymers such as PE (polyethene) may receive lower scores based on lower inherent toxicity and fewer additives (Z. Yuan et al., 2022). These scores are adopted from previous risk-ranking studies and provide a simplified, comparative means to incorporate polymer-specific toxicity into the overall exposure risk calculation.

5.6.1 Risk Assessment Framework

The framework for estimating microplastic exposure risk presented in Equation 5.3 was developed based on the foundational work of Li et al. (2020) and Zhao et al.

(2022a, b). These studies established key perspectives on microplastic exposure risk assessment, which were built upon to create a comprehensive framework for estimating exposure risks. The exposure risk (H) is represented as:

$$H = P_n \times S_n \text{ --- eqn 5.3}$$

Specifically, the framework leverages the concepts and methodologies developed by Li et al. (2020) and Zhao et al. (2022a, b) to quantify exposure risks associated with microplastic pollution. The parameter P_n is the polymer concentration, derived from the SCREEN3 model output, which simulates the dispersion of pollutants in the environment. This approach is consistent with the modelling approaches employed in previous studies.

The hazard score S_n assigned to each polymer type is based on its toxicological properties, as outlined in Table 5.3. This scoring system is informed by the hazard assessment frameworks developed by Li et al. (2020) and Zhao et al. (2022a, b), which provide a basis for evaluating the potential risks posed by different microplastic types.

Table 5. 3: MP Exposure Risk

Risk Level	Risk Index Range	Description
Level I	1-10	Low Risk
Level II	11-100	Medium Risk
Level III	101-1000	High Risk
Level IV	>1000	Danger Risk

The resulting risk index (H) is then classified into four levels of concern, as shown in Table 5.3, ranging from Level I (low risk) to Level IV (dangerous risk). This framework enables site-specific estimation of microplastic exposure risk by linking ambient concentration with polymer hazard, offering a basis for prioritising management actions and informing public health assessments.

By integrating these components, the framework presented in Equation 5.3 provides a quantitative approach to estimating microplastic exposure risks. The multiplication of P_n and S_n yields an exposure risk estimate (H) that accounts for both the

concentration and hazard potential of the polymers. This framework can be used to inform risk management strategies and policy decisions aimed at mitigating the impacts of microplastic pollution.

5.7 Results and Discussion

This study utilises WRPLOT software to generate wind rose plots, visualising seasonal wind variations and informing dispersion modelling for atmospheric microplastic (AMP) transport assessments (Kannankai & Devipriya, 2024).

During the dry season (Figure 5.5a), Wind-rose plots show a predominance of east-northeast (E/NE) winds, reflecting conditions that enhance AMP transport over longer distances due to higher average wind speeds (Bullard et al., 2021). The wind speeds range between 4.2 –6.8 m/s, with an average of 6.0 m/s. The prevailing winds predominantly come from the east-northeast (E/NE), representing 45% of the time, with an associated wind angle of 45°. Temperatures during this season are cooler, ranging from 19.1–32.0°C (average 21.0°C), with low humidity levels of 10.0 –19.0% (average 14.5%). In the wet season (Figure 5.5b), the wind rose shifts toward west-southwest (WSW) winds, 55% of the time, with an angle of 225°. The wind is associated with lower speeds, ranging from 2.8 to 4.0 m/s, with an average of 3.5 m/s.

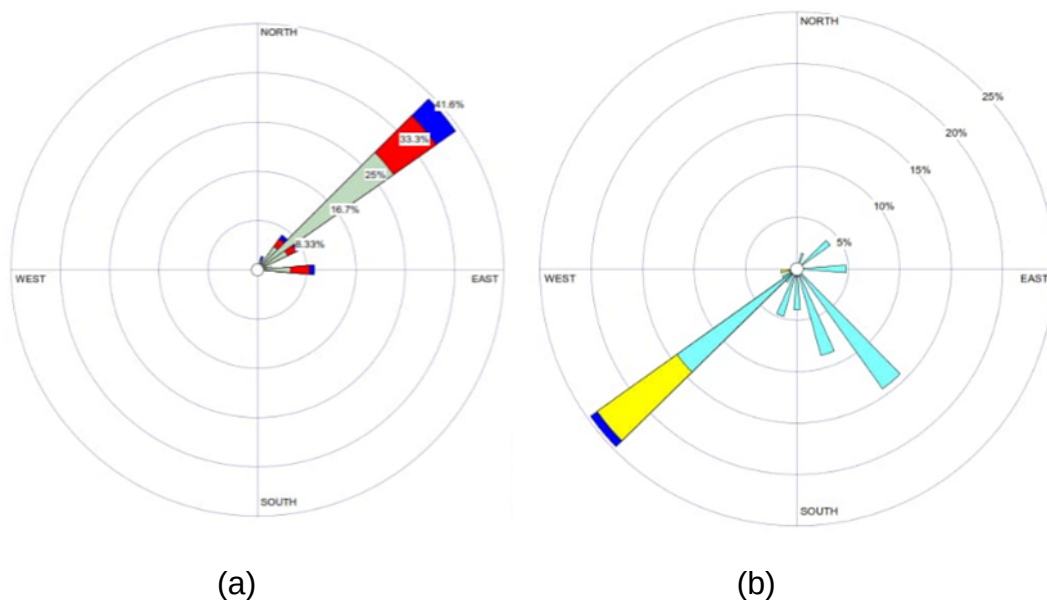


Figure 5. 5: Windrose Analysis during Dry and Wet Seasons (a) Dry Season (b) Wet Season

Key:  0.50- 2.80 m/s,  2.80- 3.00 m/s,  3.00 to 4.00 m/s, 4.00 to 5.00 m/s  and  5.00 to 6.80m/s.

These conditions may reduce transportation and increase local deposition due to rain washouts and raindrops (Abbasi, 2021; Lehmann et al., 2021). Temperatures are higher, ranging from 25–38°C (average 34.0°C), and humidity increases of 75.0–82.0% (average 80.5%). The wind rose patterns observed in this study are consistent with wind data reported in the study of air quality and pollutant emissions from a Nigerian cement plant (Adeniran et al., 2019).

Incorporating wind rose data into AMP dispersion models ensures a robust understanding of seasonal patterns, enabling more accurate predictions of transport and deposition (Kannankai & Devipriya, 2024). This visual tool also helps identify optimal locations for downwind monitoring and validate dispersion model assumptions based on prevailing wind directions.

5.7.1 Microplastics Downwind Concentration Profile

The SCREEN3 microplastic simulation at distances downwind of the waste disposal site reveals distinct seasonal patterns. During the dry season, distinct transport and deposition trends were observed for fibres, fragments, and films (Figure 5.6).

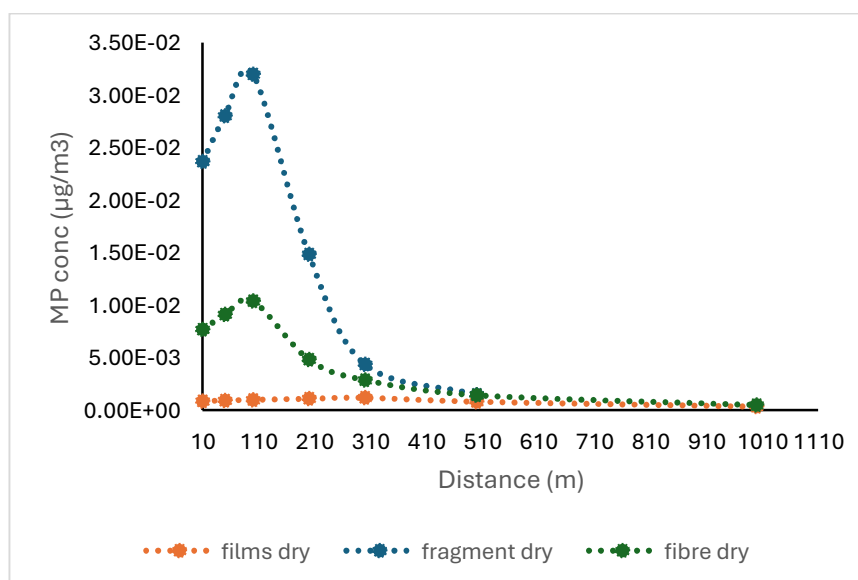


Figure 5. 6: Dry Season Microplastic Downwind Concentration Profiles

Downwind measurements were taken at offset distances ranging from 10 meters to 1000 meters from the midpoint of the north-eastern boundary of the disposal site area (P. Kumar et al., 2024). The peak concentrations occurred at mid-field distances of 100 m for fibres ($1.04 \times 10^{-2} \mu\text{g}/\text{m}^3$) and fragments ($3.20 \times 10^{-2} \mu\text{g}/\text{m}^3$), and 91 m for films ($1.17 \times 10^{-3} \mu\text{g}/\text{m}^3$).

Beyond these points, concentrations decreased sharply (310m), with significant reductions at field zones of 500 m and 1000 m. In contrast, the wet season reveals different concentration patterns due to altered meteorological conditions (Figure 5.7). Peak concentrations for all microplastic types occurred at midfield of 100 m for fibres ($2.28 \times 10^{-2} \mu\text{g}/\text{m}^3$), fragments ($6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$), and films ($2.41 \times 10^{-3} \mu\text{g}/\text{m}^3$). The concentrations declined steeply over shorter distances (210m) in the wet season.

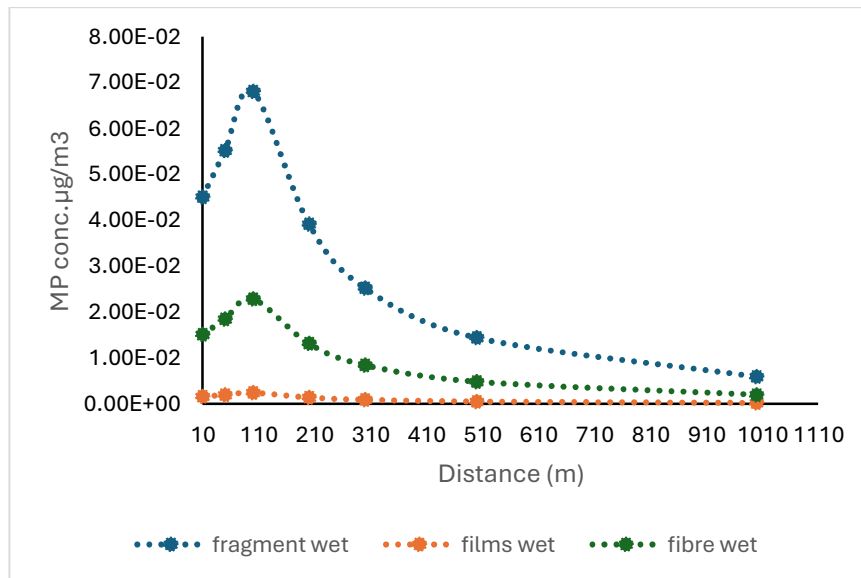


Figure 5. 7: Wet Season Microplastic Downwind Concentration Profiles

A comparison of the dry and wet seasons reveals notable differences in microplastic concentrations. For fibres, the peak concentration during the wet season ($2.28 \times 10^{-2} \mu\text{g}/\text{m}^3$) was approximately 2.2-fold higher than during the dry season ($1.04 \times 10^{-2} \mu\text{g}/\text{m}^3$). Similarly, fragments showed a 2.1-fold increase in the wet season ($6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$) compared to the dry season ($3.20 \times 10^{-2} \mu\text{g}/\text{m}^3$). Films exhibited a 2.1-fold increase in the wet season ($2.41 \times 10^{-3} \mu\text{g}/\text{m}^3$) compared to the dry season ($1.17 \times 10^{-3} \mu\text{g}/\text{m}^3$).

The observed higher concentrations during the wet season may result from various atmospheric factors. Stability class D, characteristic of the wet season, typically

corresponds to neutral atmospheric conditions that reduce vertical mixing (T. Hu & Yoshie, 2020). This limited mixing can trap microplastics near the surface, increasing ground-level concentrations (Heather Wylie, 2021). Additionally, a lower mixing height during the wet season (1120 m) restricts the vertical dispersion of microplastics, further contributing to higher concentrations near the source (Heather Wylie, 2021). Precipitation may also play a role by impeding downwind transport and causing microplastics to settle closer to their emission point (Lehmann et al., 2021). In the context of a disposal site, this means that instead of AMPs being carried over long distances to surrounding communities or ecosystems, a significant portion of the airborne microplastics will be deposited back onto the site or its immediate vicinity. This phenomenon concentrates the pollution near its source, potentially increasing the localised ecological and health risks for that area.

In contrast, the dry season's less stable atmospheric conditions (Stability Class C), higher wind speed (6 m/s), and greater mixing height (1920 m) likely promote broader dispersion, leading to more gradual declines (dilution) in concentration with distance from the source (Murthy et al., 2020). These observations are based on the outputs of the SCREEN3 dispersion model, a tool used in this study to estimate the transport and dispersion of airborne microplastics.

It is important to highlight that this is based on the available literature; no prior study has specifically investigated the effect of mixing height on airborne microplastics using methods comparable to those employed in this study. This gap in the research limits the empirical basis for the interpretations presented here. Further research and experimental validation are necessary to bridge this knowledge gap and refine the model's application to microplastic transport studies.

5.7.2 Microplastic Time-dependent Concentrations

The result of SCREEN3 modelling estimates the worst-case exposure scenario, assuming maximum 1-hour concentrations (Table 5.4). According to these estimates, the dry season fibre concentrations peaked at $1.07 \times 10^{-2} \mu\text{g}/\text{m}^3$, while fragment concentrations reached $3.30 \times 10^{-2} \mu\text{g}/\text{m}^3$. In contrast, the wet season saw substantially higher concentrations, with fibres reaching $2.28 \times 10^{-2} \mu\text{g}/\text{m}^3$ and fragments at $6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$. Film concentrations also showed seasonal variations, ranging from $1.23 \times 10^{-3} \mu\text{g}/\text{m}^3$ in the dry season to $2.41 \times 10^{-3} \mu\text{g}/\text{m}^3$ in the wet season.

Table 5. 4: Microplastic Time Dependent Concentrations

Type	Season	Distance Max (m)	1-hr Con.($\mu\text{g}/\text{m}^3$)	3-hr Con. ($\mu\text{g}/\text{m}^3$)	8-hr Con.($\mu\text{g}/\text{m}^3$)	24-hr Con.($\mu\text{g}/\text{m}^3$)	Annual Con.($\mu\text{g}/\text{m}^3$)
Fibre	Dry	107	1.07×10^{-2}	9.62×10^{-3}	7.48×10^{-3}	4.28×10^{-3}	8.55×10^{-4}
	Wet	100	2.28×10^{-2}	2.05×10^{-2}	1.60×10^{-2}	9.13×10^{-3}	1.83×10^{-3}
Fragment	Dry	107	3.30×10^{-2}	2.97×10^{-2}	2.31×10^{-2}	1.32×10^{-2}	2.64×10^{-3}
	Wet	100	6.81×10^{-2}	6.13×10^{-2}	4.76×10^{-2}	2.72×10^{-2}	5.45×10^{-3}
Films	Dry	107	1.23×10^{-3}	1.10×10^{-3}	8.59×10^{-4}	4.91×10^{-4}	9.82×10^{-5}
	Wet	100	2.41×10^{-3}	2.17×10^{-3}	1.69×10^{-3}	9.65×10^{-4}	1.93×10^{-4}

Microplastic concentrations during the wet season surpassed those in the dry season, with fragment levels reaching $6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$, the highest recorded amount. This highlights the significant risks associated with microplastic pollution. The peak one-hour (Worst-Case) concentrations were detected at distances between 100 and 107 meters downwind of the waste disposal site.

The duration of exposure was also a key factor, as concentrations significantly decreased over longer periods. These worst-case estimates highlight the potential for significant microplastic exposure near the waste disposal site. These study findings have implications for environmental and public health policy.

5.7.3 Comparison of Airborne Microplastic Concentrations to PM Standards

The modelled concentrations of airborne microplastics (AMPs), as estimated using SCREEN3 downwind of the disposal site, reveal distinct seasonal patterns but remain extremely low in mass concentration. The highest dry-season values recorded were $3.20 \times 10^{-2} \mu\text{g}/\text{m}^3$ for fragments and $1.04 \times 10^{-2} \mu\text{g}/\text{m}^3$ for fibres at 100 m downwind. Wet-season values were slightly higher, with fragments reaching $6.81 \times 10^{-2} \mu\text{g}/\text{m}^3$ (for details refer to Section 5.7.1), yet still well below established national thresholds for ambient particulate matter (Table 5.5).

For context, Nigeria's National Environmental Standards and Regulations Enforcement Agency (NESREA) sets 24-hour limits at $40 \mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$ and $150 \mu\text{g}/\text{m}^3$ for PM_{10} (FGN, 2021). Compared to these benchmarks, the AMP concentrations modelled in this study are several orders of magnitude lower

(approximately 3–4 orders), suggesting that microplastics, at least by mass, contribute negligibly to the overall particulate burden in the atmosphere (Table 5.5).

Table 5. 5: Comparison of Modelled AMPs with NESREA PM Standards

Comparison	AMP Value ($\mu\text{g}/\text{m}^3$)	PM Limit ($\mu\text{g}/\text{m}^3$)	Difference
Fragments (Wet Season) vs $\text{PM}_{2.5}$	6.81×10^{-2}	40	~588×
Fibres (Dry Season) vs PM_{10}	1.04×10^{-2}	150	~14,400×

However, unlike conventional PM, microplastics may pose distinct toxicological risks due to their polymer composition, surface chemistry, and potential to carry adsorbed pollutants. Therefore, while the mass-based concentrations remain low, their presence and persistence in inhalable air near open dump sites still present a concern, particularly for chronic exposure. These findings reinforce the need for dedicated monitoring of airborne microplastics and suggest that risk assessments should consider factors beyond mass concentration, such as particle type, morphology, and toxicity.

5.7.4 Exposure Risk Assessment

The exposure risk assessment was conducted based on the SCREEN3 simulation of the one-hour worst-case scenario. To account for climatic differences, the study simulated AMP dispersion during both the dry and wet seasons. This was critical because meteorological factors like wind and precipitation can significantly alter how microplastics are transported through the atmosphere. The dry seasons are associated with higher winds, are expected to show a greater potential for widespread dispersion, while the wet season, with its rain-induced deposition, would likely show a reduced transport distance.

Exposure durations ranged from acute (1-hour and 24-hour) to chronic (annual exposure). Risk indices were calculated to assess potential health impacts based on inhalation exposure, with values above 1 indicating risk. The exposure risk assessment revealed that fibre and film microplastics posed minimal health risks. Across all exposure durations and seasons, risk indices for these microplastics remained below 1 (Table 5.6).

Table 5. 6: Risk Assessment Levels and Exposure Duration of AMPs

Type	Season	Distance Max (m)	1-hr Exp ($\mu\text{g}/\text{m}^3$)	3-hr Exp ($\mu\text{g}/\text{m}^3$)	8-hr Exp ($\mu\text{g}/\text{m}^3$)	24-hr Expo ($\mu\text{g}/\text{m}^3$)	Annual Expo ($\mu\text{g}/\text{m}^3$)	Risk Level
Fibre	Dry	107	0.01	0.01	0.01	0.00	0.00	Level I
Fibre	Wet	100	0.02	0.02	0.02	0.01	0.00	Level I
Fragment	Dry	107	165.03	148.53	115.52	66.01	13.20	Level III
Fragment	Wet	100	340.57	306.56	238.05	136.03	27.26	Level III
Films	Dry	107	0.01	0.01	0.01	0.01	0.00	Level I
Films	Wet	100	0.03	0.02	0.02	0.01	0.00	Level I

This suggests that, under the simulated conditions, fibres and films are unlikely to result in significant health effects.

In contrast, fragment microplastics exhibited significantly higher risk indices, particularly during the wet season. At 100 metres downwind, wet season risk indices ranged from 136.03 $\mu\text{g}/\text{m}^3$ for 24-hour exposure to 340.57 $\mu\text{g}/\text{m}^3$ for 1-hour exposure. Even in the dry season, risk indices at 107 metres downwind ranged from 66.01 to 165.03 $\mu\text{g}/\text{m}^3$. Both scenarios categorise fragment microplastics as High Risk (Level III). The inhalation of airborne microplastics, particularly fragments, may pose potential health risks (Li et al., 2023). Fragments, characterised by irregular shapes and sharp edges, may cause mechanical irritation in the respiratory tract. Prolonged exposure could lead to:

Respiratory Inflammation: Microplastic particles may induce inflammatory responses in the airways, contributing to bronchitis or asthma exacerbation (Ding et al., 2023).

Cytotoxicity: Studies have demonstrated that microplastics can cause cellular damage, potentially compromising lung function over time.

Systemic Effects: Emerging evidence suggests that microplastics can translocate from the respiratory system into the bloodstream, posing risks to other organs, including the cardiovascular system (Hinata et al., 2023).

Oxidative Stress: The inhalation of microplastics may trigger oxidative stress, which has been linked to chronic diseases such as cancer and cardiovascular disorders (Liang et al., 2024). These health effects are particularly concerning for vulnerable populations.

5.7.5 Dispersion Patterns

The SCREEN3 model performance was evaluated against the ground measurements for MPs of fibres, fragments, and films downwind of the disposal site. The measured concentration of microplastics in MP/m³ at a ground distance was first converted to mass concentration (µg/m³) using equations 5.1 and 5.2 to facilitate comparison (Appendix C, Table 5.1A and Table 5.1B). The results demonstrate a clear trend of decreasing total microplastic (MP) number and mass concentrations (µg/m³) with increasing distance from the source, consistent with expected dispersion patterns. At 10 m from the source, the total mass concentration was 3.22×10^{-2} µg/m³, which was reduced to 5.9×10^{-3} µg/m³ at 1000 m during the dry season (Table 5.7).

In contrast, during the wet season, the microplastic concentration at 10 m was 1.51×10^{-2} µg/m³ and decreased to 8.08×10^{-3} µg/m³ at a distance of 1000 m. This trend highlights the dilution effect as particles disperse further into the atmosphere, with lower concentrations detected at greater distances. Such findings validate the anticipated relationship between source proximity and MP concentration, affirming the general principles of particle dispersion.

Table 5. 7: Dispersion Characteristics

season	Mass Concentration at 10m (µg/m ³)	Mass Concentration at 1000m (µg/m ³)	Concentration Difference (µg/m ³)	Dilution
Dry	3.22×10^{-2}	5.9×10^{-3}	2.66×10^{-2}	81.68%
Wet	1.51×10^{-2}	8.08×10^{-3}	6.52×10^{-3}	46.36%

The impact of wind dilution on microplastic dispersion was evident in both seasons. During the dry season, wind dilution (6 m/s) resulted in a pronounced decrease of approximately 81.68% in microplastic concentration as the distance from the source increased from 10 m to 1000 m. In contrast, with wind dilution at 3.5 m/s during the wet season, the microplastic concentration reduces by approximately 46.36%. These findings underscore the significance of wind dilution in mitigating microplastic dispersion, highlighting the influence of seasonal wind speed variations on the efficacy of this process.

5.7.6 Performance Comparison of Dry and Wet Season Dispersion Results

The absolute difference (%) between measured and modelled concentrations provides a straightforward and quantifiable metric to assess the performance, reliability, and model predictive accuracy (Oura et al., 2018). In this context, the accuracy of the SCREEN3 in predicting downwind airborne microplastic concentration from the disposal site under real-world environmental conditions was evaluated using the absolute difference. The absolute difference between the model-predicted concentrations and the ground-level concentrations measured downwind of the disposal site was calculated for the dry and wet seasons (Table 5.8). This metric offers insight into how well the model approximates actual field observations and helps identify the extent to which the simplified assumptions inherent in Gaussian dispersion models align with observed atmospheric behaviour across varying distances and seasonal conditions.

Notably, at 10 to 100m near-field distances, the dry and wet seasons show strong agreement between measured and modelled concentrations (Table 5.8). In the dry season, discrepancies are minimal, such as at 10 m, where the absolute difference is 4.41%, and at 100 m, where the difference is 4.96%. Conversely, during the wet season, the model performs well at these distances, with absolute differences of 2% at 10 m and 4% at 100 m.

These results reflect the model's capability to predict near-field dispersion accurately in both seasons, likely due to the relatively straightforward transport mechanisms closer to the source. In the mid-field range (200–500 m), the performance of the SCREEN3 model varies slightly between the two seasons. For the dry season,

discrepancies increase moderately, such as at 200 m, where the difference is 8.40%, and at 500 m, where it reaches 24.98%.

Table 5. 8: Model Performance in Dry and Wet Season

Distance (m)	Season	Measured Concentration ($\mu\text{g}/\text{m}^3$)	Modelled Concentration ($\mu\text{g}/\text{m}^3$)	Absolute Difference (%)
10	Dry	3.37E-02	3.22E-02	4.00
	Wet	6.04E-02	6.18E-02	2.00
50	Dry	6.22E-02	6.52E-02	5.00
	Wet	7.92E-02	7.56E-02	5.00
100	Dry	4.58E-02	4.36E-02	5.00
	Wet	8.96E-02	9.33E-02	4.00
200	Dry	3.25E-02	2.98E-02	8.00
	Wet	5.17E-02	5.37E-02	4.00
300	Dry	1.42E-02	1.76E-02	24.00
	Wet	1.89E-02	1.76E-02	7.00
500	Dry	7.92E-03	5.94E-03	25.00
	Wet	1.79E-02	1.97E-02	10.00
1000	Dry	1.60E-02	1.19E-02	26.00
	Wet	6.77E-03	8.08E-03	19.00

This could be due to the dampening effect of higher moisture levels, which may reduce the complexity of particle transport compared to the turbulence observed during the dry season (Fatahi et al., 2021).

At extended distances (1000 m), larger discrepancies are observed in both seasons, reflecting the limitations of the SCREEN3 model in predicting long-range dispersion. In the dry season, the difference at 1000 m is 25.83%, while in the wet season, it is slightly lower at 19%. This suggests that environmental factors, such as reduced wind speeds and increased humidity during the wet season, may slow particle dispersion, leading to a better match between modelled and measured concentrations compared to the dry season, where faster wind speeds and greater turbulence introduce additional variability (Snoun et al., 2023b).

The dry season is characterised by higher wind speeds, which enhance the transport of MPs but also introduce greater turbulence, leading to increased discrepancies between measured and modelled values at medium and long distances. Conversely, the wet season features reduced wind speeds and higher moisture levels, which

promote particle settling and dampen dispersion, resulting in slightly lower concentrations overall. For instance, at 10 m, the total measured concentration is $3.37 \times 10^{-2} \mu\text{g}/\text{m}^3$ in the dry season compared to $6.04 \times 10^{-2} \mu\text{g}/\text{m}^3$ in the wet season. These differences highlight the significant role of environmental conditions in influencing MP behaviour and dispersion.

The SCREEN3 model's performance was further evaluated for both wet and dry seasons, coefficient of determination (R^2), and Root Mean Squared Error (RMSE) (Table 5.9). The model demonstrated exceptional performance, as evidenced by its high coefficient of determination (R^2) values. Specifically, the model achieved an R^2 of 0.958 for wet atmospheric conditions, indicating that approximately 95.8% of the variance in the observed data can be explained by the model's predictions. Under dry atmospheric conditions, the performance was even more robust, with an R^2 value of 0.98, suggesting that 98% of the variability in the dry season data was accounted for by the model.

Complementing these strong R^2 values, the model also exhibited remarkably low root mean square error (RMSE) values. For the wet season, the RMSE was found to be $1.51 \times 10^{-5} \mu\text{g}/\text{m}^3$, while for the dry season, it was $2.02 \times 10^{-3} \mu\text{g}/\text{m}^3$. These low RMSE values signify that the model's predictions are, on average, very close to the actual observed concentrations, further underscoring its accuracy and reliability across different environmental conditions.

Table 5. 9: SCREEN3 Performance Metrics

Metric	Dry Season	Wet Season
R^2	0.96	0.98
RMSE ($\mu\text{g}/\text{m}^3$)	1.51×10^{-5}	1.8×10^{-3}

5.7.7 Comparison with other studies

A comparative analysis of this study with existing research on microplastic air dispersion modelling is challenging due to methodological inconsistencies and variations in the characterisation of emission sources. While several studies have

employed Lagrangian-based models, such as HYSPLIT and FLEXPART, to examine airborne microplastics from municipal solid waste disposal sites and other anthropogenic sources, significant knowledge gaps remain in evaluating model performance (Evangelou et al., 2024; Tatsii et al., 2024; Xiao et al., 2023). Few studies have acknowledged these challenges, highlighting the urgent need for further research to enhance the accuracy and reliability of microplastic air dispersion models (Svensson & Andersson-Sköld, 2021).

In this research, the SCREEN3 Gaussian model was employed for the first time to simulate the transport and fate of airborne microplastics. The findings were further contextualised by comparisons with studies on other pollutants, such as particulate matter (PM), offering analogous insights into pollutant dispersion dynamics.

For instance, studies on PM dispersion from a cement plant reported maximum particle concentrations within a radius of 1600–1800 m in the downwind direction, where the modelled concentrations showed a strong correlation to in-situ measured data ($R^2=0.98$). The cement plant's emissions were released from different stack heights ranging from 54 to 90 m, facilitating greater vertical dispersion before deposition (Khaniabadi et al., 2018).

In contrast, this study identified the maximum airborne microplastic concentrations at a significantly shorter distance of 100–107 m from the source, primarily due to the microplastic emissions' 1.6 m release height. This substantial difference in release heights explains the observed disparity in dispersion ranges. PM from the cement plant, emitted from higher stacks, undergoes prolonged atmospheric transport and dilution, allowing it to disperse farther before reaching peak concentrations. Conversely, the lower emission height in this study limits vertical dispersion, leading to higher ground-level concentrations closer to the source. The shorter dispersion range and lower emission height in this study highlight the unique behaviour of microplastics compared to PM. Microplastics' size variations and diverse aerodynamic properties result in quicker deposition and peak concentrations closer to the source. The SCREEN3 model demonstrated high accuracy in predicting microplastic concentrations within the range of 0.96 to 0.98 in dry and wet seasons, respectively. While the cement plant study achieved an exceptionally high correlation to measured data ($R^2=0.98$), the SCREEN3 model findings in the study equally demonstrated reasonable estimates with smaller errors. Further empirical validation is needed to confirm the accuracy of microplastic dispersion models under varying conditions.

5.8 Conclusions

This chapter employs the SCREEN3 model to assess the transport, fate, and exposure risk of airborne microplastic (AMP) emissions from an open waste disposal site. While the modelled concentrations are relatively low compared to national air quality standards for PM_{2.5} and PM₁₀, the findings are significant as they demonstrate that microplastics are present in ambient air and can be transported downwind from unregulated disposal areas. This has profound implications for environmental monitoring and public health risk assessment, particularly in developing regions where open dumping is prevalent and regulatory frameworks are limited.

Although no direct health outcome data were incorporated into this study, the exposure risk framework draws on existing toxicological hazard scores (Li et al., 2020; Zhao et al., 2022) to estimate polymer-specific risks. These estimates suggest a credible potential for inhalation exposure in nearby communities, warranting further investigation.

Key findings indicate that SCREEN3 effectively simulated AMP dispersion, with peak concentrations ranging from 10 m to 210 m downwind of the source. Wet season concentrations were up to 2.2 times higher than those recorded during the dry season, with fragment microplastics consistently showing the highest concentrations and corresponding risk levels. Risk assessment revealed that fragments exceeded Level III (high risk) in both seasons, indicating a potential public health concern. Model validation also demonstrated strong performance, with 80–98% accuracy and a robust correlation with ground measurements ($p = 0.967$, $p < 0.001$).

These findings should be interpreted as a foundational step, establishing a framework for future research into AMP exposure pathways. This study suggested the need for environmental regulation and targeted monitoring, rather than drawing definitive public health conclusions and Limitations.

5.8.1 Key Findings and Implications

Elevated Concentrations near Source: The model findings suggest high concentrations of AMPs within a 10–1000-meter corridor from the source, with peak values at around 100 meters in both dry and wet seasons.

Seasonal Variation: Wet season concentrations of all microplastic types were significantly higher than in the dry season (a 2.1-2.2-fold increase). This was a notable finding, as it's typically assumed that higher winds in the dry season would lead to greater concentrations. This study reveals a unique mechanism involving raindrop agitation and limited vertical dispersion during the wet season.

Varying Risks: While overall microplastic concentrations were lower than typical air pollution standards ($PM_{2.5}$ and PM_{10}), the study highlighted that their diverse properties make them more hazardous. The risk assessment revealed that **fragment microplastics** posed a high risk (Level III) in both seasons, with hazard index values of 165.03 (dry) and 340.57 (wet). Fibres and films had lower risk levels.

Model Performance: The SCREEN3 model proved highly effective, with an **80-96% accuracy** and a strong correlation with real-world measurements. Its simplicity and minimal data requirements make it a valuable, cost-effective tool for monitoring and managing AMPs in developing countries.

Unlike conventional air pollutants like $PM_{2.5}$ and PM_{10} . Airborne microplastics (AMPs) are highly varied in their characteristics, including their **shape**, **density**, and **polymer type**. This heterogeneity is crucial because it affects how they travel through the air, their potential to be inhaled, and their overall toxicity.

The study found that while AMP concentrations were **below existing PM standards**, assessing their risk requires a different approach. Because of their unique properties like persistence, bioaccumulation, and specific toxicity from additives, AMPs cannot be treated as a single type of pollutant. The study argues for new policies and monitoring frameworks that specifically address the distinct nature of microplastic pollution

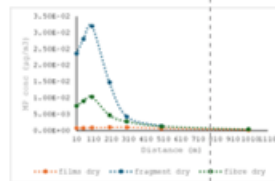
Chapter 6.0: Conclusion, Implications, Contributions, and Future Directions

This study identified and characterised the release of airborne microplastics from a typical municipal solid waste facility in Nigeria and developed a framework for using a dispersion model to assess exposure risk. The objectives of this research were successfully realised through a combination of exhaustive literature reviews, a series of fieldwork, and laboratory experiments. The principal findings and conclusions drawn from this comprehensive study are summarised below.

6.1 Key Findings and Conclusions

This study presents a novel, integrated approach for quantifying airborne microplastic (AMP) emissions from municipal solid waste (MSW) disposal sites. The study offers one of the first site-specific frameworks that links field data, laboratory analysis, and atmospheric dispersion modelling. Several prior studies have addressed airborne microplastics in urban or remote environments (Evangelidou et al., 2020c; Trainic et al., 2020), with few focusing on municipal solid waste dumpsites (Petrović et al., 2023; X. Shi et al., 2023). However, studies of airborne microplastics are largely overlooked in many regions. This research gap is pronounced particularly in developing countries like Nigeria, where open dumping is prevalent. To address this, the study developed a framework (Figure 6.1) that consolidates previously isolated elements, such as flux calculation, field sampling, and risk characterisation, into a single cohesive structure tailored for AMP studies at MSW sites. Building on the research by Hu et al. (2022) and Zapata et al. (2024), which examined AMP emissions at waste transfer stations and composting facilities, this study extends the scope to a poorly managed waste disposal site and uniquely integrates dry and wet season dynamics. Major key findings of this study reveal that airborne microplastic concentrations were higher at the disposal site during the dry season, aligning with established literature that associates increased resuspension with higher wind speeds and lower humidity (Bullard et al., 2021; Kannankai & Devipriya, 2024a).

Framework for Using the SCREEN3 Air Dispersion Model in Airborne Microplastic (AMP) Studies



STAGE 01

- Model Selection & Objective Setting
Is a Screening Model Appropriate?
- If yes, proceed to stage 2; else, advance model (E.g. AERMOD)

STAGE 02

- Characterise the source and gather input parameters. E.g., wind, temperature.
- Terrain type and receptor grid.
- Mass-based unit conversion of AMPs.

STAGE 03

- Model Processing Execution
Run the SCREEN3 model with defined inputs.
- Simulate concentration at multiple distances downwind

STAGE 04

- Data Output & Validation**
- Compare SCREEN3 model results with field data.
 - Assess correlation (R^2 , Spearman's ρ) for performance

STAGE 05

- Assess and classify risks (e.g., hazard scoring).
- Visualise dispersion maps to identify hotspots.
- Compare AMP levels with $PM_{2.5}$ and PM_{10} or other standards.
- Inform local policy and MSW management

Figure 6. 1: Framework for Using the SCREEN3 Air Dispersion Model in Airborne Microplastic (AMP) Studies

However, downwind concentrations during the wet season were up to 2.2 times higher than those observed in the dry season, a seasonal variation not previously reported with such clarity in the airborne microplastic literature. This distinction between source-zone and downwind behaviours underscores the complex interplay between meteorological conditions and microplastic transport under specific wet season dynamics.

This finding can be explained by wet-season-specific mechanisms, notably, some key factors that influenced the higher concentration may include raindrop impact, surface agitation, and runoff-driven mobilisation (X. Li et al., 2024). Raindrops may dislodge microplastic particles from exposed waste piles and compacted surfaces, spreading them locally even in low wind-driven lift. Moreover, water runoff during rainfall can concentrate microplastics in adjacent land or low-lying areas near the source (Ashrafy et al., 2023b), and they may subsequently become airborne through short-range turbulence or drying cycles. These mechanisms may explain the elevated near-source AMP concentrations observed during the wet season, particularly within 10–210 m downwind. This can be further supported by the results from SCREEN3, which predict similar peaks at the same distances.

Thus, while high wind speed remains a key factor in AMP resuspension, it is not the sole determinant; wet-season dynamics introduce alternative transport pathways. Although the precise contribution of these rainfall-driven mechanisms was not fully quantified in this study, their observed influence on near-source MP concentrations was demonstrated by Xia et al. (2020), who suggested that rainfall can significantly affect microplastic pollution in inland areas, particularly near sources (Xia et al., 2020). This underscores the need for further investigation into precipitation-induced resuspension and mobilisation processes.

Moreover, while Kannankai and Devipriya (2024) examined microplastic deposition influenced by wind direction and monsoon cycles, their work focused on long-range transport using HYSPLIT. In contrast, this study provides near-source insights, based on SCREEN3 model outputs, offering localised concentration profiles and risk estimations for up to 1000 m from the emission source.

Furthermore, Li et al. (2020) and Zhao et al. (2022) proposed risk indices based on polymer type. This study extends their approach by applying mass-based SCREEN3 outputs to real-world field data, enabling a more precise exposure assessment. The model's performance ($p = 0.967$, $p < 0.001$) and ability to classify fragment

microplastics as high-risk pollutants (Level III) under both seasonal conditions mark a notable advancement in AMP exposure modelling.

Importantly, when modelled AMP concentrations are compared with national air quality standards for particulate matter, they are several orders of magnitude lower than the NESREA 24-hour limits for PM_{2.5} (40 µg/m³) and PM₁₀ (150 µg/m³). Despite their lower mass concentrations, AMPs may still pose unique toxicological risks due to their polymeric nature, potential for long-range transport, and ability to carry chemical additives and adsorbed pollutants. This highlights the need to consider AMPs not merely as minor PM fractions, but as emerging pollutants with distinct environmental behaviours and health implications, especially in proximity to poorly managed waste sites. Thus, this study contributes to a growing body of evidence supporting the inclusion of microplastics in broader PM exposure and risk frameworks.

For the first time in this context, this study confirms that MSW sites function as both microplastic sinks and active sources, particularly during dry and windy periods. Importantly, the wet season introduces unique mobilising mechanisms that amplify near-field concentrations despite reduced resuspension. Although the mass concentrations of AMPs in this study remain significantly lower than established limits for conventional particulate matter (PM_{2.5} and PM₁₀), their polymer-specific toxicity, persistence, and potential to act as vectors for harmful chemicals warrant further consideration within broader PM exposure frameworks and air quality regulation efforts. These findings have implications for communities near dumpsites and underscore the urgent need for inclusive waste management strategies that consider dry and wet season AMP dynamics, an area largely unexplored in prior research

6.2 Airborne Microplastic (AMP) Sampling Using Low-Cost Sampler

Chapter 3 successfully demonstrated the design, development, and field validation of a low-cost, efficient airborne microplastic (AMP) sampler tailored for resource-constrained settings, fulfilling Objective 1 of this thesis. Constructed using locally available materials and simple assembly techniques, the sampler was engineered to be both accessible and scalable, particularly in developing regions where advanced monitoring infrastructure is limited. Unlike previous studies that primarily focused on conceptual designs or controlled laboratory trials (e.g., Gatson et al., 2020; Perera et

al., 2022), this research progressed beyond the design phase by systematically validating the sampler's performance through collocated field deployment at an active municipal solid waste disposal site.

The sampler produced reliable data on AMP concentrations, confirming its ability to capture a diverse range of microplastic morphologies, including fibres, fragments, and films, without compromising analytical accuracy. These outcomes demonstrate the sampler's suitability for environmental monitoring in complex, real-world settings. Moreover, the results address a critical gap in AMP sampling accessibility, offering a low-cost alternative to expensive commercial instruments often limited to well-resourced institutions (Li et al., 2020; Sun et al., 2022).

This innovation represents a practical advancement in environmental sampling technology, enabling broader participation in airborne microplastic research and laying the groundwork for more inclusive, geographically representative studies. As such, it aligns with recent calls for democratised environmental monitoring tools that support data generation in underrepresented regions (Prata et al., 2021; Rochman & Hoellein, 2020).

6.3 Microplastic Flux Estimation Using Modified Fick's Law

Chapter 4 centred on refining and applying flux estimation models to quantify airborne microplastic (AMP) emissions from municipal waste disposal sites, thereby advancing Objective 2 of this thesis work. The study integrates Fick's Law of diffusion with environmental parameters such as temperature, wind speed, and particle-specific dynamics. This chapter presents a novel, process-based approach to understanding microplastic resuspension and emission.

For the first time, this study estimated emission factors (EFs) for airborne microplastics from an active waste disposal site while incorporating site-specific parameters, such as soil porosity, polymer type, and seasonal meteorological conditions. Unlike previous studies that often apply generic or assumed values for AMP emission estimation (e.g., Prata et al., 2020), this work provides an empirically grounded and context-specific EF framework. This advancement enhances the precision of AMP emission inventories and improves their relevance for modelling and policy purposes in similar environmental settings.

The flux calculations revealed significant seasonal variations, with higher emissions recorded during the dry season, primarily driven by elevated wind speeds and reduced

soil moisture, factors conducive to particle lift-off and atmospheric dispersion. These findings are consistent with prior studies identifying wind-driven resuspension as a key transport mechanism for microplastics (Liu et al., 2023; Evangeliou et al., 2024).

This refined approach to AMP flux and EF estimation validates the importance of site-specific modelling and lays the foundation for more accurate predictions of AMP dispersion and deposition. The chapter's contributions are particularly relevant for terrestrial waste environments in low- and middle-income countries, where open dumping remains prevalent and empirical data are scarce.

6.4 Modelling and Risk Assessment of Airborne Microplastic Dispersion

Chapter 5 employed the SCREEN3 dispersion model to simulate transport and evaluate airborne microplastics (AMPs) exposure risks from municipal solid waste (MSW) disposal sites. The model effectively predicted downwind concentration patterns, with peak AMP levels observed between 10 and 210 m from the source. Although the dry season showed higher emissions at the source due to increased wind speeds, wet season concentrations were up to 2.2 times higher at downwind locations. This contrast indicates the influence of rain-induced mobilisation mechanisms, a dynamic not widely addressed in existing AMP dispersion literature. Zapata et al. (2024) highlighted elevated AMP concentrations at composting sites in England, but their research did not incorporate mass-based risk modelling or dispersion analysis. Unlike Kannankai and Devipriya (2024), who used HYSPLIT to track long-range AMP trajectories over monsoonal patterns, this chapter concentrates on near-source characterisation and health risk classification, providing site-specific insights particularly relevant to urban waste contexts with inadequate waste management systems.

The polymer-specific risk assessment revealed that fragment microplastics posed the highest exposure risks, with hazard indices exceeding Level III (high risk) in both seasons. While AMP concentrations remained significantly lower than NESREA's national PM limits ($PM_{2.5} = 40 \mu\text{g}/\text{m}^3$; $PM_{10} = 150 \mu\text{g}/\text{m}^3$), their potential toxicity, persistence, and ability to adsorb hazardous chemicals reinforce their relevance as distinct contributors to particulate matter exposure. This supports a prior call to include microplastics in broader air quality assessments (Sridharan et al., 2021).

The chapter further demonstrates SCREEN3 adaptability for AMPs. The demonstrated framework using SCREEN3 offers a practical modelling approach for resource-limited settings, filling a methodological gap in current AMP dispersion research. The SCREEN3 model validation indicates 80–96% predictive accuracy and strong correlation with measured ground downwind concentrations ($\rho = 0.967$, $p < 0.001$). This study provides original data and a replicable framework for risk-based monitoring of AMPs, with potential applications in environmental health policy, especially in regions with unregulated waste disposal practices.

6.5 Key Limitations

This study advances our understanding of airborne microplastic (AMP) emissions, transport, and impacts; however, the study faced several limitations that warrant consideration. One notable limitation of the study is that the findings do not align with clinical data or population density patterns, which could have offered valuable insights into the health and demographic consequences of airborne microplastic (AMP) exposure. This omission was primarily due to the lack of available health surveillance data linked to environmental exposure to airborne microplastics in the study area and elsewhere, as well as constraints in obtaining ethical clearance for health-related data. As a result, the study has not established direct correlations between AMP concentrations and respiratory or cardiovascular outcomes, limiting its applicability in public health risk estimation.

Second, the study's meteorological data relied on measurements from an anemometer at the study site, potentially introducing localised biases compared to broader, high-resolution atmospheric datasets. Access to such comprehensive meteorological data in developing countries like Nigeria is often constrained due to multiple systemic barriers (Isioye et al., 2022). Most weather and atmospheric data are owned or regulated by government agencies and may not be readily accessible or affordable for independent academic research (Hussaini & Matazu, 2023), making it difficult to obtain reliable, continuous regional meteorological data. These constraints limited the integration of more complex meteorological inputs, which could have improved the generalisability and accuracy of the dispersion modelling.

Third, while the study effectively captures short- to medium-range dispersion patterns of airborne microplastics (AMPs), it does not address long-range transport beyond the immediate vicinity of the disposal site. This is primarily due to the scope of the

SCREEN3 model, which is optimised for local-scale applications and operates under a steady-state assumption. While the modelling approach suits this study's objectives and provides valuable site-specific insights, it may not fully capture the impact of the transport and fate of AMPs on the regional or global scale. Recognising this limitation helps to position the findings within their intended context and identifies an opportunity for future research to expand on this foundation using more complex, time-resolved models.

Additionally, the analysis did not consider the potential influence of waste composition or management practices, which could significantly affect AMP emissions and dispersal. Sampling of the waste stream was not conducted due to logistical constraints and limited access to operational waste segregation records at the disposal site. As a result, this study didn't directly link specific microplastic types to their polymeric or waste-type origins. While this does not invalidate the concentration or dispersion data presented, it limits the interpretation of source-specific emission dynamics.

However, it is essential to acknowledge the limitations of the sampling methodology employed in this study. The use of low-cost samplers may have introduced biases in the collected samples, potentially affecting the representativeness of the MP populations. For instance, a low-cost sampler might not capture smaller MPs as efficiently as more advanced sampling equipment. This could lead to underestimations of MP abundance or biased size distributions.

On the other hand, FTIR can provide a fair estimate of microplastic quantities, aligning with this study's objective. For sophisticated analysis requiring detailed polymer characterisation, an instrument with higher precision may be needed. Polymer identification posed challenges, especially for smaller or degraded particles. Although visual sorting combined with microscopic techniques was sufficient for morphological classification, confirmatory polymer analysis using FTIR spectroscopy was limited by sample size, contamination risk, and instrument sensitivity (Primpke et al., 2017, 2020). The accuracy of FTIR spectroscopy can be further enhanced by complementary analytical techniques, such as Raman spectroscopy, which provides more detailed characterisation of polymers. Additionally, integrating machine learning (ML) and artificial intelligence (AI) algorithms can improve the analysis and

identification of microplastics. These computational methods can help process large datasets, identify patterns, and enhance the accuracy of polymer classification.

Other complementary methods that can be employed include:

- Pyrolysis gas chromatography-mass spectrometry (Py-GC-MS) for detailed chemical analysis of polymers
- Scanning electron microscopy (SEM) coupled with energy-dispersive spectroscopy (EDS) for high-resolution imaging and elemental analysis
- Thermal analysis techniques, such as differential scanning calorimetry (DSC) and thermogravimetry (TGA), for studying thermal properties of polymers.

The combination of LCS for sampling and FTIR for analysis highlights the trade-offs between cost-effective methodologies and the potential limitations in data quality. Future research can build upon these findings by exploring more advanced sampling and analytical techniques to improve the accuracy and reliability of microplastic research.

Addressing and considering these limitations in future research will strengthen the applicability and relevance of the findings. Furthermore, the data were collected from only one dumpsite, which may not be representative of other dumpsites across Nigeria. As a result, the findings may not be generalisable to the entire country, highlighting the need for further research and data collection from multiple sites.

6.6 Recommendations for Further Studies

This research has enhanced our understanding of airborne microplastic (AMP) emissions, transport, and dispersion from municipal solid waste (MSW) sites. The interdisciplinary approach undertaken navigated through environmental engineering, atmospheric science, analytical chemistry, and environmental modelling. While significant progress has been made, several challenges remain, providing a framework for further investigation. The suggestions for future studies are outlined below.

Refinement of Low-Cost Sampling Methods: Future work should focus on improving the design, standardisation, and performance of low-cost airborne microplastic samplers. This includes developing automated or semi-automated calibration systems, enhancing filter retention capacity for a broader range of particle

sizes, and improving resistance to environmental variability (e.g. humidity and temperature fluctuations). Additionally, locally sourcing or fabricating components such as fine-pore filters would reduce dependency on imported materials and increase the accessibility and sustainability of AMP sampling, particularly in low- and middle-income countries.

Modification of flux models: Studies should explore the efficacy of modified flux models to account for additional environmental parameters, such as precipitation and humidity, in estimating microplastic emissions during wet seasons. Further research into how rainfall impacts microplastic transport and deposition patterns, particularly near MSW sites in regions like Nigeria. These could provide greater insight into seasonal transport dynamics. These investigations would assess how raindrop agitation and rain-wash processes influence AMP concentrations in localised environments.

The interactions between AMPs and atmospheric components, including the potential for microplastics to act as vectors for bacterial or viral transport, warrant further study. Research on AMP concentrations in high-risk areas, such as near open burning sites or highly polluted waterways, would provide critical insights into human health risks. Furthermore, comparative studies on AMP deposition rates in remote and urban environments could expound variations in fibre and fragment dispersal patterns, further refining source-receptor relationships.

Studies focusing on microplastic flux and resuspension from soil under seasonal variations, particularly in humid and tropical regions like Nigeria, should be prioritised. The impact of soil characteristics, such as moisture content and organic matter, on AMP resuspension during dry and wet seasons should also be examined. Furthermore, accessible dispersion models tailored for emerging pollutants with validation against empirical data could help advance predictive capabilities for their environmental impact.

Integration of Advanced Modelling with Field Validation; While SCREEN3 proved effective for short-range dispersion estimation, future studies should apply more sophisticated atmospheric dispersion models (e.g., AERMOD, HYSPLIT, CALPUFF) to capture long-range transport and temporally dynamic behaviours of microplastics. These models should be calibrated and validated using real-world concentration data, ideally collected across multiple seasons and site types. This would enable more

accurate exposure predictions and support regional-scale assessments of AMP dispersion.

Long-Term Studies on Microplastic Exposure and Health Effects: There is an urgent need for longitudinal studies examining the chronic health impacts of inhaled microplastics, particularly in populations living near unmanaged waste sites. Such research should explore respiratory and cardiovascular outcomes through clinical or epidemiological approaches, combined with environmental exposure risk. Linking AMP concentrations with population density, land use, and health records could inform public health guidelines and contribute to the development of ambient air quality standards that explicitly account for microplastics.

Advancing Analytical Techniques for Polymer Identification; Future research should aim to enhance the accuracy, consistency, and scalability of analytical methods employed for microplastic identification and classification. While traditional techniques such as microscopy and Fourier Transform Infrared Spectroscopy (FTIR) provide valuable insights, they are often limited by particle size sensitivity, throughput, and operator bias. In this context, throughput refers to the number of samples or particles that can be analysed within a given period. Many conventional methods are time-consuming and labour-intensive, which can hinder large-scale or long-term monitoring efforts.

To address these limitations, studies should incorporate advanced spectroscopic methods, including Raman spectroscopy, pyrolysis-GC-MS, and thermal desorption techniques, which facilitate more precise chemical fingerprinting of microplastic polymers, particularly at smaller size fractions and within complex environmental matrices. Furthermore, the integration of machine learning (ML) and artificial intelligence (AI)-driven image recognition presents significant potential for automating particle classification, thereby reducing analysis time and improving reproducibility. AI models trained on annotated datasets can enhance the recognition of microplastic particle size estimation and even make preliminary predictions of polymer type from visual and spectral data. As such, tools become increasingly accessible, future studies should investigate the development of open-source platforms or cloud-based AI models to support widespread application, especially in resource-limited contexts. To conclude, establishing standardised workflows for contamination control, sample preparation, and spectral library referencing will be crucial for ensuring cross-study comparability and building a global evidence base on airborne microplastic pollution.

The field-scale application of these findings is crucial: The results of this research should be implemented to develop mitigation strategies, including improved waste management practices and the establishment of buffer zones near municipal solid waste (MSW) sites. Furthermore, investigating the effectiveness of these strategies during the seasonal assessment of microplastic pollution emissions in Nigeria will ensure that practical solutions are tailored to the specific conditions of each site. Lastly, improvements in policy frameworks for AMP monitoring and regulation are essential. National guidelines should be aligned with international standards to address AMP pollution more effectively. Adopting recommendations from global organisations, such as UNEP, to fund and prioritise AMP research and remediation initiatives in Nigeria would provide a pathway toward sustainable environmental management.

6.7 Contribution to Knowledge

This study makes several original contributions to the field of airborne microplastic (AMP) research, particularly within the context of developing regions. Its innovations span methodological design, flux estimation, and dispersion modelling, offering significant advancements beyond previous studies.

Firstly, the research introduces and validates a Low-Cost Sampler (LCS) designed for AMP collection, filling a major gap in global monitoring capabilities. Unlike earlier conceptual attempts (e.g., Gaston et al., 2020; Perera et al., 2022), which lacked rigorous performance testing, this study provides a quantitative validation of the LCS against a commercial SKC sampler using co-located sampling. The LCS achieved 94.12% accuracy and a strong correlation ($\rho = 0.976$) with the reference device, making it the first verified low-cost sampler in AMP studies with practical applicability in resource-limited settings. This innovation democratises AMP research and facilitates scalable environmental monitoring even with limited funding or infrastructure shortages.

Secondly, the study refines flux estimation methodology by adapting Fick's law to calculate AMP emissions from municipal solid waste disposal sites (MSWDS). While prior research (e.g., Zhou et al., 2017; Evangelou et al., 2024) has noted AMP resuspension, this work is among the first to quantitatively integrate local environmental variables, such as wind speed, soil porosity, and temperature, into a

modified diffusion-based model. This results in more accurate, site-specific flux estimates for diverse microplastic types (fibres, fragments, and films), generating a new class of microplastic emission factors (EFs) in both mass ($\mu\text{g/day}$) and particle-based ($\text{MP/m}^2/\text{s}$) units. These emissions factors provide essential inputs for simulation models and represent a critical step toward developing standardised AMP inventories for air quality research.

Thirdly, this is the first known study to employ the SCREEN3 Gaussian dispersion model in the context of airborne microplastics. While SCREEN3 has been extensively used for conventional air pollutants (e.g., Zhong et al., 2011), there is no prior literature that has applied it to simulate AMP transport. This novel application offers a computationally efficient tool for simulating AMP movement across short- to medium-range distances, with results showing strong agreement with field data ($\rho = 0.967$, $R^2 = 0.96\text{--}0.98$). Unlike long-range models (e.g., HYSPLIT, FLEXPART), SCREEN3 provides accessible and rapid output, making it particularly beneficial in low-income countries.

A significant contribution of this study is the identification of a dual seasonal pattern in AMP concentration profiles, on-site AMP levels peaked during the dry season due to increased resuspension under high wind speeds, consistent with established literature (e.g., Al-Zawaidah et al., 2024b). In contrast, downwind concentrations peaked during the wet season, a previously undocumented trend in AMP research. This unique finding introduces the concept of a “rain trap” mechanism, wherein precipitation dislodges surface-bound microplastics while also restricting vertical dispersion, leading to enhanced near-surface accumulation. Previous studies (e.g., Hu et al., 2022; Kannankai & Devipriya, 2024) acknowledged seasonal variability, but none described this two-phase dynamic with quantitative support or model validation.

The study also recommends establishing a buffer zone of 100–200 m around MSWDS to decrease the risk of AMP exposure to neighbouring communities. This practical guideline is grounded in SCREEN3 outputs and provides a science-based mitigation strategy, especially relevant for urban planning and public health in high-risk regions.

In summary, this research:

- Validates a low-cost AMP sampling tool, enabling widespread environmental monitoring

- Refines AMP emission modelling using modified Fick's law with site-specific parameters
- Introduces SCREEN3 into AMP research, bridging an important methodological gap
- Identifies a novel wet-season AMP accumulation mechanism, advancing environmental exposure theory
- Recommends evidence-based buffer zones to protect vulnerable populations.

These contributions offer a substantive advancement over existing literature and provide a foundation for future policy, modelling, and health risk assessments related to airborne microplastic pollution

7.0 References

- Abad López, A. P., Trilleras, J., Arana, V. A., Garcia-Alzate, L. S., & Grande-Tovar, C. D. (2023). Atmospheric microplastics: exposure, toxicity, and detrimental health effects. *RSC Advances*, 13(11), 7468–7489. <https://doi.org/10.1039/D2RA07098G>
- Abbasi, S. (2021). Microplastics washout from the atmosphere during a monsoon rain event. *Journal of Hazardous Materials Advances*, 4, 100035. <https://doi.org/10.1016/j.hazadv.2021.100035>
- Abbasi, S., Keshavarzi, B., Moore, F., Turner, A., Kelly, F. J., Dominguez, A. O., & Jaafarzadeh, N. (2019). Distribution and potential health impacts of microplastics and microrubbers in air and street dusts from Asaluyeh County, Iran. *Environmental Pollution*, 244, 153–164. <https://doi.org/10.1016/j.envpol.2018.10.039>
- Abbasi, S., Rezaei, M., Ahmadi, F., & Turner, A. (2022). Atmospheric transport of microplastics during a dust storm. *Chemosphere*, 292, 133456. <https://doi.org/10.1016/j.chemosphere.2021.133456>
- Abbasi, S., Turner, A., Sharifi, R., Nematollahi, Mohammad. J., Keshavarzifard, M., & Moghtaderi, T. (2022). Microplastics in the school classrooms of Shiraz, Iran. *Building and Environment*, 207, 108562. <https://doi.org/10.1016/j.buildenv.2021.108562>
- Abubakar, I. R., Maniruzzaman, K. M., Dano, U. L., AlShihri, F. S., AlShammari, M. S., Ahmed, S. M. S., Al-Gehlani, W. A. G., & Alrawaf, T. I. (2022). Environmental Sustainability Impacts of Solid Waste Management Practices in the Global South. *International Journal of Environmental Research and Public Health*, 19(19), 12717. <https://doi.org/10.3390/ijerph191912717>
- Acharya, K. P., & Pathak, S. (2019). Applied Research in Low-Income Countries: Why and How? *Frontiers in Research Metrics and Analytics*, 4. <https://doi.org/10.3389/frma.2019.00003>
- Adeniran, J. A., Yusuf, R. O., Fakinle, B. S., & Sonibare, J. A. (2019). Air quality assessment and modelling of pollutants emission from a major cement plant complex in Nigeria. *Atmospheric Pollution Research*, 10(1), 257–266. <https://doi.org/10.1016/j.apr.2018.07.010>
- Adeoye, M., Rahimzadeh, S., Taylor, S., Shrikhande, S., Perel, P., Shah, A., Di Cesare, M., & Miller, M. R. (2024). The Impact of Air Pollution on Cardiovascular Health Outcomes in African Populations. *JACC: Advances*, 3(12), 101371. <https://doi.org/10.1016/j.jacadv.2024.101371>
- Ado Umar, Y., Musa, U., Ibrahim, I., Muhammad, Y., Aliyu, A., Nabegu, B., Adamu, F., Dakata, G., Ado, Y., & Bello, U. (2014). Overview Of The Physical And Human Setting Of Kano Region, Nigeria Adamu Mustapha Overview Of The Physical And Human Setting Of Kano Region, Nigeria. In *Journal of Geography* (Vol. 1, Issue 5). <https://www.researchgate.net/publication/343859343>
- Adomat, Y., Kahl, M., Musche, F., & Grischek, T. (2022). Evaluation of microplastics sediment sampling techniques—efficiency of common methods and new

- approaches. *Microplastics and Nanoplastics*, 2(1), 27. <https://doi.org/10.1186/s43591-022-00047-x>
- Ageel, H. K., Harrad, S., & Abdallah, M. A.-E. (2022a). Occurrence, human exposure, and risk of microplastics in the indoor environment. *Environmental Science: Processes & Impacts*, 24(1), 17–31. <https://doi.org/10.1039/D1EM00301A>
- Ageel, H. K., Harrad, S., & Abdallah, M. A.-E. (2022b). Occurrence, human exposure, and risk of microplastics in the indoor environment. *Environmental Science: Processes & Impacts*, 24(1), 17–31. <https://doi.org/10.1039/D1EM00301A>
- Ainali, N. M., Kalaronis, D., Kontogiannis, A., Evgenidou, E., Kyzas, G. Z., Yang, X., Bikiaris, D. N., & Lambropoulou, D. A. (2021). Microplastics in the environment: Sampling, pretreatment, analysis and occurrence based on current and newly-exploited chromatographic approaches. *Science of The Total Environment*, 794, 148725. <https://doi.org/10.1016/j.scitotenv.2021.148725>
- Akdogan, Z., & Guven, B. (2024). Modeling the settling and resuspension of microplastics in rivers: Effect of particle properties and flow conditions. *Water Research*, 264, 122181. <https://doi.org/10.1016/j.watres.2024.122181>
- AKINBILE, C. O. (2012). Environmental impact of landfill on groundwater quality and agricultural soils in Nigeria. *Soil and Water Research*, 7(1), 18–26. <https://doi.org/10.17221/4/2011-SWR>
- Akintola, O. O., Adeyemi, G. O., Bodede, A. I., Adekoya, O., & Babatunde, K. O. (2021). Hydrochemical Assessment of Groundwater around Lapite Dumpsite for Irrigation Water Quality in Ibadan, Southwestern Nigeria. *Journal of Bioresource Management*, 8(2), 98–108. <https://doi.org/10.35691/JBM.1202.0184>
- Al Harraq, A., Brahana, P. J., Arcemont, O., Zhang, D., Valsaraj, K. T., & Bharti, B. (2022). Effects of Weathering on Microplastic Dispersibility and Pollutant Uptake Capacity. *ACS Environmental Au*, 2(6), 549–555. <https://doi.org/10.1021/acsenvironau.2c00036>
- Al-Azzawi, M. S. M., Kefer, S., Weißer, J., Reichel, J., Schwaller, C., Glas, K., Knoop, O., & Drewes, J. E. (2020). Validation of Sample Preparation Methods for Microplastic Analysis in Wastewater Matrices—Reproducibility and Standardization. *Water*, 12(9), 2445. <https://doi.org/10.3390/w12092445>
- Aldekheel, M., Farahani, V. J., Tohidi, R., Altuwayjiri, A., & Sioutas, C. (2023). Development and performance evaluation of a two-stage cascade impactor equipped with gelatin filter substrates for the collection of multi-sized particulate matter. *Atmospheric Environment*, 294, 119493. <https://doi.org/10.1016/j.atmosenv.2022.119493>
- Alijagic, A., Suljević, D., Fočak, M., Sulejmanović, J., Šehović, E., Särndahl, E., & Engwall, M. (2024). The triple exposure nexus of microplastic particles, plastic-associated chemicals, and environmental pollutants from a human health perspective. *Environment International*, 188, 108736. <https://doi.org/10.1016/j.envint.2024.108736>
- Alimi, O. S., Farner Budarz, J., Hernandez, L. M., & Tufenkji, N. (2018a). Microplastics and Nanoplastics in Aquatic Environments: Aggregation, Deposition, and Enhanced Contaminant Transport. *Environmental Science & Technology*, 52(4), 1704–1724. <https://doi.org/10.1021/acs.est.7b05559>

- Alimi, O. S., Farner Budarz, J., Hernandez, L. M., & Tufenkji, N. (2018b). Microplastics and Nanoplastics in Aquatic Environments: Aggregation, Deposition, and Enhanced Contaminant Transport. *Environmental Science & Technology*, 52(4), 1704–1724. <https://doi.org/10.1021/acs.est.7b05559>
- Alimov, Z. B., Kusakari, H., & Okuda, T. (2021). Development of A Low-cost Simultaneous Low Volume Air Sampler Controlled with Sonic Venturi. *Asian Journal of Atmospheric Environment*, 15(1), 2020072. <https://doi.org/10.5572/ajae.2020.072>
- Allen, D., Allen, S., Abbasi, S., Baker, A., Bergmann, M., Brahney, J., Butler, T., Duce, R. A., Eckhardt, S., Evangeliou, N., Jickells, T., Kanakidou, M., Kershaw, P., Laj, P., Levermore, J., Li, D., Liss, P., Liu, K., Mahowald, N., ... Wright, S. (2022a). Microplastics and nanoplastics in the marine-atmosphere environment. *Nature Reviews Earth & Environment*, 3(6), 393–405. <https://doi.org/10.1038/s43017-022-00292-x>
- Allen, D., Allen, S., Abbasi, S., Baker, A., Bergmann, M., Brahney, J., Butler, T., Duce, R. A., Eckhardt, S., Evangeliou, N., Jickells, T., Kanakidou, M., Kershaw, P., Laj, P., Levermore, J., Li, D., Liss, P., Liu, K., Mahowald, N., ... Wright, S. (2022b). Microplastics and nanoplastics in the marine-atmosphere environment. *Nature Reviews Earth & Environment*, 3(6), 393–405. <https://doi.org/10.1038/s43017-022-00292-x>
- Allen, S., Allen, D., Baladima, F., Phoenix, V. R., Thomas, J. L., Le Roux, G., & Sonke, J. E. (2021). Evidence of free tropospheric and long-range transport of microplastic at Pic du Midi Observatory. *Nature Communications*, 12(1). <https://doi.org/10.1038/s41467-021-27454-7>
- Allen, S., Allen, D., Moss, K., Le Roux, G., Phoenix, V. R., & Sonke, J. E. (2020). Examination of the ocean as a source for atmospheric microplastics. *PLoS ONE*, 15(5), 1–14. <https://doi.org/10.1371/journal.pone.0232746>
- Allen, S., Allen, D., Phoenix, V. R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S., & Galop, D. (2019). Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience*, 12(5), 339–344. <https://doi.org/10.1038/s41561-019-0335-5>
- AL-SHAMMARY, A. A. G., KOUZANI, A. Z., KAYNAK, A., KHOO, S. Y., NORTON, M., & GATES, W. (2018). Soil Bulk Density Estimation Methods: A Review. *Pedosphere*, 28(4), 581–596. [https://doi.org/10.1016/S1002-0160\(18\)60034-7](https://doi.org/10.1016/S1002-0160(18)60034-7)
- Al-Zawaidah, H., Kooi, M., Hoitink, T., Vermeulen, B., & Waldschläger, K. (2024a). Mapping Microplastic Movement: A Phase Diagram to Predict Nonbuoyant Microplastic Modes of Transport at the Particle Scale. *Environmental Science & Technology*, 58(40), 17979–17989. <https://doi.org/10.1021/acs.est.4c08128>
- Al-Zawaidah, H., Kooi, M., Hoitink, T., Vermeulen, B., & Waldschläger, K. (2024b). Mapping Microplastic Movement: A Phase Diagram to Predict Nonbuoyant Microplastic Modes of Transport at the Particle Scale. *Environmental Science & Technology*, 58(40), 17979–17989. <https://doi.org/10.1021/acs.est.4c08128>
- Aminah, I. S., & Ikejima, K. (2023). Potential sources of microplastic contamination in laboratory analysis and a protocol for minimising contamination. *Environmental Monitoring and Assessment*, 195(7), 808. <https://doi.org/10.1007/s10661-023-11410-7>

- Andrady, A. L. (2011). Microplastics in the marine environment. *Marine Pollution Bulletin*, 62(8), 1596–1605. <https://doi.org/10.1016/j.marpolbul.2011.05.030>
- AQD-ODEQ. (2017). *Air Dispersion Modeling Guidelines For Oklahoma Air Quality Permits*.
- AQIEG. (2002). *AIR QUALITY IMPACT EVALUATION GUIDELINES*. <http://www.anr.state.vt.us/dec/air/>
- Aragaw, T. A. (2021a). Microplastic pollution in African countries' water systems: a review on findings, applied methods, characteristics, impacts, and managements. *SN Applied Sciences*, 3(6), 629. <https://doi.org/10.1007/s42452-021-04619-z>
- Aragaw, T. A. (2021b). Microplastic pollution in African countries' water systems: a review on findings, applied methods, characteristics, impacts, and managements. *SN Applied Sciences*, 3(6), 629. <https://doi.org/10.1007/s42452-021-04619-z>
- Aramendia, J., García-Velasco, N., Amigo, J. M., Izagirre, U., Seifert, A., Soto, M., & Castro, K. (2024). Evidence of internalized microplastics in mussel tissues detected by volumetric Raman imaging. *Science of The Total Environment*, 914, 169960. <https://doi.org/10.1016/j.scitotenv.2024.169960>
- Arciszewska, C., & McClatchey, J. (2001). The importance of meteorological data for modelling air pollution using ADMS-Urban. *Meteorological Applications*, 8(3), S1350482701003103. <https://doi.org/10.1017/S1350482701003103>
- Arogundade, S. (2022a). *A Glance at Biggest Dumpsites in Nigeria*. BioEnergy Consult Powering a Greener Future. <https://www.bioenergyconsult.com/tag/major-landfills-in-nigeria/>
- Arogundade, S. (2022b). *A Glance at Biggest Dumpsites in Nigeria*. BioEnergy Consult Powering a Greener Future. <https://www.bioenergyconsult.com/tag/major-landfills-in-nigeria/>
- Ashrafy, A., Liza, A. A., Islam, M. N., Billah, M. M., Arafat, S. T., Rahman, M. M., & Rahman, S. M. (2023a). Microplastics Pollution: A Brief Review of Its Source and Abundance in Different Aquatic Ecosystems. *Journal of Hazardous Materials Advances*, 9, 100215. <https://doi.org/10.1016/j.hazadv.2022.100215>
- Ashrafy, A., Liza, A. A., Islam, M. N., Billah, M. M., Arafat, S. T., Rahman, M. M., & Rahman, S. M. (2023b). Microplastics Pollution: A Brief Review of Its Source and Abundance in Different Aquatic Ecosystems. *Journal of Hazardous Materials Advances*, 9, 100215. <https://doi.org/10.1016/j.hazadv.2022.100215>
- Azari, A., Vanoirbeek, J. A. J., Van Belleghem, F., Vleeschouwers, B., Hoet, P. H. M., & Ghosh, M. (2023a). Sampling strategies and analytical techniques for assessment of airborne micro and nano plastics. *Environment International*, 174, 107885. <https://doi.org/10.1016/j.envint.2023.107885>
- Azari, A., Vanoirbeek, J. A. J., Van Belleghem, F., Vleeschouwers, B., Hoet, P. H. M., & Ghosh, M. (2023b). Sampling strategies and analytical techniques for assessment of airborne micro and nano plastics. *Environment International*, 174, 107885. <https://doi.org/10.1016/j.envint.2023.107885>
- Azari, A., Vanoirbeek, J. A. J., Van Belleghem, F., Vleeschouwers, B., Hoet, P. H. M., & Ghosh, M. (2023c). Sampling strategies and analytical techniques for assessment of airborne micro and nano plastics. *Environment International*, 174, 107885. <https://doi.org/10.1016/j.envint.2023.107885>

- Babayemi, J. O., Ogundiran, M. B., Weber, R., & Osibanjo, O. (2018). Initial inventory of plastics imports in Nigeria as a basis for more sustainable management policies. *Journal of Health and Pollution*, 8(18). <https://doi.org/10.5696/22156-9614-8.18.1>
- Bakels, L., Tatsii, D., Tipka, A., Thompson, R., Dütsch, M., Blaschek, M., Seibert, P., Baier, K., Bucci, S., Cassiani, M., Eckhardt, S., Groot Zwaafink, C., Henne, S., Kaufmann, P., Lechner, V., Maurer, C., Mulder, M. D., Pisso, I., Plach, A., ... Stohl, A. (2024). FLEXPART version 11: improved accuracy, efficiency, and flexibility. *Geoscientific Model Development*, 17(21), 7595–7627. <https://doi.org/10.5194/gmd-17-7595-2024>
- Bakir, A., Doran, D., Silburn, B., Russell, J., Archer-Rand, S., Barry, J., Maes, T., Limpenny, C., Mason, C., Barber, J., & Nicolaus, E. E. M. (2023). A spatial and temporal assessment of microplastics in seafloor sediments: A case study for the UK. *Frontiers in Marine Science*, 9. <https://doi.org/10.3389/fmars.2022.1093815>
- BAROUTIAN, S., MOHEBBI, A., & GOHARRIZI, A. (2006). Measuring and modeling particulate dispersion: A case study of Kerman Cement Plant. *Journal of Hazardous Materials*, 136(3), 468–474. <https://doi.org/10.1016/j.jhazmat.2006.01.050>
- Baruah, A., Sharma, A., Sharma, S., & Nagraik, R. (2022). An insight into different microplastic detection methods. *International Journal of Environmental Science and Technology*, 19(6), 5721–5730. <https://doi.org/10.1007/s13762-021-03384-1>
- Bekwapu, M., Imeh, P., & Mbu, D. (2024). Impact of Plastic Waste Recycling on Sustainable Environmental Management in Nigeria. *IIARD International Journal of Geography & Environmental Management*, 10(12). <https://doi.org/10.56201/ijgem.v10.no12.2024.pg58.71>
- Belioka, M.-P., & Achilias, D. S. (2024a). How plastic waste management affects the accumulation of microplastics in waters: a review for transport mechanisms and routes of microplastics in aquatic environments and a timeline for their fate and occurrence (past, present, and future). *Water Emerging Contaminants & Nanoplastics*, 3(2). <https://doi.org/10.20517/wecn.2024.09>
- Belioka, M.-P., & Achilias, D. S. (2024b). The Effect of Weathering Conditions in Combination with Natural Phenomena/Disasters on Microplastics' Transport from Aquatic Environments to Agricultural Soils. *Microplastics*, 3(3), 518–538. <https://doi.org/10.3390/microplastics3030033>
- Ben Stride, Abolfathi, S., Bending, G. D., & Pearson, J. (2024). Quantifying microplastic dispersion due to density effects. *Journal of Hazardous Materials*, 466, 133440. <https://doi.org/10.1016/j.jhazmat.2024.133440>
- Bhat, A. S., & Kumar, A. (2011). Application of the United States environmental protection agency's AERSCREEN to the air quality problems. *Environmental Progress & Sustainable Energy*, 30(2), 135–140. <https://doi.org/10.1002/ep.10569>
- Bhat, M. A. (2023). Indoor microplastics: a comprehensive review and bibliometric analysis. *Environmental Science and Pollution Research*, 30(58), 121269–121291. <https://doi.org/10.1007/s11356-023-30902-0>

- Bhat, M. A. (2024). Unveiling the overlooked threat: Macroplastic pollution in indoor markets in an urban city. *Case Studies in Chemical and Environmental Engineering*, 9, 100558. <https://doi.org/10.1016/j.cscee.2023.100558>
- Bhat, M. A., & Gaga, E. O. (2024). Unveiling the invisible: first discovery of micro- and nanoplastic size segregation in indoor commercial markets using a cascade impactor. *Air Quality, Atmosphere & Health*. <https://doi.org/10.1007/s11869-024-01624-0>
- Bîrcă, A., Gherasim, O., Grumezescu, V., & Grumezescu, A. M. (2019). Introduction in thermoplastic and thermosetting polymers. In *Materials for Biomedical Engineering* (pp. 1–28). Elsevier. <https://doi.org/10.1016/B978-0-12-816874-5.00001-3>
- Blair, R. M., Waldron, S., & Gauchotte-Lindsay, C. (2019). Average daily flow of microplastics through a tertiary wastewater treatment plant over a ten-month period. *Water Research*, 163, 114909. <https://doi.org/10.1016/j.watres.2019.114909>
- Blair, R. M., Waldron, S., Phoenix, V., & Gauchotte-Lindsay, C. (2017). Micro- and Nanoplastic Pollution of Freshwater and Wastewater Treatment Systems. *Springer Science Reviews*, 5(1–2), 19–30. <https://doi.org/10.1007/s40362-017-0044-7>
- Bodor, A., Feigl, G., Kolossa, B., Mészáros, E., Laczi, K., Kovács, E., Perei, K., & Rákhely, G. (2024). Soils in distress: The impacts and ecological risks of (micro)plastic pollution in the terrestrial environment. *Ecotoxicology and Environmental Safety*, 269, 115807. <https://doi.org/10.1016/j.ecoenv.2023.115807>
- Borah, S. J., Gupta, A. K., Kumar, V., Jhajharia, P., Singh, P. P., Kumar, P., Kumar, R., Dubey, K. K., & Gupta, A. (2024). The Peril of Plastics: Atmospheric Microplastics in Outdoor, Indoor, and Remote Environments. *Sustainable Chemistry*, 5(2), 149–162. <https://doi.org/10.3390/suschem5020011>
- Borges-Ramírez, M. M., Mendoza-Franco, E. F., Escalona-Segura, G., & Osten, J. R. (2020a). Plastic density as a key factor in the presence of microplastic in the gastrointestinal tract of commercial fishes from Campeche Bay, Mexico. *Environmental Pollution*, 267, 115659. <https://doi.org/10.1016/j.envpol.2020.115659>
- Borges-Ramírez, M. M., Mendoza-Franco, E. F., Escalona-Segura, G., & Osten, J. R. (2020b). Plastic density as a key factor in the presence of microplastic in the gastrointestinal tract of commercial fishes from Campeche Bay, Mexico. *Environmental Pollution*, 267, 115659. <https://doi.org/10.1016/j.envpol.2020.115659>
- Brander, S. M., Renick, V. C., Foley, M. M., Steele, C., Woo, M., Lusher, A., Carr, S., Helm, P., Box, C., Cherniak, S., Andrews, R. C., & Rochman, C. M. (2020a). Sampling and Quality Assurance and Quality Control: A Guide for Scientists Investigating the Occurrence of Microplastics Across Matrices. *Applied Spectroscopy*, 74(9), 1099–1125. <https://doi.org/10.1177/0003702820945713>
- Brander, S. M., Renick, V. C., Foley, M. M., Steele, C., Woo, M., Lusher, A., Carr, S., Helm, P., Box, C., Cherniak, S., Andrews, R. C., & Rochman, C. M. (2020b). Sampling and Quality Assurance and Quality Control: A Guide for Scientists

- Investigating the Occurrence of Microplastics Across Matrices. *Applied Spectroscopy*, 74(9), 1099–1125. <https://doi.org/10.1177/0003702820945713>
- Brander, S. M., Renick, V. C., Foley, M. M., Steele, C., Woo, M., Lusher, A., Carr, S., Helm, P., Box, C., Cherniak, S., Andrews, R. C., & Rochman, C. M. (2020c). Sampling and Quality Assurance and Quality Control: A Guide for Scientists Investigating the Occurrence of Microplastics Across Matrices. *Applied Spectroscopy*, 74(9), 1099–1125. <https://doi.org/10.1177/0003702820945713>
- Bullard, J. E., Ockelford, A., O'Brien, P., & McKenna Neuman, C. (2021). Preferential transport of microplastics by wind. *Atmospheric Environment*, 245, 118038. <https://doi.org/10.1016/j.atmosenv.2020.118038>
- Bullard, J. E., Zhou, Z., Davis, S., & Fowler, S. (2023). Breakdown and Modification of Microplastic Beads by Aeolian Abrasion. *Environmental Science & Technology*, 57(1), 76–84. <https://doi.org/10.1021/acs.est.2c05396>
- Burgos-Aceves, M. A., Abo-Al-Ela, H. G., & Faggio, C. (2021). Physiological and metabolic approach of plastic additive effects: Immune cells responses. *Journal of Hazardous Materials*, 404(PA), 124114. <https://doi.org/10.1016/j.jhazmat.2020.124114>
- Burnley, S. J. (2007). A review of municipal solid waste composition in the United Kingdom. *Waste Management*, 27(10), 1274–1285. <https://doi.org/10.1016/j.wasman.2006.06.018>
- Butu, A. W., & Mshelia, S. S. (2014). MUNICIPAL SOLID WASTE DISPOSAL AND ENVIRONMENTAL ISSUES IN KANO METROPOLIS, NIGERIA. In *British Journal of Environmental Sciences* (Vol. 2, Issue 1). www.ea-journals.org
- Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X., & Chen, Q. (2017a). Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence. *Environmental Science and Pollution Research*, 24(32), 24928–24935. <https://doi.org/10.1007/s11356-017-0116-x>
- Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X., & Chen, Q. (2017b). Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence. *Environmental Science and Pollution Research*, 24(32), 24928–24935. <https://doi.org/10.1007/s11356-017-0116-x>
- Caldwell, J., Taladriz-Blanco, P., Lehner, R., Lubskyy, A., Ortuso, R. D., Rothen-Rutishauser, B., & Petri-Fink, A. (2022). The micro-, submicron-, and nanoplastic hunt: A review of detection methods for plastic particles. *Chemosphere*, 293, 133514. <https://doi.org/10.1016/j.chemosphere.2022.133514>
- Camalier, L., Eberly, S., Miller, J., & Papp, M. (2007). *Guideline on the Meaning and the Use of Precision and Bias Data Required by 40 CFR Part 58 Appendix A*.
- Campanale, C., Savino, I., Pojar, I., Massarelli, C., & Uricchio, V. F. (2020). A Practical Overview of Methodologies for Sampling and Analysis of Microplastics in Riverine Environments. *Sustainability*, 12(17), 6755. <https://doi.org/10.3390/su12176755>
- Castañeda, R. A., Avlijas, S., Simard, M. A., & Ricciardi, A. (2014). Microplastic pollution in St. Lawrence River sediments. *Canadian Journal of Fisheries and Aquatic Sciences*, 71(12), 1767–1771. <https://doi.org/10.1139/cjfas-2014-0281>

- Cerutti, M., Scientific, T. F., & Madison, V. R. (2019). (a) (b) *Microplastic Analysis with FTIR and Raman Spectroscopy SAMPLE PREPARATION*.
- Chang, N., Chen, L., Wang, N., Cui, Q., Qiu, T., Zhao, S., He, H., Zeng, Y., Dai, W., Duan, C., & Fang, L. (2024). Unveiling the impacts of microplastic pollution on soil health: A comprehensive review. *Science of The Total Environment*, 951, 175643. <https://doi.org/10.1016/j.scitotenv.2024.175643>
- Chapman, E., Liddle, C. R., Williams, B., Hilmer, E., Quick, L. J., Garcia, A. G., Suárez, D. C., White, D., Bunting, M. J., Walker, P., Cabaneros, S. M. S., Kinnersley, R., Hansen, M. F., Atherall, C. A., & Rotchell, J. M. (2024). Airborne microplastic monitoring: Developing a simplified outdoor sampling approach using pollen monitoring equipment. *Journal of Hazardous Materials*, 480, 136129. <https://doi.org/10.1016/j.jhazmat.2024.136129>
- Chen, Q., Shi, G., Revell, L. E., Zhang, J., Zuo, C., Wang, D., Le Ru, E. C., Wu, G., & Mitrano, D. M. (2023). Long-range atmospheric transport of microplastics across the southern hemisphere. *Nature Communications*, 14(1), 7898. <https://doi.org/10.1038/s41467-023-43695-0>
- Chen, Q., Wang, Q., Zhang, C., Zhang, J., Dong, Z., & Xu, Q. (2021a). Aging simulation of thin-film plastics in different environments to examine the formation of microplastic. *Water Research*, 202, 117462. <https://doi.org/10.1016/j.watres.2021.117462>
- Chen, Q., Wang, Q., Zhang, C., Zhang, J., Dong, Z., & Xu, Q. (2021b). Aging simulation of thin-film plastics in different environments to examine the formation of microplastic. *Water Research*, 202, 117462. <https://doi.org/10.1016/j.watres.2021.117462>
- Chen, S., Li, Y., Mawhorter, C., & Legoski, S. (2021). Quantification of microplastics by count, size and morphology in beverage containers using Nile Red and ImageJ. *Journal of Water and Health*, 19(1), 79–88. <https://doi.org/10.2166/WH.2020.171>
- Cheng, Y.-L., Zhang, R., Tisinger, L., Cali, S., Yu, Z., Chen, H. Y., & Li, A. (2022). Characterization of microplastics in sediment using stereomicroscopy and laser direct infrared (LDIR) spectroscopy. *Gondwana Research*, 108, 22–30. <https://doi.org/10.1016/j.gr.2021.10.002>
- Chew, B. S., Pimentel Contreras, R., McCartney, M. M., Borrás, E., Kenyon, N. J., & Davis, C. E. (2022). A low cost, easy-to-assemble, open-source modular mobile sampler design for thermal desorption analysis of breath and environmental VOCs. *Journal of Breath Research*, 16(3), 036005. <https://doi.org/10.1088/1752-7163/ac6c9f>
- Chia, R. W., Lee, J.-Y., Cha, J., & Rodríguez-Seijo, A. (2024). Methods of soil sampling for microplastic analysis: a review. *Environmental Chemistry Letters*, 22(1), 227–238. <https://doi.org/10.1007/s10311-023-01652-9>
- Childers, H., McCarty, E., Mosely, C., Riepenhoff, D., & Wright, L. (2023). Portable On-Site Optical Detection and Quantification of Microplastics. *2023 Waste-Management Education Research Conference (WERC)*, 1–6. <https://doi.org/10.1109/WERC58028.2023.10310827>
- Choi, H., Lee, I., Kim, H., Park, J., Cho, S., Oh, S., Lee, M., & Kim, H. (2022). Comparison of Microplastic Characteristics in the Indoor and Outdoor Air of Urban

- Areas of South Korea. *Water, Air, & Soil Pollution*, 233(5), 169. <https://doi.org/10.1007/s11270-022-05650-5>
- Choudhury, S., Deb, K., Paul, S., Chakraborty, B. B., & Goswami, S. (2023). Sampling and Detection of Microplastics in the Atmosphere. In *Microplastics in the Ecosphere* (pp. 113–124). Wiley. <https://doi.org/10.1002/9781119879534.ch7>
- Chukwuka, A. V., Adegboyegun, A. D., Oluwale, F. V., Oni, A. A., Omogbemi, E. D., & Adeogun, A. O. (2024). Microplastic dynamics and risk projections in West African coastal areas: Developing a vulnerability index, adverse ecological pathways, and mitigation framework using remote-sensed oceanographic profiles. *Science of The Total Environment*, 953, 175963. <https://doi.org/10.1016/j.scitotenv.2024.175963>
- Circelli, L., Cheng, Z., Garwood, E., Yuksel, K., Di Iorio, E., Angelico, R., & Colombo, C. (2024). Comparison of ATR-FTIR and NIR spectroscopy for identification of microplastics in biosolids. *Science of The Total Environment*, 916, 170215. <https://doi.org/10.1016/j.scitotenv.2024.170215>
- Clark, R. A., & Shaver, M. P. (2024). Depolymerization within a Circular Plastics System. *Chemical Reviews*, 124(5), 2617–2650. <https://doi.org/10.1021/acs.chemrev.3c00739>
- Cole, M., Lindeque, P., Halsband, C., & Galloway, T. S. (2011). Microplastics as contaminants in the marine environment: A review. *Marine Pollution Bulletin*, 62(12), 2588–2597. <https://doi.org/10.1016/j.marpolbul.2011.09.025>
- Colzi, I., Renna, L., Bianchi, E., Castellani, M. B., Coppi, A., Pignattelli, S., Loppi, S., & Gonnelli, C. (2022). Impact of microplastics on growth, photosynthesis and essential elements in Cucurbita pepo L. *Journal of Hazardous Materials*, 423. <https://doi.org/10.1016/j.jhazmat.2021.127238>
- Connan, O., Smith, K., Organo, C., Solier, L., Maro, D., & Hébert, D. (2013). Comparison of RIMPUFF, HYSPLIT, ADMS atmospheric dispersion model outputs, using emergency response procedures, with 85Kr measurements made in the vicinity of nuclear reprocessing plant. *Journal of Environmental Radioactivity*, 124, 266–277. <https://doi.org/10.1016/j.jenvrad.2013.06.004>
- Cook, S., Chan, H.-L., Abolfathi, S., Bending, G. D., Schäfer, H., & Pearson, J. M. (2020). Longitudinal dispersion of microplastics in aquatic flows using fluorometric techniques. *Water Research*, 170, 115337. <https://doi.org/10.1016/j.watres.2019.115337>
- Correia Prata, J. (2017). *Airborne microplastics: Consequences to human health?* <https://doi.org/10.1016/j.envpol.2017.11.043>
- Cowger, W., Steinmetz, Z., Gray, A., Munno, K., Lynch, J., Hapich, H., Primpke, S., De Frond, H., Rochman, C., & Herodotou, O. (2021). Microplastic Spectral Classification Needs an Open Source Community: Open Specy to the Rescue! *Analytical Chemistry*, 93(21), 7543–7548. <https://doi.org/10.1021/acs.analchem.1c00123>
- Crawford, C. B., & Quinn, B. (2017a). Microplastic identification techniques. In *Microplastic Pollutants* (pp. 219–267). Elsevier. <https://doi.org/10.1016/b978-0-12-809406-8.00010-4>

- Crawford, C. B., & Quinn, B. (2017b). Microplastic identification techniques. In *Microplastic Pollutants* (pp. 219–267). Elsevier. <https://doi.org/10.1016/b978-0-12-809406-8.00010-4>
- Cristiane Thé, P., & Michael Johnson, Mas. A. (2024). *User's Guide Screening Air Dispersion Model (SCREEN3)*. <http://www.weblakes.com>
- Crutchett, T. W., & Bornt, K. R. (2024). A simple overflow density separation method that recovers >95% of dense microplastics from sediment. *MethodsX*, 12, 102638. <https://doi.org/10.1016/j.mex.2024.102638>
- Cunningham, E. M., Rico Seijo, N., Altieri, K. E., Audh, R. R., Burger, J. M., Bornman, T. G., Fawcett, S., Gwinnett, C. M. B., Osborne, A. O., & Woodall, L. C. (2022). The transport and fate of microplastic fibres in the Antarctic: The role of multiple global processes. *Frontiers in Marine Science*, 9. <https://doi.org/10.3389/fmars.2022.1056081>
- D Allen, S Allen, J Sonke, V. P. (2019). *Allen: Atmospheric transport and our planetary boundary...* - Google Scholar. Springer Water.
- da Costa, J. P., Avellan, A., Tubić, A., Duarte, A. C., & Rocha-Santos, T. (2024). Understanding Interface Exchanges for Assessing Environmental Sorption of Additives from Microplastics: Current Knowledge and Perspectives. *Molecules*, 29(2), 333. <https://doi.org/10.3390/molecules29020333>
- da Silva, V. H., Murphy, F., Amigo, J. M., Stedmon, C., & Strand, J. (2020). Classification and Quantification of Microplastics (<100 µm) Using a Focal Plane Array–Fourier Transform Infrared Imaging System and Machine Learning. *Analytical Chemistry*, 92(20), 13724–13733. <https://doi.org/10.1021/acs.analchem.0c01324>
- Daffi, R. E., Chaimang, A. N., & Alfa, M. I. (2020). Environmental Impact of Open Burning of Municipal Solid Wastes Dumps in Parts of Jos Metropolis, Nigeria. *Journal of Engineering Research and Reports*, 30–43. <https://doi.org/10.9734/jerr/2020/v12i317083>
- Dauda, G., & Ali, A. M. (2024). Delineating leachate-groundwater interaction at Gyadi-Gyadi dumpsite, Kano, using natural electromagnetic (EM) field detector and Vertical Electrical Sounding (VES). *Geosystems and Geoenvironment*, 3(4), 100303. <https://doi.org/10.1016/j.geogeo.2024.100303>
- Davoli, E., Fattore, E., Paiano, V., Colombo, A., Palmiotto, M., Rossi, A. N., Il Grande, M., & Fanelli, R. (2010). Waste management health risk assessment: A case study of a solid waste landfill in South Italy. *Waste Management*, 30(8–9), 1608–1613. <https://doi.org/10.1016/j.wasman.2009.10.013>
- Dawson, A. L., Santana, M. F. M., Nelis, J. L. D., & Motti, C. A. (2023). Taking control of microplastics data: A comparison of control and blank data correction methods. *Journal of Hazardous Materials*, 443, 130218. <https://doi.org/10.1016/j.jhazmat.2022.130218>
- De Frond, H., Cowger, W., Renick, V., Brander, S., Primpke, S., Sukumaran, S., Elkhatab, D., Barnett, S., Navas-Moreno, M., Rickabaugh, K., Vollnhals, F., O'Donnell, B., Lusher, A., Lee, E., Lao, W., Amarpuri, G., Sarau, G., & Christiansen, S. (2023). What determines accuracy of chemical identification when using microspectroscopy for the analysis of microplastics? *Chemosphere*, 313, 137300. <https://doi.org/10.1016/j.chemosphere.2022.137300>

- Dehghani, S., Moore, F., & Akhbarizadeh, R. (2017). Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. *Environmental Science and Pollution Research*, 24(25), 20360–20371. <https://doi.org/10.1007/s11356-017-9674-1>
- Ding, J., Sun, C., He, C., Li, J., Ju, P., & Li, F. (2021). Microplastics in four bivalve species and basis for using bivalves as bioindicators of microplastic pollution. *Science of The Total Environment*, 782, 146830. <https://doi.org/10.1016/j.scitotenv.2021.146830>
- Dinku, T. (2019). Challenges with availability and quality of climate data in Africa. In *Extreme Hydrology and Climate Variability* (pp. 71–80). Elsevier. <https://doi.org/10.1016/B978-0-12-815998-9.00007-5>
- Dispersion, A. (2021). ADMS-Screen.
- Donaldson, K., & Tran, C. L. (2002). INFLAMMATION CAUSED BY PARTICLES AND FIBERS. *Inhalation Toxicology*, 14(1), 5–27. <https://doi.org/10.1080/089583701753338613>
- Dong, H., Wang, X., Xu, L., Ding, J., & Wania, F. (2023). A Flow-through Passive Sampler for Microplastics in Air. *Environmental Science & Technology*, 57(6), 2362–2370. <https://doi.org/10.1021/acs.est.2c07016>
- Dorau, K., Hoppe, M., Rückamp, D., Köser, J., Scheeder, G., Scholz, K., & Fries, E. (2023). Status quo of operation procedures for soil sampling to analyze microplastics. *Microplastics and Nanoplastics*, 3(1), 15. <https://doi.org/10.1186/s43591-023-00063-5>
- dos Santos Cerqueira, J., de Albuquerque, H. N., & de Assis Salviano de Sousa, F. (2019). Atmospheric pollutants: modeling with Aermid software. *Air Quality, Atmosphere and Health*, 12(1), 21–32. <https://doi.org/10.1007/s11869-018-0626-9>
- Dris, R. (2017). *First assessement of sources and fate of macro and micro plastics in urban hydrosystems : Case of Paris megacity*. 248.
- Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V., & Tassin, B. (2017a). A first overview of textile fibers, including microplastics, in indoor and outdoor environments. *Environmental Pollution*, 221, 453–458. <https://doi.org/10.1016/j.envpol.2016.12.013>
- Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V., & Tassin, B. (2017b). A first overview of textile fibers, including microplastics, in indoor and outdoor environments. *Environmental Pollution*, 221, 453–458. <https://doi.org/10.1016/j.envpol.2016.12.013>
- Dris, R., Gasperi, J., Saad, M., Mirande, C., & Tassin, B. (2016). Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin*, 104(1–2), 290–293. <https://doi.org/10.1016/j.marpolbul.2016.01.006>
- Duan, J., Bolan, N., Li, Y., Ding, S., Atugoda, T., Vithanage, M., Sarkar, B., Tsang, D. C. W., & Kirkham, M. B. (2021). Weathering of microplastics and interaction with other coexisting constituents in terrestrial and aquatic environments. *Water Research*, 196, 117011. <https://doi.org/10.1016/j.watres.2021.117011>

- Dumbili, E., & Henderson, L. (2020). The challenge of plastic pollution in Nigeria. In *Plastic Waste and Recycling* (pp. 569–583). Elsevier. <https://doi.org/10.1016/B978-0-12-817880-5.00022-0>
- Dümichen, E., Barthel, A.-K., Braun, U., Bannick, C. G., Brand, K., Jekel, M., & Senz, R. (2015). Analysis of polyethylene microplastics in environmental samples, using a thermal decomposition method. *Water Research*, 85, 451–457. <https://doi.org/10.1016/j.watres.2015.09.002>
- Duru, R. U., Ikpeama, E. E., & Ibekwe, J. A. (2019). Challenges and prospects of plastic waste management in Nigeria. *Waste Disposal & Sustainable Energy*, 1(2), 117–126. <https://doi.org/10.1007/s42768-019-00010-2>
- Dybwad, M., Skogan, G., & Blatny, J. M. (2014). Comparative Testing and Evaluation of Nine Different Air Samplers: End-to-End Sampling Efficiencies as Specific Performance Measurements for Bioaerosol Applications. *Aerosol Science and Technology*, 48(3), 282–295. <https://doi.org/10.1080/02786826.2013.871501>
- Ebere, E. C., Wirnkor, V. A., Ngozi, V. E., & Chukwuemeka, I. S. (2019). Macrodebris and microplastics pollution in Nigeria: first report on abundance, distribution and composition. *Environmental Analysis Health and Toxicology*, 34(4), e2019012. <https://doi.org/10.5620/eaht.e2019012>
- Ebrahimi, P., Abbasi, S., Pashaei, R., Bogusz, A., & Oleszczuk, P. (2022). Investigating impact of physicochemical properties of microplastics on human health: A short bibliometric analysis and review. *Chemosphere*, 289, 133146. <https://doi.org/10.1016/j.chemosphere.2021.133146>
- Edo, C., Fernández-Alba, A. R., Vejsnæs, F., van der Steen, J. J. M., Fernández-Piñas, F., & Rosal, R. (2021). Honeybees as active samplers for microplastics. *Science of The Total Environment*, 767, 144481. <https://doi.org/10.1016/j.scitotenv.2020.144481>
- Elbir, T., Dincer, F., & Muezzinoglu, A. (2007). Evaluation of Measured And Predicted Odor Concentrations around A Meat Packaging And Rendering Plant. *Environmental Engineering Science*, 24(3), 313–320. <https://doi.org/10.1089/ees.2005.0105>
- Emenike, E. C., Okorie, C. J., Ojeyemi, T., Egbemhenghe, A., Iwuozor, K. O., Saliu, O. D., Okoro, H. K., & Adeniyi, A. G. (2023). From oceans to dinner plates: The impact of microplastics on human health. *Heliyon*, 9(10), e20440. <https://doi.org/10.1016/j.heliyon.2023.e20440>
- Eneh, O. C., & Agbazue, V. C. (2011). Protection of Nigeria's Environment: A Critical Policy Review. *Journal of Environmental Science and Technology*, 4(5), 490–497. <https://doi.org/10.3923/jest.2011.490.497>
- Enyoh, C. E., Verla, A. W., Ohiagu, F. O., & Enyoh, E. C. (2023). Progress and future perspectives of microplastic research in Nigeria. *International Journal of Environmental Analytical Chemistry*, 103(9), 1971–1981. <https://doi.org/10.1080/03067319.2021.1887161>
- Enyoh, C. E., Verla, A. W., Verla, E. N., Ibe, F. C., & Amaobi, C. E. (2019). Airborne microplastics: a review study on method for analysis, occurrence, movement and risks. In *Environmental Monitoring and Assessment* (Vol. 191, Issue 11, pp. 1–17). Springer International Publishing. <https://doi.org/10.1007/s10661-019-7842-0>

- EPA. (1992). *Screening Procedures for Estimating the Air Quality Impact of Stationary Sources, Revised*.
- EPA. (1995). *SCREEN3 Model User's Guide*.
- EPA. (1999). *Compendium of Methods for the Determination of Inorganic Compounds in Ambient Air Compendium Method IO-2.1 SAMPLING OF AMBIENT AIR FOR TOTAL SUSPENDED PARTICULATE MATTER (SPM) AND PM 10 USING HIGH VOLUME (HV) SAMPLER Method IO-2.1 Acknowledgments*.
- EPA. (2021). *AERSCREEN User's Guide*.
- EPA. (2024). Air Quality Dispersion Modeling - Screening Models. *Air Quality Dispersion Modeling - Screening Models*.
- Esders, E. M., Georgi, C., Babel, W., Held, A., & Thomas, C. K. (2024). Vertical concentrations gradients and transport of airborne microplastics in wind tunnel experiments. *Aerosol Research*, 2(2), 235–243. <https://doi.org/10.5194/ar-2-235-2024>
- Esmaeilbeigi, M., Kazemi, A., Gholizadeh, M., & Rezaeiye, R. D. (2023). Microplastics and heavy metals contamination in Atropus atropos and associated health risk assessment in the northwest of the Persian Gulf, Iran. *Regional Studies in Marine Science*, 57, 102750. <https://doi.org/10.1016/j.rsma.2022.102750>
- Eso, R., Kasmia, S., & Mursyid, A. (2017). SIMULATION OF TERMINAL VELOCITY AND VISCOSITY OF FLUID USING ANALYSIS VIDEO WITH TRACKER AND MODELLUS. In *International Journal of Education and Research* (Vol. 5, Issue 10). www.ijern.com
- Essen, C. (2019). FG to ban single user plastic. *The Guarding*.
- Evangelou, N., Grythe, H., Klimont, Z., Heyes, C., Eckhardt, S., Lopez-Aparicio, S., & Stohl, A. (2020a). Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications*, 11(1), 3381. <https://doi.org/10.1038/s41467-020-17201-9>
- Evangelou, N., Grythe, H., Klimont, Z., Heyes, C., Eckhardt, S., Lopez-Aparicio, S., & Stohl, A. (2020b). Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications*, 11(1), 3381. <https://doi.org/10.1038/s41467-020-17201-9>
- Evangelou, N., Grythe, H., Klimont, Z., Heyes, C., Eckhardt, S., Lopez-Aparicio, S., & Stohl, A. (2020c). Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications*, 11(1). <https://doi.org/10.1038/s41467-020-17201-9>
- Evangelou, N., Tichý, O., Eckhardt, S., Zwaafink, C. G., & Brahney, J. (2022). Sources and fate of atmospheric microplastics revealed from inverse and dispersion modelling: From global emissions to deposition. *Journal of Hazardous Materials*, 432, 128585. <https://doi.org/10.1016/j.jhazmat.2022.128585>
- Evangelou, I., Tatsii, D., Bucci, S., & Stohl, A. (2024). Atmospheric Resuspension of Microplastics from Bare Soil Regions. *Environmental Science & Technology*, 58(22), 9741–9749. <https://doi.org/10.1021/acs.est.4c01252>
- Falakdin, P., Lopez-Rosales, A., Andrade, J., Terzaghi, E., Di Guardo, A., & Muniategui-Lorenzo, S. (2024). Comparison of microplastic type, size, and composition in atmospheric and foliage samples in an urban scenario.

- Environmental Pollution*, 349, 123911.
<https://doi.org/10.1016/j.envpol.2024.123911>
- Farid, M. A. A., Norrrahim, M. N. F., Lease, J., & Andou, Y. (2024). *Microplastic Reduction in Municipal Solid Waste (MSW) Landfill* (pp. 57–86).
https://doi.org/10.1007/978-3-031-55665-4_3
- Fatahi, M., Akdogan, G., Dorfling, C., & Van Wyk, P. (2021). Numerical Study of Microplastic Dispersal in Simulated Coastal Waters Using CFD Approach. *Water*, 13(23), 3432. <https://doi.org/10.3390/w13233432>
- Feenstra, B., Papapostolou, V., Hasheminassab, S., Zhang, H., Boghossian, B. Der, Cocker, D., & Polidori, A. (2019). Performance evaluation of twelve low-cost PM2.5 sensors at an ambient air monitoring site. *Atmospheric Environment*, 216, 116946. <https://doi.org/10.1016/j.atmosenv.2019.116946>
- Fernandes, A. C. (2021). Reductive depolymerization as an efficient methodology for the conversion of plastic waste into value-added compounds. *Green Chemistry*, 23(19), 7330–7360. <https://doi.org/10.1039/D1GC01634B>
- Ferreira, C. S. S., Adama-Ajonye, O., Ikenna, A. E., & Kalantari, Z. (2023). Groundwater quality in the vicinity of a dumpsite in Lagos metropolis, Nigeria. *Geography and Sustainability*, 4(4), 379–390.
<https://doi.org/10.1016/j.geosus.2023.09.005>
- FGN. (2021). Federal Republic of Nigeria Official Gazette. In *Federal Republic of Nigeria Official Gazette*.
- FGN Gazette. (2009). *FEDERAL REPUBLIC OF NIGERIA OFFICIAL GAZETTE*.
- Fries, E., Dekiff, J. H., Willmeyer, J., Nuelle, M. T., Ebert, M., & Remy, D. (2013). Identification of polymer types and additives in marine microplastic particles using pyrolysis-GC/MS and scanning electron microscopy. *Environmental Sciences: Processes and Impacts*, 15(10), 1949–1956.
<https://doi.org/10.1039/c3em00214d>
- Galafton, C., Thonemann, N., & Vijver, M. G. (2025). It is time to develop characterization factors for terrestrial plastic pollution impacts on ecosystems in life cycle impact assessment – a systematic review identifying knowledge gaps. *The International Journal of Life Cycle Assessment*, 30(5), 994–1010.
<https://doi.org/10.1007/s11367-025-02446-7>
- Galloway, T. S. (2015). Micro- and Nano-plastics and Human Health. In *Marine Anthropogenic Litter* (pp. 343–366). Springer International Publishing.
https://doi.org/10.1007/978-3-319-16510-3_13
- Gan, Q., Cui, J., & Jin, B. (2023). Environmental microplastics: Classification, sources, fates, and effects on plants. *Chemosphere*, 313, 137559.
<https://doi.org/10.1016/j.chemosphere.2022.137559>
- Gao, W., Deng, X.-J., Zhang, J., Qi, L., Zhao, X.-Q., & Zhang, P.-Y. (2023a). Assessment of quality control measures in the monitoring of microplastic: a critical review. *Environmental Pollutants and Bioavailability*, 35(1).
<https://doi.org/10.1080/26395940.2023.2203349>
- Gao, W., Deng, X.-J., Zhang, J., Qi, L., Zhao, X.-Q., & Zhang, P.-Y. (2023b). Assessment of quality control measures in the monitoring of microplastic: a critical review. *Environmental Pollutants and Bioavailability*, 35(1).
<https://doi.org/10.1080/26395940.2023.2203349>

- Gardon, T., Reisser, C., Soye, C., Quillien, V., & Le Moullac, G. (2018). Microplastics Affect Energy Balance and Gametogenesis in the Pearl Oyster *Pinctada margaritifera*. *Environmental Science & Technology*, 52(9), 5277–5286. <https://doi.org/10.1021/acs.est.8b00168>
- Gaston, E., Woo, M., Steele, C., Sukumaran, S., & Anderson, S. (2020). Microplastics Differ Between Indoor and Outdoor Air Masses: Insights from Multiple Microscopy Methodologies. *Applied Spectroscopy*, 74(9), 1079–1098. <https://doi.org/10.1177/0003702820920652>
- Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever made. *Science Advances*, 3(7), 25–29. <https://doi.org/10.1126/sciadv.1700782>
- Ghosh, S., Sinha, J. K., Ghosh, S., Vashisth, K., Han, S., & Bhaskar, R. (2023). Microplastics as an Emerging Threat to the Global Environment and Human Health. *Sustainability*, 15(14), 10821. <https://doi.org/10.3390/su151410821>
- Gigault, J., Halle, A. ter, Baudrimont, M., Pascal, P.-Y., Gauffre, F., Phi, T.-L., El Hadri, H., Grassl, B., & Reynaud, S. (2018a). Current opinion: What is a nanoplastic? *Environmental Pollution*, 235, 1030–1034. <https://doi.org/10.1016/j.envpol.2018.01.024>
- Gigault, J., Halle, A. ter, Baudrimont, M., Pascal, P.-Y., Gauffre, F., Phi, T.-L., El Hadri, H., Grassl, B., & Reynaud, S. (2018b). Current opinion: What is a nanoplastic? *Environmental Pollution*, 235, 1030–1034. <https://doi.org/10.1016/j.envpol.2018.01.024>
- Giordano, M. R., Malings, C., Pandis, S. N., Presto, A. A., McNeill, V. F., Westervelt, D. M., Beekmann, M., & Subramanian, R. (2021). From low-cost sensors to high-quality data: A summary of challenges and best practices for effectively calibrating low-cost particulate matter mass sensors. *Journal of Aerosol Science*, 158, 105833. <https://doi.org/10.1016/j.jaerosci.2021.105833>
- Gliaudelytė, U., Persson, M., & Daukantienė, V. (2024). Impact of textile composition, structure, and treatment on microplastic release during washing: a review. *Textile Research Journal*. <https://doi.org/10.1177/00405175241260066>
- González-Pleiter, M., Edo, C., Aguilera, Á., Viúdez-Moreiras, D., Pulido-Reyes, G., González-Toril, E., Osuna, S., de Diego-Castilla, G., Leganés, F., Fernández-Piñas, F., & Rosal, R. (2021). Occurrence and transport of microplastics sampled within and above the planetary boundary layer. *Science of the Total Environment*, 761, 143213. <https://doi.org/10.1016/j.scitotenv.2020.143213>
- Goßmann, I., Herzke, D., Held, A., Schulz, J., Nikiforov, V., Georgi, C., Evangeliou, N., Eckhardt, S., Gerdt, G., Wurl, O., & Scholz-Böttcher, B. M. (2023). Occurrence and backtracking of microplastic mass loads including tire wear particles in northern Atlantic air. *Nature Communications*, 14(1), 3707. <https://doi.org/10.1038/s41467-023-39340-5>
- Goudarzi et. al. (2017). *An assessment on dispersion of carbon monoxide from a cement factory - admin-A-10-1-85-b6aad34*.
- Gunarathne, V., Phillips, A. J., Zanoletti, A., Rajapaksha, A. U., Vithanage, M., Di Maria, F., Pivato, A., Korzeniewska, E., & Bontempi, E. (2024). Environmental pitfalls and associated human health risks and ecological impacts from landfill leachate contaminants: Current evidence, recommended interventions and future

- directions. *Science of The Total Environment*, 912, 169026. <https://doi.org/10.1016/j.scitotenv.2023.169026>
- Guo, J. J., Huang, X. P., Xiang, L., Wang, Y. Z., Li, Y. W., Li, H., Cai, Q. Y., Mo, C. H., & Wong, M. H. (2020). Source, migration and toxicology of microplastics in soil. In *Environment International*. <https://doi.org/10.1016/j.envint.2019.105263>
- Guo, J.-J., Huang, X.-P., Xiang, L., Wang, Y.-Z., Li, Y.-W., Li, H., Cai, Q.-Y., Mo, C.-H., & Wong, M.-H. (2020). Source, migration and toxicology of microplastics in soil. *Environment International*, 137, 105263. <https://doi.org/10.1016/j.envint.2019.105263>
- Gwinnett, C. M. B., Osborne, A. O., & Jackson, A. R. W. (2021a). The application of tape lifting for microplastic pollution monitoring. *Environmental Advances*, 5, 100066. <https://doi.org/10.1016/j.envadv.2021.100066>
- Gwinnett, C. M. B., Osborne, A. O., & Jackson, A. R. W. (2021b). The application of tape lifting for microplastic pollution monitoring. *Environmental Advances*, 5, 100066. <https://doi.org/10.1016/j.envadv.2021.100066>
- Hanik, N., Amstutz, V., & Zinn, M. (2019). Microplastics - From anthropogenic to natural. *Chimia*, 73(10), 841–843. <https://doi.org/10.2533/chimia.2019.841>
- Haq, A. ul, Nadeem, Q., Farooq, A., Irfan, N., Ahmad, M., & Ali, M. R. (2019). Assessment of AERMOD modeling system for application in complex terrain in Pakistan. *Atmospheric Pollution Research*, 10(5), 1492–1497. <https://doi.org/10.1016/j.apr.2019.04.006>
- Haque, F., & Fan, C. (2023). Fate of microplastics under the influence of climate change. *IScience*, 26(9), 107649. <https://doi.org/10.1016/j.isci.2023.107649>
- Hartmann, N. B., Hüffer, T., Thompson, R. C., Hassellöv, M., Verschoor, A., Daugaard, A. E., Rist, S., Karlsson, T., Brennholt, N., Cole, M., Herrling, M. P., Hess, M. C., Ivleva, N. P., Lusher, A. L., & Wagner, M. (2019). Are We Speaking the Same Language? Recommendations for a Definition and Categorization Framework for Plastic Debris. *Environmental Science & Technology*, 53(3), 1039–1047. <https://doi.org/10.1021/acs.est.8b05297>
- Heather Wylie. (2021). Atmospheric Dispersion and Pollution Transport. *Air Quality Portal*.
- Hengstmann, E., & Fischer, E. K. (2019a). Nile red staining in microplastic analysis—proposal for a reliable and fast identification approach for large microplastics. *Environmental Monitoring and Assessment*, 191(10), 612. <https://doi.org/10.1007/s10661-019-7786-4>
- Hengstmann, E., & Fischer, E. K. (2019b). Nile red staining in microplastic analysis—proposal for a reliable and fast identification approach for large microplastics. *Environmental Monitoring and Assessment*, 191(10), 612. <https://doi.org/10.1007/s10661-019-7786-4>
- Henkel, C., Hüffer, T., & Hofmann, T. (2022). Polyvinyl Chloride Microplastics Leach Phthalates into the Aquatic Environment over Decades. *Environmental Science & Technology*, 56(20), 14507–14516. <https://doi.org/10.1021/acs.est.2c05108>
- Hernandez, L. M., Yousefi, N., & Tufenkji, N. (2017). Are There Nanoplastics in Your Personal Care Products? *Environmental Science & Technology Letters*, 4(7), 280–285. <https://doi.org/10.1021/acs.estlett.7b00187>

- Herzke, D., Schmidt, N., Schulze, D., Eckhardt, S., & Evangeliou, N. (2025). *Comparison of Atmospheric Microplastic in remote and urban locations in Norway; occurrence, composition and sources*. <https://doi.org/10.5194/egusphere-egu25-20432>
- Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2012). Microplastics in the marine environment: A review of the methods used for identification and quantification. *Environmental Science and Technology*, 46(6), 3060–3075. <https://doi.org/10.1021/es2031505>
- Hinata, H., Kuwae, M., Tsugeki, N., Masumoto, I., Tani, Y., Hatada, Y., Kawamata, H., Mase, A., Kasamo, K., Sukenaga, K., & Suzuki, Y. (2023a). A 75-year history of microplastic fragment accumulation rates in a semi-enclosed hypoxic basin. *Science of The Total Environment*, 854, 158751. <https://doi.org/10.1016/j.scitotenv.2022.158751>
- Hinata, H., Kuwae, M., Tsugeki, N., Masumoto, I., Tani, Y., Hatada, Y., Kawamata, H., Mase, A., Kasamo, K., Sukenaga, K., & Suzuki, Y. (2023b). A 75-year history of microplastic fragment accumulation rates in a semi-enclosed hypoxic basin. *Science of The Total Environment*, 854, 158751. <https://doi.org/10.1016/j.scitotenv.2022.158751>
- Hong, Y., Wu, S., & Wei, G. (2023). Adverse effects of microplastics and nanoplastics on the reproductive system: A comprehensive review of fertility and potential harmful interactions. *Science of The Total Environment*, 903, 166258. <https://doi.org/10.1016/j.scitotenv.2023.166258>
- Hossain, Md. J., AftabUddin, S., Akhter, F., Nusrat, N., Rahaman, A., Sikder, M. N. A., Monwar, Md. M., Chowdhury, M. S. N., Jiang, S., Shi, H., & Zhang, J. (2022). Surface water, sediment, and biota: The first multi-compartment analysis of microplastics in the Karnafully river, Bangladesh. *Marine Pollution Bulletin*, 180, 113820. <https://doi.org/10.1016/j.marpolbul.2022.113820>
- Hu, K., Zhou, P., Yang, Y., Hall, T., Nie, G., Yao, Y., Duan, X., & Wang, S. (2022). Degradation of Microplastics by a Thermal Fenton Reaction. *ACS ES&T Engineering*, 2(1), 110–120. <https://doi.org/10.1021/acsestengg.1c00323>
- Hu, T., He, P., Yang, Z., Wang, W., Zhang, H., Shao, L., & Lü, F. (2022a). Emission of airborne microplastics from municipal solid waste transfer stations in downtown. *Science of The Total Environment*, 828, 154400. <https://doi.org/10.1016/J.SCITOTENV.2022.154400>
- Hu, T., He, P., Yang, Z., Wang, W., Zhang, H., Shao, L., & Lü, F. (2022b). Emission of airborne microplastics from municipal solid waste transfer stations in downtown. *Science of The Total Environment*, 828, 154400. <https://doi.org/10.1016/J.SCITOTENV.2022.154400>
- Hu, T., & Yoshie, R. (2020). Effect of atmospheric stability on air pollutant concentration and its generalization for real and idealized urban block models based on field observation data and wind tunnel experiments. *Journal of Wind Engineering and Industrial Aerodynamics*, 207, 104380. <https://doi.org/10.1016/j.jweia.2020.104380>
- Hu, Y., Guo, S., Shen, D., Shentu, J., Lu, L., Qi, S., Zhu, M., & Long, Y. (2024). Overlooked risk of microplastics from municipal solid waste–storage site.

- Frontiers of Environmental Science & Engineering*, 18(10), 125.
<https://doi.org/10.1007/s11783-024-1885-3>
- Huang, D., Chen, H., Shen, M., Tao, J., Chen, S., Yin, L., Zhou, W., Wang, X., Xiao, R., & Li, R. (2022). Recent advances on the transport of microplastics/nanoplastics in abiotic and biotic compartments. *Journal of Hazardous Materials*, 438, 129515.
<https://doi.org/10.1016/j.jhazmat.2022.129515>
- Huang, M., Si, C., Qiu, C., & Wang, G. (2024). Microplastics analysis: from qualitative to quantitative. *Environmental Science: Advances*, 3(12), 1652–1668.
<https://doi.org/10.1039/D4VA00244J>
- Huang, Y., He, T., Yan, M., Yang, L., Gong, H., Wang, W., Qing, X., & Wang, J. (2021). Atmospheric transport and deposition of microplastics in a subtropical urban environment. *Journal of Hazardous Materials*, 416, 126168.
<https://doi.org/10.1016/j.jhazmat.2021.126168>
- Huang, Z., Hu, B., & Wang, H. (2023). Analytical methods for microplastics in the environment: a review. *Environmental Chemistry Letters*, 21(1), 383–401.
<https://doi.org/10.1007/s10311-022-01525-7>
- Hussaini, A., & Matazu, B. M. (2023). AN OVERVIEW OF KEY IMPROVEMENTS BY THE NIGERIAN METEOROLOGICAL AGENCY FOR THE MODERNISATION OF METEOROLOGICAL SERVICES IN NIGERIA. *Science World Journal*, 18(1), 2023. www.scienceworldjournal.org
- IAQM. (2020). *IAQM u Position Statement Use of Screening Tools Position Statement- Use of Screening Tools for Preliminary Determination of the Significance of Pollutant Sources Use of Screening Tools for Preliminary Determination of the Significance of Pollutant Sources*. www.iaqm.co.uk|1
- Idowu, G. A., Oriji, A. Y., Olorunfemi, K. O., Sunday, M. O., Sogbanmu, T. O., Bodunwa, O. K., Shokunbi, O. S., & Aiyesanmi, A. F. (2024). Why Nigeria should ban single-use plastics: Excessive microplastic pollution of the water, sediments and fish species in Osun River, Nigeria. *Journal of Hazardous Materials Advances*, 13, 100409. <https://doi.org/10.1016/j.hazadv.2024.100409>
- Ihenetu, S. C., Li, G., Mo, Y., & Jacques, K. J. (2024). Impacts of microplastics and urbanization on soil health: An urgent concern for sustainable development. *Green Analytical Chemistry*, 8, 100095.
<https://doi.org/10.1016/j.greeac.2024.100095>
- Imhof, H. K., Laforsch, C., Wiesheu, A. C., Schmid, J., Anger, P. M., Niessner, R., & Ivleva, N. P. (2016). Pigments and plastic in limnetic ecosystems: A qualitative and quantitative study on microparticles of different size classes. *Water Research*, 98, 64–74. <https://doi.org/10.1016/j.watres.2016.03.015>
- Isaac Anyaogu. (2022, March). Nigeria's 32m-ton annual solid waste ripe for investments. *Businessday.Ng*.
- Isioye, O. A., Combrinck, L., Botai, J. O., & Moses, M. (2022). Assessing the impact of variations in atmospheric water vapour content over Nigeria from GNSS measurements. *South African Journal of Geomatics*, 8(1), 30–56.
<https://doi.org/10.4314/sajg.v8i1.3>
- Izquierdo, R., & Avila, A. (2012). Comparison of collection methods to determine atmospheric deposition in a rural Mediterranean site (NE Spain). *Journal of*

- Atmospheric Chemistry*, 69(4), 351–368. <https://doi.org/10.1007/s10874-012-9246-1>
- Jalón-Rojas, I., Romero-Ramírez, A., Fauquembergue, K., Rossignol, L., Cachot, J., Sous, D., & Morin, B. (2022). Effects of Biofilms and Particle Physical Properties on the Rising and Settling Velocities of Microplastic Fibers and Sheets. *Environmental Science & Technology*, 56(12), 8114–8123. <https://doi.org/10.1021/acs.est.2c01302>
- Jan Kole, P., Löhr, A. J., Van Belleghem, F. G. A. J., & Ragas, A. M. J. (2017). Wear and tear of tyres: A stealthy source of microplastics in the environment. *International Journal of Environmental Research and Public Health*, 14(10). <https://doi.org/10.3390/ijerph14101265>
- Jawad, M. A. (2018). *Evaluation of a novel biosensor for bioaerosol sampling and modelling School of Water, Energy and Environment*.
- Jayavel, S., Govindaraju, B., Michael, J. R., & Viswanathan, B. (2024a). Impacts of micro and nanoplastics on human health. *Bulletin of the National Research Centre*, 48(1), 110. <https://doi.org/10.1186/s42269-024-01268-1>
- Jayavel, S., Govindaraju, B., Michael, J. R., & Viswanathan, B. (2024b). Impacts of micro and nanoplastics on human health. *Bulletin of the National Research Centre*, 48(1), 110. <https://doi.org/10.1186/s42269-024-01268-1>
- Jenkins, T., Persaud, B. D., Cowger, W., Szigeti, K., Roche, D. G., Clary, E., Slowinski, S., Lei, B., Abeynayaka, A., Nyadjro, E. S., Maes, T., Thornton Hampton, L., Bergmann, M., Aherne, J., Mason, S. A., Honek, J. F., Rezanezhad, F., Lusher, A. L., Booth, A. M., ... Van Cappellen, P. (2022). Current State of Microplastic Pollution Research Data: Trends in Availability and Sources of Open Data. *Frontiers in Environmental Science*, 10. <https://doi.org/10.3389/fenvs.2022.912107>
- Jeyasanta, K. I., Sathish, N., Patterson, J., & Edward, J. K. P. (2020). Macro-, meso- and microplastic debris in the beaches of Tuticorin district, Southeast coast of India. *Marine Pollution Bulletin*, 154, 111055. <https://doi.org/10.1016/j.marpolbul.2020.111055>
- Joo, H., Park, K., Lee, K., & Ndegwa, P. M. (2015). Mass concentration coupled with mass loading rate for evaluating PM_{2.5} pollution status in the atmosphere: A case study based on dairy barns. *Environmental Pollution*, 207, 374–380. <https://doi.org/10.1016/j.envpol.2015.09.048>
- K, M. B., Natesan, U., R, V., R, P. K., R, R., & S, S. (2021a). Spatial distribution of microplastic concentration around landfill sites and its potential risk on groundwater. *Chemosphere*, 277. <https://doi.org/10.1016/j.chemosphere.2021.130263>
- K, M. B., Natesan, U., R, V., R, P. K., R, R., & S, S. (2021b). Spatial distribution of microplastic concentration around landfill sites and its potential risk on groundwater. *Chemosphere*, 277. <https://doi.org/10.1016/j.chemosphere.2021.130263>
- Kabir, M. S., Wang, H., Luster-Teasley, S., Zhang, L., & Zhao, R. (2023). Microplastics in landfill leachate: Sources, detection, occurrence, and removal. *Environmental Science and Ecotechnology*, 16, 100256. <https://doi.org/10.1016/j.ese.2023.100256>

- Kang, Y., Aye, L., Ngo, T. D., & Zhou, J. (2022). Performance evaluation of low-cost air quality sensors: A review. *Science of The Total Environment*, 818, 151769. <https://doi.org/10.1016/j.scitotenv.2021.151769>
- Kannankai, M. P., & Devipriya, S. P. (2024a). Atmospheric microplastic deposition in a coastal city of India: The influence of a landfill source on monsoon winds. *Science of the Total Environment*, 908. <https://doi.org/10.1016/j.scitotenv.2023.168235>
- Kannankai, M. P., & Devipriya, S. P. (2024b). Atmospheric microplastic deposition in a coastal city of India: The influence of a landfill source on monsoon winds. *Science of The Total Environment*, 908, 168235. <https://doi.org/10.1016/j.scitotenv.2023.168235>
- Käppler, A., Fischer, M., Scholz-Böttcher, B. M., Oberbeckmann, S., Labrenz, M., Fischer, D., Eichhorn, K. J., & Voit, B. (2018). Comparison of μ -ATR-FTIR spectroscopy and py-GCMS as identification tools for microplastic particles and fibers isolated from river sediments. *Analytical and Bioanalytical Chemistry*, 410(21), 5313–5327. <https://doi.org/10.1007/s00216-018-1185-5>
- KARA, G. (2020). Development of a highly sensitive sampler for atmospheric monitoring of 8-H ambient ozone concentrations. *Environmental Engineering Research*, 25(6), 938–945. <https://doi.org/10.4491/eer.2019.411>
- Karami, A., Golieskardi, A., Keong Choo, C., Larat, V., Galloway, T. S., & Salamatinia, B. (2017). The presence of microplastics in commercial salts from different countries. *Scientific Reports*, 7(1), 46173. <https://doi.org/10.1038/srep46173>
- Kaydi, N., Jorfi, S., Takdastan, A., Jaafarzadeh Haghighifard, N., & Khafaie, M. A. (2024). Source identification and apportionment of ambient air microplastics: a systematic review. *Discover Applied Sciences*, 7(1), 41. <https://doi.org/10.1007/s42452-024-06422-y>
- Kernchen, S., Löder, M. G. J., Fischer, F., Fischer, D., Moses, S. R., Georgi, C., Nölscher, A. C., Held, A., & Laforsch, C. (2022a). Airborne microplastic concentrations and deposition across the Weser River catchment. *Science of the Total Environment*, 818. <https://doi.org/10.1016/j.scitotenv.2021.151812>
- Kernchen, S., Löder, M. G. J., Fischer, F., Fischer, D., Moses, S. R., Georgi, C., Nölscher, A. C., Held, A., & Laforsch, C. (2022b). Airborne microplastic concentrations and deposition across the Weser River catchment. *Science of The Total Environment*, 818, 151812. <https://doi.org/10.1016/j.scitotenv.2021.151812>
- Kernchen, S., Schmalz, H., Löder, M. G. J., Georgi, C., Einhorn, A., Greiner, A., Nölscher, A. C., Laforsch, C., & Held, A. (2024). Atmospheric deposition studies of microplastics in Central Germany. *Air Quality, Atmosphere & Health*. <https://doi.org/10.1007/s11869-024-01571-w>
- Khan, A., Sharma, S., Chowdhury, K. R., & Sharma, P. (2022). A novel seasonal index-based machine learning approach for air pollution forecasting. *Environmental Monitoring and Assessment*, 194(6), 429. <https://doi.org/10.1007/s10661-022-10092-x>
- Khan, M. T., Cheng, Y. L., Hafeez, S., Tsang, Y. F., Yang, J., & Nawab, A. (2020). Microplastics in Wastewater. In *Handbook of Microplastics in the Environment* (pp. 1–33). Springer International Publishing. https://doi.org/10.1007/978-3-030-10618-8_39-1

- Khaniabadi, Y. O., Sicard, P., Taiwo, A. M., De Marco, A., Esmaeili, S., & Rashidi, R. (2018). Modeling of particulate matter dispersion from a cement plant: Upwind-downwind case study. *Journal of Environmental Chemical Engineering*, 6(2), 3104–3110. <https://doi.org/10.1016/j.jece.2018.04.022>
- Kibria, Md. G., Masuk, N. I., Safayet, R., Nguyen, H. Q., & Mourshed, M. (2023). Plastic Waste: Challenges and Opportunities to Mitigate Pollution and Effective Management. *International Journal of Environmental Research*, 17(1), 20. <https://doi.org/10.1007/s41742-023-00507-z>
- Kim, S.-Y., Blanco, M. N., Bi, J., Larson, T. V., & Sheppard, L. (2023). Exposure assessment for air pollution epidemiology: A scoping review of emerging monitoring platforms and designs. *Environmental Research*, 223, 115451. <https://doi.org/10.1016/j.envres.2023.115451>
- Kitole, F. A., Ojo, T. O., Emenike, C. U., Khumalo, N. Z., Elhindi, K. M., & Kassem, H. S. (2024). The Impact of Poor Waste Management on Public Health Initiatives in Shanty Towns in Tanzania. *Sustainability*, 16(24), 10873. <https://doi.org/10.3390/su162410873>
- Kjærsgaard, E. R., Hasager, F., Petters, S. S., Glasius, M., & Bilde, M. (2024). Bubble-mediated generation of airborne nanoplastic particles. *Environmental Science: Processes & Impacts*, 26(7), 1216–1226. <https://doi.org/10.1039/D4EM00124A>
- Klein, M., & Fischer, E. K. (2019a). Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. *Science of the Total Environment*, 685, 96–103. <https://doi.org/10.1016/j.scitotenv.2019.05.405>
- Klein, M., & Fischer, E. K. (2019b). Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. *Science of the Total Environment*, 685, 96–103. <https://doi.org/10.1016/j.scitotenv.2019.05.405>
- Knobloch, E., Ruffell, H., Aves, A., Pantos, O., Gaw, S., & Revell, L. E. (2021). Comparison of Deposition Sampling Methods to Collect Airborne Microplastics in Christchurch, New Zealand. *Water, Air, & Soil Pollution*, 232(4), 133. <https://doi.org/10.1007/s11270-021-05080-9>
- Koelmans, A. A., Redondo-Hasselerharm, P. E., Nor, N. H. M., de Ruijter, V. N., Mintenig, S. M., & Kooi, M. (2022a). Risk assessment of microplastic particles. *Nature Reviews Materials*, 7(2), 138–152. <https://doi.org/10.1038/s41578-021-00411-y>
- Koelmans, A. A., Redondo-Hasselerharm, P. E., Nor, N. H. M., de Ruijter, V. N., Mintenig, S. M., & Kooi, M. (2022b). Risk assessment of microplastic particles. *Nature Reviews Materials*, 7(2), 138–152. <https://doi.org/10.1038/s41578-021-00411-y>
- Kooi, M., & Koelmans, A. A. (2019). Simplifying Microplastic via Continuous Probability Distributions for Size, Shape, and Density. *Environmental Science & Technology Letters*, 6(9), 551–557. <https://doi.org/10.1021/acs.estlett.9b00379>
- Kooi, M., Primpke, S., Mintenig, S. M., Lorenz, C., Gerdt, G., & Koelmans, A. A. (2021). Characterizing the multidimensionality of microplastics across environmental compartments. *Water Research*, 202, 117429. <https://doi.org/10.1016/j.watres.2021.117429>
- Kota, S. H., Park, C., Hale, M. C., Werner, N. D., Schade, G. W., & Ying, Q. (2014). Estimation of VOC emission factors from flux measurements using a receptor

- model and footprint analysis. *Atmospheric Environment*, 82, 24–35. <https://doi.org/10.1016/j.atmosenv.2013.09.052>
- Kumar, A., Mishra, S., Pandey, R., Yu, Z. G., Kumar, M., Khoo, K. S., Thakur, T. K., & Show, P. L. (2023). Microplastics in terrestrial ecosystems: Un-ignorable impacts on soil characterises, nutrient storage and its cycling. *TrAC Trends in Analytical Chemistry*, 158, 116869. <https://doi.org/10.1016/j.trac.2022.116869>
- Kumar, A., Thakur, A. K., Gaurav, G. K., Klemeš, J. J., Sandhwar, V. K., Pant, K. K., & Kumar, R. (2023). A critical review on sustainable hazardous waste management strategies: a step towards a circular economy. *Environmental Science and Pollution Research*, 30(48), 105030–105055. <https://doi.org/10.1007/s11356-023-29511-8>
- Kumar, K., Umapathi, R., Ghoreishian, S. M., Tiwari, J. N., Hwang, S. K., Huh, Y. S., Venkatesu, P., Shetti, N. P., & Aminabhavi, T. M. (2023). Microplastics and biobased polymers to combat plastics waste. *Chemosphere*, 341, 140000. <https://doi.org/10.1016/j.chemosphere.2023.140000>
- Kumar, N., Lamba, M., Pachar, A. K., Yadav, S., & Acharya, A. (2024). Microplastics – A Growing Concern as Carcinogens in Cancer Etiology: Emphasis on Biochemical and Molecular Mechanisms. *Cell Biochemistry and Biophysics*. <https://doi.org/10.1007/s12013-024-01436-0>
- Kumar, P., Tiwari, S., Uguz, S., Li, Z., Gonzalez, J., Wei, L., Samuel, R. S., Zhang, Y., & Yang, X. (2024). Bioaerosols downwind from animal feeding operations: A comprehensive review. *Journal of Hazardous Materials*, 480, 135825. <https://doi.org/10.1016/j.jhazmat.2024.135825>
- Kurniawan, S. B. (2024). *Challenges of Microplastic Research in Asia and Africa* (pp. 73–88). https://doi.org/10.1007/978-3-031-64253-1_4
- Lai, H., Liu, X., & Qu, M. (2022). Nanoplastics and Human Health: Hazard Identification and Biointerface. *Nanomaterials*, 12(8), 1298. <https://doi.org/10.3390/nano12081298>
- Lakes. (2024). *AERSCREEN View – Modeling Fumigation*.
- Lara-Topete, G. O., Castanier-Rivas, J. D., Bahena-Osorio, M. F., Krause, S., Larsen, J. R., Loge, F. J., Mahlkecht, J., Gradilla-Hernández, M. S., & González-López, M. E. (2024). Compounding one problem with another? A look at biodegradable microplastics. *Science of The Total Environment*, 944, 173735. <https://doi.org/10.1016/j.scitotenv.2024.173735>
- Lavery, P. S., Oldham, C. E., & Ghisalberti, M. (2001). The use of Fick's First Law for predicting porewater nutrient fluxes under diffusive conditions. *Hydrological Processes*, 15(13), 2435–2451. <https://doi.org/10.1002/hyp.297>
- Lawal, K. A., Olaniyan, E., Ishiyaku, I., Hirons, L. C., Thompson, E., Talib, J., Boulton, V. L., Ogungbenro, S. B., Gbode, I. E., Ajayi, V. O., Okogbue, E. C., Adefisan, E. A., Indasi, V. S., Youds, L., Nkiaka, E., Stone, D. A., Nzekwu, R., Folorunso, O., Oyedepo, J. A., ... Woolnough, S. J. (2021). Progress and Challenges of Demand-Led Co-Produced Sub-Seasonal-to-Seasonal (S2S) Climate Forecasts in Nigeria. *Frontiers in Climate*, 3. <https://doi.org/10.3389/fclim.2021.712502>
- Lear, G., Kingsbury, J. M., Franchini, S., Gambarini, V., Maday, S. D. M., Wallbank, J. A., Weaver, L., & Pantos, O. (2021). Plastics and the microbiome: impacts and

- solutions. *Environmental Microbiome*, 16(1), 2. <https://doi.org/10.1186/s40793-020-00371-w>
- Leelőssy, Á., Molnár, F., Izsák, F., Havasi, Á., Lagzi, I., & Mészáros, R. (2014). Dispersion modeling of air pollutants in the atmosphere: a review. *Open Geosciences*, 6(3). <https://doi.org/10.2478/s13533-012-0188-6>
- Lehmann, M., Oehlschlägel, L. M., Häusl, F. P., Held, A., & Gekle, S. (2021). Ejection of marine microplastics by raindrops: a computational and experimental study. *Microplastics and Nanoplastics*, 1(1), 18. <https://doi.org/10.1186/s43591-021-00018-8>
- Lenz, R., Enders, K., Stedmon, C. A., Mackenzie, D. M. A., & Nielsen, T. G. (2015a). A critical assessment of visual identification of marine microplastic using Raman spectroscopy for analysis improvement. *Marine Pollution Bulletin*, 100(1), 82–91. <https://doi.org/10.1016/j.marpolbul.2015.09.026>
- Lenz, R., Enders, K., Stedmon, C. A., Mackenzie, D. M. A., & Nielsen, T. G. (2015b). A critical assessment of visual identification of marine microplastic using Raman spectroscopy for analysis improvement. *Marine Pollution Bulletin*, 100(1), 82–91. <https://doi.org/10.1016/j.marpolbul.2015.09.026>
- Leroy, C., Maro, D., Hébert, D., Solier, L., Rozet, M., Le Cavelier, S., & Connan, O. (2010). A study of the atmospheric dispersion of a high release of krypton-85 above a complex coastal terrain, comparison with the predictions of Gaussian models (Briggs, Doury, ADMS4). *Journal of Environmental Radioactivity*, 101(11), 937–944. <https://doi.org/10.1016/j.jenvrad.2010.06.011>
- Leslie, H. A., Brandsma, S. H., van Velzen, M. J. M., & Vethaak, A. D. (2017). Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International*, 101, 133–142. <https://doi.org/10.1016/j.envint.2017.01.018>
- Li, C., Gao, Y., He, S., Chi, H.-Y., Li, Z.-C., Zhou, X.-X., & Yan, B. (2021). Quantification of Nanoplastic Uptake in Cucumber Plants by Pyrolysis Gas Chromatography/Mass Spectrometry. *Environmental Science & Technology Letters*, 8(8), 633–638. <https://doi.org/10.1021/acs.estlett.1c00369>
- Li, D., Sheerin, E. D., Shi, Y., Xiao, L., Yang, L., Boland, J. J., & Wang, J. J. (2022). Alcohol Pretreatment to Eliminate the Interference of Micro Additive Particles in the Identification of Microplastics Using Raman Spectroscopy. *Environmental Science & Technology*, 56(17), 12158–12168. <https://doi.org/10.1021/acs.est.2c01551>
- Li, J., Liu, H., & Paul Chen, J. (2018). Microplastics in freshwater systems: A review on occurrence, environmental effects, and methods for microplastics detection. *Water Research*, 137, 362–374. <https://doi.org/10.1016/j.watres.2017.12.056>
- Li, J., Liu, Y., Gao, Y., Li, X., & Gong, Y. (2023). Study on the Extraction Method of Microplastic System in Textile Wastewater. *Polymers*, 15(6), 1394. <https://doi.org/10.3390/polym15061394>
- Li, X., Wu, F., Zhang, C., & Wang, T. (2024). The Effects of Rainfall Events on the Composition and Diversity of Microplastics on Beaches in Xiamen City on a Short-Term Scale. *Toxics*, 12(5), 375. <https://doi.org/10.3390/toxics12050375>

- Li, Y., Chen, L., Zhou, N., Chen, Y., Ling, Z., & Xiang, P. (2024). Microplastics in the human body: A comprehensive review of exposure, distribution, migration mechanisms, and toxicity. *Science of The Total Environment*, 946, 174215. <https://doi.org/10.1016/j.scitotenv.2024.174215>
- Li, Y., Shao, L., Wang, W., Zhang, M., Feng, X., Li, W., & Zhang, D. (2020). Airborne fiber particles: Types, size and concentration observed in Beijing. *Science of the Total Environment*, 705, 135967. <https://doi.org/10.1016/j.scitotenv.2019.135967>
- Li, Y., Tao, L., Wang, Q., Wang, F., Li, G., & Song, M. (2023). Potential Health Impact of Microplastics: A Review of Environmental Distribution, Human Exposure, and Toxic Effects. *Environment & Health*, 1(4), 249–257. <https://doi.org/10.1021/envhealth.3c00052>
- Liang, J., Ji, F., Abdullah, A. L. B., Qin, W., Zhu, T., Tay, Y. J., Li, Y., & Han, M. (2024a). Micro/nano-plastics impacts in cardiovascular systems across species. *Science of The Total Environment*, 942, 173770. <https://doi.org/10.1016/j.scitotenv.2024.173770>
- Liang, J., Ji, F., Abdullah, A. L. B., Qin, W., Zhu, T., Tay, Y. J., Li, Y., & Han, M. (2024b). Micro/nano-plastics impacts in cardiovascular systems across species. *Science of The Total Environment*, 942, 173770. <https://doi.org/10.1016/j.scitotenv.2024.173770>
- Liao, Z., Ji, X., Ma, Y., Lv, B., Huang, W., Zhu, X., Fang, M., Wang, Q., Wang, X., Dahlgren, R., & Shang, X. (2021). Airborne microplastics in indoor and outdoor environments of a coastal city in Eastern China. *Journal of Hazardous Materials*, 417, 126007. <https://doi.org/10.1016/j.jhazmat.2021.126007>
- Liguori, F., Moreno-Marrodán, C., & Barbaro, P. (2021). Valorisation of plastic waste via metal-catalysed depolymerisation. *Beilstein Journal of Organic Chemistry*, 17, 589–621. <https://doi.org/10.3762/bjoc.17.53>
- Lin, C., Solera Garcia, M. A., Timmis, R., & Jones, K. C. (2011). Design and laboratory testing of a new flow-through directional passive air sampler for ambient particulate matter. *Journal of Environmental Monitoring*, 13(3), 753. <https://doi.org/10.1039/c0em00210k>
- Lin, X., Zhang, S., Yang, S., Zhang, R., Shi, X., & Song, L. (2023). A landfill serves as a critical source of microplastic pollution and harbors diverse plastic biodegradation microbial species and enzymes: Study in large-scale landfills, China. *Journal of Hazardous Materials*, 457, 131676. <https://doi.org/10.1016/j.jhazmat.2023.131676>
- Liu, B., Jiang, Q., Qiu, Z., Liu, L., Wei, R., Zhang, X., & Xu, H. (2022). Process analysis of microplastic degradation using activated PMS and Fenton reagents. *Chemosphere*, 298, 134220. <https://doi.org/10.1016/j.chemosphere.2022.134220>
- Liu, C., Li, J., Zhang, Y., Wang, L., Deng, J., Gao, Y., Yu, L., Zhang, J., & Sun, H. (2019a). Widespread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. *Environment International*, 128, 116–124. <https://doi.org/10.1016/j.envint.2019.04.024>
- Liu, C., Li, J., Zhang, Y., Wang, L., Deng, J., Gao, Y., Yu, L., Zhang, J., & Sun, H. (2019b). Widespread distribution of PET and PC microplastics in dust in urban

- China and their estimated human exposure. *Environment International*, 128, 116–124. <https://doi.org/10.1016/j.envint.2019.04.024>
- Liu, K., Wang, X., Fang, T., Xu, P., Zhu, L., & Li, D. (2019). Source and potential risk assessment of suspended atmospheric microplastics in Shanghai. *Science of the Total Environment*, 675, 462–471. <https://doi.org/10.1016/j.scitotenv.2019.04.110>
- Liu, K., Wu, T., Wang, X., Song, Z., Zong, C., Wei, N., & Li, D. (2019a). Consistent Transport of Terrestrial Microplastics to the Ocean through Atmosphere. *Environmental Science and Technology*, 53(18), 10612–10619. <https://doi.org/10.1021/acs.est.9b03427>
- Liu, K., Wu, T., Wang, X., Song, Z., Zong, C., Wei, N., & Li, D. (2019b). Consistent Transport of Terrestrial Microplastics to the Ocean through Atmosphere. *Environmental Science and Technology*, 53(18), 10612–10619. <https://doi.org/10.1021/acs.est.9b03427>
- Lusher, A. (2015). Microplastics in the marine environment: Distribution, interactions and effects. In *Marine Anthropogenic Litter* (pp. 245–307). Springer International Publishing. https://doi.org/10.1007/978-3-319-16510-3_10
- Lusher, A. L., Hurley, R., Arp, H. P. H., Booth, A. M., Bråte, I. L. N., Gabrielsen, G. W., Gomiero, A., Gomes, T., Grøsvik, B. E., Green, N., Haave, M., Hallanger, I. G., Halsband, C., Herzke, D., Joner, E. J., Kögel, T., Rakkestad, K., Ranneklev, S. B., Wagner, M., & Olsen, M. (2021). Moving forward in microplastic research: A Norwegian perspective. *Environment International*, 157, 106794. <https://doi.org/10.1016/j.envint.2021.106794>
- Ma, J., & Singhirunnusorn, W. (2012). Distribution and Health Risk Assessment of Heavy Metals in Surface Dusts of Maha Sarakham Municipality. *Procedia - Social and Behavioral Sciences*, 50(July), 280–293. <https://doi.org/10.1016/j.sbspro.2012.08.034>
- Maes, T., Jessop, R., Wellner, N., Haupt, K., & Mayes, A. G. (2017). A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red. *Scientific Reports*, 7. <https://doi.org/10.1038/srep44501>
- Mahrt, L. (2010). Computing turbulent fluxes near the surface: Needed improvements. *Agricultural and Forest Meteorology*, 150(4), 501–509. <https://doi.org/10.1016/j.agrformet.2010.01.015>
- Majewsky, M., Bitter, H., Eiche, E., & Horn, H. (2016). Determination of microplastic polyethylene (PE) and polypropylene (PP) in environmental samples using thermal analysis (TGA-DSC). *Science of the Total Environment*, 568, 507–511. <https://doi.org/10.1016/j.scitotenv.2016.06.017>
- Makhdoumi, P., Naghshbandi, M., Ghaderzadeh, K., Mirzabeigi, M., Yazdanbakhsh, A., & Hossini, H. (2021). Micro-plastic occurrence in bottled vinegar: Qualification, quantification and human risk exposure. *Process Safety and Environmental Protection*, 152, 404–413. <https://doi.org/10.1016/j.psep.2021.06.022>
- Malyan, V., Kumar, V., & Sahu, M. (2023). Significance of sources and size distribution on calibration of low-cost particle sensors: Evidence from a field sampling campaign. *Journal of Aerosol Science*, 168, 106114. <https://doi.org/10.1016/j.jaerosci.2022.106114>

- Mangla, S., Pathak, G., & Bhan, V. (2025). Microplastics contamination in soil, water, air, and environmental health risk assessment. In *Microplastics* (pp. 3–30). Elsevier. <https://doi.org/10.1016/B978-0-443-29804-2.00001-9>
- Manikanda Bharath, K., Muthulakshmi, A. L., & Natesan, U. (2023). Microplastic contamination around the landfills: Distribution, characterization and threats: A review. *Current Opinion in Environmental Science & Health*, 31, 100422. <https://doi.org/10.1016/j.coesh.2022.100422>
- Manisalidis, I., Stavropoulou, E., Stavropoulos, A., & Bezirtzoglou, E. (2020). Environmental and Health Impacts of Air Pollution: A Review. *Frontiers in Public Health*, 8. <https://doi.org/10.3389/fpubh.2020.00014>
- Mansa, R., & Zou, S. (2021). Thermogravimetric analysis of microplastics: A mini review. *Environmental Advances*, 5, 100117. <https://doi.org/10.1016/j.envadv.2021.100117>
- Mark Willett, P. J. (2019). An introduction to image analysis using ImageJ. *University of Southampton*.
- Markovic, M. Z., Prokop, S., Staebler, R. M., Liggio, J., & Harner, T. (2015). Evaluation of the particle infiltration efficiency of three passive samplers and the PS-1 active air sampler. *Atmospheric Environment*, 112, 289–293. <https://doi.org/10.1016/j.atmosenv.2015.04.051>
- Martina, M., & Trini Castelli, S. (2023). Modelling the Potential Long-Range Dispersion of Atmospheric Microplastics Reaching a Remote Site. *Atmospheric Environment*, 312, 120044. <https://doi.org/10.1016/j.atmosenv.2023.120044>
- Matavos-Aramyan, S. (2024). Addressing the microplastic crisis: A multifaceted approach to removal and regulation. *Environmental Advances*, 17, 100579. <https://doi.org/10.1016/j.envadv.2024.100579>
- Mateos-Cárdenas, A., O'Halloran, J., van Pelt, F. N. A. M., & Jansen, M. A. K. (2020). Rapid fragmentation of microplastics by the freshwater amphipod *Gammarus duebeni* (Lillj.). *Scientific Reports*, 10(1). <https://doi.org/10.1038/s41598-020-69635-2>
- Materić, D., Ludewig, E., Brunner, D., Röckmann, T., & Holzinger, R. (2021). Nanoplastics transport to the remote, high-altitude Alps. *Environmental Pollution*, 288, 117697. <https://doi.org/10.1016/j.envpol.2021.117697>
- Mathias Agbo Jr. (2025, March). *The Problem with Solid Waste Management in Nigeria's Low-Income Neighbourhoods*. Earth.Org .
- Maurizi, L., Simon-Sánchez, L., Vianello, A., Nielsen, A. H., & Vollertsen, J. (2024). Every breath you take: High concentration of breathable microplastics in indoor environments. *Chemosphere*, 361, 142553. <https://doi.org/10.1016/j.chemosphere.2024.142553>
- Mbachu, O., Jenkins, G., Pratt, C., & Kaparaju, P. (2021). Enzymatic purification of microplastics in soil. *MethodsX*, 8, 101254. <https://doi.org/10.1016/j.mex.2021.101254>
- Meyers, N., Catarino, A. I., Declercq, A. M., Brenan, A., Devriese, L., Vandegehuchte, M., De Witte, B., Janssen, C., & Everaert, G. (2022). Microplastic detection and identification by Nile red staining: Towards a semi-automated, cost- and time-effective technique. *Science of The Total Environment*, 823, 153441. <https://doi.org/10.1016/j.scitotenv.2022.153441>

- Michel, A. P. M., Morrison, A. E., Preston, V. L., Marx, C. T., Colson, B. C., & White, H. K. (2020). Rapid Identification of Marine Plastic Debris via Spectroscopic Techniques and Machine Learning Classifiers. *Environmental Science & Technology*, 54(17), 10630–10637. <https://doi.org/10.1021/acs.est.0c02099>
- Miller, E., Sedlak, M., Lin, D., Box, C., Holleman, C., Rochman, C. M., & Sutton, R. (2021). Recommended best practices for collecting, analyzing, and reporting microplastics in environmental media: Lessons learned from comprehensive monitoring of San Francisco Bay. *Journal of Hazardous Materials*, 409, 124770. <https://doi.org/10.1016/j.jhazmat.2020.124770>
- Mohan, M., Bhati, S., Sreenivas, A., & Marrapu, P. (2011). Performance evaluation of AERMOD and ADMS-Urban for total suspended particulate matter concentrations in Megacity Delhi. *Aerosol and Air Quality Research*, 11(7), 883–894. <https://doi.org/10.4209/aaqr.2011.05.0065>
- Molina Rueda, E., Carter, E., L'Orange, C., Quinn, C., & Volckens, J. (2023). Size-Resolved Field Performance of Low-Cost Sensors for Particulate Matter Air Pollution. *Environmental Science & Technology Letters*, 10(3), 247–253. <https://doi.org/10.1021/acs.estlett.3c00030>
- Monteiro, S. S., Rocha-Santos, T., Prata, J. C., Duarte, A. C., Girão, A. V., Lopes, P., Cristovão, T., & da Costa, J. P. (2022). A straightforward method for microplastic extraction from organic-rich freshwater samples. *Science of The Total Environment*, 815, 152941. <https://doi.org/10.1016/j.scitotenv.2022.152941>
- Mukhanov, V. S., Litvinyuk, D. A., Sakhon, E. G., Bagaev, A. V., Veerasingam, S., & Venkatachalapathy, R. (2019). A new method for analyzing microplastic particle size distribution in marine environmental samples. *Ecologica Montenegrina*, 23, 77–86. <https://doi.org/10.37828/em.2019.23.10>
- Müller, A., Kocher, B., Altmann, K., & Braun, U. (2022). Determination of tire wear markers in soil samples and their distribution in a roadside soil. *Chemosphere*, 294, 133653. <https://doi.org/10.1016/j.chemosphere.2022.133653>
- Munno, K., Lusher, A. L., Minor, E. C., Gray, A., Ho, K., Hankett, J., T Lee, C.-F., Primpke, S., McNeish, R. E., Wong, C. S., & Rochman, C. (2023a). Patterns of microparticles in blank samples: A study to inform best practices for microplastic analysis. *Chemosphere*, 333, 138883. <https://doi.org/10.1016/j.chemosphere.2023.138883>
- Munno, K., Lusher, A. L., Minor, E. C., Gray, A., Ho, K., Hankett, J., T Lee, C.-F., Primpke, S., McNeish, R. E., Wong, C. S., & Rochman, C. (2023b). Patterns of microparticles in blank samples: A study to inform best practices for microplastic analysis. *Chemosphere*, 333, 138883. <https://doi.org/10.1016/j.chemosphere.2023.138883>
- Murashov, V., Geraci, C. L., Schulte, P. A., & Howard, J. (2021). Nano- and microplastics in the workplace. *Journal of Occupational and Environmental Hygiene*, 18(10–11), 489–494. <https://doi.org/10.1080/15459624.2021.1976413>
- Murthy, B. S., Latha, R., Tiwari, A., Rathod, A., Singh, S., & Beig, G. (2020). Impact of mixing layer height on air quality in winter. *Journal of Atmospheric and Solar-Terrestrial Physics*, 197, 105157. <https://doi.org/10.1016/j.jastp.2019.105157>
- Na, S.-H., Kim, M.-J., Kim, J., Batool, R., Cho, K., Chung, J., Lee, S., & Kim, E.-J. (2024). Fate and potential risks of microplastic fibers and fragments in water and

- wastewater treatment processes. *Journal of Hazardous Materials*, 463, 132938. <https://doi.org/10.1016/j.jhazmat.2023.132938>
- Naderi Beni, N., Karimifard, S., Gilley, J., Messer, T., Schmidt, A., & Bartelt-Hunt, S. (2023). Higher concentrations of microplastics in runoff from biosolid-amended croplands than manure-amended croplands. *Communications Earth & Environment*, 4(1), 42. <https://doi.org/10.1038/s43247-023-00691-y>
- Nalakurthi, N. V. S. R., Abimbola, I., Ahmed, T., Anton, I., Riaz, K., Ibrahim, Q., Banerjee, A., Tiwari, A., & Gharbia, S. (2024). Challenges and Opportunities in Calibrating Low-Cost Environmental Sensors. *Sensors*, 24(11), 3650. <https://doi.org/10.3390/s24113650>
- Narváez Valderrama, J. F., Baek, K., Molina, F. J., & Allan, I. J. (2016). Implications of observed PBDE diffusion coefficients in low density polyethylene and silicone rubber. *Environmental Science: Processes & Impacts*, 18(1), 87–94. <https://doi.org/10.1039/C5EM00507H>
- Nava, V., & Leoni, B. (2021). Comparison of different procedures for separating microplastics from sediments. *Water (Switzerland)*, 13(20). <https://doi.org/10.3390/w13202854>
- Nayanathara Thathsarani Pilapitiya, P. G. C., & Ratnayake, A. S. (2024). The world of plastic waste: A review. *Cleaner Materials*, 11, 100220. <https://doi.org/10.1016/j.clema.2024.100220>
- Ncube, L. K., Ude, A. U., Ogunmuyiwa, E. N., Zulkifli, R., & Beas, I. N. (2021). An Overview of Plastic Waste Generation and Management in Food Packaging Industries. *Recycling*, 6(1), 12. <https://doi.org/10.3390/recycling6010012>
- Nene, A., Sadeghzade, S., Viaroli, S., Yang, W., Uchenna, U. P., Kandwal, A., Liu, X., Somani, P., & Galluzzi, M. (2025). Recent advances and future technologies in nano-microplastics detection. *Environmental Sciences Europe*, 37(1), 7. <https://doi.org/10.1186/s12302-024-01044-y>
- Newton, E., Ersoy, D., Rodriguez, E., & Lamb, B. K. (2024). Development of Company-Specific Emission Factors with Confidence Intervals for Natural Gas Customer Meters in Southern California. *Environmental Science & Technology*, 58(16), 6954–6963. <https://doi.org/10.1021/acs.est.3c10316>
- Noiki, A. A., Afolalu, S. A., Yusuf, O. O., Emeteri, M. E., Ongbali, S. O., Oloyede, O. R., Joseph, O. O., & Banjo, S. O. (2021). Impact Assessment of the Current Waste Management Practices in Nigeria. *IOP Conference Series: Materials Science and Engineering*, 1107(1), 012172. <https://doi.org/10.1088/1757-899X/1107/1/012172>
- Nolan, C., & Amiaqm, M. (2020). *SCREENING AIR MODELLING ASSESSMENT OF EMISSIONS FROM AN ASPHALT BATCHING PLANT AT TONROE QUARRY, ARDARHAN, CO. GALWAY TOBIN Consulting Engineers Technical Report*

Prepared By.
- Nuelle, M. T., Dekiff, J. H., Remy, D., & Fries, E. (2014). A new analytical approach for monitoring microplastics in marine sediments. *Environmental Pollution*, 184, 161–169. <https://doi.org/10.1016/j.envpol.2013.07.027>

- Nwafor, N., & Walker, T. R. (2020). Plastic Bags Prohibition Bill: A developing story of crass legalism aiming to reduce plastic marine pollution in Nigeria. *Marine Policy*, 120, 104160. <https://doi.org/10.1016/j.marpol.2020.104160>
- Nyaga, M. P., Shabaka, S., Oh, S., Osman, D. M., Yuan, W., Zhang, W., & Yang, Y. (2024). Microplastics in aquatic ecosystems of Africa: A comprehensive review and meta-analysis. *Environmental Research*, 248, 118307. <https://doi.org/10.1016/j.envres.2024.118307>
- OBIDIMMA & AZUBUIKE. (2023). *Examination of the Legal Framework for Plastic Pollution Management in Nigeria* EXAMINATION OF THE LEGAL FRAMEWORK FOR PLASTIC POLLUTION MANAGEMENT IN NIGERIA*. <https://aculj.acuoyo.net/index.php/lj/article/download/21/22>
- O'Brien, S., Okoffo, E. D., O'Brien, J. W., Ribeiro, F., Wang, X., Wright, S. L., Samanipour, S., Rauert, C., Toapanta, T. Y. A., Albarracin, R., & Thomas, K. V. (2020). Airborne emissions of microplastic fibres from domestic laundry dryers. *Science of The Total Environment*, 747, 141175. <https://doi.org/10.1016/j.scitotenv.2020.141175>
- O'Brien, S., Rauert, C., Ribeiro, F., Okoffo, E. D., Burrows, S. D., O'Brien, J. W., Wang, X., Wright, S. L., & Thomas, K. V. (2023). There's something in the air: A review of sources, prevalence and behaviour of microplastics in the atmosphere. *Science of The Total Environment*, 874, 162193. <https://doi.org/10.1016/j.scitotenv.2023.162193>
- Ogunkan, D. V. (2022). Achieving sustainable environmental governance in Nigeria: A review for policy consideration. *Urban Governance*, 2(1), 212–220. <https://doi.org/10.1016/j.ugj.2022.04.004>
- Okedere, O. B., Olalekan, A. P., Fakinle, B. S., Elehinafe, F. B., Odunlami, O. A., & Sonibare, J. A. (2019). Urban air pollution from the open burning of municipal solid waste. *Environmental Quality Management*, tqem.21633. <https://doi.org/10.1002/tqem.21633>
- Oladejo, A. (2017). *Analysis of microplastics and their removal from water*.
- Oliveri Conti, G., Rapisarda, P., & Ferrante, M. (2024). Relationship between climate change and environmental microplastics: a one health vision for the platysphere health. *One Health Advances*, 2(1), 17. <https://doi.org/10.1186/s44280-024-00049-9>
- Ololade, I. A., Apata, A., Oladoja, N. A., Alabi, B. A., & Ololade, O. O. (2023). Microplastic particles in river sediments and water of southwestern Nigeria: insights on the occurrence, seasonal distribution, composition, and source apportionment. *Environmental Science and Pollution Research*, 31(1), 1314–1330. <https://doi.org/10.1007/s11356-023-31118-y>
- Oni, B. A., Ayeni, A. O., Agboola, O., Oguntade, T., & Obanla, O. (2020). Comparing microplastics contaminants in (dry and raining) seasons for Ox- Bow Lake in Yenagoa, Nigeria. *Ecotoxicology and Environmental Safety*, 198, 110656. <https://doi.org/10.1016/j.ecoenv.2020.110656>
- Oura, M., Ohba, R., Robins, A., & Kato, S. (2018). Validation Study for an Atmospheric Dispersion Model, Using Effective Source Heights Determined from Wind Tunnel Experiments in Nuclear Safety Analysis. *Atmosphere*, 9(3), 111. <https://doi.org/10.3390/atmos9030111>

- Pasquier, G., Doyen, P., Carlesi, N., & Amara, R. (2022). An innovative approach for microplastic sampling in all surface water bodies using an aquatic drone. *Heliyon*, 8(11), e11662. <https://doi.org/10.1016/j.heliyon.2022.e11662>
- Pathak, G., Nichter, M., Hardon, A., Moyer, E., Latkar, A., Simbaya, J., Pakasi, D., Taqueban, E., & Love, J. (2023). Plastic pollution and the open burning of plastic wastes. *Global Environmental Change*, 80, 102648. <https://doi.org/10.1016/j.gloenvcha.2023.102648>
- Patterson, S. L., Rusiecki, J. A., Barnes, S. L., Heller, J. M., Sutphin, J. B., & Kluchinsky, T. A. (2010). Effectiveness, suitability, and performance testing of the SKC Deployable Particulate Sampler (DPS) as compared to the currently deployed Airmetrics MiniVol portable air sampler. *Journal of Environmental Health*, 73(3), 16–22.
- Pegado, T., Andrades, R., Noletto-Filho, E., Franceschini, S., Soares, M., Chelazzi, D., Russo, T., Martellini, T., Barone, A., Cincinelli, A., & Giarizzo, T. (2024). Meso- and microplastic composition, distribution patterns and drivers: A snapshot of plastic pollution on Brazilian beaches. *Science of The Total Environment*, 907, 167769. <https://doi.org/10.1016/j.scitotenv.2023.167769>
- Pekdogan, T., Yildizhan, H., & Ameen, A. (2024). Unveiling the Air Quality Impacts of Municipal Solid Waste Disposal: An Integrative Study of On-Site Measurements and Community Perceptions. *Atmosphere*, 15(4), 410. <https://doi.org/10.3390/atmos15040410>
- Periyasamy, A. P., & Tehrani-Bagha, A. (2022). A review on microplastic emission from textile materials and its reduction techniques. *Polymer Degradation and Stability*, 199, 109901. <https://doi.org/10.1016/j.polymdegradstab.2022.109901>
- Peter Sundt, Per-Erik Schultze, F. S. (2014). Sources of microplastic- pollution to the marine environment. *Mepex, Norwegian Environment Agency*, 1–108.
- Petrović, M., Mihajlović, I., Tubić, A., & Novaković, M. (2023). Microplastics in municipal solid waste landfills. *Current Opinion in Environmental Science & Health*, 31, 100428. <https://doi.org/10.1016/j.coesh.2022.100428>
- Pham, D. T., Kim, J., Lee, S.-H., Kim, J., Kim, D., Hong, S., Jung, J., & Kwon, J.-H. (2023). Analysis of microplastics in various foods and assessment of aggregate human exposure via food consumption in Korea. *Environmental Pollution*, 322, 121153. <https://doi.org/10.1016/j.envpol.2023.121153>
- Pilechi, A., Mohammadian, A., & Murphy, E. (2022). A numerical framework for modeling fate and transport of microplastics in inland and coastal waters. *Marine Pollution Bulletin*, 184, 114119. <https://doi.org/10.1016/j.marpolbul.2022.114119>
- Pimentel, J. C., Avila, R., & Lourenço, A. G. (1975). Respiratory disease caused by synthetic fibres: a new occupational disease. *Thorax*, 30(2), 204 LP – 219. <https://doi.org/10.1136/thx.30.2.204>
- Pinder, R. W., Klopp, J. M., Kleiman, G., Hagler, G. S. W., Awe, Y., & Terry, S. (2019). Opportunities and challenges for filling the air quality data gap in low- and middle-income countries. *Atmospheric Environment*, 215, 116794. <https://doi.org/10.1016/j.atmosenv.2019.06.032>
- Pona, H. T., Xiaoli, D., Ayantobo, O. O., & Narh Daniel Tetteh. (2021). Environmental health situation in Nigeria: current status and future needs. *Heliyon*, 7(3), e06330. <https://doi.org/10.1016/j.heliyon.2021.e06330>

- Prasittisopin, L., Ferdous, W., & Kamchoom, V. (2023). Microplastics in construction and built environment. *Developments in the Built Environment*, 15, 100188. <https://doi.org/10.1016/j.dibe.2023.100188>
- Prata, J. C., Castro, J. L., da Costa, J. P., Cerqueira, M., Duarte, A. C., & Rocha-Santos, T. (2020). Airborne Microplastics. In *Handbook of Microplastics in the Environment* (pp. 1–25). Springer International Publishing. https://doi.org/10.1007/978-3-030-10618-8_37-1
- Prata, J. C., da Costa, J. P., Lopes, I., Duarte, A. C., & Rocha-Santos, T. (2020). Environmental exposure to microplastics: An overview on possible human health effects. *Science of The Total Environment*, 702, 134455. <https://doi.org/10.1016/j.scitotenv.2019.134455>
- Preston, C. A., McKenna Neuman, C. L., & Aherne, J. (2023). Effects of Shape and Size on Microplastic Atmospheric Settling Velocity. *Environmental Science & Technology*, 57(32), 11937–11947. <https://doi.org/10.1021/acs.est.3c03671>
- Primpke, S., Fischer, M., Lorenz, C., Gerdts, G., & Scholz-Böttcher, B. M. (2020). Comparison of pyrolysis gas chromatography/mass spectrometry and hyperspectral FTIR imaging spectroscopy for the analysis of microplastics. *Analytical and Bioanalytical Chemistry*, 412(30), 8283–8298. <https://doi.org/10.1007/s00216-020-02979-w>
- Primpke, S., Lorenz, C., Rascher-Friesenhausen, R., & Gerdts, G. (2017). An automated approach for microplastics analysis using focal plane array (FPA) FTIR microscopy and image analysis. *Analytical Methods*, 9(9), 1499–1511. <https://doi.org/10.1039/c6ay02476a>
- Pukos, S., Maszczyk, P., Dąbrowski, K., Zebrowski, M. L., & Babkiewicz, E. (2023). The effect of planktivorous fish on the vertical flux of polystyrene microplastics. *The European Zoological Journal*, 90(1), 401–413. <https://doi.org/10.1080/24750263.2023.2217199>
- Puteri, M. N., Gew, L. T., Ong, H. C., & Ming, L. C. (2025). Technologies to eliminate microplastic from water: Current approaches and future prospects. *Environment International*, 109397. <https://doi.org/10.1016/j.envint.2025.109397>
- Quinn, B., Murphy, F., & Ewins, C. (2017). Validation of density separation for the rapid recovery of microplastics from sediment. *Analytical Methods*, 9(9), 1491–1498. <https://doi.org/10.1039/C6AY02542K>
- Radford, F., Zapata-Restrepo, L. M., Horton, A. A., Hudson, M. D., Shaw, P. J., & Williams, I. D. (2021a). Developing a systematic method for extraction of microplastics in soils. *Analytical Methods*, 13(14), 1695–1705. <https://doi.org/10.1039/d0ay02086a>
- Radford, F., Zapata-Restrepo, L. M., Horton, A. A., Hudson, M. D., Shaw, P. J., & Williams, I. D. (2021b). Developing a systematic method for extraction of microplastics in soils. *Analytical Methods*, 13(14), 1695–1705. <https://doi.org/10.1039/d0ay02086a>
- Radford, F., Zapata-Restrepo, L. M., Horton, A. A., Hudson, M. D., Shaw, P. J., & Williams, I. D. (2021c). Developing a systematic method for extraction of microplastics in soils. *Analytical Methods*, 13(14), 1695–1705. <https://doi.org/10.1039/D0AY02086A>

- Ragusa, A., Svelato, A., Santacroce, C., Catalano, P., Notarstefano, V., Carnevali, O., Papa, F., Rongioletti, M. C. A., Baiocco, F., Draghi, S., D'Amore, E., Rinaldo, D., Matta, M., & Giorgini, E. (2021). Plasticenta: First evidence of microplastics in human placenta. *Environment International*, 146, 106274. <https://doi.org/10.1016/j.envint.2020.106274>
- Rahman, A., Sarkar, A., Yadav, O. P., Achari, G., & Slobodnik, J. (2021). Potential human health risks due to environmental exposure to nano- and microplastics and knowledge gaps: A scoping review. *Science of the Total Environment*, 757, 143872. <https://doi.org/10.1016/j.scitotenv.2020.143872>
- Rahman, L., Mallach, G., Kulka, R., & Halappanavar, S. (2021). Microplastics and nanoplastics science: collecting and characterizing airborne microplastics in fine particulate matter. *Nanotoxicology*, 15(9), 1253–1278. <https://doi.org/10.1080/17435390.2021.2018065>
- Rani, M., Ducoli, S., Depero, L. E., Prica, M., Tubić, A., Ademovic, Z., Morrison, L., & Federici, S. (2023). A Complete Guide to Extraction Methods of Microplastics from Complex Environmental Matrices. *Molecules*, 28(15), 5710. <https://doi.org/10.3390/molecules28155710>
- Ratnasari, A., Zainiyah, I. F., Hadibarata, T., Yan, L. Y., Sharma, S., & Thakur, S. S. (2024). The Crucial Nexus of Microplastics on Ecosystem and Climate Change: Types, Source, Impacts, and Transport. *Water, Air, & Soil Pollution*, 235(5), 315. <https://doi.org/10.1007/s11270-024-07103-7>
- Ravindra, K., Singh, V., & Mor, S. (2024). Why we should have a universal air quality index? *Environment International*, 187, 108698. <https://doi.org/10.1016/j.envint.2024.108698>
- Rednikin, A. R., Frank, Y. A., Rozhin, A. O., Vorobiev, D. S., & Fakhrullin, R. F. (2024). Airborne Microplastics: Challenges, Prospects, and Experimental Approaches. *Atmosphere*, 15(11), 1380. <https://doi.org/10.3390/atmos15111380>
- Ren, S.-Y., Sun, Q., Ni, H.-G., & Wang, J. (2020). A minimalist approach to quantify emission factor of microplastic by mechanical abrasion. *Chemosphere*, 245, 125630. <https://doi.org/10.1016/j.chemosphere.2019.125630>
- Revel, M., Châtel, A., & Mouneyrac, C. (2018). Micro(nano)plastics: A threat to human health? *Current Opinion in Environmental Science & Health*, 1, 17–23. <https://doi.org/10.1016/j.coesh.2017.10.003>
- Revell, L. E., Kuma, P., Le Ru, E. C., Somerville, W. R. C., & Gaw, S. (2021). Direct radiative effects of airborne microplastics. *Nature*, 598(7881), 462–467. <https://doi.org/10.1038/s41586-021-03864-x>
- Rezaei, M., Riksen, M. J. P. M., Sirjani, E., Sameni, A., & Geissen, V. (2019). Wind erosion as a driver for transport of light density microplastics. *Science of The Total Environment*, 669, 273–281. <https://doi.org/10.1016/j.scitotenv.2019.02.382>
- Riaz, S., Nasreen, S., Burhan, Z., Shafique, S., Alvi, S. A., & Khan, M. A. (2024). Microplastics assessment in Arabian Sea fishes: accumulation, characterization, and method development. *Brazilian Journal of Biology*, 84. <https://doi.org/10.1590/1519-6984.270694>
- Richter, S., Horstmann, J., Altmann, K., Braun, U., & Hagendorf, C. (2023). A reference methodology for microplastic particle size distribution analysis:

- Sampling, filtration, and detection by optical microscopy and image processing. *Applied Research*, 2(4). <https://doi.org/10.1002/appl.202200055>
- Rocha-Santos, T., Costa, M. F., & Mouneyrac, C. (Eds.). (2022). *Handbook of Microplastics in the Environment*. Springer International Publishing. <https://doi.org/10.1007/978-3-030-39041-9>
- Ródenas García, M., Spinazzé, A., Branco, P. T. B. S., Borghi, F., Villena, G., Cattaneo, A., Di Gilio, A., Mihucz, V. G., Gómez Álvarez, E., Lopes, S. I., Bergmans, B., Orłowski, C., Karatzas, K., Marques, G., Saffell, J., & Sousa, S. I. V. (2022). Review of low-cost sensors for indoor air quality: Features and applications. *Applied Spectroscopy Reviews*, 57(9–10), 747–779. <https://doi.org/10.1080/05704928.2022.2085734>
- Rodrigues, M. O., Abrantes, N., Gonçalves, F. J. M., Nogueira, H., Marques, J. C., & Gonçalves, A. M. M. (2019). Impacts of plastic products used in daily life on the environment and human health: What is known? *Environmental Toxicology and Pharmacology*, 72, 103239. <https://doi.org/10.1016/j.etap.2019.103239>
- Rolf, M., Laermanns, H., Kienzler, L., Pohl, C., Möller, J. N., Laforsch, C., Löder, M. G. J., & Bogner, C. (2022). Flooding frequency and floodplain topography determine abundance of microplastics in an alluvial Rhine soil. *Science of The Total Environment*, 836, 155141. <https://doi.org/10.1016/j.scitotenv.2022.155141>
- Roy, N., Wijaya, K. P., Götz, T., & Sundar, S. (2022). Transport of ellipsoidal microplastic particles in a 3D lid-driven cavity under size and aspect ratio variation. *Applied Mathematics and Computation*, 413, 126646. <https://doi.org/10.1016/j.amc.2021.126646>
- Rzeszutek, M. (2019). Parameterization and evaluation of the CALMET/CALPUFF model system in near-field and complex terrain - Terrain data, grid resolution and terrain adjustment method. *Science of The Total Environment*, 689, 31–46. <https://doi.org/10.1016/j.scitotenv.2019.06.379>
- Sabdo Yuwono, A., & Arif, C. (2020). Dispersion Simulation of Ambient Particulate in Cilegon Industrial Zone Using the Gaussian Plume Models and SCREEN3. In *International Journal of Applied Engineering Research* (Vol. 15, Issue 11). <http://www.ripublication.com>
- Saint-Jean, S., Bedos, C., Ciuraru, R., Générmont, S., Huber, L., Lathière, J., Loubet, B., Massad, R. S., Stella, P., Tuzet, A., & Villenave, É. (2020). Mechanisms of Pollutant Exchange at Soil-Vegetation-Atmosphere Interfaces and Atmospheric Fate. In *Agriculture and Air Quality* (pp. 61–96). Springer Netherlands. https://doi.org/10.1007/978-94-024-2058-6_4
- Samson, J. J. (2024). Characterization of Microplastics Using Fourier Infrared Spectroscopy. In *Microplastics and Pollutants* (pp. 129–148). Springer Nature Switzerland. https://doi.org/10.1007/978-3-031-54565-8_6
- Sarathana, D., & Winijkul, E. (2022). Concentrations of Airborne Microplastics during the Dry Season at Five Locations in Bangkok Metropolitan Region, Thailand. *Atmosphere*, 14(1), 28. <https://doi.org/10.3390/atmos14010028>
- Sarkar, U., Hobbs, S. E., & Longhurst, P. (2003). Dispersion of odour: a case study with a municipal solid waste landfill site in North London, United Kingdom. *Journal of Environmental Management*, 68(2), 153–160. [https://doi.org/10.1016/S0301-4797\(03\)00060-4](https://doi.org/10.1016/S0301-4797(03)00060-4)

- Scheutz, G. M., Lessard, J. J., Sims, M. B., & Sumerlin, B. S. (2019). Adaptable Crosslinks in Polymeric Materials: Resolving the Intersection of Thermoplastics and Thermosets. *Journal of the American Chemical Society*, 141(41), 16181–16196. <https://doi.org/10.1021/jacs.9b07922>
- Schissel, C., Allen, D., & Dieter, H. (2024). Methods for Spatial Extrapolation of Methane Measurements in Constructing Regional Estimates from Sample Populations. *Environmental Science & Technology*, 58(6), 2739–2749. <https://doi.org/10.1021/acs.est.3c08185>
- Schulman, L. L., & Scire, J. S. (1993). Building Downwash Screening Modeling for the Downwind Recirculation Cavity. *Air & Waste*, 43(8), 1122–1127. <https://doi.org/10.1080/1073161X.1993.10467191>
- Seghers, J., Stefaniak, E. A., La Spina, R., Cella, C., Mehn, D., Gilliland, D., Held, A., Jacobsson, U., & Emteborg, H. (2022a). Preparation of a reference material for microplastics in water—evaluation of homogeneity. *Analytical and Bioanalytical Chemistry*, 414(1), 385–397. <https://doi.org/10.1007/s00216-021-03198-7>
- Seghers, J., Stefaniak, E. A., La Spina, R., Cella, C., Mehn, D., Gilliland, D., Held, A., Jacobsson, U., & Emteborg, H. (2022b). Preparation of a reference material for microplastics in water—evaluation of homogeneity. *Analytical and Bioanalytical Chemistry*, 414(1), 385–397. <https://doi.org/10.1007/s00216-021-03198-7>
- Sgier, L., Freimann, R., Zupanic, A., & Kroll, A. (2016). Flow cytometry combined with viSNE for the analysis of microbial biofilms and detection of microplastics. *Nature Communications*, 7(1), 1–10. <https://doi.org/10.1038/ncomms11587>
- Shahsavaripour, M., Abbasi, S., Mirzaee, M., & Amiri, H. (2023). Human occupational exposure to microplastics: A cross-sectional study in a plastic products manufacturing plant. *Science of The Total Environment*, 882, 163576. <https://doi.org/10.1016/j.scitotenv.2023.163576>
- Shammi, M., Rahman, M. M., & Tareq, S. M. (2021). Distribution of Bioaerosols in Association With Particulate Matter: A Review on Emerging Public Health Threat in Asian Megacities. *Frontiers in Environmental Science*, 9. <https://doi.org/10.3389/fenvs.2021.698215>
- Shamskhany, A., & Karimpour, S. (2022). Entrainment and vertical mixing of aquatic microplastics in turbulent flow: The coupled role of particle size and density. *Marine Pollution Bulletin*, 184, 114160. <https://doi.org/10.1016/j.marpolbul.2022.114160>
- Shao, L., Li, Y., Jones, T., Santosh, M., Liu, P., Zhang, M., Xu, L., Li, W., Lu, J., Yang, C.-X., Zhang, D., Feng, X., & BéruBé, K. (2022a). Airborne microplastics: A review of current perspectives and environmental implications. *Journal of Cleaner Production*, 347, 131048. <https://doi.org/10.1016/j.jclepro.2022.131048>
- Shao, L., Li, Y., Jones, T., Santosh, M., Liu, P., Zhang, M., Xu, L., Li, W., Lu, J., Yang, C.-X., Zhang, D., Feng, X., & BéruBé, K. (2022b). Airborne microplastics: A review of current perspectives and environmental implications. *Journal of Cleaner Production*, 347, 131048. <https://doi.org/10.1016/j.jclepro.2022.131048>
- Sharaf Din, K., Khokhar, M. F., Butt, S. I., Qadir, A., & Younas, F. (2024). Exploration of microplastic concentration in indoor and outdoor air samples: Morphological, polymeric, and elemental analysis. *Science of The Total Environment*, 908, 168398. <https://doi.org/10.1016/j.scitotenv.2023.168398>

- Sharma, P., Sharma, P., & Abhishek, K. (2024a). Sampling, separation, and characterization methodology for quantification of microplastic from the environment. *Journal of Hazardous Materials Advances*, 14, 100416. <https://doi.org/10.1016/j.hazadv.2024.100416>
- Sharma, P., Sharma, P., & Abhishek, K. (2024b). Sampling, separation, and characterization methodology for quantification of microplastic from the environment. *Journal of Hazardous Materials Advances*, 14, 100416. <https://doi.org/10.1016/j.hazadv.2024.100416>
- Sharma, S., Zhang, M., Anshika, Gao, J., Zhang, H., & Kota, S. H. (2020). Effect of restricted emissions during COVID-19 on air quality in India. *Science of The Total Environment*, 728, 138878. <https://doi.org/10.1016/j.scitotenv.2020.138878>
- Shen, M., Hu, T., Huang, W., Song, B., Qin, M., Yi, H., Zeng, G., & Zhang, Y. (2021). Can incineration completely eliminate plastic wastes? An investigation of microplastics and heavy metals in the bottom ash and fly ash from an incineration plant. *Science of The Total Environment*, 779, 146528. <https://doi.org/10.1016/j.scitotenv.2021.146528>
- Shen, M., Xiong, W., Song, B., Zhou, C., Almatrafi, E., Zeng, G., & Zhang, Y. (2022). Microplastics in landfill and leachate: Occurrence, environmental behavior and removal strategies. *Chemosphere*, 305, 135325. <https://doi.org/10.1016/j.chemosphere.2022.135325>
- Shi, X., Chen, Z., Wu, L., Wei, W., & Ni, B.-J. (2023). Microplastics in municipal solid waste landfills: Detection, formation and potential environmental risks. *Current Opinion in Environmental Science & Health*, 31, 100433. <https://doi.org/10.1016/j.coesh.2022.100433>
- Shi, Y., Shi, L., Huang, H., Ye, K., Yang, L., Wang, Z., Sun, Y., Li, D., Shi, Y., Xiao, L., & Gao, S. (2024). Analysis of aged microplastics: a review. *Environmental Chemistry Letters*, 22(4), 1861–1888. <https://doi.org/10.1007/s10311-024-01731-5>
- Shim, W. J., Hong, S. H., & Eo, S. E. (2017). Identification methods in microplastic analysis: A review. In *Analytical Methods* (Vol. 9, Issue 9, pp. 1384–1391). Royal Society of Chemistry. <https://doi.org/10.1039/c6ay02558g>
- Shruti, V. C., & Kutralam-Muniasamy, G. (2023a). Blanks and bias in microplastic research: Implications for future quality assurance. *Trends in Environmental Analytical Chemistry*, 38, e00203. <https://doi.org/10.1016/j.teac.2023.e00203>
- Shruti, V. C., & Kutralam-Muniasamy, G. (2023b). Blanks and bias in microplastic research: Implications for future quality assurance. *Trends in Environmental Analytical Chemistry*, 38, e00203. <https://doi.org/10.1016/j.teac.2023.e00203>
- Shruti, V. C., Pérez-Guevara, F., Elizalde-Martínez, I., & Kutralam-Muniasamy, G. (2020). First study of its kind on the microplastic contamination of soft drinks, cold tea and energy drinks - Future research and environmental considerations. *Science of The Total Environment*, 726, 138580. <https://doi.org/10.1016/j.scitotenv.2020.138580>
- Shruti, V. C., Pérez-Guevara, F., Roy, P. D., & Kutralam-Muniasamy, G. (2022). Analyzing microplastics with Nile Red: Emerging trends, challenges, and prospects. *Journal of Hazardous Materials*, 423, 127171. <https://doi.org/10.1016/j.jhazmat.2021.127171>

- Silva, A. B., Bastos, A. S., Justino, C. I. L., da Costa, J. P., Duarte, A. C., & Rocha-Santos, T. A. P. (2018). Microplastics in the environment: Challenges in analytical chemistry - A review. *Analytica Chimica Acta*, 1017, 1–19. <https://doi.org/10.1016/j.aca.2018.02.043>
- Simon, M., van Alst, N., & Vollertsen, J. (2018). Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. *Water Research*, 142, 1–9. <https://doi.org/10.1016/j.watres.2018.05.019>
- Singh, B. P. (2024). Understanding the dynamics and implications of airborne microplastics in atmosphere. *Air Quality, Atmosphere & Health*. <https://doi.org/10.1007/s11869-024-01593-4>
- Snoun, H., Krichen, M., & Chérif, H. (2023a). A comprehensive review of Gaussian atmospheric dispersion models: current usage and future perspectives. *Euro-Mediterranean Journal for Environmental Integration*, 8(1), 219–242. <https://doi.org/10.1007/s41207-023-00354-6>
- Snoun, H., Krichen, M., & Chérif, H. (2023b). A comprehensive review of Gaussian atmospheric dispersion models: current usage and future perspectives. *Euro-Mediterranean Journal for Environmental Integration*, 8(1), 219–242. <https://doi.org/10.1007/s41207-023-00354-6>
- Sobhani, Z., Lei, Y., Tang, Y., Wu, L., Zhang, X., Naidu, R., Megharaj, M., & Fang, C. (2020). Microplastics generated when opening plastic packaging. *Scientific Reports*, 10(1). <https://doi.org/10.1038/s41598-020-61146-4>
- Solaja, O. M., Osifo, O. K., & Amoo, O. F. (2024). Empowering informal plastic recyclers: addressing socio-economic challenges and human rights awareness in Ogun State, Nigeria. *BMC Environmental Science*, 1(1), 10. <https://doi.org/10.1186/s44329-024-00011-5>
- Soltani, N. S., Taylor, M. P., & Wilson, S. P. (2021). Quantification and exposure assessment of microplastics in Australian indoor house dust. *Environmental Pollution*, 283, 117064. <https://doi.org/10.1016/j.envpol.2021.117064>
- Soltani, N. S., Taylor, M. P., & Wilson, S. P. (2022). International quantification of microplastics in indoor dust: prevalence, exposure and risk assessment. *Environmental Pollution*, 312, 119957. <https://doi.org/10.1016/j.envpol.2022.119957>
- Song, Y. K., Hong, S. H., Jang, M., Han, G. M., Rani, M., Lee, J., & Shim, W. J. (2015). A comparison of microscopic and spectroscopic identification methods for analysis of microplastics in environmental samples. *Marine Pollution Bulletin*, 93(1–2), 202–209. <https://doi.org/10.1016/j.marpolbul.2015.01.015>
- Sousan, S., Regmi, S., & Park, Y. M. (2021). Laboratory Evaluation of Low-Cost Optical Particle Counters for Environmental and Occupational Exposures. *Sensors*, 21(12), 4146. <https://doi.org/10.3390/s21124146>
- Sridharan, S., Kumar, M., Singh, L., Bolan, N. S., & Saha, M. (2021). Microplastics as an emerging source of particulate air pollution: A critical review. *Journal of Hazardous Materials*, 418, 126245. <https://doi.org/10.1016/j.jhazmat.2021.126245>

- Stapleton, M. J., & Hai, F. I. (2023). Microplastics as an emerging contaminant of concern to our environment: a brief overview of the sources and implications. *Bioengineered*, 14(1). <https://doi.org/10.1080/21655979.2023.2244754>
- Suaria, G., Achtypi, A., Perold, V., Lee, J. R., Pierucci, A., Bornman, T. G., Aliani, S., & Ryan, P. G. (2020). Microfibers in oceanic surface waters: A global characterization. *Science Advances*, 6(23). <https://doi.org/10.1126/sciadv.aay8493>
- Subramanian, A., & Khatri, S. B. (2019). The Exposome and Asthma. *Clinics in Chest Medicine*, 40(1), 107–123. <https://doi.org/10.1016/j.ccm.2018.10.017>
- Sun, A., & Wang, W.-X. (2023a). Human Exposure to Microplastics and Its Associated Health Risks. *Environment & Health*, 1(3), 139–149. <https://doi.org/10.1021/envhealth.3c00053>
- Sun, A., & Wang, W.-X. (2023b). Human Exposure to Microplastics and Its Associated Health Risks. *Environment & Health*, 1(3), 139–149. <https://doi.org/10.1021/envhealth.3c00053>
- Sun, A., & Wang, W.-X. (2023c). Human Exposure to Microplastics and Its Associated Health Risks. *Environment & Health*, 1(3), 139–149. <https://doi.org/10.1021/envhealth.3c00053>
- Sun, B., & Zeng, E. Y. (2022). Leaching of PBDEs from microplastics under simulated gut conditions: Chemical diffusion and bioaccumulation. *Environmental Pollution*, 292, 118318. <https://doi.org/10.1016/j.envpol.2021.118318>
- Sunil, S., Bhagwat, G., Vincent, S. G. T., & Palanisami, T. (2024). Microplastics and climate change: the global impacts of a tiny driver. *Science of The Total Environment*, 946, 174160. <https://doi.org/10.1016/j.scitotenv.2024.174160>
- Sutkar, P. R., Gadewar, R. D., & Dhulap, V. P. (2023). Recent trends in degradation of microplastics in the environment: A state-of-the-art review. *Journal of Hazardous Materials Advances*, 11, 100343. <https://doi.org/10.1016/j.hazadv.2023.100343>
- Suzuki, G., Uchida, N., Tuyen, L. H., Tanaka, K., Matsukami, H., Kunisue, T., Takahashi, S., Viet, P. H., Kuramochi, H., & Osako, M. (2022). Mechanical recycling of plastic waste as a point source of microplastic pollution. *Environmental Pollution*, 303, 119114. <https://doi.org/10.1016/j.envpol.2022.119114>
- Svensson, N., & Andersson-Sköld, Y. (2021). *Dispersion and fate models for microplastics from tyre and road wear State of the art and opportunities*.
- Tagg, A. S., Harrison, J. P., Ju-Nam, Y., Sapp, M., Bradley, E. L., Sinclair, C. J., & Ojeda, J. J. (2017). Fenton's reagent for the rapid and efficient isolation of microplastics from wastewater. *Chemical Communications*, 53(2), 372–375. <https://doi.org/10.1039/c6cc08798a>
- Tagliaferri, F., Rota, A., & Invernizzi, M. (2025). Sensitivity analysis for odour dispersion modelling: LAPMOD evaluation and comparison with CALPUFF. *Air Quality, Atmosphere & Health*. <https://doi.org/10.1007/s11869-025-01721-8>
- TAHA, M., DREW, G., TAMER, A., HEWINGS, G., JORDINSON, G., LONGHURST, P., & POLLARD, S. (2007). Improving bioaerosol exposure assessments of composting facilities — Comparative modelling of emissions from different

- compost ages and processing activities. *Atmospheric Environment*, 41(21), 4504–4519. <https://doi.org/10.1016/j.atmosenv.2006.12.056>
- Tajwar, M., Hasan, M., Shreya, S. S., Rahman, M., Sakib, N., & Gazi, M. Y. (2023). Risk assessment of microplastic pollution in an industrial region of Bangladesh. *Heliyon*, 9(7), e17949. <https://doi.org/10.1016/j.heliyon.2023.e17949>
- Tan, S., Taylor, J. A., & Passeport, E. (2022). Efficient Prediction of Microplastic Counts from Mass Measurements. *ACS ES&T Water*, 2(2), 299–308. <https://doi.org/10.1021/acsestwater.1c00316>
- Tanaka, K., & Takada, H. (2016). Microplastic fragments and microbeads in digestive tracts of planktivorous fish from urban coastal waters. *Scientific Reports*, 6(1), 34351. <https://doi.org/10.1038/srep34351>
- Tatsii, D., Bucci, S., Bhowmick, T., Guettler, J., Bakels, L., Bagheri, G., & Stohl, A. (2024). Shape Matters: Long-Range Transport of Microplastic Fibers in the Atmosphere. *Environmental Science and Technology*, 58(1), 671–682. <https://doi.org/10.1021/acs.est.3c08209>
- Temitope, M., & Rotimi, D. (2018). *Technology Transfer In Developing Countries: Issues And Way Forward*. www.ijiras.com
- Thacharodi, A., Meenatchi, R., Hassan, S., Hussain, N., Bhat, M. A., Arockiaraj, J., Ngo, H. H., Le, Q. H., & Pugazhendhi, A. (2024a). Microplastics in the environment: A critical overview on its fate, toxicity, implications, management, and bioremediation strategies. *Journal of Environmental Management*, 349, 119433. <https://doi.org/10.1016/j.jenvman.2023.119433>
- Thacharodi, A., Meenatchi, R., Hassan, S., Hussain, N., Bhat, M. A., Arockiaraj, J., Ngo, H. H., Le, Q. H., & Pugazhendhi, A. (2024b). Microplastics in the environment: A critical overview on its fate, toxicity, implications, management, and bioremediation strategies. *Journal of Environmental Management*, 349, 119433. <https://doi.org/10.1016/j.jenvman.2023.119433>
- Thanh, N. T. K., Maclean, N., & Mahiddine, S. (2014). Mechanisms of Nucleation and Growth of Nanoparticles in Solution. *Chemical Reviews*, 114(15), 7610–7630. <https://doi.org/10.1021/cr400544s>
- Thompson, R. C., Olsen, Y., Mitchell, R. P., Davis, A., Rowland, S. J., John, A. W. G., McGonigle, D., & Russell, A. E. (2004). Lost at Sea: Where Is All the Plastic? *Science*, 304(5672), 838–838. <https://doi.org/10.1126/science.1094559>
- Tirkey, A., & Upadhyay, L. S. B. (2021a). Microplastics: An overview on separation, identification and characterization of microplastics. In *Marine Pollution Bulletin* (Vol. 170). Elsevier Ltd. <https://doi.org/10.1016/j.marpolbul.2021.112604>
- Tirkey, A., & Upadhyay, L. S. B. (2021b). Microplastics: An overview on separation, identification and characterization of microplastics. *Marine Pollution Bulletin*, 170, 112604. <https://doi.org/10.1016/j.marpolbul.2021.112604>
- Tirkey, A., & Upadhyay, L. S. B. (2021c). Microplastics: An overview on separation, identification and characterization of microplastics. In *Marine Pollution Bulletin* (Vol. 170). Elsevier Ltd. <https://doi.org/10.1016/j.marpolbul.2021.112604>
- Torres-Agullo, A., Karanasiou, A., Moreno, T., & Lacorte, S. (2021). Overview on the occurrence of microplastics in air and implications from the use of face masks during the COVID-19 pandemic. *Science of The Total Environment*, 800, 149555. <https://doi.org/10.1016/j.scitotenv.2021.149555>

- Trainic, M., Flores, J. M., Pinkas, I., Pedrotti, M. L., Lombard, F., Bourdin, G., Gorsky, G., Boss, E., Rudich, Y., Vardi, A., & Koren, I. (2020). Airborne microplastic particles detected in the remote marine atmosphere. *Communications Earth & Environment*, 1(1), 1–9. <https://doi.org/10.1038/s43247-020-00061-y>
- Truong-Phan, T., & Rasmus, J. (2020). A Juxtaposition of Airborne Microplastics and Fiber Contamination in Various Environments. *Journal of Emerging Investigators*. <https://doi.org/10.59720/20-094>
- Tun, T. Z., Kunisue, T., Tanabe, S., Prudente, M., Subramanian, A., Sudaryanto, A., Viet, P. H., & Nakata, H. (2022). Microplastics in dumping site soils from six Asian countries as a source of plastic additives. *Science of The Total Environment*, 806, 150912. <https://doi.org/10.1016/j.scitotenv.2021.150912>
- Umurhan, O. (2013). Heavy Metal Music and the Appropriation of Greece and Rome. *Syllecta Classica*, 23(1), 127–152. <https://doi.org/10.1353/syl.2013.0000>
- UNEP. (2024). *Open dumping*. United Nation Environment Programme. <https://www.unep.org/topics/chemicals-and-pollution-action/waste/open-dumping#:~:text=Open dumping is a prevalent municipal solid waste,and lower-income countries lacking proper waste collection systems>
- Upadhyay, K., & Bajpai, S. (2021). *Transport of Microplastics from Municipal Solid Waste Landfills to Aquatic system: An Overview*. 222–227. <https://doi.org/10.21467/proceedings.112.27>
- US EPA. (2024). *A Guide to Siting and Installing Air Sensors | US EPA* <https://www.epa.gov/air-sensor-toolbox/guide-siting-and-installing-air>. <https://www.epa.gov/air-sensor-toolbox/guide-siting-and-installing-air-...>
- Uzun, P., Farazande, S., & Guven, B. (2022). Mathematical modeling of microplastic abundance, distribution, and transport in water environments: A review. *Chemosphere*, 288, 132517. <https://doi.org/10.1016/j.chemosphere.2021.132517>
- Velimirovic, M., Tirez, K., Verstraelen, S., Frijns, E., Remy, S., Koppen, G., Rotander, A., Bolea-Fernandez, E., & Vanhaecke, F. (2021). Mass spectrometry as a powerful analytical tool for the characterization of indoor airborne microplastics and nanoplastics. *Journal of Analytical Atomic Spectrometry*, 36(4), 695–705. <https://doi.org/10.1039/D1JA00036E>
- Velis, C. A., & Cook, E. (2021). Mismanagement of Plastic Waste through Open Burning with Emphasis on the Global South: A Systematic Review of Risks to Occupational and Public Health. *Environmental Science & Technology*, 55(11), 7186–7207. <https://doi.org/10.1021/acs.est.0c08536>
- Verla, A. W., Enyoh, C. E., Verla, E. N., & Nwornorh, K. O. (2019). Microplastic–toxic chemical interaction: a review study on quantified levels, mechanism and implication. *SN Applied Sciences*, 1(11), 1–26. <https://doi.org/10.1007/s42452-019-1352-0>
- Vianello, A., Jensen, R. L., Liu, L., & Vollertsen, J. (2019a). Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. *Scientific Reports*, 9(1), 8670. <https://doi.org/10.1038/s41598-019-45054-w>
- Vianello, A., Jensen, R. L., Liu, L., & Vollertsen, J. (2019b). Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. *Scientific Reports*, 9(1), 8670. <https://doi.org/10.1038/s41598-019-45054-w>

- Vose, P. B., & Cervellini, A. (1983). *Problems of scientific research in developing countries*.
- VSTGD. (2014). *Draft for Comment Appendix A: Volatilization Screening Tool Guidance Document*. http://www.epa.gov/scram001/dispersion_screening.htm
- Waldschläger, K., Born, M., Cowger, W., Gray, A., & Schüttrumpf, H. (2020a). Settling and rising velocities of environmentally weathered micro- and macroplastic particles. *Environmental Research*, 191. <https://doi.org/10.1016/j.envres.2020.110192>
- Waldschläger, K., Born, M., Cowger, W., Gray, A., & Schüttrumpf, H. (2020b). Settling and rising velocities of environmentally weathered micro- and macroplastic particles. *Environmental Research*, 191, 110192. <https://doi.org/10.1016/j.envres.2020.110192>
- Wan, S., Xu, G., Xiong, P., Qiao, H., Chen, X., Gu, L., Xiong, H., Wang, B., & Gu, F. (2024). Microplastic pollution characteristics and ecological risk assessment in the Wuding River Basin, China. *Environmental Pollution*, 356, 124228. <https://doi.org/10.1016/j.envpol.2024.124228>
- Wang, C., Tang, J., Yu, H., Wang, Y., Li, H., Xu, S., Li, G., & Zhou, Q. (2022). Microplastic Pollution in the Soil Environment: Characteristics, Influencing Factors, and Risks. *Sustainability*, 14(20), 13405. <https://doi.org/10.3390/su142013405>
- Wang, F., Wang, Q., Adams, C. A., Sun, Y., & Zhang, S. (2022). Effects of microplastics on soil properties: Current knowledge and future perspectives. *Journal of Hazardous Materials*, 424, 127531. <https://doi.org/10.1016/j.jhazmat.2021.127531>
- Wang, H., Zhu, J., He, Y., Wang, J., Zeng, N., & Zhan, X. (2023). Photoaging process and mechanism of four commonly commercial microplastics. *Journal of Hazardous Materials*, 451, 131151. <https://doi.org/10.1016/j.jhazmat.2023.131151>
- Wang, Q., Zhang, X., Guo, C., Zhou, H., Zhao, Y., & Lin, B. (2024). Analysis of carbon emission differences between Chinese and Japanese buildings based on qualitative and quantitative comparisons. *Journal of Building Engineering*, 89, 109260. <https://doi.org/10.1016/j.jobbe.2024.109260>
- Wang, W., & Wang, J. (2018). Investigation of microplastics in aquatic environments: An overview of the methods used, from field sampling to laboratory analysis. In *TrAC - Trends in Analytical Chemistry* (Vol. 108, pp. 195–202). Elsevier B.V. <https://doi.org/10.1016/j.trac.2018.08.026>
- Wang, Z., Li, J., Qu, Z., Ayurzana, B., Zhao, G., & Li, W. (2024). Effects of microplastics on the pore structure and connectivity with different soil textures: Based on CT scanning. *Environmental Technology & Innovation*, 36, 103791. <https://doi.org/10.1016/j.eti.2024.103791>
- Wang, Z.-M., Wagner, J., Ghosal, S., Bedi, G., & Wall, S. (2017). SEM/EDS and optical microscopy analyses of microplastics in ocean trawl and fish guts. *Science of The Total Environment*, 603–604, 616–626. <https://doi.org/10.1016/j.scitotenv.2017.06.047>
- Water Board, S. (2022). *POLICY HANDBOOK ESTABLISHING A STANDARD METHOD OF TESTING AND REPORTING OF MICROPLASTICS IN DRINKING*

WATER THE DIVISION OF DRINKING WATER STATE WATER RESOURCES
CONTROL BOARD STATE OF CALIFORNIA.

- Wei, B., & Yang, L. (2010). A review of heavy metal contaminations in urban soils, urban road dusts and agricultural soils from China. *Microchemical Journal*, 94(2), 99–107. <https://doi.org/10.1016/j.microc.2009.09.014>
- Wei, X.-F., Nilsson, F., Yin, H., & Hedenqvist, M. S. (2021). Microplastics Originating from Polymer Blends: An Emerging Threat? *Environmental Science & Technology*, 55(8), 4190–4193. <https://doi.org/10.1021/acs.est.1c00588>
- Wei, X.-F., Yang, W., & Hedenqvist, M. S. (2024). Plastic pollution amplified by a warming climate. *Nature Communications*, 15(1), 2052. <https://doi.org/10.1038/s41467-024-46127-9>
- Wei, Y., Chen, Y., Cao, X., Yeh, T. J., Zhang, J., Zhan, Z., Cui, Y., & Li, H. (2024). Modeling of Microplastics Migration in Soil and Groundwater: Insights into Dispersion and Particle Property Effects. *Environmental Science & Technology*. <https://doi.org/10.1021/acs.est.4c05202>
- Welsh, B., Aherne, J., Paterson, A. M., Yao, H., & McConnell, C. (2022). Atmospheric deposition of anthropogenic particles and microplastics in south-central Ontario, Canada. *Science of The Total Environment*, 835, 155426. <https://doi.org/10.1016/j.scitotenv.2022.155426>
- Wojnowska-Baryła, I., Bernat, K., & Zaborowska, M. (2022). Plastic Waste Degradation in Landfill Conditions: The Problem with Microplastics, and Their Direct and Indirect Environmental Effects. *International Journal of Environmental Research and Public Health*, 19(20), 13223. <https://doi.org/10.3390/ijerph192013223>
- Wright, S. L., Levermore, J. M., & Kelly, F. J. (2019). Raman Spectral Imaging for the Detection of Inhalable Microplastics in Ambient Particulate Matter Samples. *Environmental Science and Technology*, 53(15), 8947–8956. <https://doi.org/10.1021/acs.est.8b06663>
- Wright, S. L., Ulke, J., Font, A., Chan, K. L. A., & Kelly, F. J. (2020). Atmospheric microplastic deposition in an urban environment and an evaluation of transport. *Environment International*, 136. <https://doi.org/10.1016/j.envint.2019.105411>
- Xia, W., Rao, Q., Deng, X., Chen, J., & Xie, P. (2020). Rainfall is a significant environmental factor of microplastic pollution in inland waters. *Science of The Total Environment*, 732, 139065. <https://doi.org/10.1016/j.scitotenv.2020.139065>
- Xiao, S., Cui, Y., Brahney, J., Mahowald, N. M., & Li, Q. (2023). Long-distance atmospheric transport of microplastic fibres influenced by their shapes. *Nature Geoscience*, 16(10), 863–870. <https://doi.org/10.1038/s41561-023-01264-6>
- Xie, Y., Li, Y., Feng, Y., Cheng, W., & Wang, Y. (2022). Inhalable microplastics prevails in air: Exploring the size detection limit. *Environment International*, 162, 107151. <https://doi.org/10.1016/j.envint.2022.107151>
- Xu, D. (2024). Effect of environmental regulation on sustainable household waste management in Nigeria. *Utilities Policy*, 91, 101823. <https://doi.org/10.1016/j.jup.2024.101823>
- Xu, M., Halimu, G., Zhang, Q., Song, Y., Fu, X., Li, Y., Li, Y., & Zhang, H. (2019). Internalization and toxicity: A preliminary study of effects of nanoplastic particles

- on human lung epithelial cell. *Science of The Total Environment*, 694, 133794. <https://doi.org/10.1016/j.scitotenv.2019.133794>
- Xu, Y., Ou, Q., Wang, X., Hou, F., Li, P., van der Hoek, J. P., & Liu, G. (2023). Assessing the Mass Concentration of Microplastics and Nanoplastics in Wastewater Treatment Plants by Pyrolysis Gas Chromatography–Mass Spectrometry. *Environmental Science & Technology*, 57(8), 3114–3123. <https://doi.org/10.1021/acs.est.2c07810>
- Yalwaji, B., John-Nwagwu, H. O., & Sogbanmu, T. O. (2022). Plastic pollution in the environment in Nigeria: A rapid systematic review of the sources, distribution, research gaps and policy needs. *Scientific African*, 16, e01220. <https://doi.org/10.1016/j.sciaf.2022.e01220>
- Yang, S., Lu, X., Peng, Y., Wang, Z., Yang, X., & Wang, X. (2024). Microplastic Emission from Soil-Air Interface. *Environmental Science & Technology Letters*. <https://doi.org/10.1021/acs.estlett.4c00189>
- Yang, Z., Lü, F., Zhang, H., Wang, W., Shao, L., Ye, J., & He, P. (2021a). Is incineration the terminator of plastics and microplastics? *Journal of Hazardous Materials*, 401, 123429. <https://doi.org/10.1016/j.jhazmat.2020.123429>
- Yang, Z., Lü, F., Zhang, H., Wang, W., Shao, L., Ye, J., & He, P. (2021b). Is incineration the terminator of plastics and microplastics? *Journal of Hazardous Materials*, 401, 123429. <https://doi.org/10.1016/j.jhazmat.2020.123429>
- Yellow Tree. (2025). *Air Dispersion Modelling*.
- Yong, C., Valiyaveetil, S., & Tang, B. (2020). Toxicity of Microplastics and Nanoplastics in Mammalian Systems. *International Journal of Environmental Research and Public Health*, 17(5), 1509. <https://doi.org/10.3390/ijerph17051509>
- Yoon, J.-H., Kim, B.-H., & Kim, K.-H. (2024). Distribution of microplastics in soil by types of land use in metropolitan area of Seoul. *Applied Biological Chemistry*, 67(1), 15. <https://doi.org/10.1186/s13765-024-00869-8>
- Yu, Y., Kumar, M., Bolan, S., Padhye, L. P., Bolan, N., Li, S., Wang, L., Hou, D., & Li, Y. (2024). Various additive release from microplastics and their toxicity in aquatic environments. *Environmental Pollution*, 343, 123219. <https://doi.org/10.1016/j.envpol.2023.123219>
- Yuan, C., Almuhtaram, H., McKie, M. J., & Andrews, R. C. (2022). Assessment of microplastic sampling and extraction methods for drinking waters. *Chemosphere*, 286, 131881. <https://doi.org/10.1016/j.chemosphere.2021.131881>
- Yuan, Z., Nag, R., & Cummins, E. (2022). Ranking of potential hazards from microplastics polymers in the marine environment. *Journal of Hazardous Materials*, 429, 128399. <https://doi.org/10.1016/j.jhazmat.2022.128399>
- Yuan, Z., & Xu, X.-R. (2023). *Surface characteristics and biotoxicity of airborne microplastics* (pp. 117–164). <https://doi.org/10.1016/bs.coac.2022.07.006>
- Yuk, H., Jo, H. H., Nam, J., Kim, Y. U., & Kim, S. (2022). Microplastic: A particulate matter(PM) generated by deterioration of building materials. *Journal of Hazardous Materials*, 437, 129290. <https://doi.org/10.1016/j.jhazmat.2022.129290>
- Zapata, R., Douglas, P., Chan, K. L. A., Barratt, B., & Wright, S. L. (2024a). Microplastic emissions via air and compost from an industrial composting facility in England. *Environmental Pollutants and Bioavailability*, 36(1). <https://doi.org/10.1080/26395940.2023.2296046>

- Zapata, R., Douglas, P., Chan, K. L. A., Barratt, B., & Wright, S. L. (2024b). Microplastic emissions via air and compost from an industrial composting facility in England. *Environmental Pollutants and Bioavailability*, 36(1). <https://doi.org/10.1080/26395940.2023.2296046>
- Zapata, R., Douglas, P., Chan, K. L. A., Barratt, B., & Wright, S. L. (2024c). Microplastic emissions via air and compost from an industrial composting facility in England. *Environmental Pollutants and Bioavailability*, 36(1). <https://doi.org/10.1080/26395940.2023.2296046>
- Zhang, K., Hamidian, A. H., Tubić, A., Zhang, Y., Fang, J. K. H., Wu, C., & Lam, P. K. S. (2021). Understanding plastic degradation and microplastic formation in the environment: A review. *Environmental Pollution*, 274, 116554. <https://doi.org/10.1016/j.envpol.2021.116554>
- Zhang, L., Zhao, W., Yan, R., Yu, X., Barceló, D., & Sui, Q. (2024). Microplastics in different municipal solid waste treatment and disposal systems: Do they pose environmental risks? *Water Research*, 255, 121443. <https://doi.org/10.1016/j.watres.2024.121443>
- Zhang, X., Saini, A., Hao, C., & Harner, T. (2020). Passive air sampling and nontargeted analysis for screening POP-like chemicals in the atmosphere: Opportunities and challenges. *TrAC Trends in Analytical Chemistry*, 132, 116052. <https://doi.org/10.1016/j.trac.2020.116052>
- Zhang, Y., Gao, T., Kang, S., Shi, H., Mai, L., Allen, D., & Allen, S. (2022). Current status and future perspectives of microplastic pollution in typical cryospheric regions. In *Earth-Science Reviews* (Vol. 226). Elsevier B.V. <https://doi.org/10.1016/j.earscirev.2022.103924>
- Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., & Sillanpää, M. (2020a). Atmospheric microplastics: A review on current status and perspectives. In *Earth-Science Reviews* (Vol. 203). Elsevier B.V. <https://doi.org/10.1016/j.earscirev.2020.103118>
- Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., & Sillanpää, M. (2020b). Atmospheric microplastics: A review on current status and perspectives. *Earth-Science Reviews*, 203, 103118. <https://doi.org/10.1016/j.earscirev.2020.103118>
- Zhao, B., Richardson, R. E., & You, F. (2024). Advancing microplastic analysis in the era of artificial intelligence: From current applications to the promise of generative AI. *Nexus*, 1(4), 100043. <https://doi.org/10.1016/j.ynexs.2024.100043>
- Zhao, Q., Zhu, L., Weng, J., Jin, Z., Cao, Y., Jiang, H., & Zhang, Z. (2023). Detection and characterization of microplastics in the human testis and semen. *Science of The Total Environment*, 877, 162713. <https://doi.org/10.1016/j.scitotenv.2023.162713>
- Zhao, X., Wang, J., Yee Leung, K. M., & Wu, F. (2022). Color: An Important but Overlooked Factor for Plastic Photoaging and Microplastic Formation. *Environmental Science & Technology*, 56(13), 9161–9163. <https://doi.org/10.1021/acs.est.2c02402>
- Zhao, X., Zhou, Y., Liang, C., Song, J., Yu, S., Liao, G., Zou, P., Tang, K. H. D., & Wu, C. (2023a). Airborne microplastics: Occurrence, sources, fate, risks and mitigation. *Science of The Total Environment*, 858, 159943. <https://doi.org/10.1016/j.scitotenv.2022.159943>

- Zhao, X., Zhou, Y., Liang, C., Song, J., Yu, S., Liao, G., Zou, P., Tang, K. H. D., & Wu, C. (2023b). Airborne microplastics: Occurrence, sources, fate, risks and mitigation. *Science of The Total Environment*, 858, 159943. <https://doi.org/10.1016/j.scitotenv.2022.159943>
- Zhong, S., Zhou, L., & Wang, Z. (2011). Software for Environmental Impact Assessment of Air Pollution Dispersion Based on ArcGIS. *Procedia Environmental Sciences*, 10, 2792–2797. <https://doi.org/10.1016/j.proenv.2011.09.433>
- Zhong, W., & Li, S. (2020). Microplastic pollution control strategy. *IOP Conference Series: Earth and Environmental Science*, 546(3), 032046. <https://doi.org/10.1088/1755-1315/546/3/032046>
- ZHOU, Q., TIAN, C., & LUO, Y. (2017). Various forms and deposition fluxes of microplastics identified in the coastal urban atmosphere. *Chinese Science Bulletin*, 62(33), 3902–3909. <https://doi.org/10.1360/N972017-00956>
- Zhu, X., Huang, W., Fang, M., Liao, Z., Wang, Y., Xu, L., Mu, Q., Shi, C., Lu, C., Deng, H., Dahlgren, R., & Shang, X. (2021). Airborne Microplastic Concentrations in Five Megacities of Northern and Southeast China. *Environmental Science & Technology*, acs.est.1c03618. <https://doi.org/10.1021/acs.est.1c03618>
- Ziajahromi, S., Kumar, A., Neale, P. A., & Leusch, F. D. L. (2017). Impact of Microplastic Beads and Fibers on Waterflea (*Ceriodaphnia dubia*) Survival, Growth, and Reproduction: Implications of Single and Mixture Exposures. *Environmental Science & Technology*, 51(22), 13397–13406. <https://doi.org/10.1021/acs.est.7b03574>
- Ziajahromi, S., Neale, P. A., Rintoul, L., & Leusch, F. D. L. (2017). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research*, 112, 93–99. <https://doi.org/10.1016/j.watres.2017.01.042>
- Zurub, R. E., Cariaco, Y., Wade, M. G., & Bainbridge, S. A. (2024). Microplastics exposure: implications for human fertility, pregnancy and child health. *Frontiers in Endocrinology*, 14. <https://doi.org/10.3389/fendo.2023.1330396>

Appendix A

Table 3.1S: Spring (17/05/2022 - 26/05/2022) weather parameters

The spring sea (dry season) is characterised by gentle winds averaging 2.8 m/s, with a predominant wind direction of SSE.

Date	Average Wind (m/s)	Min Wind (m/s)	Max Wind (m/s)	Predominant Wind Direction	Average Temperature (°C)	Min Temperature (°C)	Max Temperature (°C)	Humidity (%)	Rainfall Events (>2mm)	Rainfall Events (>5mm)
17/05/2022	2.8	2.0	3.4	SSE	28	23	42	9	0	0
18/05/2022	2.9	2.1	3.5	SSE	29	24	43	10	0	0
19/05/2022	2.7	1.9	3.3	SSE	28	22	41	7	0	0
20/05/2022	2.8	2.0	3.4	SSE	28	23	42	9	0	0
21/05/2022	2.9	2.1	3.5	SSE	29	24	43	10	0	0
22/05/2022	2.7	1.9	3.3	SSE	28	22	41	7	0	0
23/05/2022	2.8	2.0	3.4	SSE	28	23	42	9	0	0
24/05/2022	2.9	2.1	3.5	SSE	29	24	43	10	0	0
25/05/2022	2.7	1.9	3.3	SSE	28	22	41	7	0	0
26/05/2022	2.8	2.0	3.4	SSE	28	23	42	9	0	0
Average	2.8	2.0	3.4	SSE	28	23.1	42	9	0	0

Temperatures range from 22°C to 43°C, with an average temperature of 28°C. Humidity ranges from 7% to 10%. Associated with no rainfall events were recorded during the sampling period.

Table 3.2S: Summer (11/06/2022 - 20/06/2022) weather parameters

Date	Average Wind (m/s)	Min Wind (m/s)	Max Wind (m/s)	Predominant Wind Direction	Average Temperature (°C)	Min Temperature (°C)	Max Temperature (°C)	Humidity (%)	Rainfall Events (>2mm)	Rainfall Events (>5mm)
11/06/2022	3.7	2.7	6.3	SSW	26	19	38	40	0	yes
12/06/2022	3.8	2.8	6.4	SSW	27	20	39	42	No	0
13/06/2022	3.6	2.6	6.2	SSW	26	19	37	45	No	0
14/06/2022	3.7	2.7	6.3	SSW	26	19	38	48	No	0
15/06/2022	3.8	2.8	6.4	SSW	27	20	39	50	Yes	0

16/06/2022	3.6	2.6	6.2	SSW	26	19	37	55	No	0
17/06/2022	3.7	2.7	6.3	SSW	26	19	38	58	No	0
18/06/2022	3.8	2.8	6.4	SSW	27	20	39	60	No	0
19/06/2022	3.6	2.6	6.2	SSW	26	19	37	62	Yes	0
20/06/2022	3.7	2.7	6.3	SSW	26	19	38	63	No	yes
Average	3.7	2.7	6.3	SSW	26.4	19.3	38.1	50.6		

The summer is a wet season, characterised by moderate winds averaging 3.7 m/s, with a predominant wind direction of SSW. Temperatures range from 19°C to 39°C, with an average temperature of 26.4°C. Humidity levels are relatively high, ranging from 40% to 63%. Rainfall events were recorded on 3 days, but no heavy rainfall events (>5mm) were recorded during the sampling period.

Table 3.3S: Autumn (14/10/2022 - 23/10/2022) weather parameters

The autumn season is a wet season, with winds averaging 3.3 m/s, with a predominant wind direction of SSE.

Date	Average Wind (m/s)	Min Wind (m/s)	Max Wind (m/s)	Predominant Wind Direction	Average Temperature (°C)	Min Temperature (°C)	Max Temperature (°C)	Humidity (%)	Rainfall Events (>2mm)	Rainfall Events (>5mm)
14/10/2022	3.3	1.4	5.6	SSE	30	15	37	55	Yes	0
15/10/2022	3.4	1.5	5.7	SSE	31	16	38	57	No	0
16/10/2022	3.2	1.3	5.5	SSE	29	14	36	60	No	0
17/10/2022	3.3	1.4	5.6	SSE	30	15	37	61	Yes	0
18/10/2022	3.4	1.5	5.7	SSE	31	16	38	63	No	0
19/10/2022	3.2	1.3	5.5	SSE	29	14	36	65	No	0
20/10/2022	3.3	1.4	5.6	SSE	30	15	37	66	No	0
21/10/2022	3.4	1.5	5.7	SSE	31	16	38	63	No	0
22/10/2022	3.2	1.3	5.5	SSE	29	14	36	62	Yes	0
23/10/2022	3.3	1.4	5.6	SSE	30	15	37	61	No	0
Average	3.3	1.4	5.6	SSE	30.2	15.1	37.2	60.8		

Temperatures range from 14°C to 38°C, with an average temperature of 30.2°C. Humidity levels are moderate, ranging from 55% to 66%. Sampling was paused for 4 days due to rainfall events, and no heavy rainfall events (>5mm) were recorded during the sampling period.

Table 3.4S: Winter (2/12/2022 - 11/12/2022) weather parameters

The winter is a dry season, marked by stronger winds averaging 4.2 m/s, with a predominant wind direction of NE-E.

Date	Average Wind (m/s)	Min Wind (m/s)	Max Wind (m/s)	Predominant Wind Direction	Average Temperature (°C)	Min Temperature	Max Temperature (°C)	Humidity (%)	Rainfall Events (>2mm)	Rainfall Events (>5mm)
2/12/2022	4.2	2.9	7.2	NE-E	17	9	32	0	0	0
3/12/2022	4.3	3.0	7.3	NE-E	18	10	33	0	0	0
4/12/2022	4.1	2.8	7.1	NE-E	16	8	31	0	0	0
5/12/2022	4.2	2.9	7.2	NE-E	17	9	32	0	0	0
6/12/2022	4.3	3.0	7.3	NE-E	18	10	33	0	0	0
7/12/2022	4.1	2.8	7.1	NE-E	16	8	31	0	0	0
8/12/2022	4.2	2.9	7.2	NE-E	17	9	32	0	0	0
9/12/2022	4.3	3.0	7.3	NE-E	18	10	33	0	0	0
10/12/2022	4.1	2.8	7.1	NE-E	16	8	31	0	0	0
11/12/2022	4.2	2.9	7.2	NE-E	17	9	32	0	0	0
Average	4.2	2.9	7.2	NE-E	17.2	9.1	32.2	0	0	0

Temperatures range from 8°C to 33°C, with an average temperature of 17.2°C. Humidity was extremely low at 0%. No rainfall events were recorded during the sampling period, as sampling was paused on rainy days.

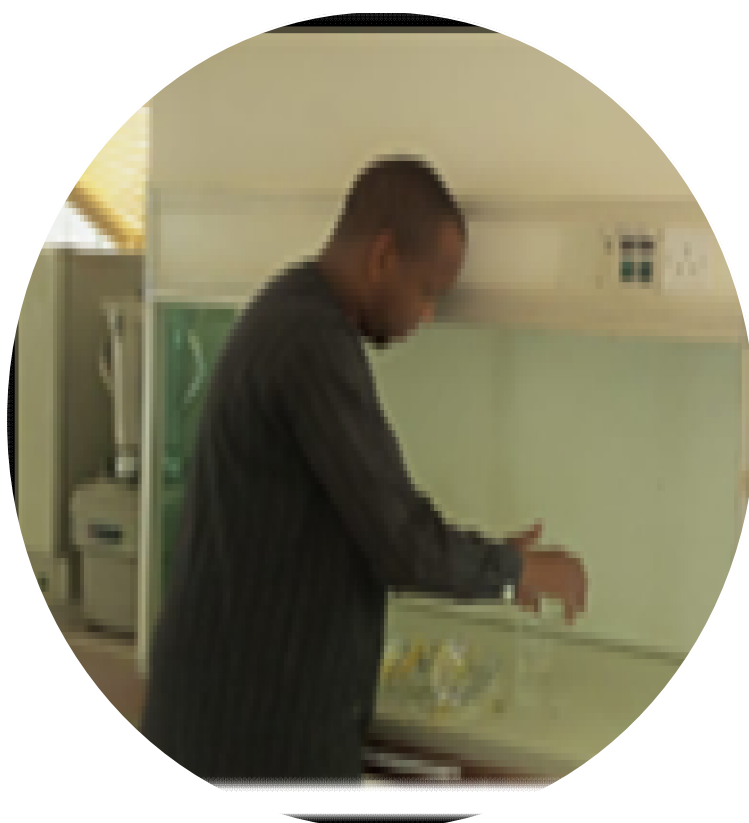


Figure 3.1S Sampling Processing within the Laminar Flow Hood

3.1A: Microplastics in Blank Samples

Tables 3.5S and 3.6S show the distribution of microplastics in procedural blanks to account for contamination during the sampling process. Table 3.5S shows the microplastics breakdown by season, while the second Table 3.6S categorises them by blank type (Lab, Field, and RO). Both tables reveal that the total count of microplastics in procedural blanks is 13 particles, indicating that contamination was minimal and consistent across different seasons and blank types.

Tables (3.5S and 3.36) also show that Lab blanks had the highest count of microplastics (8), followed by Field blanks (3) and RO blanks (2) (Miller et al., 2021). Fibre was the most common type of microplastic found in Lab blanks while fragments were predominant in the field, the RO water took the lead in films. The consistency in microplastic counts across seasons and blank types suggests that the procedural blanks effectively accounted for contamination, ensuring the accuracy of the microplastic analysis. (P. Sharma et al., 2024b).

Table 3.5S: Microplastics in blanks by Season

Season	Fibre	Fragment	Film	Total
Summer	1	1	0	2
Autumn	1	1	1	3
Spring	1	0	1	2
Winter	3	1	1	5
Total	7	3	3	13

Table 3.6S: Microplastics in Blanks

Blank Type	Fibre	Fragment	Film	Total
Lab	5	1	2	8
Field	1	2	0	3
RO	0	0	3	3
Total	7	3	3	13

Text 3.2A: Performance Evaluation of Low-Cost Sampler (LCS) against Reference Sampler (Ref): Precision Measures

When comparing the performance of a Low-Cost Sampler (LCS) to a Reference Sampler (Ref), precision measures are crucial to assess the accuracy and reliability of the LCS. Here's how MBE, RMSE, and R2 can be applied in this context (Feenstra et al., 2019) :

Mean Bias Error (MBE)

MBE measures the average difference between LCS and Ref concentrations.

Formula: $MBE = (1/n) * \sum (LCS - Ref)$ -----Eqn. 3.1S

- A positive MBE indicates that the LCS overestimates concentrations compared to the Ref.
- A negative MBE indicates that the LCS underestimates concentrations.
- An MBE close to zero suggests minimal bias between LCS and Ref.

Root Mean Square Error (RMSE)

RMSE quantifies the magnitude of errors between LCS and Ref concentrations.

Formula: $RMSE = \sqrt{[(1/n) * \sum (LCS - Ref)^2]}$ -----Eqn. 3.2S

- Lower RMSE values indicate better agreement between LCS and Ref.

Correlation Coefficients (R2)

R2 measures the strength and direction of the linear relationship between LCS and Ref concentrations.

Formula: $R^2 = 1 - [\sum (Ref - LCS)^2 / \sum (Ref - Mean(Ref))^2]$ -----3.3S

- R2 ranges from 0 to 1, where:
 - 0 indicates no correlation between LCS and Ref

- o 1 indicates perfect correlation

By calculating MBE, RMSE, and R2. The following performance can be identified if the:

- If the LCS is biased compared to the Ref (MBE)
- The magnitude of errors between LCS and Ref (RMSE)
- The strength of the linear relationship between LCS and Ref (R2)

These precision measures enable informed decisions on the suitability of the LCS for specific applications, potential calibration needs, and overall performance improvements.

Table 3.7S: Airborne Microplastic Abundance Across the Seasons (Reference Sampler N=40)

season		Date	Start	End	Fibre	Fragment	Film	Total	MP/m3
Spring	SP-R01	17/5/2022	9.00am	5.00pm	3.0	12.0	3.0	18.0	7.5
	SP-R02	18/5/2022	9.00am	5.00pm	2.0	13.0	3.0	18.0	7.5
	SP-R03	19/5/2022	9.00am	5.00pm	4.0	10.0	3.0	17.0	7.1
	SP-R04	20/5/2022	9.00am	5.00pm	3.0	13.0	3.0	19.0	7.9
	SP-R05	21/5/2022	9.00am	5.00pm	4.0	15.0	3.0	22.0	7.5
	SP-R06	22/5/2022	9.00am	5.00pm	3.0	13.0	3.0	19.0	7.9
	SP-R07	23/5/2022	9.00am	5.00pm	3.0	15.0	3.0	21.0	8.8
	SP-R08	24/5/2022	9.10am	5.10pm	3.0	16.0	2.0	21.0	8.8
	SP-R09	25/5/2022	9.00am	5.00pm	3.0	17.0	3.0	23.0	9.6
	SP-R10	26/5/2022	9.00am	5.00pm	2.0	17.0	4.0	23.0	9.6
Summer	SM-R01	11/6/2022	9.00am	5.00pm	5.0	18.0	2.0	25.0	10.4
	SM-R02	12/6/2022	9.00am	5.00pm	4.0	15.0	5.0	24.0	10.0
	SM-R03	13/06/2022	9.10am	5.10pm	4.0	15.0	5.0	24.0	10.0
	SM-R04	14/06/2022	9.00am	5.00pm	4.0	15.0	5.0	24.0	10.0
	SM-R05	15/06/2022	9.00am	5.00pm	4.0	16.0	4.0	24.0	10.0
	SM-R06	16/06/2022	9.00am	5.00pm	4.0	14.0	6.0	24.0	10.0
	SM-R07	17/06/2022	9.00am	5.00pm	3.0	13.0	5.0	21.0	8.8
	SM-R08	18/06/2022	9.00am	5.00pm	4.0	13.0	5.0	22.0	9.2
	SM-R09	19/06/2022	9.00am	5.00pm	4.0	13.0	6.0	23.0	9.6
	SM-R10	20/06/2022	9.00am	5.00pm	4.0	11.0	6.0	21.0	8.8

Autumn	AT-R01	14/10/2022	9.00am	5.00pm	9.0	17.0	3.0	29.0	12.1
	AT-R02	15/10/2022	9.00am	5.00pm	8.0	18.0	3.0	29.0	12.1
	AT-R03	16/10/2022	9.00am	5.00pm	8.0	18.0	6.0	32.0	13.3
	AT-R04	17/10/2022	9.00am	5.00pm	8.0	16.0	5.0	29.0	12.1
	AT-R05	18/10/2022	9.00am	5.00pm	7.0	17.0	6.0	30.0	11.7
	AT-R06	19/10/2022	9.00am	5.00pm	9.0	16.0	5.0	30.0	11.3
	AT-R07	20/10/2022	9.00am	5.00pm	9.0	17.0	6.0	32.0	12.5
	AT-R08	21/10/2022	9.00am	5.00pm	9.0	15.0	5.0	29.0	12.1
	AT-R09	22/10/2022	9.00am	5.00pm	8.0	21.0	4.0	33.0	12.1
	AT-R10	23/10/2022	9.00am	5.00pm	9.0	17.0	8.0	34.0	13.3
Winter	WT-R01	2/12/2022	9.00am	5.00pm	8.0	20.0	4.0	32.0	13.3
	WT-R02	3/12/2022	9.00am	5.00pm	7.0	22.0	6.0	35.0	14.6
	WT-R03	4/12/2022	9.00am	5.00pm	9.0	21.0	5.0	35.0	14.6
	WT-R04	5/12/2022	9.00am	5.00pm	6.0	26.0	5.0	37.0	15.4
	WT-R05	6/12/2022	9.00am	5.00pm	9.0	23.0	4.0	36.0	13.3
	WT-R06	7/12/2022	9.00am	5.00pm	8.0	20.0	5.0	33.0	13.8
	WT-R07	8/12/2022	9.00am	5.00pm	9.0	21.0	8.0	38.0	11.7
	WT-R08	9/12/2022	9.00am	5.00pm	9.0	18.0	5.0	32.0	13.3
	WT-R09	10/12/2022	9.00am	5.00pm	9.0	19.0	6.0	34.0	12.9
	WT-R10	11/12/2022	9.00am	5.00pm	9.0	20.0	7.0	36.0	15.0

Table 3.8S: Airborne Microplastic Abundance across the Seasons (Low-cost Sampler N=40)

Season	Filter ID	Date	Start	End	Fibre	Fragment	Film	Total	MP/m3
Spring	SP- L01	17/5/2022	9.00 am	5.00 pm	4	12	3	19	7.9
	SP- L02	18/5/2022	9.00 am	5.00 pm	3	13	2	18	7.5
	SP- L03	19/5/2022	9.00 am	5.00 pm	3	11	1	15	6.3
	SP- L04	20/5/2022	9.00 am	5.00 pm	4	13	2	19	7.9
	SP- L05	21/5/2022	9.00 am	5.00 pm	2	13	2	17	7.1
	SP- L06	22/5/2022	9.00 am	5.00 pm	4	12	2	18	7.5
	SP- L07	23/5/2022	9.00 am	5.00 pm	4	13	3	20	8.3
	SP- L08	24/5/2022	9.10 am	5.10 pm	4	13	2	19	7.9
	SP- L09	25/5/2022	9.00 am	5.00 pm	3	13	5	21	8.8
	SP- L10	26/5/2022	9.00 am	5.00 pm	4	12	5	21	8.8
Summer	SM-L01	11/6/2022	11.00 am	7.00 pm	4	12	5	21	8.8
	SM-L02	12/6/2022	9.00 am	5.00 pm	4	14	3	21	8.8
	SM-L03	13/06/2022	9.10 am	5.10 pm	4	14	3	21	8.8
	SM-L04	14/06/2022	9.00 am	5.00 pm	6	12	4	22	9.2

	SM-L05	15/06/2022	9.00 am	5.00 pm	3	14	5	22	9.2
	SM-L06	16/06/2022	9.00 am	5.00 pm	5	13	4	22	9.2
	SM-L07	17/06/2022	9.00 am	5.00 pm	4	14	5	23	9.6
	SM-L08	18/06/2022	9.00 am	5.00 pm	5	16	4	25	10.4
	SM-L09	19/06/2022	9.00 am	5.00 pm	6	14	3	23	9.6
	SM-L10	20/06/2022	10.00 am	6.00 pm	5	14	4	23	9.6
Autumn	AT-L01	14/10/2022	9.00 am	5.00 pm	6	15	8	29	12.1
	AT-L02	15/10/2022	9.00 am	5.00 pm	8	16	5	29	12.1
	AT-L03	16/10/2022	9.00 am	5.00 pm	9	16	5	30	12.5
	AT-L04	17/10/2022	9.00 am	5.00 pm	11	9	2	22	9.2
	AT-L05	18/10/2022	9.00 am	5.00 pm	9	15	4	28	11.7
	AT-L06	19/10/2022	9.00 am	5.00 pm	6	15	4	25	10.4
	AT-L07	20/10/2022	9.00 am	5.00 pm	7	14	5	26	10.8
	AT-L08	21/10/2022	9.00 am	5.00 pm	7	18	5	30	12.5
	AT-L09	22/10/2022	9.00 am	5.00 pm	7	19	6	32	13.3
	AT-L10	23/10/2022	9.00 am	5.00 pm	12	16	7	35	14.6
Winter	WT-L01	2/12/2022	9.00 am	5.00 pm	5	22	7	34	14.2
	WT-L02	3/12/2022	9.00 am	5.00 pm	6	22	6	34	14.2
	WT-L03	4/12/2022	9.00 am	5.00 pm	5	23	7	35	14.6
	WT-L04	5/12/2022	9.00 am	5.00 pm	9	21	5	35	14.6
	WT-L05	6/12/2022	9.00 am	5.00 pm	5	23	6	34	14.2
	WT-L06	7/12/2022	9.00 am	5.00 pm	4	23	7	34	14.2
	WT-L07	8/12/2022	9.00 am	5.00 pm	8	22	5	35	14.6
	WT-L08	9/12/2022	9.00 am	5.00 pm	5	23	6	34	14.2
	WT-L09	10/12/2022	9.00 am	5.00 pm	7	23	5	35	14.6
	WT-L10	11/12/2022	9.00 am	5.00 pm	6	23	6	35	14.6

Table 3.9S: Airborne Microplastic Particle Sizes Distribution Across the Seasons (Reference Sampler N=40)

Seasons	ID	0-20µm	20-50µm	50-75µm	75-100µm	100-150µm	150-200µm	200-1010µm	Sum
Spring	SP-R01	2	2	2	2	5	2	3	18
	SP-R02	1	2	2	3	3	3	2	16
	SP-R03	2	2	1	3	3	3	3	17
	SP-R04	1	1	3	3	6	5	5	24
	SP-R05	3	1	3	4	7	5	3	26
	SP-R06	2	2	2	3	3	3	3	18

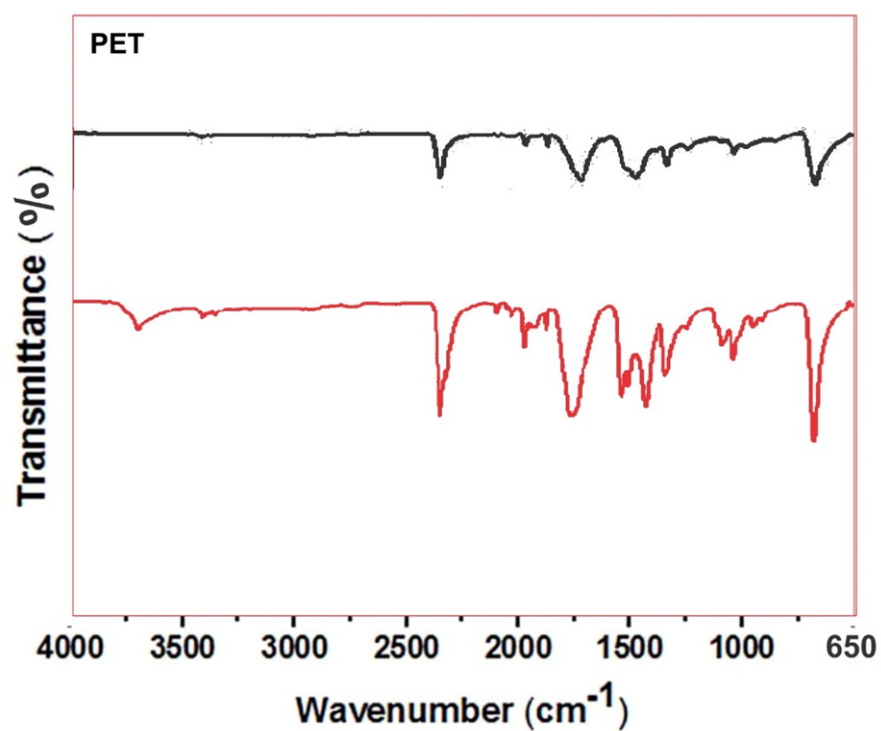
	SP-R07	3	2	3	3	3	3	3	20
	SP-R08	2	2	2	3	3	3	5	20
	SP-R09	3	2	3	3	5	5	4	25
	SP-R10	3	3	1	3	4	4	3	21
		22	19	22	30	42	36	34	205
Summer	SM-C01	3	1	2	2	5	7	5	25
	SM-C02	1	3	2	4	6	4	5	25
	SM-C03	1	3	3	3	6	2	2	20
	SM-C04	3	3	3	3	7	4	3	26
	SM-C05	1	3	2	2	6	7	3	24
	SM-C06	2	2	3	3	8	5	4	27
	SM-C07	2	2	2	3	4	4	4	21
	SM-C08	1	1	1	3	6	5	6	23
	SM-C09	2	1	2	4	7	5	5	26
	SM-C10	2	2	2	2	7	5	3	23
		18	21	22	29	62	48	40	240
Autumn	AT-C01	2	3	3	4	8	8	5	33
	AT-C02	1	2	4	3	9	9	6	34
	AT-C03	3	3	3	3	7	7	7	33
	AT-C04	2	3	3	4	7	4	5	28
	AT-C05	3	3	3	5	7	3	7	31
	AT-C06	1	5	3	3	7	5	5	29
	AT-C07	2	3	3	3	5	9	5	30
	AT-C08	2	2	3	3	7	5	5	27
	AT-C09	3	3	3	3	6	4	5	27
	AT-C10	3	4	3	4	6	9	5	34
		22	31	31	35	69	63	55	306
Winter	WT-C01	4	2	1	5	8	7	7	34
	WT-C02	3	5	3	4	7	8	8	38
	WT-C03	3	3	2	3	7	7	7	32
	WT-C04	5	3	3	5	6	7	8	37
	WT-C05	3	4	2	4	7	7	6	33
	WT-C06	3	1	2	3	8	8	8	33
	WT-C07	2	3	3	5	5	7	5	30
	WT-C08	3	2	3	4	8	8	6	34
	WT-C09	2	5	3	3	7	7	5	32
	WT-C10	3	2	1	5	7	7	7	32
	total	31	30	23	41	70	73	67	335
	Overall	93	101	98	135	243	220	196	1086

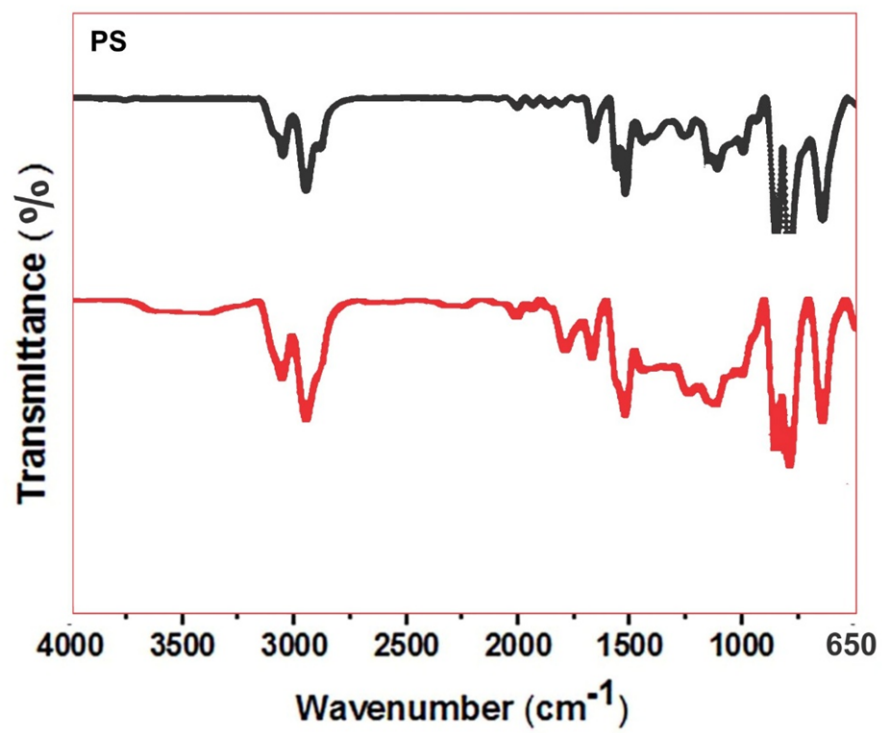
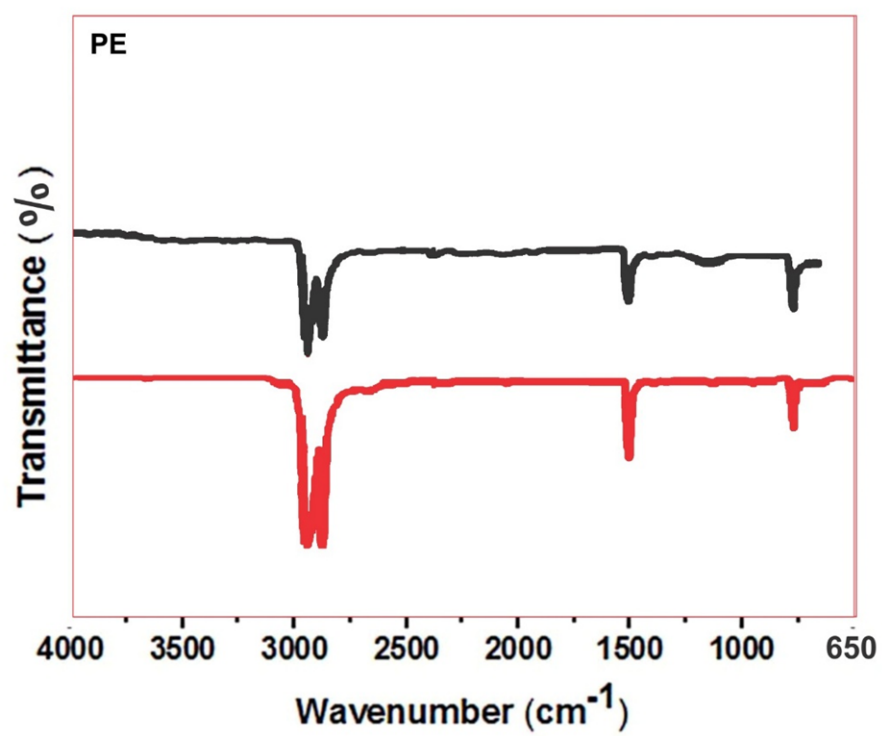
Table 3.10S: Airborne Microplastic Particle Sizes Distribution Across the Seasons (Low-cost Sampler N=40)

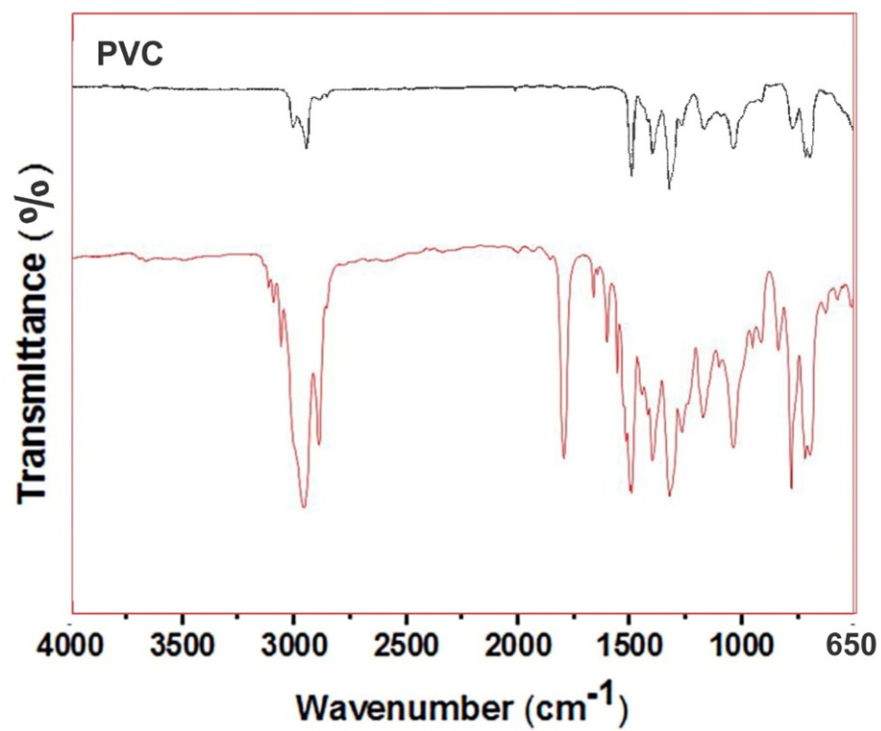
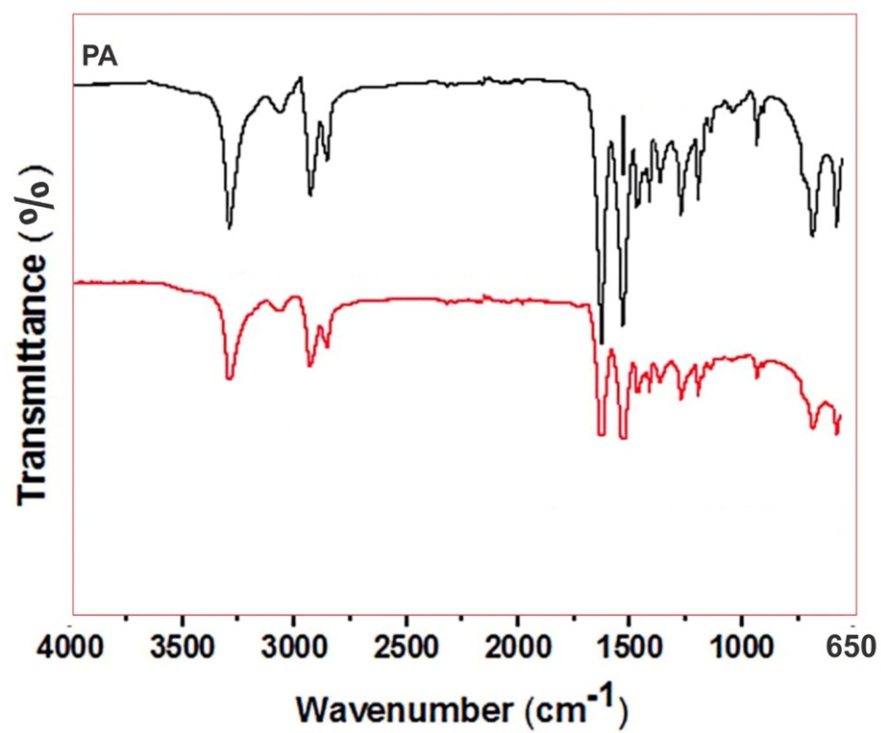
Seasons	Filter ID	0-20µm	20-50µm	50-75µm	75-100µm	100-150µm	150-200µm	200-1010µm	Sum
	SP- L01	2	1	3	3	4	3	3	19
Spring	SP- L02	1	3	2	3	2	5	2	18
	SP- L03	1	1	1	1	5	3	3	15
	SP- L04	2	2	3	1	5	5	1	19
	SP- L05	1	1	1	1	5	4	4	17
	SP- L06	2	1	2	3	5	3	2	18
	SP- L07	1	1	2	3	5	5	3	20
	SP- L08	2	2	0	3	5	4	3	19
	SP- L09	1	1	1	2	5	6	5	21
	SP- L10	4	1	1	0	5	7	3	21
		17	14	16	20	46	45	29	187
Summer	SM-L01	0	3	2	2	5	0	9	21
	SM-L02	0	3	3	3	6	5	1	21
	SM-L03	0	1	1	2	8	8	1	21
	SM-L04	0	5	1	4	5	5	2	22
	SM-L05	2	2	3	2	2	7	4	22
	SM-L06	1	3	3	2	5	5	3	22
	SM-L07	1	1	1	5	6	5	4	23
	SM-L08	1	2	2	6	6	5	3	25
	SM-L09	3	3	3	3	4	5	2	23
	SM-L10	3	3	3	2	5	3	4	23
		11	26	22	31	52	48	33	223
Autumn	AT-L01	1	2	1	2	7	8	8	29
	AT-L02	1	1	1	5	9	5	7	29
	AT-L03	1	1	5	2	9	5	7	30
	AT-L04	2	2	2	2	3	4	7	22
	AT-L05	2	5	4	4	5	4	4	28
	AT-L06	2	3	2	4	5	5	4	25
	AT-L07	2	3	3	3	7	4	4	26
	AT-L08	1	1	2	3	9	7	7	30
	AT-L09	1	1	5	1	8	8	8	32
	AT-L10	3	3	3	5	7	7	7	35

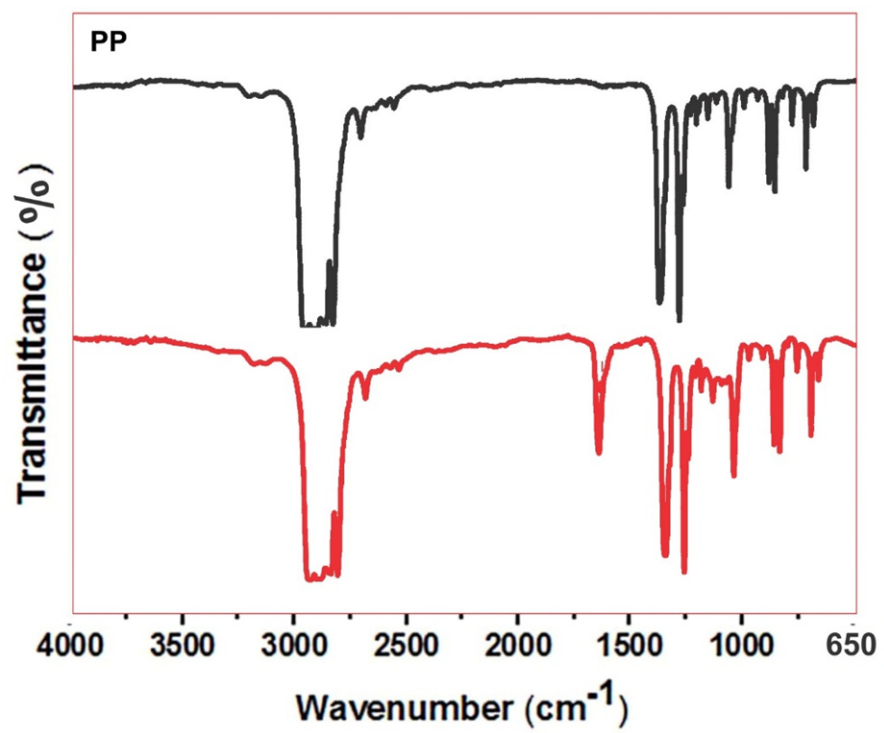
		16	22	28	31	69	57	63	286
Winter	WT-L01	3	3	3	3	5	8	9	34
	WT-L02	5	4	3	2	9	5	6	34
	WT-L03	4	5	2	3	8	9	4	35
	WT-L04	3	2	5	3	10	7	5	35
	WT-L05	4	1	2	5	8	8	6	34
	WT-L06	2	3	7	6	6	5	5	34
	WT-L07	6	1	0	5	5	7	11	35
	WT-L08	2	3	2	5	9	5	8	34
	WT-L09	5	2	0	9	8	5	6	35
	WT-L10	4	7	2	7	3	6	6	35
	total	38	31	26	48	71	65	66	345
	Overall	82	93	92	130	238	215	191	1041

FTIR Polymer Spectra









Appendix B

Table 4.1a: Airborne Samples for Dry Season Upwind

Date	Date	Sample ID	MP Fibres	MP Fragments	MP Films	Total
2/2/2021	9.00 am -6.00 pm	UPD01	4.00	1.00	3.00	8.00
3/2/2021	9.00 am -6.00 pm	UPD02	7.00	2.00	3.00	12.00
4/2/2021	9.00 am -6.00 pm	UPD03	7.00	2.00	2.00	11.00
5/2/2021	9.00 am -6.00 pm	UPD04	6.00	1.00	3.00	10.00
6/2/2021	9.00 am -6.00 pm	UPD05	5.00	1.00	4.00	10.00
7/2/2021	9.00 am -6.00 pm	UPD06	5.00	2.00	3.00	10.00
8/2/2021	9.00 am -6.00 pm	UPD07	6.00	1.00	3.00	10.00
9/2/2021	9.00 am -6.00 pm	UPD08	6.00	1.00	2.00	9.00
10/2/2021	9.00 am -6.00 pm	UPD09	6.00	2.00	2.00	10.00
11/2/2021	9.00 am -6.00 pm	UPD10	5.00	3.00	4.00	12.00
	Total		57.00	16.00	29.00	102.00
	Dry	MP/m3	2.38	0.67	1.21	4.25
	Std		0.90	0.66	0.70	1.17

Table 4.1b: Airborne Samples for Wet Season Upwind

Date	Time		MP Fibres	MP Fragments	MP Films	Total
01/07/2021	9.00 am -5.00 pm	UPW01	3.00	1.00	1.00	5.00
02/07/2021	9.00 am -5.00 pm	UPW02	3.00	1.00	2.00	6.00
03/07/2021	9.00 am -5.00 pm	UPW03	5.00	1.00	2.00	8.00
04/07/2021	9.00 am -5.00 pm	UPW04	3.00	2.00	1.00	6.00
05/07/2021	9.00 am -7.00 pm	UPW05	3.00	2.00	2.00	7.00
06/07/2021	9.00 am -5.00 pm	UPW06	2.00	1.00	1.00	4.00
07/07/2021	9.00 am -5.00 pm	UPW07	3.00	1.00	2.00	6.00
08/07/2021	9.00 am -5.00 pm	UPW08	5.00	2.00	2.00	9.00
09/07/2021	9.00 am -5.00 pm	UPW09	5.00	2.00	1.00	8.00
10/07/2021	9.00 am -5.00 pm	UPW10	3.00	1.00	3.00	7.00
	Total		35.00	14.00	17.00	66.00
	Wet	MP/m3	1.46	0.58	0.71	2.75
	Std		1.02	0.49	0.64	1.43

Table 4.2a: Airborne Samples for Dry Season On-site

Date	Duration	Filter ID	MP Fibres	MP Fragments	MP Films	Total
14/02/2021	9.00 am -5.00 pm	DON01	12.00	24.00	4.00	40.00
16/02/2021	9.00 am -5.00 pm	DON02	9.00	26.00	7.00	42.00
18/02/2021	9.00 am -5.00 pm	DON03	9.00	22.00	5.00	36.00
20/02/2021	9.00 am -5.00 pm	DON04	7.00	20.00	5.00	32.00
22/02/2021	9.00 am -5.00 pm	DON05	7.00	18.00	5.00	30.00
24/02/2021	9.00 am -5.00 pm	DON06	9.00	23.00	5.00	37.00
26/02/2021	9.00 am -5.00 pm	DON07	5.00	20.00	5.00	30.00
28/02/2021	9.00 am -5.00 pm	DON08	13.00	23.00	7.00	43.00
02/03/2021	9.00 am -5.00 pm	DON09	7.00	20.00	2.00	29.00
04/03/2021	9.00 am -5.00 pm	DON10	9.00	20.00	5.00	34.00
06/03/2021	9.00 am -5.00 pm	DON11	8.00	20.00	4.00	32.00
08/03/2021	9.00 am -5.00 pm	DON12	7.00	24.00	5.00	36.00
10/03/2021	9.00 am -5.00 pm	DON13	9.00	23.00	5.00	37.00
12/03/2021	10.00 am -7.00 pm	DON14	7.00	24.00	4.00	35.00
14/03/2021	9.00 am -5.00 pm	DON15	5.00	22.00	7.00	34.00
16/03/2021	9.00 am -5.00 pm	DON16	8.00	22.00	6.00	36.00
18/03/2021	9.00 am -5.00 pm	DON17	5.00	23.00	5.00	33.00
20/03/2021	9.00 am -5.00 pm	DON18	7.00	20.00	4.00	31.00
22/03/2021	9.00 am -5.00 pm	DON19	7.00	23.00	6.00	36.00
24/03/2021	9.00 am -5.00 pm	DON20	6.00	20.00	3.00	29.00
26/03/2021	9.30 am -5.30 pm	DON21	5.00	21.00	6.00	32.00
		Total	161.0	458.0	105.0	724.00
	Dry	MP/m3	3.19	9.09	2.08	14.37
		Std	2.08	1.92	1.23	3.87
		Min		12.08	Max	17.92

Table 4.2b: Airborne Samples for Wet Season On-site

Date	Duration	Filter ID	MP Fibres	MP Fragments	MP Films	Total
16/07/2021	9.00 am -5.00 pm	WON01	7.0	16.0	3.0	26.0
18/07/2021	9.00 am -5.00 pm	WON02	4.0	17.0	3.0	24.0
20/07/2021	9.00 am -5.00 pm	WON03	7.0	17.0	1.0	25.0
22/07/2021	9.00 am -5.00 pm	WON04	5.0	15.0	5.0	25.0

24/07/2021	9.00 am -5.00 pm	WON05	5.0	16.0	7.0	28.0
26/07/2021	9.00 am -5.00 pm	WON06	5.0	15.0	7.0	27.0
28/07/2021	9.00 am -5.00 pm	WON07	2.0	17.0	9.0	28.0
30/07/2021	9.00 am -5.00 pm	WON08	2.0	19.0	3.0	24.0
01/08/2021	9.00 am -5.00 pm	WON09	7.0	15.0	1.0	23.0
03/08/2021	9.00 am -5.00 pm	WON10	6.0	19.0	1.0	26.0
05/08/2021	9.00 am -5.00 pm	WON11	6.0	18.0	5.0	29.0
07/08/2021	9.00 am -5.00 pm	WON12	5.0	21.0	3.0	29.0
09/08/2021	9.00 am -5.00 pm	WON13	7.0	20.0	3.0	30.0
11/08/2021	10.00 am -7.00 pm	WON14	9.0	23.0	5.0	37.0
13/08/2021	9.00 am -5.00 pm	WON15	6.0	16.0	5.0	27.0
15/08/2021	9.00 am -5.00 pm	WON16	5.0	15.0	7.0	27.0
17/08/2021	9.00 am -5.00 pm	WON17	3.0	17.0	5.0	25.0
19/08/2021	9.00 am -5.00 pm	WON18	7.0	19.0	3.0	29.0
21/08/2021	9.00 am -5.00 pm	WON19	5.0	15.0	4.0	24.0
23/08/2021	9.00 am -5.00 pm	WON20	5.0	19.0	3.0	27.0
25/08/2021	9.30 am -5.30 pm	WON21	5.0	21.0	4.0	30.0
		Total	113.0	370.0	87.0	570.0
	Wet	MP/m3	2.24	7.34	1.73	11.31
		Std	1.68	2.28	2.08	3.00
			Min	9.58	Max	15.42

Table 4.3a: MP in Soil Samples for Dry Season On-site

Date	Sample ID	Fibers	Fragments	Films	Total
visit 1 14/02/2021	OND1P1	35.00	65.00	23.00	123.00
	OND1P2	34.00	61.00	20.00	115.00
	OND1P3	32.00	65.00	28.00	125.00
	OND1P4	34.00	64.00	21.00	119.00
	OND1P5	34.00	76.00	22.00	132.00
	OND1P6	35.00	59.00	25.00	119.00
	OND1P7	34.00	58.00	24.00	116.00
visit 2 6/03/2021	OND2P1	35.00	76.00	26.00	137.00
	OND2P2	35.00	63.00	28.00	126.00

	OND2P3	35.00	59.00	21.00	115.00
	OND2P4	35.00	73.00	23.00	131.00
	OND2P5	34.00	61.00	23.00	118.00
	OND2P6	37.00	60.00	26.00	123.00
	OND2P7	39.00	70.00	28.00	137.00
visit 3 26/03/2021	OND3P1	39.00	64.00	25.00	128.00
	OND3P2	39.00	77.00	25.00	141.00
	OND3P3	34.00	72.00	25.00	131.00
	OND3P4	33.00	71.00	23.00	127.00
	OND3P5	35.00	69.00	22.00	126.00
	OND3P6	37.00	65.00	23.00	125.00
	OND3P7	36.00	69.00	25.00	130.00
	Total	741.00	1397.00	506.00	2644.00

Table 4.3b: MP in Soil Samples for Wet Season On-site

Date	Sample ID	Fibers	Fragments	Films	Total
visit 1 16/07/2021	ONW1P1	40.00	67.00	33.00	140.00
	ONW1P2	44.00	76.00	25.00	145.00
	ONW1P3	45.00	67.00	31.00	143.00
	ONW1P4	41.00	54.00	30.00	125.00
	ONW1P5	35.00	75.00	28.00	138.00
	ONW1P6	45.00	58.00	22.00	125.00
	ONW1P7	46.00	57.00	33.00	136.00
visit 2 5/08/2021	ONW2P1	39.00	79.00	23.00	141.00
	ONW2P2	39.00	69.00	25.00	133.00
	ONW2P3	44.00	77.00	25.00	146.00
	ONW2P4	45.00	79.00	23.00	147.00
	ONW2P5	44.00	76.00	22.00	142.00
	ONW2P6	48.00	59.00	23.00	130.00
	ONW2P7	44.00	78.00	25.00	147.00
visit 3 25/08/2021	ONW3P1	45.00	49.00	21.00	115.00
	ONW3P2	41.00	59.00	21.00	121.00
	ONW3P3	47.00	60.00	28.00	135.00
	ONW3P4	36.00	67.00	31.00	134.00

	ONW3P5	41.00	69.00	34.00	144.00
	ONW3P6	41.00	79.00	31.00	151.00
	ONW3P7	43.00	73.00	30.00	146.00
	Total	893.00	1427.00	564.00	2884.00

Table 4.4a: Dry Season Bulk Density

Sample #	Mass (g)	Volume (cm³)	Bulk Density (g/cm³)	Bulk Density (kg/m³)	Porosity (e)
1	7.84	5.50	1.39	1420.00	0.48
2	8.96	6.20	1.44	1440.00	0.46
3	6.96	4.90	1.42	1420.00	0.46
4	9.28	6.50	1.46	1430.00	0.45
5	8.16	5.70	1.43	1430.00	0.46
6	7.36	5.10	1.44	1440.00	0.46
7	8.64	6.00	1.44	1440.00	0.46
Avg.	8.17	5.70	1.43	1431.43	0.46

Table 4.4b: Wet Season Bulk Density

Sample #	Mass (g)	Volume (cm³)	Bulk Density (g/cm³)	Bulk Density (kg/m³)	Porosity (e)
1	9.52	5.6	1.7	1700	0.36
2	10.72	6.4	1.67	1670	0.37
3	8.8	5.2	1.69	1690	0.36
4	11.44	6.7	1.71	1710	0.35
5	10.24	6	1.7	1700	0.36
6	9.28	5.4	1.72	1720	0.35
7	10.96	6.5	1.68	1680	0.37
Avg.	10.14	5.97	1.70	1695.71	0.36

Table 4.5a: Microplastics in Blanks by Season

Season	Fibre	Fragment	Film	Total
Dry	3	2	3	8
Wet	2	2	2	6
Total	5	4	5	14

Table 4.5b: Microplastics in Blanks

Blank Type	Fibre	Fragment	Film	Total
Lab	2	1	0	3
Field	3	3	3	9
RO	1	0	1	2
Total	5	4	3	14

Text 4.1

The selection of Fick's law for microplastic flux estimation was based on its well-established framework in environmental science, which effectively models diffusion processes. Fick's law, which describes the diffusion process as the movement of particles from an area of higher concentration to an area of lower concentration proportional to the concentration gradient, has been widely accepted (Fick, 1855). This fundamental principle has been successfully applied in various environmental studies, including Mechanisms of Pollutant Exchange at Soil-Vegetation-Atmosphere Interfaces and Atmospheric Fate (Saint-Jean et al., 2020), and estimation of porewater nutrient fluxes under diffusive conditions from the soil-water interface (Lavery et al., 2001).

Additionally, previous research has demonstrated the versatility of Fick's law in understanding the behaviour of microplastics and associated pollutants. For instance,

Fick's law has been used to determine the leaching of Polybrominated Diphenyl Ethers (PBDEs) from microplastics under simulated gut conditions (B. Sun & Zeng, 2022). Moreover, researchers have applied Fick's law to measure PBDE diffusion coefficients in Low-Density Polyethylene (LDPE) and silicone rubber (Narváez Valderrama et al., 2016). Furthermore, Fick's law has been employed to understand interface exchanges for assessing the environmental sorption of additives from microplastics (da Costa et al., 2024). This empirical validation in the context of this study enhances the credibility of the findings. Fick's law is adaptable to various conditions, making it suitable for this study's objectives. Seasonal variations in temperature, humidity, precipitation, wind patterns, soil texture, and vegetation cover, are considered. This research aims to quantify microplastic flux in wet and dry seasons.

Text 4.2

ρ -soil (Bulk density of soil, kg/m³):

The bulk density measures the mass per unit volume of soil, including solid particles and pore spaces. The amount of material available per unit volume of soil affects the resuspension of microplastics. Denser soil, containing more mass, prevents resuspension, leading to low levels of microplastic particles being released into the air (C. Wang et al., 2022). The parameter ρ -soil ensures that the model considers the physical availability of microplastics in the soil, making the equation more representative of real-world conditions.

C-soil (Concentration of microplastics in soil, MP/g):

This parameter represents the initial source strength of microplastic contamination in the soil. The higher the concentration of microplastics in the soil, the more particles are available for potential emission into the atmosphere.

e (Void ratio, dimensionless):

The void ratio describes the ratio of pore space to solid material in the soil. A higher void ratio indicates more space between soil particles. Soil void magnitude impacts microplastic transport; higher voids promote resuspension, whereas lower voids suppress it. The void ratio is crucial for describing how easily microplastics can migrate from the soil to the air, as it affects both particle mobility and aeration of the soil (A. Kumar, Mishra, et al., 2023).

C-air (Concentration of microplastics in air, MP/m³):

The concentration of microplastics in the air influences the net flux of particles from soil to air. The gradient between soil and air concentrations drives diffusion-like

processes, where microplastics move from high concentration (soil) to low concentration (air).

H (Height of flux measurement, m):

The height at which the flux measurements affect the microplastic concentration disperses in the atmosphere. The greater the height, the longer the distance microplastics travel further, and they may disperse or settle before reaching that point (Pukos et al., 2023)(Pukos et al., 2023). The Incorporation of the height in the model accounts for the vertical dispersion of particles, which is crucial for accurate predictions of airborne microplastic concentrations at various altitudes.

Overall Scientific Justification:

This equation accounts for **mass balance** and **gradient-driven transport**, where microplastic particles move from regions of higher concentration (soil) to lower concentration (air) due to environmental factors like soil properties, air properties, and the difference in concentrations. The equation's parameters consider the source strength from the soil and the transport dynamics of air concentration and vertical dispersion. The proposed equation enables a more precise prediction of the rate of change in airborne microplastic concentration.

Table 4.6a: Particles Distribution of Airborne MPs On-site Dry Season

Fibres	Fibres	Fragments	Films	Total
>5-20	20.00	70.00	40.00	130.00
20-50	22.00	71.00	39.00	132.00
50-75	16.00	78.00	15.00	109.00
75-100	60.00	81.00	15.00	156.00
100-150	60.00	24.00	10.00	94.00
150-200	38.00	17.00	2.00	57.00
200-1010	34.00	9.00	3.00	46.00
Total	250.00	350.00	124.00	724.00
MP/m3	4.96	6.94	2.46	14.37

Table 4.6b: Particles Distribution of Airborne MPs On-site Wet Season

Size Range (µm)	Fibres	Fragments	Films	Total
>5-20	22.00	59.00	19.00	100.00

20-50	15.00	50.00	15.00	80.00
50-75	10.00	40.00	10.00	60.00
75-100	17.00	48.00	18.00	83.00
100-150	25.00	55.00	10.00	90.00
150-200	10.00	60.00	10.00	80.00
200-1010	14.00	58.00	5.00	77.00
Total	113.00	370.00	87.00	570.00
MP/m3	2.24	7.34	1.73	11.31

Table 4.7a: Particles Distribution of Airborne MPs Upwind Dry Season

Size Range (µm)	Fibres	Fragments	Films	Total
>5-20	11.00	3.00	5.00	19.00
20-50	11.00	3.00	2.00	16.00
50-75	12.00	3.00	5.00	20.00
75-100	7.00	2.00	4.00	13.00
100-150	4.00	3.00	4.00	11.00
150-200	4.00	1.00	4.00	9.00
200-1010	8.00	1.00	5.00	14.00
Total	57.00	16.00	29.00	102.00

Table 4.7b: Particles Distribution of Airborne MPs Upwind Wet Season

Size Range (µm)	Fibres	Fragments	Films	Total
>5-20	5.00	2.00	2.00	9.00
20-50	5.00	2.00	3.00	10.00
50-75	6.00	3.00	3.00	12.00
75-100	6.00	3.00	2.00	11.00
100-150	5.00	2.00	3.00	10.00
150-200	5.00	1.00	2.00	8.00
200-1010	3.00	1.00	2.00	6.00
Total	35.00	14.00	17.00	66.00

Appendix C

Table 5.1A: Dry Season Conversion of MP/m³ to Mass concentration in µg/m³

Distance	Shapes	MP	Mass (ug)	MP/m3	Mass (ug/m3)	Model Value (ug/m3)
10	Fibre	7	2.02E-02	2.92	8.43E-03	7.68E-03
	fragment	19	5.49E-02	7.92	2.29E-02	2.37E-02
	Films	2	5.78E-03	0.83	2.41E-03	8.81E-04
	Sum MP	27	8.09E-02	11.67	3.37E-02	3.22E-02
	Total Mass (ug)	0.078	8.09E-02			
50	Fibre	3	2.24E-02	1.25	9.33E-03	9.10E-03
	fragment	12	8.96E-02	5.00	3.73E-02	2.80E-02
	Films	5	3.73E-02	2.08	1.56E-02	2.80E-02
	Sum MP	28	1.60E-01	8.33	6.22E-02	6.52E-02
	Total Mass (ug)	0.209	1.49E-01			
100	Fibre	7	2.33E-02	2.92	9.72E-03	1.04E-02
	fragment	22	7.33E-02	9.17	3.06E-02	3.20E-02
	Films	4	1.33E-02	1.67	5.56E-03	1.19E-03
	Sum MP	30	1.00E-01	13.75	4.58E-02	4.36E-02
	Total Mass (ug)	0.1	1.10E-01			
200	fibre	5	3.00E-02	2.08	1.25E-02	1.44E-02
	fragment	6	3.60E-02	2.50	1.50E-02	1.31E-02
	films	2	1.20E-02	0.83	5.00E-03	2.23E-03
	Sum MP	12	7.80E-02	5.42	3.25E-02	2.98E-02
	Total Mass (ug)	0.072	7.80E-02			
300	fibre	5	1.55E-02	2.08	6.44E-03	8.81E-03
	fragment	2	6.18E-03	0.83	2.58E-03	3.28E-04
	films	4	1.24E-02	1.67	5.15E-03	8.45E-03
	Sum MP	10	4.20E-02	4.58	1.42E-02	1.76E-02
	Total Mass (ug)	0.0309	3.40E-02			
500	fibre	1	3.17E-03	0.42	1.32E-03	1.42E-03
	fragment	4	1.27E-02	1.67	5.28E-03	4.36E-03
	films	1	3.17E-03	0.42	1.32E-03	1.62E-04
	Sum MP	6	4.98E-02	2.50	7.92E-03	5.94E-03

	Total Mass (ug)	0.019	1.90E-02			
1000	fibre	1	4.80E-03	0.42	2.00E-03	1.42E-03
	fragment	2	9.61E-03	0.83	4.00E-03	4.36E-03
	films	1	4.80E-03	0.42	2.00E-03	1.62E-04
	Sum MP	4	4.80E-03	1.67	8.01E-03	5.94E-03
	Total Mass (ug)	0.999	1.92E-02			

Table 5.1B: Wet Season Conversion of MP/m³ to Mass concentration in µg/m³

Distance	Shapes	MP	Mass (ug)	MP/m3	Mass (ug/m3)	Model Value (ug/m3)
10	fibre	9.00	5.02E-02	3.75	2.09E-02	1.51E-02
	fragment	12.00	6.69E-02	5.00	2.79E-02	4.51E-02
	films	5.00	2.79E-02	2.08	1.16E-02	1.60E-03
	Sum MP	26.00	0.145	10.83	6.04E-02	6.18E-02
	Total Mass (ug)	0.145	0.145			
50	fibre	6.00	5.70E-02	2.50	2.38E-02	1.85E-02
	fragment	12.00	1.14E-01	5.00	4.75E-02	5.52E-02
	films	2.00	1.90E-02	0.83	7.92E-03	1.95E-03
	Sum MP	20.00	0.19	8.33	7.92E-02	7.56E-02
	Total Mass (ug)	0.19	1.90E-01			
100	fibre	7.00	5.38E-02	2.92	2.24E-02	2.28E-02
	fragment	20.00	1.54E-01	8.33	6.40E-02	6.81E-02
	films	1.00	7.68E-03	0.42	3.20E-03	2.41E-03
	Sum MP	28.00	0.22	11.67	8.96E-02	9.33E-02
	Total Mass (ug)	0.215	0.22			
200	fibre	5	2.96E-02	2.08	1.23E-02	1.31E-02
	fragment	15.00	8.87E-02	6.25	3.69E-02	3.92E-02
	films	1.00	5.91E-03	0.42	2.46E-03	1.38E-03
	Sum MP	21.00	0.124	8.75	5.17E-02	5.37E-02
	Total Mass (ug)	0.124	0.124			
300	fibre	5.00	1.13E-02	2.08	4.72E-03	8.81E-03
	fragment	14.00	3.17E-02	5.83	1.32E-02	3.28E-04
	films	1.00	2.27E-03	0.42	9.44E-04	8.45E-03
	Sum MP	20.00	0.045	8.33	1.89E-02	1.76E-02
	Total Mass (ug)	0.045	0.045			

500	fibre	5.00	1.02E-02	2.08	4.26E-03	4.83E-03
	fragment	15.00	3.07E-02	6.25	1.28E-02	1.44E-02
	films	1.00	2.05E-03	0.42	8.53E-04	5.08E-04
	Sum MP	21.00	4.30E-02	8.75	1.79E-02	1.97E-02
	Total Mass (ug)	0.043	0.04			
1000	fibre	4.00	3.42E-03	1.67	1.43E-03	1.98E-03
	fragment	14.00	1.20E-02	5.83	4.99E-03	5.90E-03
	films	1.00	8.56E-04	0.42	3.57E-04	2.08E-04
	Sum MP	19.00	1.63E-02	7.92	6.77E-03	8.08E-03
	Total Mass (ug)	0.0163	0.0163			