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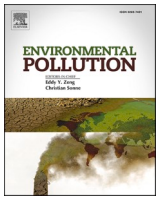

Microplastics in the air: Weather and polymer influences on deposition trends across a rural–urban gradient☆

Gbotemi A. Adediran [a,*], Victoria Taylor [b], Alexandra Howard [c], Paul G. Whitehead [b], Jocelyne M.R. Hughes [b]

[a] School of Earth and Environment, University of Leeds, Leeds, S2 9JT, UK
[b] School of Geography and the Environment, University of Oxford, Oxford, OX1 3QY, UK
[c] UK Centre for Ecology & Hydrology, Maclean Building, Wallingford, OX10 8BB, UK

A B S T R A C T

Atmospheric microplastics are an emerging concern, yet their deposition dynamics across different landscapes and weather conditions remain poorly understood. We investigated microplastic deposition along a rural-to-urban gradient in England, sampling Wytham Woods (rural), Summertown (suburban), and Oxford City (urban) every 2–3 days from May to July 2023. Using high-resolution μFTIR spectroscopy, we quantified 21 polymer types across four size fractions (25–50 μm, 50–75 μm, 75–100 μm, and >100 μm) and analysed their deposition patterns in relation to weather variables. Deposition rates varied from 12 to 500 particles/m^2/day, with Wytham Woods recording the highest overall deposition and Oxford City exhibiting the greatest polymer diversity. The 25–50 μm size fraction dominated in all sites, comprising up to 99 % of total deposition during high-concentration events. Polymer prevalence varied by site, with polyethylene terephthalate most abundant in Wytham Woods, polyethylene in Summertown, and ethylene vinyl alcohol in Oxford City. Weather conditions influenced deposition trends. Higher atmospheric pressure suppressed deposition, while increased wind speed and winds from the northeast enhanced it. Rainfall reduced overall deposition but increased the proportion of larger microplastics (50–75 μm). These findings challenge the assumption that urban areas consistently experience the highest microplastic loads, emphasising the impact of weather patterns on microplastic dispersion and deposition. This study highlights the need for further research into long-term deposition patterns of microplastics, focusing on specific polymer types and sizes, and their relationship with short-term and seasonal weather variations across diverse landscapes.

1. Introduction

Plastics have become indispensable in modern life, serving crucial roles in diverse applications ranging from packaging and construction to electronics and healthcare (Andrady and Neal, 2009). Their durability, versatility, and cost-effectiveness have driven a dramatic increase in global plastic production, with more than 370 million tons being manufactured annually (Walker and Fequet, 2023). However, this widespread use has come with environmental penalties. The generation of plastic waste has escalated, with millions of tons entering landfills and natural environments each year (Pilapitiya and Ratnayake, 2024). A challenging aspect of plastic waste is its degradation into microplastics - tiny plastic particles that range in size from 1 μm to 5 mm (Frias and Nash, 2019).

Microplastics in the environment primarily arise from the breakdown of larger plastic items, a process driven by environmental factors such as ultraviolet radiation, mechanical abrasion, and chemical degradation (Sutkar et al., 2023). However, microplastics can also be introduced into the environment at any stage of their production, use, and disposal (Bank and Hansson, 2019). Widely used plastics, including polyethylene, polypropylene, polyvinyl chloride, and polyurethane are prevalent in packaging, construction materials and textiles (Lam et al., 2018). These materials are major contributors to the environmental load of microplastics, which are diverse in size, shape, and chemical makeup. Microplastics are now ubiquitous, infiltrating both terrestrial and aquatic ecosystems and even becoming airborne (Wright et al., 2020).

Airborne particle transport is influenced by fluid dynamics, turbulence, and gravitational settling, with particles dispersing through advection and diffusion caused by turbulence and Brownian motion (Guha, 2008). Their travel distance depends on size, density, shape, and atmospheric conditions (Tsuda et al., 2013). Fine particulate matter ($PM_{2.5}$, ≤2.5 μm) can remain suspended for days to weeks, travelling thousands of kilometres due to its low settling velocity (Fang et al., 2018). In contrast, coarser particles (PM_{10}, ≤10 μm) settle more quickly,

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* Corresponding author.
E-mail address: G.A.Adediran@leeds.ac.uk (G.A. Adediran).

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typically travelling tens to hundreds of kilometres before deposition (Hudda et al., 2014). Studies have shown that PM_{10} can be transported from desert regions to distant continents, while $PM_{2.5}$ from urban and industrial sources persists in the atmosphere, contributing to regional and global pollution (Van Der Does et al., 2018; Zou et al., 2017).

These transport mechanisms also apply to airborne microplastics, with smaller fragments remaining suspended for extended periods and dispersing over vast distances. Due to their low density and small size, microplastics can be carried by wind and atmospheric currents, facilitating their global distribution (Shao et al., 2022). This widespread dispersal has led to microplastic deposition in even the most remote and pristine environments, highlighting their far-reaching environmental impact (Allen et al., 2019).

Urban environments, characterised by high population density, extensive road networks, and industrial activities, are believed to have higher concentrations of airborne pollutants compared to rural areas (Konarski et al., 2006; Van Der Zee et al., 1998). However, the gradient of microplastic deposition from rural to urban areas remains poorly understood. Moreover, while weather conditions such as wind speed and precipitation are known to affect the deposition of airborne pollutants, their impact specifically on microplastic deposition has yet to be fully elucidated (O'Brien et al., 2023).

The study of atmospheric microplastics is crucial to addressing the broader challenge of planetary plastic pollution (Villarrubia-Gómez et al., 2024), which the United Nations seeks to address through its forthcoming treaty on plastic pollution. Despite their significant environmental impact, secondary atmospheric microplastics, arising from the fragmentation of larger plastic materials, are often underrepresented in policy frameworks, including the draft UN treaty (Diana et al., 2024). This gap underscores the urgent need for targeted research and systematic monitoring to characterise microplastic transport dynamics, and atmospheric deposition patterns.

Current studies on atmospheric microplastic deposition have revealed considerable variability in deposition rates, spanning several orders of magnitude. For instance, measurements in Paris reported deposition rates of 29–280 particles/m^2/day, in Hamburg, Germany, 136–512 particles/m^2/day, and in central London, UK, 575–1008 particles/m^2/day (Dris et al., 2015; Klein and Fischer, 2019; Wright et al., 2020). While these variations could be attributed to differences in location and proximity to pollution sources, inconsistencies in sampling methodologies and analytical approaches may also play a significant role.

The most precise method for quantifying microplastics involves combining particle detection with chemical composition analysis using micro-spectroscopic techniques such as Fourier-transform infrared (μFTIR) and Raman spectroscopy (Schymanski et al., 2021). However, the accuracy of these techniques is influenced by particle size. In a global interlaboratory comparison, those using μFTIR or Raman spectroscopy achieved a 92 % recovery rate and correct identification for pure microplastic standards larger than 50 μm (De Frond et al., 2022). However, accuracy dropped to 33 % for particles smaller than 20 μm. Additionally, up to 22 % of μFTIR analyses mistakenly identified natural materials as plastic, while this rate increased to 83 % for Raman spectroscopy (De Frond et al., 2022).

Given these challenges, this study aims to use μFTIR at a spectral resolution of 25 μm, the most accurate available, to investigate the abundance of atmospheric microplastic deposition across a rural-to-urban gradient for a locality in England. The research specifically focuses on analysing the behaviour of 21 polymer types across four major size fractions (25–50 μm, 50–75 μm, 75–100 μm, and >100 μm), under varying weather conditions, with rigorous use of appropriate field and laboratory blanks to ensure accuracy.

We hypothesise that.

1. Atmospheric microplastic deposition will be lowest in the rural environment of Wytham Woods and highest in the urban centre of Oxford City.
2. Microplastics in the 25–50 μm size fraction will be the most abundant across all sites when compared to the other (50–75 μm, 75–100 μm, and >100 μm) size fractions.
3. Polypropylene and polyethylene, the world's most commonly used plastics, will be the dominant polymer types found across the gradient.
4. Variations in weather conditions will drive the abundance and behaviour of atmospheric microplastic deposition.

2. Materials and methods

2.1. Sampling locations

Three sampling locations - Wytham Woods, Summertown, and Oxford Central - were selected to represent rural, suburban, and urban environments in Southeastern England (Fig. S1). Wytham Woods is a 1000-acre area of semi-natural woodland, renowned as one of the most studied woodlands globally. This site hosts a diverse array of habitats, including oak forests, mixed broadleaf woodlands, and grasslands, rendering it a significant area for the conservation of nature (Kirby et al., 2014). Summertown, a residential suburb approximately two miles north of Oxford city centre, is characterised by tree-lined streets and a mix of Victorian and modern homes. With a population of ~7000 residents, Summertown serves as a transition zone between the urban core of Oxford and the more rural areas to the north, making it a valuable site for studies on suburban air pollution gradients. Oxford City, known for its prestigious university, is densely populated with ~165,200 residents. The city features a mix of historical and modern buildings and busy roads, making it an ideal location for urban air quality studies. These sampling sites were strategically selected to be within 5 km of three meteorological stations: Wytham Woods Station, Radcliffe Meteorological Station, and the Earth Observation Data Group Station at the University of Oxford (Fig. S1).

2.2. Controlling microplastic contamination

To mitigate microplastic contamination during sampling and analysis, the sampling unit and most laboratory equipment were constructed from non-plastic materials. The only exception was wash bottles, made from uncommon, non-shedding fluorinated-ethylene-propylene. Equipment and glassware were meticulously cleaned using natural fibre brushes to prevent the introduction of microplastics during the washing process. Following cleaning, all items were rinsed multiple times with 0.7 μm filtered reverse osmosis (RO) water and air-dried within a safety cabinet equipped with a high-efficiency particulate air filter, which removes 99.999 % of particles larger than 0.3 μm, thereby preventing airborne contamination. To minimise contamination from synthetic fibres, personnel wearing 100 % cotton lab coats conducted all sampling and processing activities.

2.3. Sampling procedure

Sampling units consisted of an aluminium funnel (diameter 18 cm) inserted into a 2 L glass jar, securely housed within a wooden box (Fig. S2). This setup, with an effective sampling area of 0.02544 m^2, was designed to collect both dry and wet depositions, including microplastics suspended in rainwater. To minimise contamination from non-atmospheric sources, the sampling units at Wytham Woods and Summertown were mounted on metal stands, ensuring sample collection occurred 1.6 m above ground level, and the sampling unit for Oxford City was installed on the rooftop of the School of Geography and the Environment at the University of Oxford (Figs. S3–S4). For all sites, samples were collected every 2–3 days per site for five weeks (31 May –

July 3, 2023), providing higher temporal resolution than weekly sampling for this short-term study.

At the end of sampling, ~400 mL of 0.7 μm GF/F filtered RO water was used to rinse the aluminium funnel into the sampling jar. The funnel was then removed, and the sampling jar was replaced with a clean jar. The used sampling jar with the sample was immediately covered with aluminium foil, secured with a metal lid, and transported to the laboratory for analysis. This technique limited the number of sampling jars to two per site, reducing contamination risk and minimising microplastic loss during sample transfers. This sampling procedure followed established methodologies (Fan et al., 2022; Wright et al., 2020) to ensure consistency for cross-study comparisons.

2.4. Preparation of full experimental blanks

Due to the pervasive nature of microplastics, generating experimental blanks that accurately reflect both field sampling and laboratory handling was essential. Field blank sampling units were set up similarly to the main experimental units, as described in Fig. S3a, but with the aluminium funnel covered to prevent atmospheric deposition of microplastics. These blanks effectively accounted for potential microplastic contamination at every stage of the experiment, including field sampling, transportation, laboratory processing, and analysis. A total of 14 field blanks were generated for this study.

2.5. Blank correction and limit of detection

For each polymer and size range, the mean blank value was subtracted from the raw count for a sample. The limit of detection (LOD) for the blank-corrected sample was defined as 3.3 × the standard deviation of the blanks as recommended by AOAC International. If the blank corrected value was above the LOD, it was acknowledged as being detected.

2.6. Microplastic analysis by μFTIR

The samples collected from the three sites were relatively clean. Preliminary tests indicated that traditional methods, such as Fenton reaction digestion and zinc chloride density separation, did not enhance microplastic detection. A spike recovery test using polystyrene (30–90 μm, 0.96–1.05 g/cm^3) and polyamide (30–50 μm, 1.13 g/cm^3) showed comparable recovery rates between untreated samples and 0.7 μm GF/F filtered RO water. To avoid contamination from reagents and potential microplastic fragmentation during treatment, the samples were analysed without undergoing chemical digestion.

For μFTIR analysis, the entire sample was transferred from the sampling jar using a glass pipette onto a silver membrane filter (Sterlitech, Washington, USA; 25 mm diameter, 3 μm pore size) placed under a 10 mm internal diameter silicone washer to define the filtration area. The glass jar was rinsed with 0.7 μm filtered RO water to ensure complete transfer of all particles onto the filter. Microplastics within the filtration area were then identified and quantified at a pixel size of 25 μm using a PerkinElmer Spotlight 400 linear array imaging μFTIR spectrometer, which collected spectra across a wavenumber range of 4000–700 cm^{-1}. A background spectrum of the silver filter was subtracted from the data. Automated spectral matching was performed using the Purency Microplastics Finder software (Hufnagl et al., 2019), which employs machine learning algorithms to eliminate operator bias, providing polymer-specific particle counts and detailed information on the two-dimensional characteristics of each particle.

2.7. Polymers quantified

This study quantified 21 polymers: acrylonitrile butadiene styrene (ABS), cellulose acetate (CA), ethylene vinyl acetate (EVAc), ethylene vinyl alcohol (EVOH), polyacrylonitrile (PAN), polyamide (PA), polybutylene terephthalate (PBT), polycarbonate (PC), polyethylene (PE), polyether ether ketone (PEEK), polyoxymethylene (POM), polyphenylene sulfone (PPSU), polysulfone (PSU), polyethylene terephthalate (PET), polymethyl methacrylate (PMMA), polylactic acid (PLA), polypropylene (PP), polystyrene (PS), polyurethane (PU), polyvinyl chloride (PVC), and silicone (SI). These were selected based on their widespread use and documented occurrence in atmospheric samples (Abbasi et al., 2024; Klein and Fischer, 2019).

Equation (1) was used to calculate the daily microplastic deposition rate *(R),* following the method described by Fan et al. (2022).

$$R = \frac{N}{A \times t} \quad \text{Equation 1}$$

Where R: Deposition rate (particles/m^2/day), N: Total microplastic particles collected, A: Sampling area (m^2), and t: Sampling duration (day).

2.8. Meteorological data

Meteorological data were sourced from two long-term monitoring stations. At Wytham Woods, data were collected from a weather station operated as part of the UK Environmental Change Network. The station is equipped with automated sensors that measure key atmospheric variables at 10-min intervals. Instrumentation is regularly calibrated and maintained, and data quality is ensured through standardised validation protocols (Rennie et al., 2020). Complementary meteorological data were sourced from the rooftop weather station at the University of Oxford's Atmospheric Physics Building, operated by the Earth Observation Data Group (EODG, 2024; Evans, 2014). This Instromet Met4Net system is rooftop-mounted and measures key atmospheric variables every 0.5 s, averaged over 10 s, and logged at 2-min intervals. Quality control is maintained via factory calibration and cross-validation against the nearby Radcliffe Meteorological station (Evans, 2014). Although atmospheric pressure data were not available for the Wytham Woods site, measurements of temperature, precipitation, relative humidity, wind speed, and wind direction were consistently recorded across all three sites. Daily averages were calculated to produce a consistent daily meteorological dataset.

3. Results

3.1. Spatial and temporal deposition patterns

Atmospheric microplastic deposition exhibited considerable spatial and temporal variability (Fig. 1). Deposition rates ranged from 13.0 to 500.2 particles/m^2/day in Wytham Wood, 13.1 to 181.8 particles/m^2/day in Summertown, and 12.1 to 272.2 particles/m^2/day in Oxford City (Fig. 1a).

Pronounced temporal variations in microplastic deposition were observed across different particle sizes. In high deposition scenarios, where rates exceed 75 particles/m^2/day, the 25–50 μm size fraction (the smallest studied) constituted 99 % of total deposition (Fig. 1b). This 25–50 μm size fraction consistently dominated across all sites, with average deposition rates of 117.9 particles/m^2/day in Wytham Wood, 71.0 particles/m^2/day in Summertown, and 91.1 particles/m^2/day in Central Oxford (Fig. 2c). While Wytham Wood exhibited the highest deposition of 25–50 μm particles, Summertown recorded the highest deposition rates for the 50–75 μm and 75–100 μm fractions. Central Oxford, conversely, showed the highest rates for particles larger than 100 μm.

3.2. Polymer composition and diversity

Of the 21 polymers analysed, PC, PEEK, POM, PPSU, and PSU were undetected. Analysis of the deposition dynamics for the 16 detectable

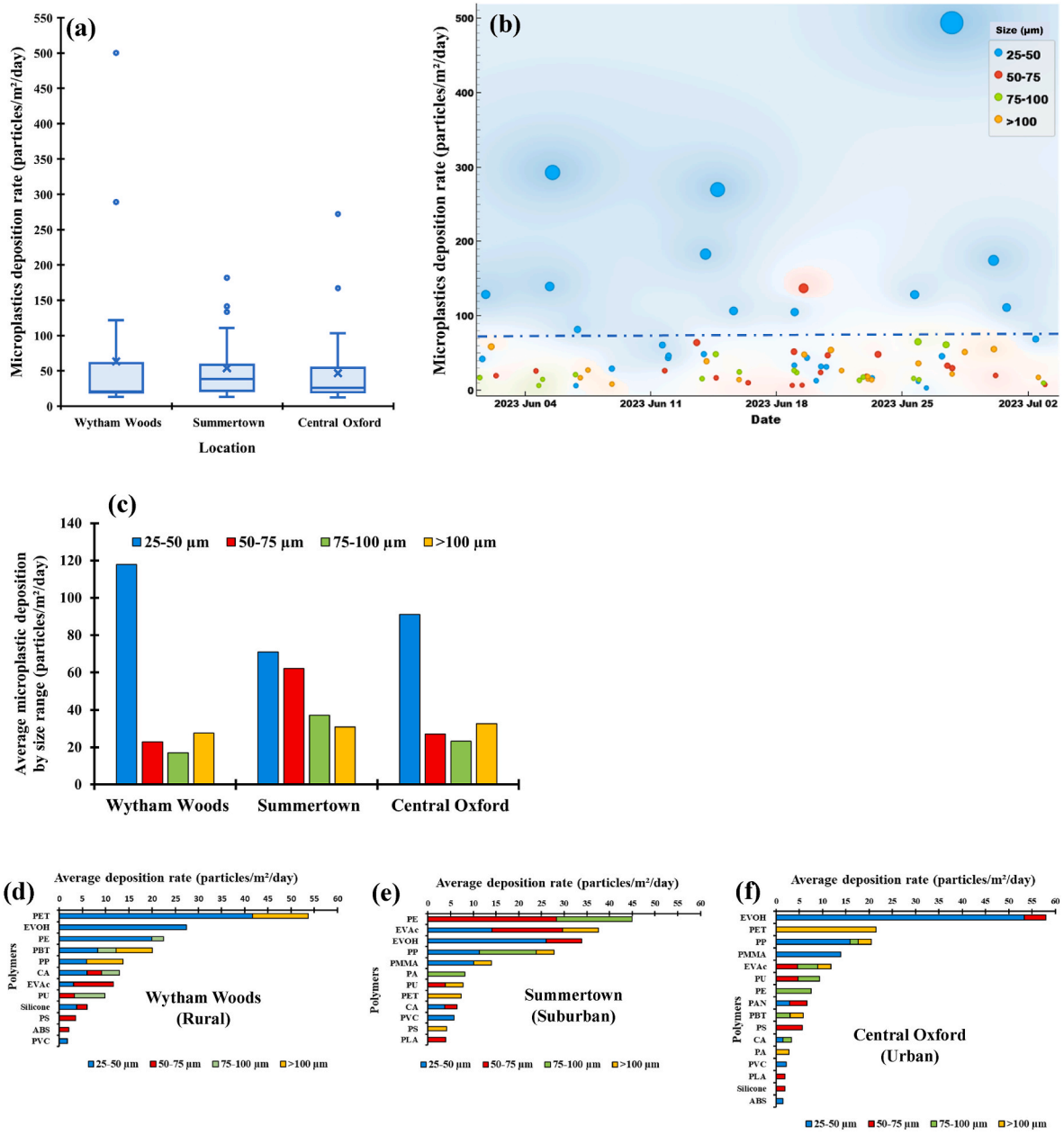

Fig. 1. Spatial and temporal distribution of atmospheric microplastic deposition per square meter of land surface per day. Panel (a) presents the atmospheric microplastic deposition rates across different sites, while (b) tracks the temporal dynamics of microplastic deposition with respect to particle size. Panel (c) shows the average microplastic deposition categorised by site and size, and panels (d), (e), and (f) detail the average deposition rates for specific polymers. The bigger the size of the symbols in (b) the more microplastics there are.

polymers revealed spatial heterogeneity in polymer-specific deposition rates.

The dominant polymer varied across locations: PET in Wytham Wood, PE in Summertown, and EVOH in Oxford City, with average deposition rates of 53.6, 44.9, and 57.9 particles/m^2/day, respectively (Fig. 1d–f). In Wytham Wood, PET, EVOH, PE, PBT, and PP collectively constituted 74 % of the total deposition. In Summertown, PE, EVAc, EVOH, PP, and PMMA were dominant, accounting for 78 %. For Oxford City, EVOH, PET, PP, PMMA, and EVAc were the primary polymers, comprising 72 % of the microplastic deposition. Polymer diversity was highest in Oxford City, with 16 different types detected, compared to 12 types in both Wytham Wood and Summertown.

3.3. Influence of weather parameters on deposition

Atmospheric Pressure: During the study period, atmospheric pressure ranged from 1006.57 to 1021.85 hPa, revealing two distinct regimes in relation to microplastic deposition: a low-pressure regime (1006.57–1013.28 hPa) and a high-pressure regime (1015.45–1021.85 hPa) (Fig. S5a). As atmospheric pressure increased from 1010.38 hPa to 1017.76 hPa, the average microplastic deposition rate dropped from 62.2 to 36.15 particles/m^2/day. Throughout this transition, the 25–50 µm size fraction consistently dominated, representing 48.5 % of particles at lower pressure and 50.9 % at higher pressure (Fig. S5b). The increase in atmospheric pressure also shifted the dominant polymer type from EVOH to EVAc and reduced polymer diversity from 15 to 13 types. Under low-pressure conditions, EVOH, PP, PET, and PE made up 73 % of deposited polymers, whereas under high-pressure conditions, the same

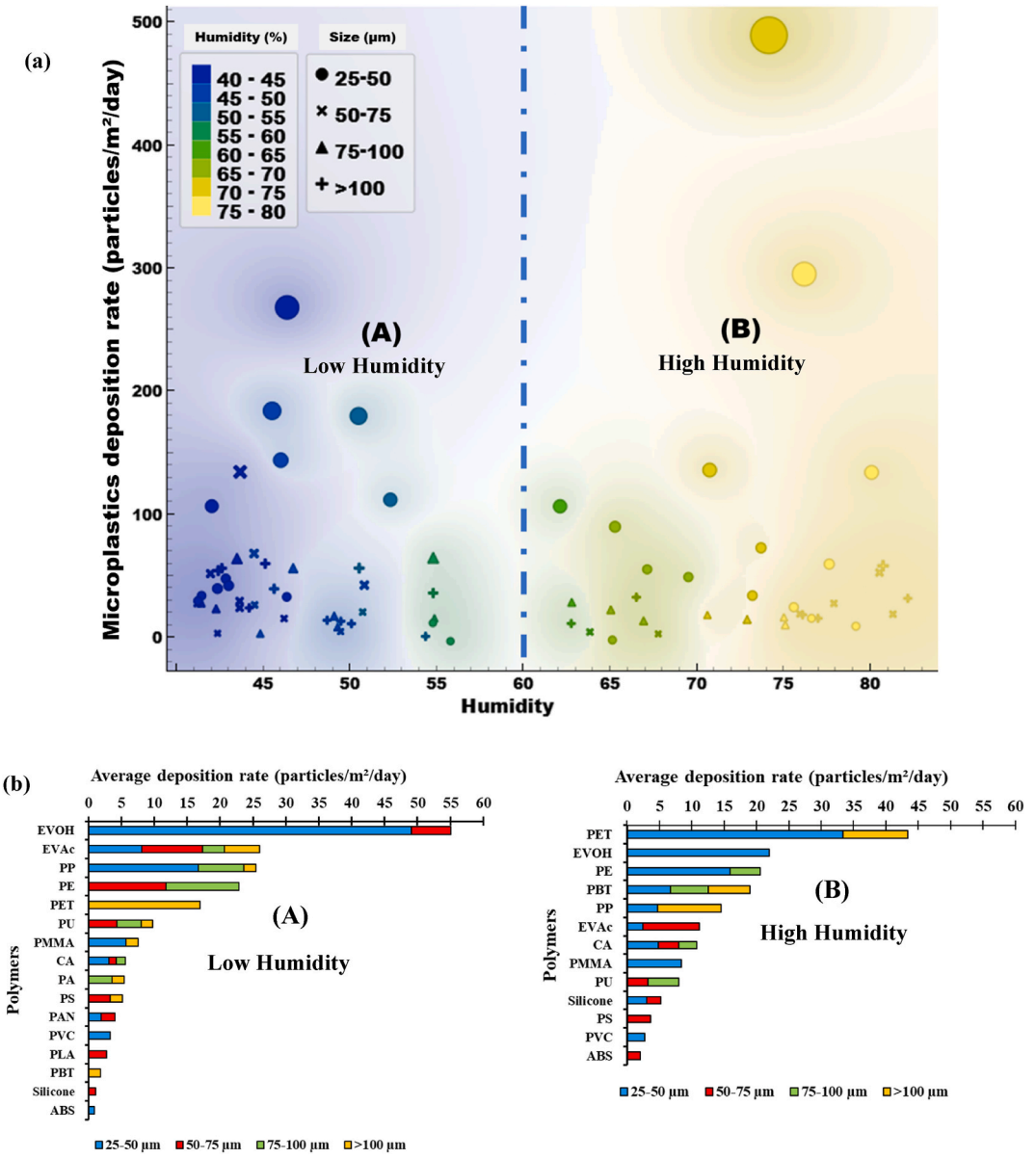

Fig. 2. (A) Atmospheric microplastic deposition rates across a relative humidity range of 41.49 %–80.86 %. (b) Average deposition rates by polymer size and type under low (A: 41.49 %–55.07 %) and high (B: 62.98 %–80.86 %) humidity conditions. The bigger the size of the symbols in (a) the more microplastics there are (particles/m^2/day).

percentage comprised EVAc, PMMA, PE, PU, and EVOH.

Relative Humidity: Relative humidity, ranging from 41.49 % to 80.86 %, also influenced deposition patterns, exhibiting two distinct regimes: low (41.49 %–55.07 %) and high (62.98 %–80.86 %) (Fig. 2a). As average humidity increased from 46.66 % to 72.73 %, the average rate of microplastic deposition rose from 51.70 to 58.27 particles/m^2/day. Throughout this shift, 25–50 μm microplastics remained the dominant size fraction, constituting 45.87 % of particles at lower humidity and 60.73 % at higher humidity (Fig. 2b). The increase in humidity also altered the polymer profile, shifting the dominant type from EVOH to PET and reducing polymer diversity from 16 to 13 types. Under low humidity conditions, EVOH, EVAc, PP, PE, and PET accounted for 70 % of deposited polymers, while under high humidity, PET, EVOH, PE, PBT, PP, and EVAc made up the same percentage.

Precipitation: Precipitation conditions varied from dry periods to heavy rainfall (up to 18.59 mm) (Fig. S6a). Under low rainfall (0–6.23 mm) conditions, deposition rates were higher but decreased from 55.86 to 44.92 particles/m^2/day as rainfall increased (14.8–18.59 mm). With increasing rainfall, the dominant microplastic size shifted from 25 to 50 μm (55 % of deposits under low rainfall) to 50–25 μm (45 % under high rainfall) (Fig. S6b). The rise in rainfall also altered the polymer profile, replacing EVOH with PET as the dominant type and reducing polymer diversity from 16 to 9 types. Under low rainfall conditions, EVOH, PET, EVAc, PP, and PE constituted 70 % of deposited polymers, while under high rainfall, PE, EVOH, PET, and silicone accounted for the same percentage.

Temperature: Temperatures during the study ranged from 9.8 to 20.1 °C, with microplastic deposition showing distinct patterns at low (9.8–14.1 °C) and high (15.3–20.1 °C) temperature regimes (Fig. 3a). As the average temperature increased from 11.6 °C to 17.7 °C, the deposition rate of microplastics rose from 50.4 to 55.8 particles/m^2/day. Throughout this temperature shift, 25–50 μm microplastics remained the dominant size fraction, accounting for 61.5 % of particles at lower temperatures and 48.8 % at higher temperatures (Fig. 3b). The temperature rise also changed the polymer signature, with the dominant types shifting from PE to EVOH, and polymer diversity expanding from

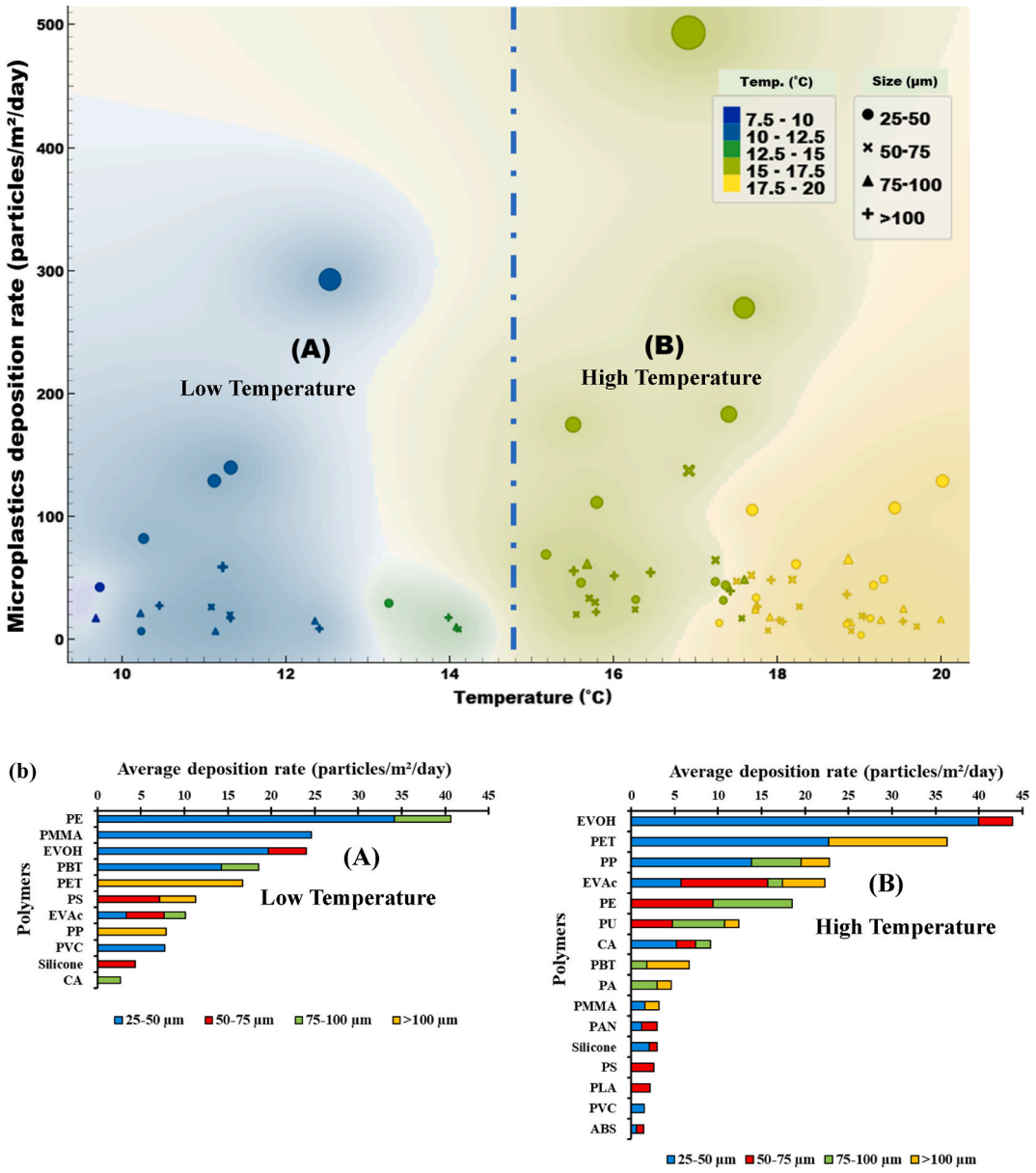

Fig. 3. (A) Atmospheric microplastic deposition rates across a temperature range of 9.8–20.1 °C. (b) Average deposition rates by polymer size and type under low (A: (9.8–14.1 °C)) and high (B: 15.3–20.1 °C) temperature conditions. The bigger the size of the symbols in (a) the more microplastics there are (particles/m^2/day).

11 to 16 types. At lower temperatures, PE, PMMA, EVOH, PBT, and PET made up 70 % of the deposited polymers, while at higher temperatures, EVOH, PET, PP, EVAc, and PE constituted the same proportion.

***Wind Speed*:** Wind speeds during the study varied from 0.18 to 2.46 m/s, with microplastics showing distinct deposition patterns at low (0.18–1.41 m/s) and high (1.5–2.46 m/s) wind speeds (Fig. 4a). As wind speeds increased from an average of 0.78 m/s to 1.90 m/s, the microplastic deposition rate more than doubled, rising from 46.32 to 87.18 particles/m^2/day. Throughout this transition, 25–50 µm microplastics remained the dominant size fraction, constituting 43.30 % of particles at lower wind speeds and 71.63 % at higher wind speeds (Fig. 4b). The shift in wind speed also altered the polymer composition, with PET overtaking EVOH as the dominant polymer, and polymer diversity decreasing from 16 to 12 types. Under low wind conditions, EVOH, EVAc, PP, PE, PET, and PMMA made up 70 % of the deposited polymers, while at higher wind speeds, PET, EVOH, and PE dominated, accounting for the same proportion.

***Wind Direction*:** Wind directions during the study varied between 29.75° and 224.47°, resulting in distinct microplastic deposition patterns at the Northeast (NE; 29.75°–72.3°), Southeast (SE; 116.88°–148.14°), and Southwest (SW; 205.13°–259.47°) directions (Fig. 5a). Microplastic deposition was highest under NE winds, averaging 61.8 particles/m^2/day, decreasing to 39.9 particles/m^2/day with SE winds, and then rising to 53.4 particles/m^2/day as winds shifted to the SW. Throughout these changes, 25–50 µm microplastics consistently dominated, comprising 60.31 % of particles in NE, 39.53 % in SE, and 47.1 % in SW directions (Fig. 5b). The shift in wind direction also influenced the polymer signature, with EVOH as the dominant polymer in the NE, while PET took precedence under SE and SW winds. Polymer diversity remained relatively constant, with 13 types detected under NE winds and 12 types under both SE and SW conditions. NE winds favoured the deposition of EVOH, PP, PE, PMMA, and PBT, accounting for 74 % of the total, while SE winds predominantly carried PET, EVOH, PP, PAN, EVAc, and PU. In the SW direction, PET, EVOH, PE, and EVAc made up a similar proportion of the deposited polymers.

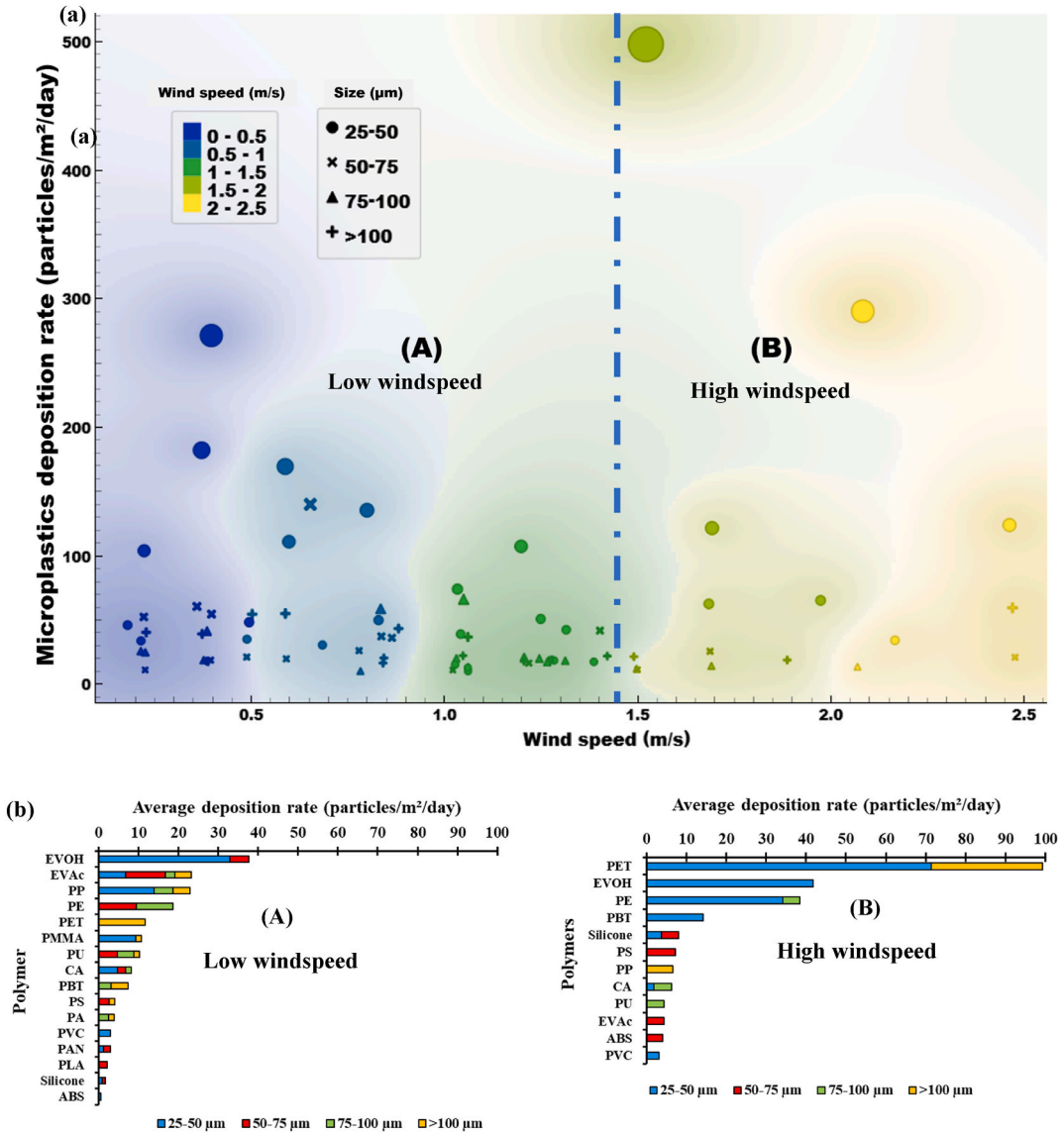

Fig. 4. (A) Atmospheric microplastic deposition rates across a windspeed range of 0.18–2.46 m/s. (b) Average deposition rates by polymer size and type under low (A: 0.18–1.41 m/s) and high (B: 1.5–2.46 m/s) windspeed conditions. The bigger the size of the symbols in (a) the more microplastics there are (particles/m^2/day).

3.4. Overall deposition trends and key influences

Overall, an increase in wind speed (from 0.78 m/s to 1.90 m/s), a shift in prevailing wind direction (from southeast to southwest), and elevated levels of relative humidity (from 46.66 % to 72.73 %) and ambient temperature (from 11.6 °C to 17.7 °C) resulted in atmospheric microplastic deposition increasing by 88.2 %, 33.8 %, 12.7 %, and 10.7 %, respectively.

4. Discussion

The atmospheric microplastic deposition rate of 12–500 particles/m^2/day observed in our study falls within the range reported in Paris (29–280 particles/m^2/day) and Hamburg (136–512 particles/m^2/day) (Dris et al., 2015; Klein and Fischer, 2019). However, it is significantly lower than the 575–1008 particles/m^2/day recorded in central London in 2020, despite London being geographically closer to Oxfordshire (Wright et al., 2020). The higher deposition rates in London could be attributed to the city's denser population, likely resulting in increased microplastic emissions from human activities. However, differences in analytical methodologies may also explain these variations. Unlike Wright's study, which employed Nile Red staining and visualisation for microplastic identification and performed FTIR analysis on selected particles only (Wright et al., 2020), our study subjected all particles in a sample to FTIR analysis. This approach minimises potential errors that could arise from estimating total microplastic concentrations based on a subset of particles.

Furthermore, the use of appropriate experimental blanks in our study allowed us to establish a robust limit of detection (LOD) (Bertil and Örnemark, 2014). We employed a LOD calculation that theoretically ensures a significance level of 0.05 for both false-negative and false-positive error rates, meaning that if a sample's microplastic concentration is at the LOD level, the probability of a false positive result, indicating the presence of microplastics when there are none, is only 5 % (Armbruster and Pry, 2008). In this study, the recovery rate for PS and PA standards, with sizes ranging from 30 to 90 μm, was between 56 % and 90 %. Similar recovery rates have been observed in other studies that estimated spike recovery (Adediran et al., 2024). However, the absence of appropriate standards for different polymer types across various size ranges limits the effectiveness of spike recovery, rendering positive controls merely indicative. Despite these constraints, our research adhered strictly to the quality standards for microplastic

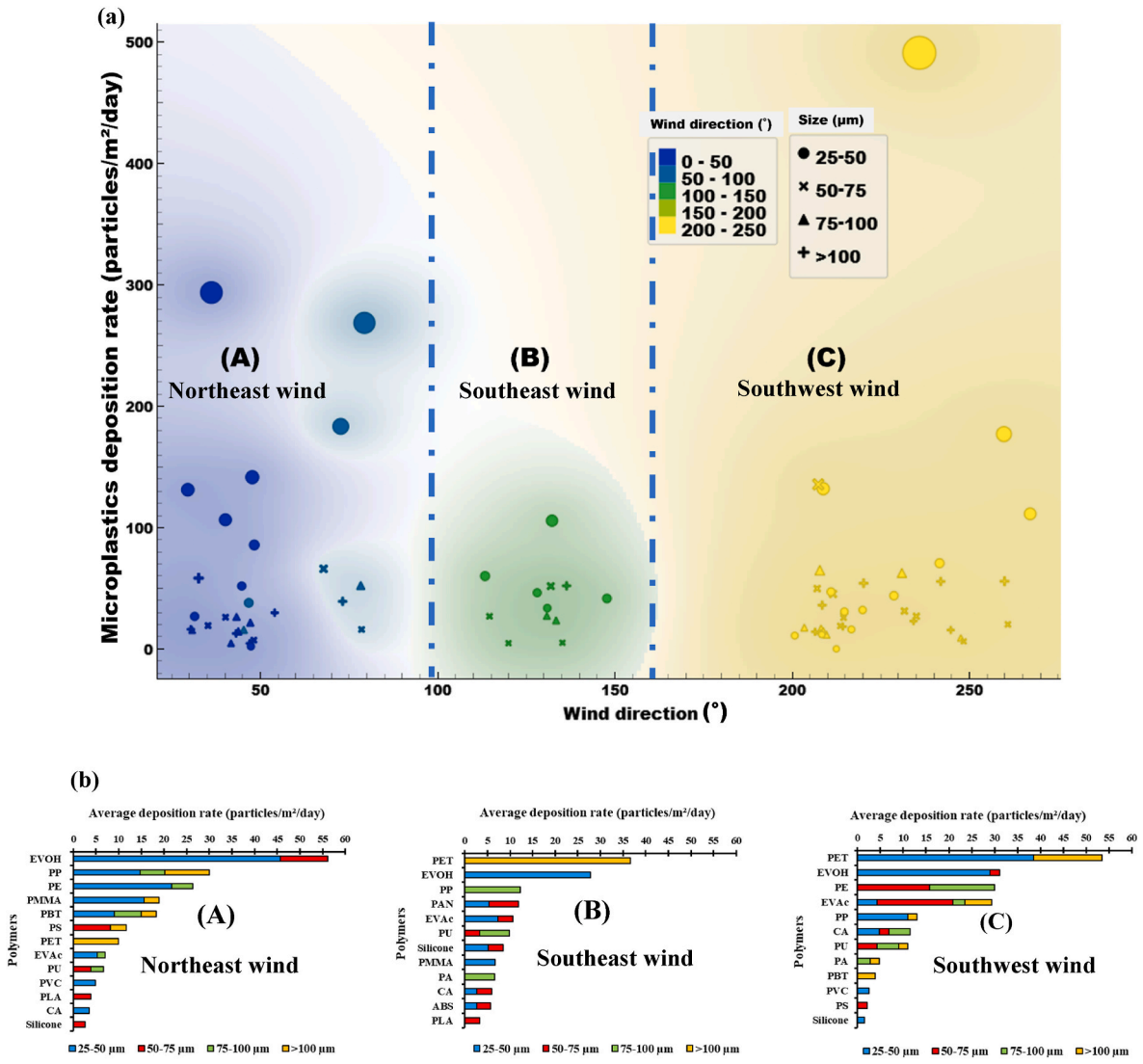

Fig. 5. (A) Atmospheric microplastic deposition rates in relation to wind direction, ranging from 29.75° to 224.47° (Northeast to Southwest). (b) Average deposition rates by polymer size and type under Northeast (A: 29.75°–72.3°), Southeast (B: 116.88°–148.14°) and Southwest (B: 205.13°–259.47°) wind direction. The bigger the size of the symbols in (a) the more microplastics there are (particles/m^2/day).

analysis in environmental samples (Koelmans et al., 2019), ensuring meticulous blank corrections and the use of positive controls.

Contrary to our initial hypothesis, the highest microplastic deposition was observed in the rural area of Wytham Wood, rather than in the urban core of Oxford City. This finding indicates that the distribution of atmospheric microplastics is not exclusively driven by proximity to urban centres but may also be significantly influenced by factors such as local meteorological conditions and landscape features. A similar pattern was reported by Klein and Fischer (2019) in Hamburg, where higher microplastic deposition rates were observed in a rural forest area, a phenomenon they attributed to the "comb-out effect" of trees removing microplastics from the atmosphere. The elevated deposition rates in Wytham Wood may also be attributed to this comb-out effect, given its proximity to dense forest vegetation. Additionally, wind patterns are likely contributing to the transportation of microplastics from surrounding regions into this rural setting, leading to their subsequent deposition.

Despite the urban site in Oxford Central not exhibiting the highest overall microplastic deposition rates, it showed the greatest diversity of polymer types. This observation suggests that urban environments, with their wide variety of plastic sources, contribute to a more heterogeneous mixture of microplastics in the atmosphere. In contrast, the suburban site in Summertown recorded intermediate deposition levels but had the highest rates for specific size fractions, particularly those in the 50–100 μm range. This pattern reflects the transitional nature of suburban areas, which are likely influenced by both urban and rural sources of microplastic pollution.

The varying dominance of certain polymers at different sites highlights the diverse sources of microplastic pollution. At the rural location, the prevalence of PET suggests significant contributions from agricultural and consumer products, as PET is commonly used in items like agricultural films, plastic bottles, food containers, and polyester fabrics (Tan et al., 2023; Tsironi et al., 2022). In the suburban area, PE was most abundant, likely reflecting its extensive use in everyday items such as plastic bags, packaging, and insulation materials for construction (Agarwal and Gupta, 2017; Alam et al., 2018). Meanwhile, EVOH was most common in the urban area, indicating significant contributions from industrial and automotive activities, as EVOH is frequently utilised in fuel system components and high-performance food packaging (Maes et al., 2018; Mokwena and Tang, 2012).

Our results affirmed the hypothesis that microplastics in the smallest size fraction (25–50 μm) would be the most abundant across all sites. This result aligns with prior research demonstrating smaller microplastic particles' prevalence due to their extended airborne persistence

and potential for long-distance transport (O'Brien et al., 2023; Zhang et al., 2020). This size fraction's dominance may also be attributed to the higher likelihood of smaller microplastics being generated from the fragmentation of larger particles (Andrady and Koongolla, 2022).

Furthermore, weather conditions were found to influence the deposition pattern of atmospheric microplastics, aligning with our expectations. High atmospheric pressure was associated with reduced deposition rates, potentially due to the stabilisation of the atmosphere, which may limit vertical mixing and transport of microplastics. This observation aligns with previous research findings indicating that atmospheric stability under high-pressure conditions tends to suppress vertical air movements, thereby reducing the deposition of particulate matter, including microplastics (Dris et al., 2016; Wesely and Hicks, 2000). Conversely, lower pressure conditions, often linked with increased wind activity, were associated with higher deposition rates. The increased wind speeds under these conditions likely enhance the uplift and horizontal transport of microplastics, leading to greater deposition (Allen et al., 2019; Wesely and Hicks, 2000).

The association between increased humidity and higher microplastic deposition rates could be attributed to the hygroscopic nature of certain microplastics, which could enhance their aggregation and subsequent deposition (Bain and Preston, 2021; Szewc et al., 2021). However, high rainfall led to a notable decrease in deposition rates, as precipitation likely scavenges microplastics from the atmosphere, depositing them elsewhere. This phenomenon is consistent with findings that rainfall can effectively remove airborne particles, including microplastics, from the atmosphere (Jia et al., 2022; Szewc et al., 2021).

Wind speed emerged as a critical factor influencing microplastic deposition, with higher speeds increasing deposition rates, particularly for smaller particles. This underscores the role of wind in the horizontal transport of microplastics, enabling their distribution across long distances. Such findings are consistent with prior research that highlights the significant impact of wind on the distribution and deposition of airborne microplastics (Rezaei et al., 2019; Zhang et al., 2020). Additionally, wind direction played a crucial role in determining where microplastics were deposited, with certain wind patterns, particularly those from the northeast, being linked to higher deposition rates. This suggests that specific regional sources or atmospheric transport pathways are responsible for the observed patterns, a conclusion supported by previous studies on atmospheric circulation and pollutant transport (Ding et al., 2022; Evangeliou et al., 2020).

Our findings highlight the need to better understand the fate and transport of deposited atmospheric microplastics within the broader environment. A key uncertainty is the extent to which these particles are subsequently mobilised into aquatic systems, particularly within the Thames River catchment. In the Oxford Reach of the River Thames, Whitehead et al. (2021) quantified substantial microplastic loads, primarily linked to agricultural runoff and point source pollution. However, the potential contribution from atmospheric deposition has yet to be quantified. Future research will address this gap by incorporating our atmospheric deposition data into the INCA-Plastics model, providing a more comprehensive picture of microplastic fluxes and transport pathways within the Thames system.

5. Conclusion

This study offers significant insights into the intricate dynamics of atmospheric microplastic deposition across a rural-to-urban gradient. Contrary to expectations, the highest deposition rates were observed in the rural setting of Wytham Wood, rather than in the urban core of Oxford City. This suggests that local meteorological factors, particularly wind speed and direction, along with landscape characteristics like forest cover, play a pivotal role in influencing microplastic deposition. The consistent dominance of the 25–50 μm size fraction across all sites highlights the critical role of fine particles in atmospheric deposition processes. Furthermore, the variation in polymer types, with PET being most common in rural areas and PE and EVOH prevalent in suburban and urban environments, underscores the complex and diverse origins of microplastic pollution.

The implications of these findings are far-reaching for both environmental management and public health. The elevated levels of microplastic deposition in rural areas indicate that microplastics are not limited to urban environments, highlighting the need for broader monitoring efforts that encompass various landscape types. The ubiquitous presence of smaller microplastics raises concerns about potential health risks, especially in relation to respiratory exposure. Our results underscore the importance of further research to unravel the relationships between environmental factors and microplastic deposition. Future studies would focus on the long-term effects of polymer types, particle size distributions, and local meteorological conditions on microplastic dynamics in different ecosystems. Such research is vital to inform the development of effective strategies to mitigate the impact of microplastics on the environment and public health.

CRediT authorship contribution statement

Gbotemi A. Adediran: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Supervision, Writing – original draft, Writing – review & editing. **Victoria Taylor:** Conceptualization, Formal analysis, Investigation, Methodology. **Alexandra Howard:** Formal analysis, Investigation, Methodology. **Paul G. Whitehead:** Investigation, Project administration, Resources, Supervision, Writing – review & editing. **Jocelyne M.R. Hughes:** Conceptualization, Resources, Supervision, Writing – review & editing.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2025.127388.

Data availability

Data will be made available on request.

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