

Correlative SEM–Raman Microscopy of Inhalable Particles Sampled during Indoor Football on Artificial Turf with Crumb Rubber and Olive-Stone Infills

Maike Stange

Federal Institute for Occupational Safety and Health

Monica Andreassen

Norwegian Institute of Public Health

Torben Peters

Federal Institute for Occupational Safety and Health

Bryan Lauw

Federal Institute for Occupational Safety and Health

Carmen Thim

Federal Institute for Occupational Safety and Health

Nico Dziurawitz

Federal Institute for Occupational Safety and Health

Asmus Meyer-Plath

Federal Institute for Occupational Safety and Health

Stefanie Klar

Federal Institute for Occupational Safety and Health

Igor Snapkow

Norwegian Institute of Public Health

Torunn Kringlen Ervik

National Institute of Occupational Health (STAMI)

Hubert Dirven

Norwegian Institute of Public Health

Raymond H.H. Pieters

Utrecht University

Berit Granum

Norwegian Institute of Public Health

Dirk Broßell

`brossell.dirk@buaa.bund.de`

Research Article

Keywords: artificial turf, turf infill, micro- and nanoplastics, indoor sports, inhalation exposure, single-particle analysis, correlative microscopy

Posted Date: October 22nd, 2026

DOI: <https://doi.org/10.21203/rs.3.rs-11040166/v1>

License: © ⓘ This work is licensed under a Creative Commons Attribution 4.0 International License.
[Read Full License](#)

Additional Declarations: No competing interests reported.

Correlative SEM–Raman Microscopy of Inhalable Particles Sampled during Indoor Football on Artificial Turf with Crumb Rubber and Olive-Stone Infills

Maike Stange^{1*}, Monica Andreassen^{2*}, Torben Peters¹, Bryan Lauw¹, Carmen Thim¹, Nico Dziurrowitz¹, Asmus Meyer-Plath¹, Stefanie Klar¹, Igor Snapkow², Torunn Kringlen Ervik⁴, Hubert Dirven², Raymond H.H. Pieters³, Berit Granum^{2**}, Dirk Broßell^{1,5**}

*Shared first authorship; **Shared last authorship

¹ Federal Institute for Occupational Safety and Health (BAuA), Division Hazardous Substances and Biological Agents, Berlin, Germany

² Norwegian Institute of Public Health (NIPH), Department of Chemical Toxicology, Oslo, Norway

³ Institute for Risk Assessment Sciences, Utrecht University, Utrecht, Netherlands

⁴ National Institute of Occupational Health (STAMI), Oslo, Norway

⁵ Charité – Universitätsmedizin Berlin, corporate member of Freie Universität Berlin and Humboldt-Universität zu Berlin, Berlin, Germany

E-mail: broßell.dirk@baua.bund.de

Abstract

Artificial turf systems are potential emission sources for micro- and nanoplastic Particles (MNP), as mechanical abrasion and material degradation may release polymer-derived particles. When installed in indoor halls, emitted particles become highly relevant as athletes become exposed to accumulating dust concentrations. This study investigates airborne particle emissions from indoor artificial football pitches with different infill materials and quantifies the resulting inhalation exposure under real-life conditions during matches in two Norwegian sports halls. One hall used conventional crumb rubber infill derived from recycled tyres, the other a biodegradable infill made from crushed olive stone and quartz (olive sand).

Comprehensive dust particle analysis was performed by applying a correlative microscopy workflow that combined scanning electron microscopy, artificial-intelligence-based particle classification, and μ -Raman spectroscopy to chemically identify and quantify individual airborne particles. Despite contributions from indoor and outdoor environments, this approach enabled to differentiate infill-derived particles from background aerosol components. Both infill systems contributed to airborne particle emissions. Olive-stone-related particles were clearly identified indoors but were absent outdoors, whereas tyre-derived rubber particles occurred in both environments, indicating multiple sources including outdoor traffic and potential infill abrasion. Particulate matter (PM₁₀) concentrations were higher in the hall with olive stone infill (58.4 $\mu\text{g}/\text{m}^3$) than in the hall with crumb rubber infill (6.0 $\mu\text{g}/\text{m}^3$). In the former, fungal spores were detected, suggesting that organic infill can become a source of health-relevant bioaerosols.

These findings show that replacing crumb rubber with a biologically-derived infill does not eliminate inhalative particle exposure but changes its composition. They underline the need for exposure-based evaluation of materials subject to continuous wear, as sustainable alternatives should not be assumed inherently safer without comprehensive characterisation of their chemical and biological emissions.

40 **Keywords:** artificial turf; turf infill; micro- and nanoplastics; indoor sports; inhalation exposure; single-particle analysis;
41 correlative microscopy

42

43 **1. Introduction**

44 Artificial turf systems are widely used in indoor and outdoor sports facilities, offering greater durability and lower
45 maintenance requirements compared to natural grass. A key component of these systems is the infill material, which stabilizes
46 the turf and influences its playing characteristics. Crumb rubber, typically derived from recycled car tyres, is the predominant
47 infill material used [1–3]. Sports grounds with such turf systems have increasingly been recognized as significant sources of
48 particle emissions, arising from mechanical abrasion and material aging. In outdoor facilities, weathering, precipitation, and
49 surface runoff further govern the emission and dispersion of these particles [4, 5].

50 Particles derived from crumb rubber are chemically consistent with tyre wear particles (TWPs); however, TWPs are
51 conventionally defined by their origin from urban road traffic rather than from artificial turf systems [6, 7]. They typically
52 consist of elastomers - often manufacturer-specific blends of various polymers - along with carbon black as a filler material, as
53 well as heavy metals, polycyclic aromatic hydrocarbons, and other additives, which can be toxic either individually or in
54 combination. [8, 9]. TWPs themselves have become part of the ubiquitous particulate matter burden affecting humans in both
55 urban and rural environments [10], and are frequently discussed within the context of micro- and nanoplastic (MNP) pollution
56 [11–14].

57 The pulmonary system, lacking the thick protective barriers characteristic of the skin, offers a vast surface area that facilitates
58 rapid particle uptake into the bloodstream [15, 16]. Deposition in the deeper lung (alveolar region) is most probable for particles
59 with aerodynamic diameters smaller than approximately 2.5 μm ($\text{PM}_{2.5}$), while deposition in the tracheobronchial region
60 predominates for particles smaller than approximately 10 μm (PM_{10}) [17]. While several studies have quantified crumb rubber-
61 derived particle losses to soils, drainage systems, or seepage water from outdoor sports fields [18], direct measurements of
62 aerosol emissions remain scarce, and human exposure has largely been inferred rather than quantified.

63 Size distributions of atmospheric MNPs are often reported only down to approximately 10 μm , owing to the detection limits
64 of commonly employed size-sensitive, direct-reading aerosol instruments such as optical particle counters and light microscopy
65 [19], which may partly explain why inhalation exposure to MNPs has received limited attention to date. A recent meta-study,
66 however, reported an exponential increase in globally observed atmospheric MNP concentrations with decreasing particle size
67 down to 1 μm [20], suggesting that inhalation may represent a major exposure pathway for humans [21].

Indoor football halls represent a unique exposure scenario, in which mechanical abrasion, player movement, and limited air exchange may enhance particle release and suspension [3, 4, 22]. Notably, studies on indoor sports environments report increased inhalation doses due to elevated respiratory ventilation rates during exercise, but they lack specific characterization of infill-derived particles [23]. Quantitative data on particle release from artificial turf infill under indoor conditions—and the resulting inhalation exposure of football players - remain limited [24].

Building on evidence of environmental contamination, recent regulatory developments, notably the EU restriction on intentionally added microplastics under REACH, explicitly identify artificial turf infill as a major emission source and mandate its phase-out by 2031, underscoring the relevance of exposure-oriented research [25]. As alternatives, biodegradable organic materials - such as refined crushed olive stones, cork, walnut shells, wood fibre, or rice husks - are increasingly used, representing potentially more sustainable solutions [22, 26]. However, the transition from polymer-based to bio-based infills introduces new uncertainties, as these materials may differ not only in chemical composition but also in their propensity to generate inhalable particles and to support biological processes such as microbial growth—as already demonstrated for other bio-based materials used in construction [27–29].

A major challenge in this context is the lack of analytical approaches that combine size-resolved particle detection with reliable chemical identification at the single-particle level. As noted above, particles released locally from crumb rubber infill and TWPs transported into the sampling site are likely indistinguishable based on physical or chemical characteristics alone, hindering source-specific exposure assessment.

Similar challenges exist in workplace exposure assessment for mixtures of airborne fibres and particles, where scanning electron microscopy (SEM)-based strategies have been developed to identify and quantify fibres based on morphological properties in cases where their origin is known [30]. However, for complex mixtures of particles and fibres of unknown origin and composition, purely SEM-based approaches are insufficient. Instead, analytical techniques capable of resolving particle chemistry are required.

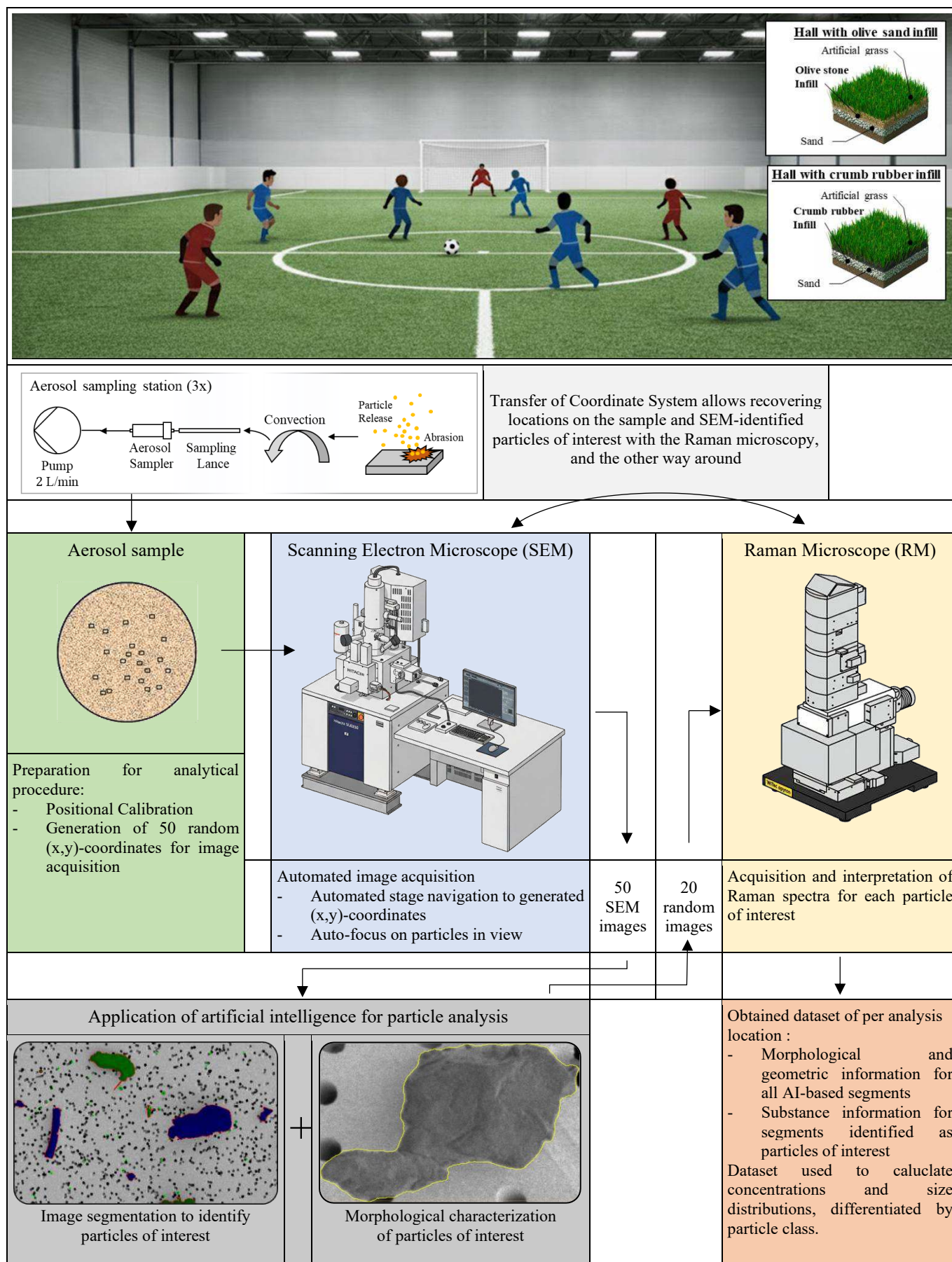
SEM secondary electron imaging enables fast, high-resolution imaging, and energy-dispersive X-ray spectroscopy (EDS) allows classification of many inorganic particles based on elemental composition. However, for polymer-based particles such as crumb rubber, TWPs, or other MNPs, identification remains challenging due to their less distinctive elemental signatures. In this context, Raman microscopy has gained importance in MNP research [31–33], as it enables identification of chemical groups and molecular structures even at the micro- and nanoscale, including for polymers and carbonaceous materials [34, 35].

The present study aims to fill this data gap by analysing aerosol samples collected during football matches in indoor sports halls to assess player exposure to particles emitted from turf infill during play, while differentiating these from background particles. Two sports halls in Norway were investigated: one with olive sand infill (refined crushed olive stones mixed with

98 sand) and one with crumb rubber infill. This work is part of a broader study in which blood and urine samples were collected
99 before and after matches to investigate potential biomarkers of exposure alongside chemical analyses of air samples (manuscript
100 in preparation).

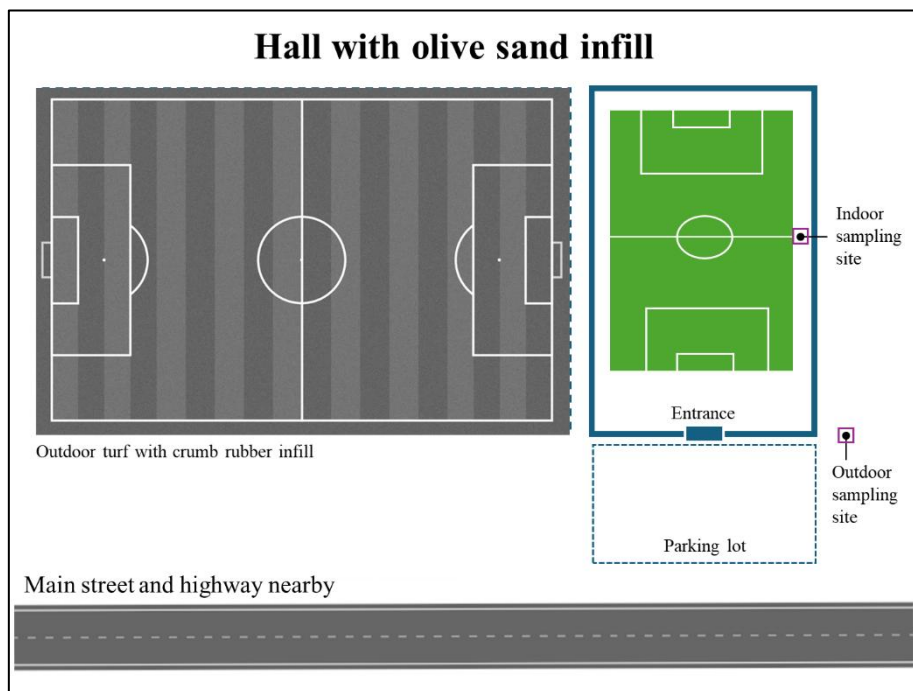
101 Specifically, this study aims to characterize inhalable airborne particles generated or resuspended during indoor football
102 matches and to compare particle profiles between the two infill systems. A correlative microscopy approach is applied to aerosol
103 samples, combining SEM imaging, AI-assisted particle detection, and μ -Raman microscopy, as previously introduced in a
104 workplace exposure study [36]. Briefly, SEM is used to acquire high-resolution images, which are analysed using an artificial
105 neural network to identify particles and characterize their morphology. Particle coordinates are then transferred to Raman
106 microscopy for chemical identification.

107 By combining morphological and chemical information, particles are assigned to predefined classes based on their chemical
108 composition (e.g., MNPs, soot). This approach enables quantification of particle counts by class, estimation of size
109 distributions, and extrapolation to number and mass concentrations for the PM_{10} and $PM_{2.5}$ aerosol fractions. The results are
110 evaluated to identify potential emission sources, with particular emphasis on particles originating from the respective infill
111 materials.



112 Figure 1: Workflow of the Correlative SEM-Raman Microscopy Procedure. Detailed description in the text of section 2.4.

(a)



(b)

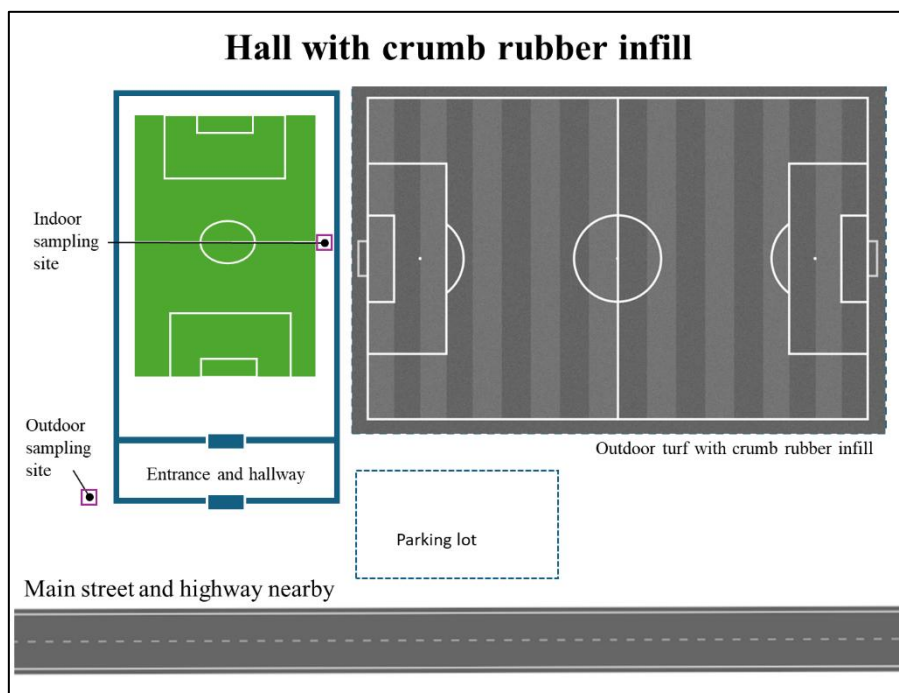


Figure 2: (a) Schematic overview of Hall with olive sand infill and its surroundings, including adjacent outdoor artificial turf field, entrance, parking lot, and two main streets/highways 500 m in distance to the hall with annual average daily traffic between 7200 and 8600 vehicles (source: <https://vegkart.atlas.vegvesen.no/>). (b) Schematic overview of Hall with crumb rubber infill and its surroundings, including entrance with hallway, adjacent outdoor artificial turf field, parking lot, and nearby main streets/highways, about 500 m and 1500 m in distance to the hall with annual average daily traffic 21,000 and 30,000 vehicles, respectively (source: [37]).

113
114
115
116
117
118
119

120
121
122
123
124
125

126
127
128
129
130
131
132
133

134
135
136
137
138
139

140
141

2. Methods

Figure 1 shows an overview of the procedural steps for the exposure assessment of football players to particulate matter, from aerosol sampling setup to the correlative microscopy strategy applied to the aerosol samples. In this section, each subsection describes the procedural steps highlighted in the figure in more detail. Section 2.1 describes the sampling sites, i.e., the indoor football halls; the aerosol sampling procedure is described in section 2.2 and 2.3; and the correlative microscopy routine is described in section 2.4, whose analytical steps shown in Figure 1 are detailed in the respective subsections.

2.1. Sampling sites

The study was conducted in two indoor artificial turf football halls in Norway during autumn 2022. The halls were selected to compare particle emissions from artificial turf infills under contrasting environmental and operational conditions. The hall with olive sand infill was situated in a rural area approximately 60 km outside Oslo, while the hall with crumb rubber infill was located in a more urban setting near the city centre of Oslo. Both halls were adjacent to outdoor artificial turf arenas with sand and crumb rubber infill, which could represent a potential source of background particle contamination.

The spatial configurations of the halls are illustrated in Figure 2 (a) and (b). In the hall with olive sand infill, the artificial turf field was directly accessible from the main entrance, with the outdoor sampling site positioned adjacent to the entrance; two main streets or highways were located approximately 500 m from the hall. The annual average daily traffic on each of these roads varied between 7200 and 8600 vehicles [37]. In the hall with crumb rubber infill, access to the indoor turf was via an entrance hallway that served as a transitional space between the outdoor environment and the playing field, with the outdoor sampling site located at the entrance to this hallway; two main streets or highway were located approximately 500 m and 1500 m from this hall. The annual average daily traffic was 21,000 and 30,000 vehicles, respectively [37]. Thus, there was more traffic on the roads close to the rubber granulate hall compared to the olives stone hall.

Each hall contained an artificial football field measuring 60 × 40 m, surfaced with polyethylene artificial grass. The infill composition differed between the halls: one hall used a mixture of crushed olive stones and sand, while the other used crumb rubber and sand, applied as separate layers. For clarity, the halls are hereafter referred to as the "hall with olive sand infill" and the "hall with crumb rubber infill," respectively. Prior to the aerosol sampling campaign, granules of the infill materials were collected and analysed using Raman spectroscopy to provide reference spectra for identifying particles in aerosol samples with Raman microscopy (μ -Raman).

Ventilation conditions differed between the halls. In the olive sand hall, inlet and outlet ventilation rates were 5100 m³/h and 6900 m³/h, respectively, resulting in a net negative pressure indoors. In contrast, the crumb rubber hall had balanced inlet and

142 outlet ventilation rates of 24,000 m³/h. This difference in ventilation rate and pressure balance may influence the exchange of
143 indoor and outdoor air and, consequently, background particle concentrations at the two sites.

144 The turf in the olive sand hall was not subjected to regular cleaning, whereas the turf in the crumb rubber hall was typically
145 cleaned on a regular basis. However, cleaning was paused for several weeks prior to the study to ensure comparable conditions
146 between the two sites.

147 Ambient conditions, including temperature and relative humidity measured during sampling, are summarized in Table S1 in
148 the Supplemental Information.

149 2.2. *Aerosol sampling procedures*

150 Indoor aerosol sampling was performed during scheduled sport activities (two consecutive football matches; total duration
151 210 minutes including breaks) to reflect realistic use conditions. In each hall, three inhalable dust samplers (IOM samplers
152 designed to trap particles up to 100 µm in aerodynamic diameter, selected because of their plastic-free configuration) connected
153 to Casella Apex 2 sampling pumps were positioned at the midpoint of one long side of the hall, approximately 2 m from the
154 field boundary. The samplers were mounted at 1.7 m above ground level and spaced approximately 0.1 m apart (edge-to-edge).
155 Pumps were operated at a constant flow rate of 2 L/min for 210 minutes. Gold-coated track-etched polycarbonate membrane
156 filters (25 mm diameter, 10% porosity, 0.8 µm pore size; 60 nm gold coating on the top and 40 nm on the bottom, reducing the
157 pore size to approximately 0.6 µm, manufactured by APC, Eschborn, Germany) were used with metal support grids instead of
158 without cellulose support pads. The selected sampler did not allow for the mounting both supports. Excluding the support pads
159 reduced flow resistance, thereby preserving pump capacity and enabling continuous sampling throughout the entire sampling
160 period. This configuration was validated in a pilot study, though it resulted in a characteristic cross-shaped area of markedly
161 lower particle deposition (approaching zero) on the filter surface (see Figure 3), likely reflecting reduced local filtration
162 efficiency where the filter was unsupported; this pattern was accounted for during particle imaging and analysis. Note the
163 thicker gold top coating compared to the standardly used 40 nm, which was specifically manufactured in order to significantly
164 reduce the evanescent Raman signal of the polycarbonate filter material and minimize etaloning affecting the signal [38]. Pumps
165 were calibrated immediately before and after sampling using an Aalborg rotameter (adjusted to 2007.3 mL/min), with negligible
166 flow drift observed between calibrations. Near-complete collection efficiency can be assumed for particles >0.8 µm, whereas
167 efficiency for smaller particles is unknown and expected to decrease markedly in the nanoscale range. Outdoor sampling was
168 conducted outside each hall, also with three samplers placed ca. 0.1 m apart, on separate days for logistical reasons, using the
169 same setup (three samplers, 2 L/min, 210 minutes), mounted on a metal mesh fence 10–15 m from the building at 1.7 m height.

170

2.3. *Sample processing and quality control*

Before aerosol sampling, the gold-coated track-etched polycarbonate filters were placed in the IOM sample heads using stainless steel tweezers (cleaned with 96% ethanol), in a dedicated filter cabinet. The sample heads including filters were then covered in aluminium foil and stored in paper card boxes until sampling. After aerosol sampling, the filter heads were disconnected from the pumps, carefully embedded in aluminium foil, and transported to the Norwegian laboratory for dismounting of the filters. There, all filters (i.e., loaded filters, field blank filters and laboratory blank filters, see below) were individually transferred to a glass petri dish, sealed, and stored at room temperature until transportation to the microscopy laboratory in Germany, approximately six months later. Before transportation, each filter was removed from the glass petri dish and mounted on a pedestal with copper tape in a customised container for filter transport.

In addition to the loaded filters, the experimental set-up included 2 field blanks and 2 laboratory blanks from each indoor experimental day (for outdoor sampling, only 1 field blank and 1 laboratory blank were included), amounting to 9 blank filters in total across the study. The field blanks were handled in the same way as the loaded filters during the experiment; however, the pumps were not switched on and the aluminium foil covering the filter heads was not removed. The laboratory blanks were handled only inside the filter cabinet in the lab, i.e., they were transferred to a glass petri dish in parallel with the loaded filters and field blanks and subsequently transferred to the customised containers for filter transport.

To ensure blinding, all filters, i.e., including loaded filters, field blanks, and laboratory blanks, were assigned sample identification numbers, and no information regarding hall type or indoor/outdoor sampling status was provided for the subsequent SEM and Raman analyses. Sample preparation for correlative microscopy was conducted in a workbench with a HEPA-filtered air sheath flow preventing particles entering the workbench. Only metal tools were used to process samples for analysis.

2.4. *Correlative microscopy procedure*

The correlative microscopy (CM) procedure involved scanning electron microscopy (SEM), AI-assisted particle analysis, and μ -Raman microscopy (RM), followed by statistical extrapolation to derive airborne particle number and mass concentrations. The workflow comprised (i) positional calibration and coordinate system definition (Section 2.4.1), (ii) automated SEM image acquisition at randomised locations (Section 2.4.2), (iii) AI-assisted particle segmentation and morphological characterization (Section 2.4.3), (iv) Raman-based chemical identification of selected particles (Section 2.4.4), (v) classification of particles into predefined chemical classes (Section 2.4.5), and (vi) statistical extrapolation for exposure assessment (Section 2.4.6). The whole procedure is outlined in Figure 1.

2.4.1. *Positional calibration and coordinate system definition*

200 Samples were marked at four edges to enable spatial calibration for re-localisation. A Cartesian coordinate system was
201 established using custom stage navigation software ("TiNa"), developed for the Bruker SEM interface. The stage was manually
202 navigated to align the SEM field of view with the four markings, defining four reference (x,y)-coordinates that established the
203 spatial and angular alignment of the sample relative to the microscope stage. This calibration enabled reproducible repositioning
204 of the sample in SEM and Raman microscopes, achieving spatial matching within a few micrometres. For Raman microscopy,
205 additional fine calibration was performed by aligning the unique random pore pattern of the filter around a particle of interest
206 visible both in SEM and optical (Raman) images. This allows spatial correlation accuracies below 100 nm.

207 2.4.2. *Automated SEM image acquisition at randomised locations*

208 SEM analysis was conducted using a SU8230 (Hitachi, Tokyo, Japan). Within the area defined by the calibration coordinates,
209 the TiNa software generated 50 random (x,y)-positions and automatically navigated the stage so that the field of view was
210 centred at each coordinate. An autofocus algorithm optimised image sharpness at the substrate level before acquisition.
211 Secondary electron (SE) images were recorded at: accelerating voltage: 0.8 kV, image resolution: 5120×3840 pixels, pixel
212 size: 24.8 nm (corresponding to a nominal magnification of 1000 $\times$). This procedure yielded 50 unique micrographs of
213 equivalent size, resolution, and sharpness.

214 2.4.3. *AI-assisted particle analysis to identify particles of interest*

215 Particle analysis was performed using the custom software "FibreDetect" [39, 40]. Binary semantic segmentation of SE
216 images employed a pre-trained artificial neural network (ANN), which classified pixel clusters as "non-filter" and thereby
217 identified particulate objects. Initial segmentation performance was suboptimal; therefore, the ANN was iteratively re-trained
218 using manually annotated SEM images from the training repository of the Bundesanstalt für Arbeitsschutz und Arbeitsmedizin
219 (BAuA) (indoor and outdoor aerosol samples with diverse particle morphologies). Segmentation artefacts and undetected
220 particles were corrected manually and incorporated into subsequent training cycles until learning stagnation was observed.
221 Manual verification and minor corrections remained necessary.

222 For each segmented object, morphological parameters were calculated, including: projected area, minimum and maximum
223 Feret diameters, morphological classification (e.g., particle vs. fibre), and spatial coordinates. Particles of interest (POIs) were
224 selected based on a minimum projected area threshold of approximately $0.5 \mu\text{m}^2$ ($\approx$ equivalent circular diameter of $0.8 \mu\text{m}$,
225 corresponding to filter pore size). This threshold was applied because the aerosol sampling collection efficiency for smaller
226 particles was uncertain as mentioned above. Furthermore, smaller particles and weak Raman scatterers can require inefficiently
227 long integration times, posing a risk to damage the particles or the filter substrate with the excitation laser. This can lead to
228 false-positive carbon-related peaks in organic material. Particles below this threshold were labelled *Not classified* and excluded
229 from chemical analysis.

230

231

232 2.4.4. Raman analysis of particles of interest

233 From the 50 unique SEM images acquired per sample; 20 images were randomly selected for Raman analysis. The
234 coordinates of particles of interest (POIs) identified in the SEM images were transferred to the Raman microscope, and spatial
235 alignment was achieved as described previously.

236 Raman microscopy was performed using an Alpha 300 RM apyron system (WITec, Ulm, Germany) equipped with a Zeiss
237 LD EC Epiplan-Neofluar HD 100×/0.75 NA objective (Zeiss, Oberkochen, Germany). Video-camera imaging provided an
238 additional magnification factor of 11.6×. Raman spectra were acquired using a 532 nm laser at laser powers of 1–2 mW. The
239 scattered light was collected into an optical fibre coupled to a spectrometer of 300 mm focal-length equipped with a 600 lines
240 mm⁻¹ grating blazed at 550 nm and an EMCCD camera (Newton DU970P-BVF, Oxford Instruments GmbH, Wiesbaden,
241 Germany).

242 Spectral acquisition times were adjusted to achieve an adequate signal-to-noise ratio where possible. Acquisition conditions
243 were adjusted to avoid visible particle damage, excessive fluorescence, or detector saturation.

244 Spectral identification was performed using the TrueMatch module of the Project software (Version 7.0, WITec, Ulm,
245 Germany). Raman spectra were compared against: (a) a study-specific reference library, including tyre-derived rubber particles
246 assumed to have originated from the infill and crushed olive-stone infill material; (b) the commercial ST Japan Raman database
247 [41] as well the open access RRUFF database [42]; and (c) additional reference spectra obtained from collected materials,
248 including human hair, skin flakes, mineral particles, and non-black rubber materials.

249 A preliminary hit quality index (HQI) acceptance threshold of 70% was used for spectral matching, calculated using the
250 Pearson correlation coefficient against a reference spectrum. Automatic database matching was only one of the criteria for
251 identification. All spectra were visually inspected, and assignments were accepted only when the main diagnostic bands,
252 supporting secondary bands, particle morphology, and library match were mutually consistent. Spectra with strong
253 fluorescence, low signal-to-noise ratio, incomplete diagnostic bands, or conflicting spectral features were assigned
254 conservatively to broader classes, *Other*, *Unknown*, or *Not classified*, according to the criteria described below.

255 Identification of tyre-derived rubber particles required careful interpretation because tyre formulations differ between
256 manufacturers and may be altered by aging, wear, and environmental exposure. Characteristic features included the carbon
257 black signal, with the G-band around 1582 cm⁻¹ attributed to graphitic carbon and the D-band around 1345 cm⁻¹ associated
258 with structural disorder in the carbon matrix. Rubber-related bands in the fingerprint region and near 2900 cm⁻¹ were considered

supportive but were often partially masked by carbon black. Particles were therefore classified as tyre-derived rubber only when carbon black features, rubber-related spectral contributions, particle morphology, and comparison with the study-specific tyre reference spectra collectively supported this classification.

2.4.5. Particle classification criteria

Particles were classified into predefined chemical classes based on Raman spectral features supported by morphological characteristics (see Table 1).

A particle class was reported separately only when at least three particles of the same class were detected within the analysed Raman subset. Classes represented by fewer than three particles were not extrapolated individually and were grouped under *Other*, unless the available spectra were insufficient for reliable identification, in which case particles were assigned to *Unknown*. Particles were assigned to one of the following main chemical classes: *Tyre-derived rubber*, *Olive stone*, *MNP*, *Quartz*, *Mineral*, *Natural*, *Spore*, *Soot*. In addition, three procedural categories — *Other*, *Unknown*, and *Not classified* — were used to hold particle classifications that could not be assigned to one of the compositional classes, either due to low particle counts, inconclusive spectra, or exclusion based on size (see above).

Rubber-based particles originating from the crumb-rubber infill were expected to be spectrally similar to tyre wear particles, as both materials are derived from carbon-black-filled automotive tyre rubber. Because this compositional similarity limits the ability of Raman spectroscopy to distinguish crumb-rubber-infill particles from traffic-derived tyre wear particles, such particles were assigned to the generalized class *Tyre-derived rubber* rather than to a more specific source category.

Particles were assigned to the *Olive stone* class when their Raman spectra showed high similarity to spectra acquired from the study-specific olive-stone reference material. These assignments were further checked by visual inspection of the main diagnostic bands and by comparison with particle morphology.

The *MNP* class would include synthetic polymer particles such as polyethylene, polypropylene, polystyrene, polyethylene terephthalate, polyamide, polyvinyl chloride, polymethyl methacrylate, polyurethane, acrylonitrile-butadiene-styrene, styrene-butadiene-based non-tyre rubber, and other synthetic polymeric fragments or fibres. Particles containing polymer spectra but dominated by fillers, pigments, soot, or additives were classified as *MNP* only when polymer-specific Raman bands were clearly identifiable.

The *Quartz* class was assigned separately when diagnostic quartz bands were present, particularly the characteristic band near 464 cm⁻¹, together with supporting spectral and morphological evidence.

The *Mineral* class included inorganic non-quartz particles such as aluminosilicates, feldspar-like particles, clay-like particles, mica-like particles, carbonates such as calcite or dolomite, sulphate minerals such as gypsum, metal oxides including

288 titanium dioxide or iron oxides, glass-like amorphous silicates, ceramic or cementitious particles, and unresolved inorganic
289 mineral dust. Non-quartz mineral particles were assigned to the mineral class when Raman spectra and morphology indicated
290 inorganic mineral material but did not justify reliable species-level identification.

291 *Spores* were also reported separately when their morphology and Raman spectra were consistent with biological spore-like
292 particles. *Fungal spore* identity was further confirmed by a complementary cultivation-based analysis, described separately in
293 Section 2.4.

294 The *Natural* class included non-synthetic organic background particles of probable biological or biogenic origin, including
295 keratinaceous material such as skin flakes and hair fragments, cellulose-rich particles, plant-derived fragments, lignocellulosic
296 material, paper- or cotton-like fibres, and unresolved natural organic matter. Although biogenic in origin, olive-stone and spore
297 particles were reported as separate classes rather than grouped under *Natural* (see above).

298 The *Soot* class was assigned to carbonaceous particles showing dominant disordered carbon features, including D- and G-
299 band signals, without sufficient evidence for a rubber matrix or polymer-specific bands.

300 The *Other* class included particles for which a Raman-based assignment was possible but which did not belong to one of the
301 predefined main classes or occurred below the threshold for separate class-level reporting.

302 The *Unknown* class included particles for which a Raman spectrum was obtained but no confident assignment could be
303 made, either because the spectral match quality index was below the 70% threshold or because visual inspection did not support
304 the chemical identity hypothesis derived from the spectrum.

305 Table 1 lists all particle classifications, the primary method of identification and which reference spectra were used as well
306 as a qualitative assessment of the confidence in the classification results and possible materials the particle could have been
307 confused with.

308

309 Table 1: Summary of particle classes, classification-confidence scheme, possible confusions.

Particle class	Primary identification method	Reference spectrum used	Confidence level	Possible confusion
<i>Tyre-derived rubber</i>	Raman, supported by morphology and size	Study-specific crumb rubber and black tyre-wear reference spectra	Medium–High	carbon-black-rich particles, soot-stained polymer fragments
<i>Olive stone</i>	Raman	Study-specific olive-stone reference; ST Japan Raman database	High	Other lignocellulosic or plant-derived particles
<i>MNP</i>	Raman	ST Japan Raman database; polymer reference spectra	High	Fluorescent particles (only with high laser powers), degraded/weathered polymers,
<i>Quartz</i>	Raman	ST Japan Raman database; RRUFF database	High	none
<i>Mineral</i>	Raman, supported by morphology	ST Japan Raman database; RRUFF database	Medium–High	Pigments, glass, ceramic/cementitious particles, mixed inorganic-organic particles
<i>Natural</i>	Raman, supported by morphology	Skin flake reference; ST Japan Raman database; Customized reference spectra database	Medium	Autofluorescent organic matter, degraded biological material, natural fibres, lignocellulosic particles
<i>Spore</i>	Morphology, supported by Raman;	Spore-like morphology; ST Japan Raman database	High	pollen fragments, other biological particles
<i>Soot</i>	Raman, supported by morphology	ST Japan Raman database; Customized reference spectra database	High	combustion-derived carbonaceous particles, charred organic matter
<i>Other</i>	Raman	ST Japan Raman database or additional reference spectra	Variable / class-specific	Particles outside predefined target classes; rare or ambiguous identified materials
<i>Unknown</i>	Raman attempted	No confident match or no visually reliable assignment	Not applicable	Low signal-to-noise spectra, fluorescence, mixed particles, incomplete spectra, non-Raman active particles
<i>Not classified</i>	Size	Not applicable	Not applicable	carbon-black-rich particles, soot-stained polymer fragments

310

311 2.4.6. Data processing for exposure assessment

312 The CM dataset comprised particle counts per class, projected areas, Feret diameters, and chemical identity. Statistical
313 extrapolation from analysed filter areas to airborne concentrations followed the methodology outlined in CEN/TS 18117:2025-
314 04 [43]. Average number concentrations were derived as:

315
$$\text{Number concentration} = \frac{\text{Particle Count}}{\text{Sampling Volume}} \quad (\text{Eq. 1})$$

316 In order to determine the *particle count* on the whole filter, far too many SEM images would be required to be analysed.

317 Hence, number concentrations were determined from the object counts within the analysed sample area, the number of SEM
318 images multiplied by their individual area (~127×95 μm²). To account for both counting statistics and environmental
319 heterogeneity between the sampling replicates (n=3), uncertainties for quantitative data are reported as the standard deviation
320 (SD) of the mean across the three replicates, rather than extrapolated Poisson confidence intervals.

Particle masses were estimated based on the product of approximated individual particle volumes and presumed densities of the materials respective for each particle class. Because SEM provides only two-dimensional projections, particle volumes were geometrically approximated as prolate ellipsoids of revolution, assuming rotational symmetry about the major axis:

$$V_e = \frac{\pi}{6} d_{f,\min}^2 \cdot d_{f,\max}, \quad (\text{Eq. 4})$$

where $d_{f,\min}$ and $d_{f,\max}$ are the minimal and maximal Feret diameter. Note that this approximation tends to overestimate to the true volume. Particles were also separated based on their volume-equivalent diameters, the diameter of a sphere with equivalent volume. The combined particle masses of all particles yielded the total particle mass, particles with volume-equivalent diameters smaller than 10 μm yielded the PM_{10} -equivalent mass fraction, for particles smaller than 2.5 μm the $\text{PM}_{2.5}$ -equivalent mass fraction. Note that PM fractions are defined by aerodynamic diameter, which cannot be determined from particle properties measured by SEM alone. Normalization by the product of specific sampling volume and analysed area as described above yielded the respective particle mass concentrations.

Size distributions were obtained by binning the available data of volume equivalent diameters for each particle class. Bin widths were calculated according to the Freedman-Diaconis rule [44].

Size datasets of each set of samples (i.e., three aerosol samples per sampling location) were pooled in order to increase the data sample size. Based on visual inspection of the number of distinguishable modes in the binned size distribution, either a mono- or bimodal lognormal probability density function was fitted to the data [45].

2.5. Spore analysis

To identify species of spore-forming fungi, one air sampling filter, together with a corresponding negative control filter, was randomly selected from each of the two halls (olive sand infill and crumb rubber infill). Filters were placed face-down onto DG18 agar plates supplemented with chloramphenicol (Thermo Scientific, Schwerte, Germany) to suppress bacterial growth, and incubated at 25 °C for eight days. Plates exhibiting fungal growth were subsequently sealed and sent to a specialised, accredited analytical laboratory (Umweltmykologie GmbH, Berlin, Germany) for species identification based on morphological characterization by light microscopy.

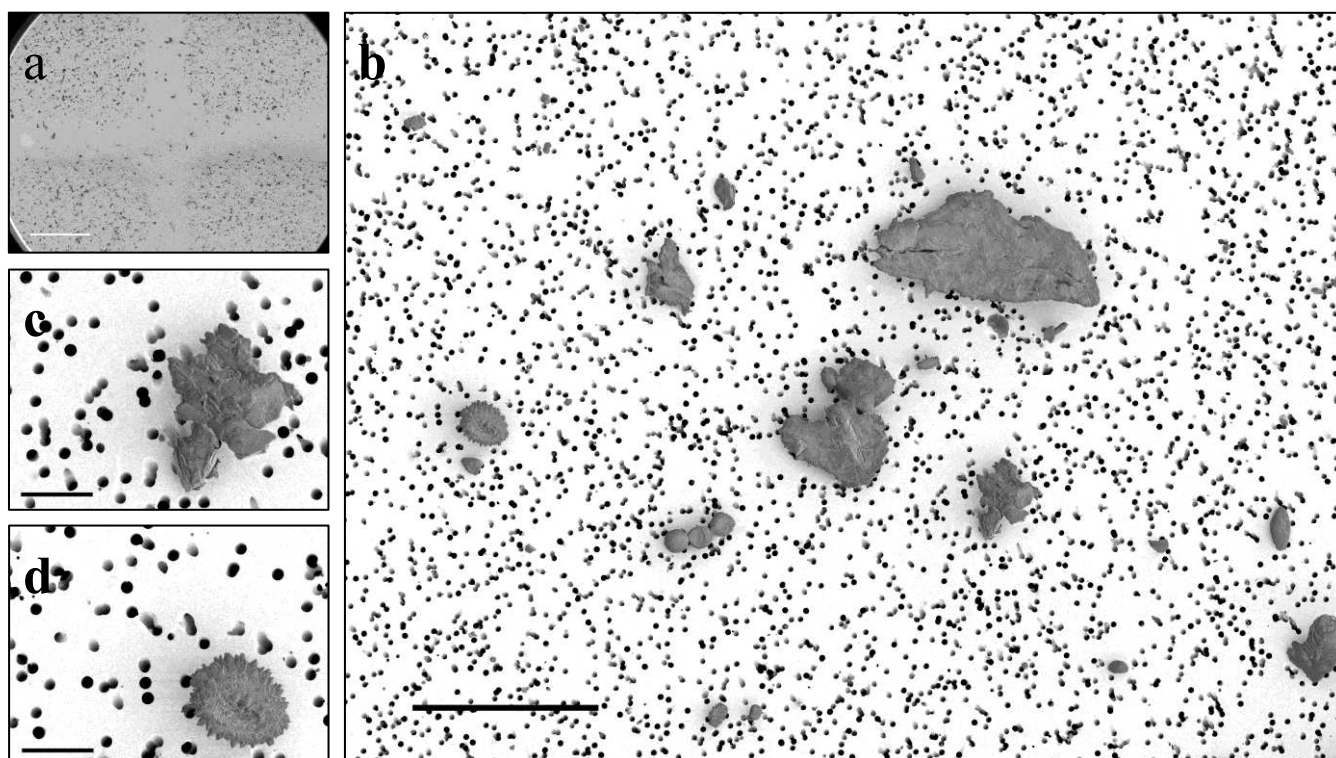


Figure 3: SEM micrographs from indoor aerosol sample collected at football hall with olive sand infill. a) Sampling method generated cross-shaped zone with low particle count and thus inhomogeneous particle deposition density. This pattern was present on all samples, reducing the effective collection area. Scale bar 250 μm . b) Particle collected in football hall with olive sand infill are of arbitrary shape and surface properties. Scale bar 30 μm . c) Unidentified particle of arbitrary shape, Scale bar 5 μm . d) Particle presumed to be biological, later identified as spore of most likely *Aspergillus glaucus*. Scale bar 5 μm .

3. Results

3.1. Sample contamination and cross-contamination

Blank filters were analysed by SEM to assess the presence of background particles, with detected particles further characterized by μ -Raman microscopy to determine their chemical composition. A total of four laboratory blanks, three indoor blanks (one from the hall with olive sand infill and two from the hall with crumb rubber infill), and outdoor blank filters were analysed following the same procedure as the main samples.

All blank filters contained a small number of particles (between 5 and 20 in ca. 10,000 μm^2 sample area) of varying chemical composition, predominantly cellulose-based materials, soot, and other natural particles. Tyre-derived rubber particles (between 0 and 5) were also detected, consistent with their presence as background particles (e.g., ambient tyre wear). A small number of particles were additionally identified as varnish.

No olive-stone particles were detected in any blank sample, indicating that contamination from infill material during sample preparation was negligible.

Overall, these findings suggest that contamination by ambient particles during sample preparation was possible but low relative to particle counts observed in the aerosol samples. Cross-contamination between samples was considered unlikely.

3.2. SEM images

Figure 3a shows a low-magnification overview image of one of the indoor aerosol samples from the hall with olive sand infill. A cross-shaped area of low particle density was visible on each sample. This pattern was attributed to the absence of a cellulose backing filter beneath the sampling filter, which would otherwise have compensated for flow patterns generated by the underlying metal support grid. The absence of cellulose filter reduced the flow through the pores directly above the grid, producing a projection of lower deposition probability. SEM micrographs were only included in the analysis if they were acquired outside this affected area. As a result, the effective sample area was reduced by approximately 6% on average, based on the measured size of the affected zone relative to the total sample area across all filters.

Figure 3b shows a representative SEM micrograph illustrating the high heterogeneity in particle size, shape, and surface texture observed across samples. Two examples are highlighted in Figure 3c and 3d: the former shows a particle of arbitrary shape that could not be identified based on morphology alone, while the latter was presumed to be a biological particle based on its spheroidal shape and repeating surface features. Such characteristics are unlikely to arise outside of natural, biological processes.

3.3. Chemical identification and classification of particles of interest by Raman spectroscopy

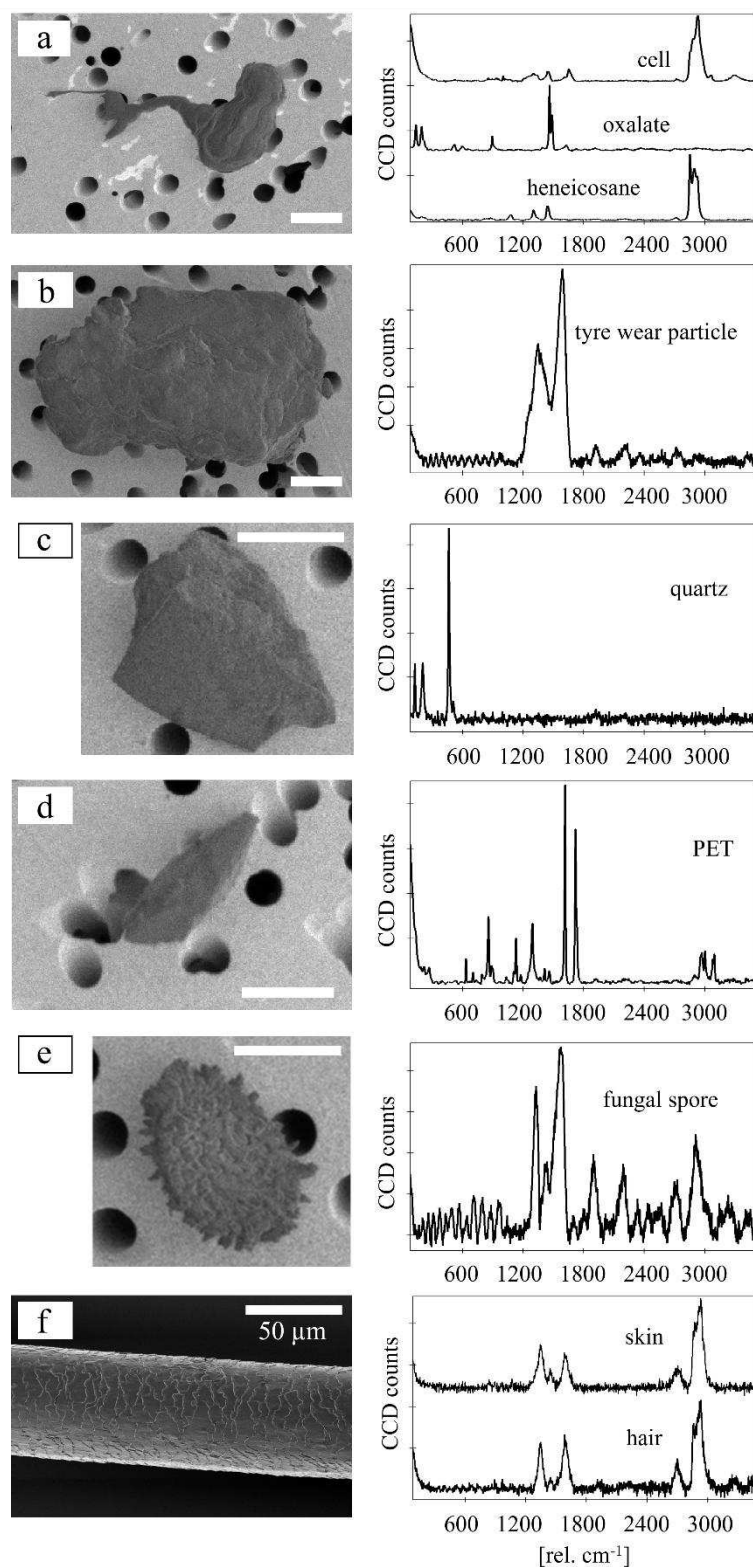
Figure 4 shows examples of particles found in the two halls, along with their corresponding Raman spectra. In Figure 4a, spectra from olive stone granules are compared with reference spectra from the ST Japan database and previously measured reference samples of the infill material. Key Raman peaks were associated with specific components, including biological cells, oxalate (a compound commonly found in olive stones), and heneicosane, a waxy compound showing a high degree of spectral similarity. The majority of biological cell spectra showed features in the fingerprint region ($400\text{--}1800\text{ cm}^{-1}$), corresponding to proteins, carbohydrates, nucleic acids, and lipids. A stronger peak at approximately 2900 cm^{-1} , characteristic of phospholipids in cellular membranes, was also observed. A broad peak around 2100 cm^{-1} , within the cell-silent region between 1800 cm^{-1} and 2800 cm^{-1} of most biological samples, could not be clearly assigned but reappeared consistently in several olive stone reference samples.

Some spectra from tyre-derived rubber particles exhibited small peaks within the fingerprint region ($400\text{--}1800\text{ cm}^{-1}$), a characteristic peak near 2900 cm^{-1} (CH_2 stretching vibrations), and no peaks in the silent region. Spectra of tyre-derived rubber particles further revealed the characteristic broad, overlapping D- and G-bands of carbon black at 1348 cm^{-1} and 1582 cm^{-1} , respectively. This carbon black signal masked the typical rubber-related peak pattern within the fingerprint region, as shown in Figure 4b, but enabled clear identification of these particles as tyre-derived rubber.

As shown in Figure 4c, quartz sand particles exhibited a prominent quartz peak at 465 cm^{-1} in the fingerprint region, along with smaller peaks at 129 and 205 cm^{-1} , corresponding to the DQ12 quartz standard from the JRC repository. Reference spectra

388 for specific polymers, obtained from the ST Japan database, were used to identify micro- and nanoplastic particles, with PET
389 particles (e.g., Figure 4d) being the most prevalent. Figure 4e presents an exemplary spectrum acquired from one of the spores
390 found in the hall with olive sand infill.

391 Finally, the spectra of skin flakes and human hair (Figure 4f) differed markedly from those of intact, living cells, showing
392 stronger signals at the carbon peaks (1348, 1582, and 2700 cm^{-1}). These signals are likely attributable to fatty acids and proteins
393 present within the skin's protective barrier, consistent with observations reported in previous studies [46].



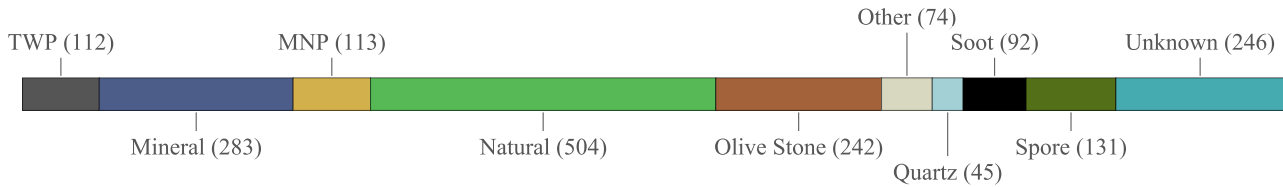
394

395 *Figure 4: Left: SEM images of a) an olive stone particle, b) a tyre-derived rubber particle, c) a quartz particle, d) a PET MNP particle, e) a*
 396 *fungal spore found in Hall with olive sand infill and f) image of a human hair. Right: Associated Raman spectra of the above-mentioned*
 397 *particles. a) distinct Raman spectra from olive stones, b) spectra from tyre wear particles, c) quartz particle, d) PET particle, e) from a fungal*
 398 *spore and f) spectra from human skin and hair as reference spectra for natural occurring particles at contact sport locations. Raman spectra*
 399 *in CCD counts over relative cm⁻¹. Scale bar: 2 μm (unless otherwise stated).*

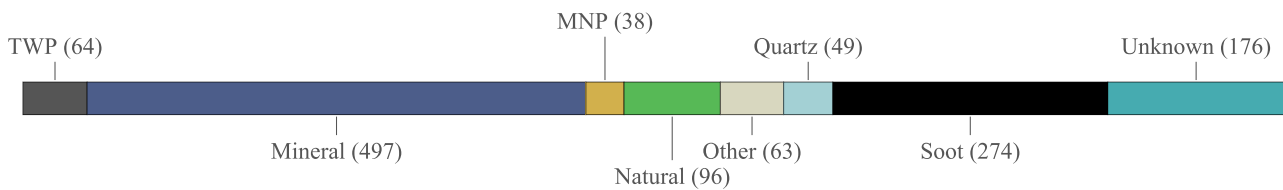
Particle Number Concentration [$10^3/\text{m}^3$]

Hall with olive sand infill

Indoor (Total: 5480, Not classified: 66 %)

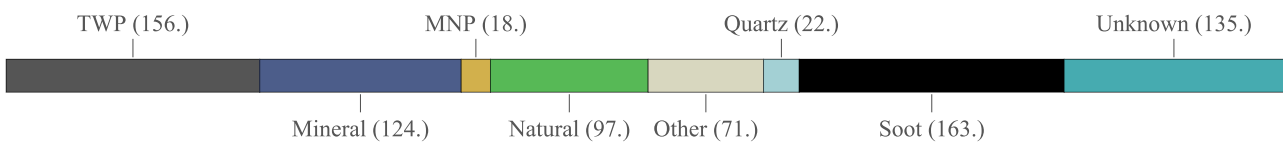


Outdoor (Total: 9597, Not classified: 87 %)



Hall with crumb rubber infill

Indoor (Total: 5252, Not classified: 85 %)



Outdoor (Total: 12025, Not classified: 90 %)

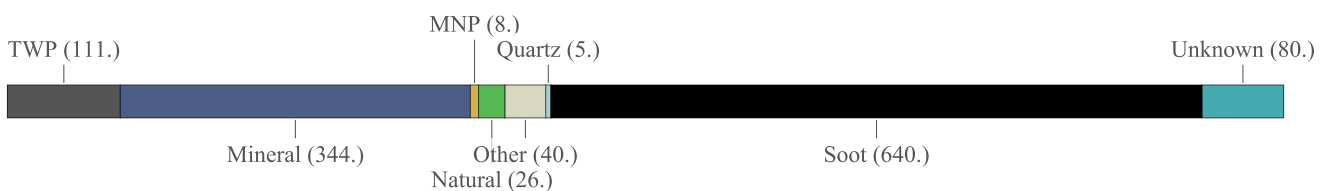


Figure 5: Particle number concentrations for each particle class as stacked bar charts for which each bar size was generated on the basis of its percentage of the total particle concentration minus the "Not classified" concentration, left out for better visibility.

3.4. Particle number concentrations

Figure 5 shows the fractional contribution of each particle class to the total particle number concentration, presented as stacked bar charts. Each bar segment reflects the percentage of the total particle concentration excluding the "Not classified" fraction, which was omitted for improved visibility. Quantitative results shown in the callouts represent the rounded means $\pm$ rounded standard deviations. Table S2 in the Supplemental Information provides a comprehensive summary of the exact values.

405 For the hall with olive sand infill, the mean indoor total particle number concentration was approximately 56% of the outdoor
406 concentration, a difference heavily driven by the highly variable '*Not Classified*' fraction. However, when unclassified particles
407 (mainly nanoparticles) were excluded, the remaining indoor concentration exceeded the outdoors by nearly 50%. Olive stone-
408 related particles were found exclusively in indoor samples, contributing substantially to the overall indoor aerosol burden.
409 Quartz particles, a common component of urban aerosols, were present in both environments at comparable concentrations.
410 Natural particles, primarily human skin flakes, were approximately five times more abundant indoors than outdoors, consistent
411 with the presence of players and staff, though high variability between replicates warrants caution. Tyre-derived rubber particles
412 were approximately 75% higher indoors, and MNPs were found in notably higher concentrations indoors, i.e., roughly triple
413 the outdoor levels.

414 For the hall with crumb rubber infill, a similar trend was observed for total particles, with the indoor concentration reaching
415 only about 44% of the outdoor concentration, again dominated by the '*Not Classified*' fraction. In this case, however, the indoor
416 concentration remained nearly 40% lower than the outdoor concentration even after excluding unclassified particles: this
417 contrasts with the pattern observed in the olive sand hall. The fractional contribution of tyre-derived rubber particles was similar
418 indoors and outdoors, though the absolute concentration was approximately 40% higher indoors. Natural particles and quartz
419 particles both showed roughly four-fold higher concentrations indoors compared to outdoors. Particles identified as MNPs were
420 detected in all samples, but in relatively small amounts, with indoor levels being more than double those found outdoors.).

421 3.5. Particle sizes

422 Size distributions were compared for particle classes present both indoors and outdoors, i.e., *Tyre-derived rubber* particles,
423 *Mineral*, *MNP*, *Natural*, *Quartz*, and *Soot* particles. For both halls, indoor particle sizes were generally larger than outdoor
424 particle sizes. This suggests that larger particles accumulated over time, due to their greater mass and their tendency to settle
425 on the turf, and were subsequently resuspended through player activity.

426 Figure 6 shows particle volume-equivalent diameter count distributions for tyre-derived rubber particles found on indoor
427 and outdoor samples from both halls. While all size distributions showed a peak in the 1–5 μm range, the indoor particle size
428 distribution for the crumb rubber hall might exhibited a second, much smaller peak around 10 μm . All other particle size
429 distributions were monomodal, regardless of sampling location. Table S2 in the Supplemental Information provides particle
430 sizes as mean volume-equivalent diameters with corresponding standard deviations.

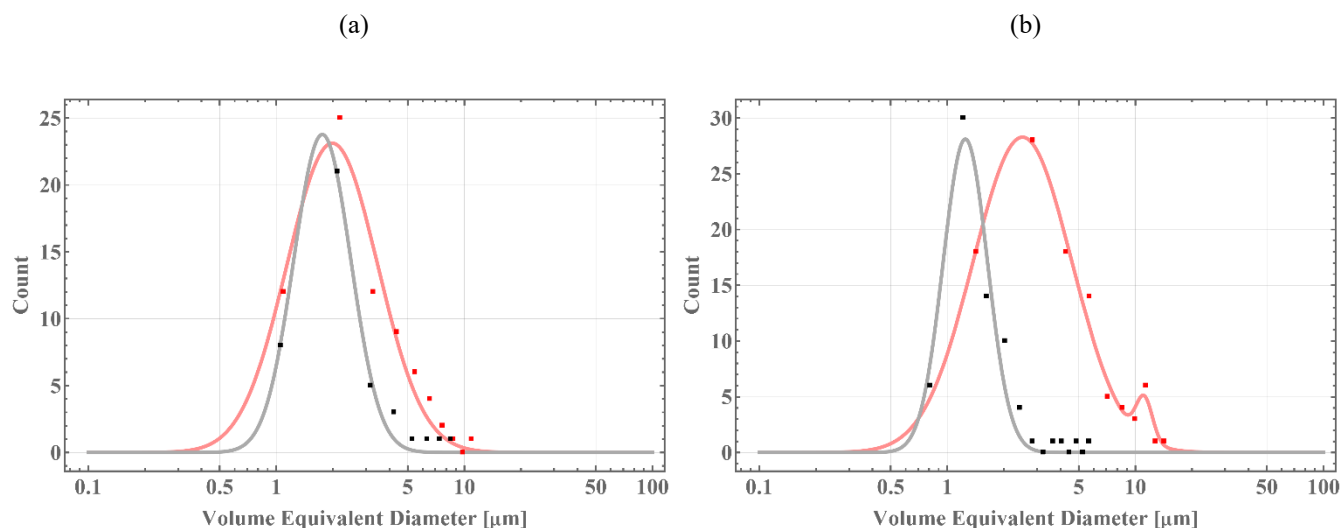


Figure 6: Particle size distributions of collected tyre-derived rubber particles in the hall with olive sand infill (a) and crumb rubber infill (b), with black curves representing the outdoor aerosol and red the indoor aerosol. Curves are lognormal fits of the binned size data. Note the possible minor population of larger particles which is more pronounced for the indoor aerosols of the crumb rubber infill hall (b), leading in to a visible 'bump' in the data. The data was thus fitted with a bimodal distribution.

3.6. Particle Mass Concentrations

Figure 7 displays the estimated particle mass concentrations ($\mu\text{g}/\text{m}^3$, logarithmic scale) for each particle class, differentiated by size fraction ($\text{PM}_{2.5}$ in full colour, PM_{10} in lighter colour, and total mass in the lightest colour), for indoor and outdoor environments in the halls with olive sand and crumb rubber infill. The *All Objects* category represents the sum of all individual particle classes. Table S2 in the Supplemental Information provides a comprehensive summary of the quantitative results, including mass concentrations for all size fractions.

Across all four panels, the *All Objects* category exhibited the highest total mass concentration, spanning approximately two to three orders of magnitude from $\text{PM}_{2.5}$ to total mass, indicating that the bulk of particle mass is contained in the coarser size fractions rather than the fine $\text{PM}_{2.5}$ range.

In the olive sand hall (indoor), the *Olive Stone* category showed the highest total mass concentration among individual classes, exceeding $1000 \mu\text{g}/\text{m}^3$, with a pronounced increase from $\text{PM}_{2.5}$ to total mass, indicating a predominantly coarse distribution. *Mineral* and *Natural* displayed the next-highest total concentrations, though with visibly larger $\text{PM}_{2.5}$ contributions, suggesting a greater fine-fraction presence. The *Tyre-derived rubber* particles, *MNPs*, *Quartz*, *Soot*, *Spore*, and *Unknown* categories showed substantially lower concentrations across all fractions, with *Tyre-derived rubber* particles and *Soot* displaying relatively small differences between $\text{PM}_{2.5}$ and total mass. In the corresponding outdoor environment, the *Mineral* category exhibited the highest concentrations across all size fractions, following a pattern similar to indoor conditions but with reduced total mass overall. *Natural* and *Other* showed moderate concentrations, while the *Soot* category displayed a relatively small increase from $\text{PM}_{2.5}$ to total mass, indicating a predominantly fine outdoor size range. *Tyre-derived rubber* particle class concentrations remained low and comparable between environments.

450 In the crumb rubber hall (indoor), overall concentrations were markedly lower than in the olive sand hall, with the
451 total mass of *All Objects* reaching only the order of 100 $\mu\text{g}/\text{m}^3$. The *Other* and *Natural* categories displayed the highest total
452 concentrations among individual classes, followed by tyre-derived rubber particles and Mineral, while *MNPs*, *Quartz*, *Soot*,
453 and *Unknown* remained comparatively low. For the *Other* class, the elevated total mass was driven predominantly by a single
454 very large particle. *Tyre-derived rubber* particles showed a relatively small difference between $\text{PM}_{2.5}$ and total mass, indicating
455 a higher relative fine-fraction contribution, consistent with release from the infill. In the outdoor environment of the crumb
456 rubber hall, concentrations were lower still, with *Soot* exhibiting the highest concentration among individual classes and a
457 distinctly small difference between $\text{PM}_{2.5}$ and total mass, indicating that soot is concentrated almost entirely within the fine
458 fraction. *Mineral*, *Other*, and *Tyre-derived rubber* particles showed moderate concentrations, while the remaining classes were
459 low across all fractions.

460 Across both hall types, the coarse-dominated classes (*Olive Stone*, *Mineral*, *Natural*) consistently showed pronounced
461 increases from $\text{PM}_{2.5}$ to total mass, whereas *Soot* and *Tyre-derived rubber* particles consistently showed comparatively small
462 differences, indicating a size-dependent distribution that differs systematically between particle classes. Mean total
463 concentrations trended higher in the olive sand hall than in the crumb rubber hall, and generally appeared higher indoors than
464 outdoors. However, the high standard errors associated with several particle classes reflect significant environmental
465 heterogeneity between sampling replicates, meaning absolute differences should be interpreted with caution.

466 The three reported metrics differ in robustness. Because $\text{PM}_{2.5}$ mass concentrations aggregate over a large number of
467 particles, they are less susceptible to the extreme sample-to-sample variations seen in total mass. PM_{10} and total mass
468 concentrations are increasingly influenced by the coarse tail of the size distribution—total mass most severely, since it has no
469 upper size limit—and are therefore more sensitive to the presence or absence of individual large particles in a given sample.
470 Total mass values should accordingly be read as order-of-magnitude indicators rather than precise concentrations. While still
471 subject to environmental heterogeneity, $\text{PM}_{2.5}$ is treated as the more robust metric for comparison with air quality guidelines
472 and between samples, with PM_{10} occupying an intermediate position.

473 3.7. Composition of Particle Mass

474 Figure 8 presents the mass fraction composition of particulate matter across the three size classes (Total, PM_{10} , and $\text{PM}_{2.5}$)
475 for the olive sand infill hall and the crumb rubber infill hall, each paired with corresponding outdoor measurements. In contrast
476 to the absolute concentrations in Figure 7, this representation expresses the relative contribution of each particle class to total
477 mass, independent of absolute magnitude.

478 In the olive sand hall (indoor), the Total and PM_{10} fractions were dominated by *Olive Stone*, consistent with both the crushed-
479 olive-stone infill composition and the pronounced coarse distribution of this class observed in Figure 7. *Olive Stone* constituted

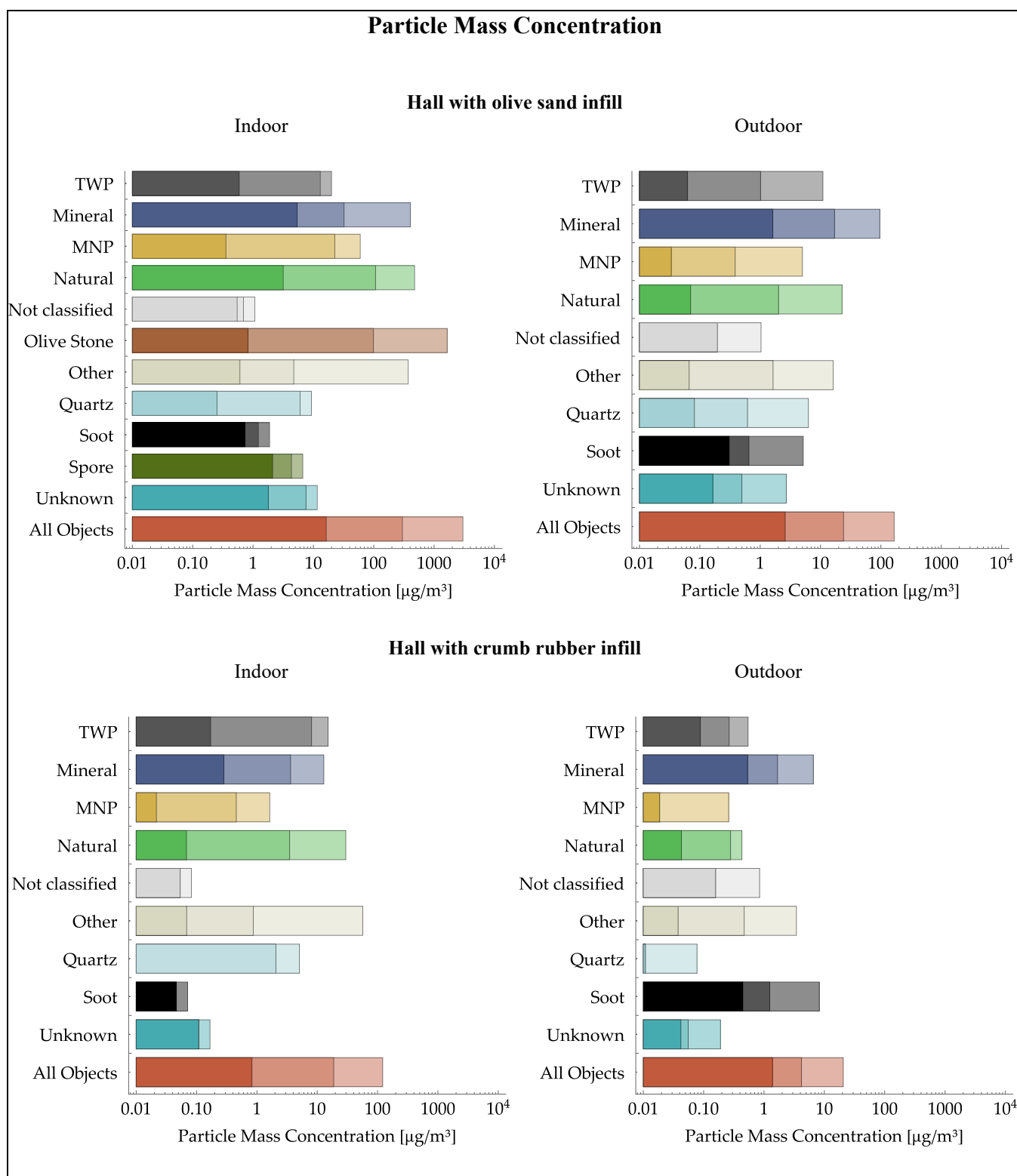


Figure 7: Estimated particle mass concentrations for each particle class as stacked bar charts, differentiated by $\text{PM}_{2.5}$ (full colour), PM_{10} (slightly lighter colour) and total mass concentration (light colour)

only a minor fraction of $\text{PM}_{2.5}$, confirming its concentration in the coarse and fine dust fractions. Quartz, a known constituent of the olive sand infill, contributed a minor but consistent share across all size fractions. Tyre-derived rubber particles were present in the PM_{10} and $\text{PM}_{2.5}$ fractions but constituted only minor proportions of total mass, in agreement with their low

absolute concentrations. Outdoor measurements showed a markedly different composition, with *Mineral* particles representing the dominant fraction across all size classes, consistent with the outdoor concentration pattern in Figure 7. The *Soot* fraction increased proportionally in PM_{2.5} relative to the Total and PM₁₀ fractions, reflecting its predominantly fine size range. Tyre-derived rubber particles contributed only a small fraction of the outdoor mass across all size classes, comparable in magnitude to the fraction observed indoors at this location.

In the crumb rubber hall (indoor), the Total and PM₁₀ fractions were characterized by substantial contributions from *Natural* particles. Quartz and tyre-derived rubber particles yielded relatively large contributions, consistent with the composition of the crumb rubber infill. The tyre-derived rubber particle fraction increased progressively with decreasing particle size, becoming the largest single contributor in PM₁₀ and the second largest in PM_{2.5}. This pattern contrasts with the olive sand hall, where tyre-derived rubber particles remained a minor fraction across all size classes, and is consistent with the higher relative fine-fraction contribution noted in Figure 7. Outdoor measurements at the crumb rubber hall were dominated by *Soot* across all size fractions, with the *Soot* proportion increasing further in PM_{2.5}. Mineral and particles allocated to the *Not classified* class contributed secondary shares of total mass, while tyre-derived rubber particle and *Natural* fractions remained comparatively minor across all size classes.

Across both hall types, outdoor composition was consistently dominated by *Mineral* or *Soot* fractions with minimal tyre-derived rubber particle contribution — both typical background particle classes found in urban areas. In contrast, indoor compositions reflected the specific infill material used in each hall, with *Olive Stone* and tyre-derived rubber particle contributions showing distinct size-dependent trends between the two infill types.

3.8. Spore analysis results

Two distinct mould species were identified in air samples collected from the hall with olive sand. No spores were found on the aerosol samples collected in the hall with crumb rubber infill. Filters were incubated on DG18 agar at 25 °C for eight days, after which samples exhibiting fungal growth were sealed and dispatched to the accredited laboratory (Umweltmykologie GmbH, Berlin), where morphological identification was performed on the day of receipt. Fungal isolates cultured on growth medium were identified by morphological differentiation, assessed both macroscopically under a stereomicroscope (colony morphology) and by light microscopy (microscopic structures). These species were *Aspergillus glaucus* and a *Penicillium* species, most likely *Penicillium crustosum*,

Change of Dust Mass Composition with Particle Size

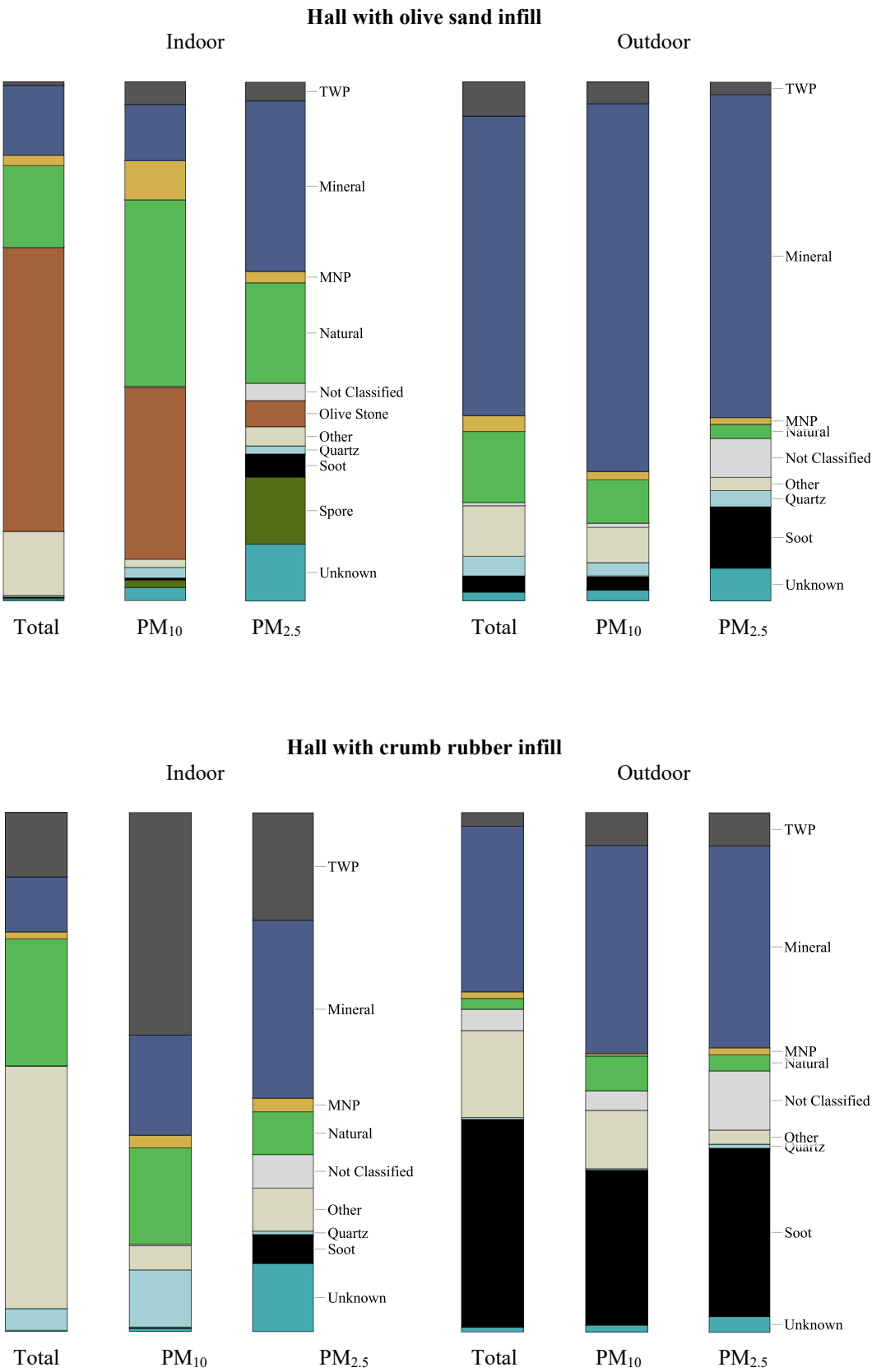


Figure 8: Estimated particle mass concentrations for each particle class as stacked bar charts, differentiated by PM₁₀, PM_{2.5} and total mass concentration. Bar size is its percentage of the total particle mass concentration.

510 4. Discussion

511 4.1. Insights from μ -Raman spectroscopy

512 Tyre-derived rubber particles posed a particular challenge due to the variability of both synthetic and natural variants, arising
513 from differing constituents, mixing ratios, or possibly surface-adsorbed chemicals. Nevertheless, these spectra were overall
514 consistent with rubber spectra reported in the literature and databases and could be identified by their broad G- and D-band
515 carbon signal in combination with their morphology.

516 For olive stone particles, a recurring signal at 2100 cm^{-1} (Figure 4) may be caused by the autofluorescence of the cellular walls
517 of plant cells and olive oil, as well as the degradation products of pigments within them [47]. However, other explanations for
518 this nonspecific signal in the 2100 cm^{-1} region are possible. Firstly, the background fluorescence of lignocellulose and other
519 cell wall components, such as pigments, can produce a broad background signal that may appear as a large, broad band in the
520 cell-silent region [48]. Although anthocyanins, which is a part of the pigment content in olive fruit, can strongly quench the
521 fluorescence of cell wall components, they themselves emit high levels of fluorescence when excited at 532 nm , as their
522 excitation maximum lies around 520 nm [49]. This results in an irregular, unspecific broad peak. A second possible explanation
523 is that fungi other than mould live saprophytically on the crushed olive stones, decaying the dead material. These fungi are
524 known to produce a large number of alkynyl-containing compounds [50], which give a strong Raman signal in the otherwise
525 empty region around 2100 cm^{-1} . Thirdly, as we lack information on the production steps, the presence of alkyl groups [51] in
526 fungicide chemicals used during pre-treatment of the olive stone particles cannot be excluded.

527 MNPs of different polymer types and quartz particles appeared morphologically similar but exhibited distinct and clearly
528 identifiable Raman spectra. Particles of natural origin were more difficult to classify further, as the Raman spectra of different
529 cell types differ only slightly and require high spectral quality to reliably distinguish subtle variations. The category "natural"
530 therefore included all spectra originating from known biological sources, spectra attributable to human hair or skin particles,
531 and particles exhibiting similar spectral and morphological characteristics.

532 4.2. Infill as particle source

533 In the hall with olive sand infill, particles were evidently released from the infill, as supported by the presence of olive
534 stone-related particles indoors that were absent outdoors.

535 For the hall with crumb rubber infill, this interpretation was more difficult, given that tyre-derived rubber particles were
536 found both indoors and outdoors. As a generalized class, tyre-derived rubber particles comprise both traffic-generated tyre wear
537 particles (TWPs) and particles derived from the crumb rubber infill, which cannot be distinguished spectroscopically. Thus, to
538 identify the infill as a particle source, other sources that might have led to the convective transport of tyre-derived rubber
539 particles into the hall must first be discussed. The outdoor number concentration of tyre-derived rubber particles at the hall with

crumb rubber infill was higher than at the hall with olive sand infill, which can be attributed to its more urban location. In general, data specifically on TWP concentrations are sparse, as many studies report on general road dust, which also comprises mineral, soot, and other particles. At a motorway in the Rhein-Main area of Germany, PM_{10} mass concentrations decreased with increasing distance from the road, from $3.8 \mu\text{g}/\text{m}^3$ (3.5 m) to $0.95 \mu\text{g}/\text{m}^3$ (30.5 m) [52]. The reason for this is the wide size distribution of TWPs, ranging from sub-micron up to $100 \mu\text{m}$, as reported by various studies [7]. Large particles ($>10 \mu\text{m}$) usually have short settling times [53] but contribute a large portion of the total particle mass, resulting in a drastic decrease of mass concentration with distance from their source. TWPs also typically increase in number concentration with decreasing particle size [54], peaking around $2 \mu\text{m}$ [55]. Both halls were located approximately 500 m from nearby motorways; therefore, smaller TWP particles could be expected. In fact, the mean particle sizes were around $1.5 \mu\text{m}$ in both cases. It should, however, be noted that the traffic load on the nearby motorway was much higher at the hall with crumb rubber infill than at the hall with olive sand infill. PM_{10} mass concentrations were in a similar range to the 30.5 m result in the previously mentioned study. In Switzerland, a PM_{10} mass concentration of $1.5 \mu\text{g}/\text{m}^3$ was measured for TWPs 50 m from the road [56]. Similar values have also been reported for Sweden [57], Oslo, Norway [58] and Hamburg, Germany [59]. Our findings compare well with these reported results. Note that, because our workflow reliably captured only particles above ca. $0.8 \mu\text{m}$, the sub-micron fraction is undercounted; given that TWP number concentrations rise with decreasing size, our data likely underestimate the actual concentration of the smallest particles.

In addition to tyre-derived rubber particles (and quartz) being coincidentally released due to abrasive processes during football play, TWPs from an outdoor source transported indoors, or generated during earlier play and settled onto the turf, could have been resuspended into the air. At least in the hall with olive sand infill, where tyre-derived rubber particles would not be expected to stem from abrasive processes, this would explain both their presence and their slightly larger size. Similar to the road dust discussion above, larger particles would have settled more efficiently on the indoor turf after being transported in by ventilation and open doors [60]. Abrasion as a particle release process in the crumb rubber hall was likely an additional but not predominant source, as hinted by the bimodal tyre-derived rubber particle size distribution (Figure 6b), with one large peak at ca. $3 \mu\text{m}$ and another at ca. $10 \mu\text{m}$. The former would be associated with blown-up older TWP dust, most likely transported in, and the latter with particles released by the crumb rubber infill due to abrasion. Note that this interpretation is speculative, as studies with similar analytical depth yielding size distributions specifically for particles released from the turf are lacking. Particle size data from laboratory rubber surface abrasion scenarios yielded mean particle sizes of the released aerosol of ca. $3 \mu\text{m}$ for soft rubber tyre rolling/braking wear [61] and $10 \mu\text{m}$ and larger for rubber composite abrasion [62]. These results indicate that sizes of tyre-derived rubber particles released from the infill are expected to be in the $10 \mu\text{m}$ -range. Nevertheless, even in the hall with crumb rubber infill, tyre-derived rubber particles may have predominantly originated from traffic as TWPs. Notably, the net negative indoor pressure in the olive sand hall (outlet $6900 \text{ m}^3/\text{h}$ vs. inlet $5100 \text{ m}^3/\text{h}$) might have promoted

infiltration of outdoor air through doors and leaks, providing a plausible pathway for outdoor tyre-derived rubber particles to enter and subsequently settle onto the turf, where the comparatively low air-exchange rate (discussed below) would further favour their retention.

Higher indoor concentrations of natural particles, such as skin flakes, were expected due to human presence. Mineral particles and soot were likely transported from the urban environment through the ventilation. The fact that the hall with crumb rubber infill was located in a more urban environment results in a ca. 2× higher soot number concentration compared to the hall with olive stone infill. Indoors, this difference becomes more pronounced, reaching 3× the levels of soot. This is consistent with the substantially higher ventilation rate at the crumb rubber hall (balanced inlet and outlet rates of 24,000 m³/h) compared to the olive sand hall (inlet 5100 m³/h, outlet 6900 m³/h), which would promote greater exchange of indoor and outdoor air and thus more efficient transport of urban-derived soot into the hall. Notably, this same high air-exchange rate would be expected to dilute indoor-generated particles more effectively. However, the fact that soot proved to be enriched indoors reinforces its predominantly outdoor origin. Conversely, particle classes associated with local release and resuspension (e.g., tyre-derived rubber particles and quartz) were enriched indoors in the olive sand hall, which is consistent with, though not attributable to the lower ventilation rate that would favour their accumulation. MNP release from the grass blades was not evident in either hall, as MNPs on the aerosol samples exhibited a variety of polymer bases, not predominantly polyethylene. At both halls, indoor concentrations were slightly higher than outdoor levels. As MNPs are common constituents of urban aerosol, they were most likely transported into the halls. Additional indoor sources are also plausible, including abrasion from shoes and clothing of players and spectators, as well as the use of plastic-based sports equipment, indicated by higher indoor MNP levels determined for both halls. Overall, the MNP concentrations measured both indoors and outdoors were approximately one or more orders of magnitude higher than values reported in the literature [63–66]. However, the referenced studies relied on aerosol measurement techniques (namely, either sampling methods or direct-reading instruments) with comparatively low collection or counting efficiencies for particles <10 µm, which likely explains this discrepancy. Accordingly, MNP air concentration might be systematically underestimated.

Finally, both halls were situated next to outdoor football fields with artificial turf and crumb rubber infills. Both could also have been sources of tyre-derived rubber particles carried inside. Particles originating from these sources would also most likely have been on the smaller side, since larger particles would not travel as far.

4.3. Fine dust

Because health-based air quality guidelines are defined for standardized exposure periods, short-term indoor exposure concentrations were converted to 24-h averages, allowing assessment of potential health risks in relation to cumulative daily

600 exposure rather than isolated peak concentrations alone. For this conversion, an exposure duration of 3 h indoors per day was
601 assumed, with outdoor/background exposure during the remaining 21 h.

602 In the hall with olive sand infill, estimated indoor PM₁₀ mass concentrations were substantially higher than outdoor levels,
603 reaching approximately 298.6 µg/m³ compared with 24.1 µg/m³ outdoors. This corresponds to an estimated 24-h average PM₁₀
604 concentration of 58.4 µg/m³, which is higher than the World Health Organization guideline value of 45 µg/m³ [67]. Despite
605 elevated estimated indoor PM_{2.5} concentrations (16.4 µg/m³ indoors versus 2.6 µg/m³ outdoors), the estimated 24-h average
606 PM_{2.5} concentration was 4.3 µg/m³ and therefore remained below the recommended limit of 15 µg/m³. However, WHO PM
607 guidelines are mass-based ambient standards for the general population under chronic exposure, derived from epidemiology on
608 combustion-related and urban PM. Here they serve only as benchmarks for contextualising the measured concentrations; they
609 are not used to assess whether the dust exposure in either hall posed an actual health risk.

610 In contrast, indoor 24-h averages for estimated PM₁₀ and PM_{2.5} concentrations in the hall with crumb rubber infill were lower
611 than the guideline values. PM₁₀ values were 18.8 µg/m³ indoors and 4.2 µg/m³ outdoors, and PM_{2.5} values were 0.8 µg/m³
612 indoors and 1.4 µg/m³ outdoors, resulting in 24-h averages of 6.0 µg/m³ and 1.3 µg/m³ for PM₁₀ and PM_{2.5}, respectively.
613 Notably, a converse trend was observed for indoor PM_{2.5}: it was lower than outdoor in this hall, which is consistent with the
614 high ventilation rate (Section 4.2) efficiently diluting indoor-generated fine particles.

615 Estimated outdoor concentrations for PM₁₀ and PM_{2.5} were lower than the most recently published year-average for urban
616 areas in Norway from 2024, at 15–25 µg/m³ (PM₁₀) and 6.1–6.6 µg/m³ (PM_{2.5}) [68].

617 **4.4. Presence of *Aspergillus* spores in the olive stone infill hall**

618 The term mould refers to fungi that produce hyphae and spores. Two mould taxa were identified in air samples collected
619 from the football hall containing organic infill of shredded olive stones and quartz sand: *Aspergillus glaucus* and a *Penicillium*
620 species, most likely *P. crustosum*. Their presence is relevant due to the potential health implications for players, facility
621 personnel, fans and visitors, including children and elderly persons.

622 *Aspergillus* is known for its environmental resilience and its ability to colonize dry, organic substrates, suggesting that
623 growth on olive stone-based turf infill is plausible, as already shown for other bio-based materials used in construction [27–
624 29]. In addition, *A. glaucus* is among several fungal species reported to degrade synthetic polymers through the secretion of
625 extracellular enzymes, highlighting its ability to utilise diverse carbon sources [69]. Although this does not imply degradation
626 of the rubber turf material in the present study, as it could not be determined in Hall 2, it further demonstrates the ecological
627 versatility of the species. Even though it generally poses limited risk to immunocompetent individuals, *A. glaucus* can cause

628 severe infections in immunocompromised persons and is frequently regarded as a bioindicator of indoor climate or moisture-
629 related problems.

630 The second identified species, *Penicillium crustosum*, is known to produce tremorgenic neurotoxins such as *penitrem A*.
631 While most documented intoxication cases are associated with ingestion, the health risks associated with inhalation exposure
632 are less clear, and assessing actual inhalation risk is beyond the scope of this study. The co-occurrence of *Aspergillus* and
633 *Penicillium* species in indoor environments typically indicates organic-rich and potentially moisture-affected conditions. This
634 underscores the importance of systematic monitoring of fungal contamination in sports facilities, though such spores have also
635 been reported to originate from air conditioning systems [70].

636 *Penicillium* species are frequently associated with lipid-rich substrates; therefore, colonization of the olive stone-based infill
637 is a theoretical possibility. Indeed, *P. crustosum* has been successfully cultivated on olive stone-derived substrates in bioprocess
638 research [71].

639 If olive stone powder becomes colonised by *P. crustosum* or other moulds, mechanical disturbance of the infill material may
640 facilitate spore detachment and aerosolization into dust and indoor air. This mechanism is well described in studies of fungal
641 ecology and indoor bioaerosol dynamics [72]. However, because our biological analysis was limited to a single airborne sample
642 filter (n=1) per hall, and the infill material was not directly sampled, further targeted investigation is required to determine
643 whether the infill acts as a local source for fungal growth.

644 Airborne fungal spore concentrations are typically expressed in CFU/m³ (colony-forming units per cubic metre). No health-
645 based limit values for exposure to airborne biological agents currently exist; to nonetheless assess the level of airborne exposure,
646 the German Technical Rules for Biological Agents (TRBA) no. 400 compares workplace concentrations with average
647 concentrations in ambient air [73]. A concentration greater than 100,000 CFU/m³ is classified as "high" exposure. It must be
648 emphasised, however, that we did not perform viable-spore (CFU) quantification: the TRBA 400 value is a viable-count
649 threshold, whereas our measurements are total particle counts, and the two cannot be directly compared. Viable concentrations
650 would need to be determined using dedicated protocols, e.g. those published by the German BAuA [74] or American CDC
651 [75]. With this caveat, total particle concentrations in the hall with olive sand infill exceeded the 100,000 units/m³ reference
652 level. If viable spore concentrations were of a comparable order of magnitude—which our data cannot confirm—an increased
653 health hazard could not be excluded, even for the short, intermittent exposure of a weekly football match, and would call for
654 hygiene measures such as improved cleaning and ventilation protocols. Given that an estimated 3–10 % of the population shows
655 heightened sensitivity or allergic responses to fungal allergens [76], sensitised individuals among the players could plausibly
656 experience health effects.

No spores were detected on the aerosol samples collected in the hall with crumb rubber infill. Although the two halls differed in several respects and no direct causal comparison can be drawn from a single facility each, this observation indicates that bio-based infills should not be presumed to be categorically safer than crumb rubber with respect to inhalative particle exposure. Their organic composition may support fungal colonization and thereby introduce an exposure pathway not present with synthetic infill, warranting dedicated assessment before such materials are adopted as a health-motivated substitute.

4.5. Limitations and Outlook

As with any study, several limitations should be considered when interpreting these findings. First, the results describe particle exposure rather than the resulting health risk. The analysis identified and quantified particles by size and chemical class but did not evaluate their toxicological properties, such as chemical leachability, oxidative potential, or biological response upon inhalation [77–79]. As risk is a function of both exposure and hazard, the present findings are best understood as an exposure assessment that provides a basis for subsequent toxicological evaluation; the term "potentially hazardous particles" used throughout refers to material identity and respirable size, not to a demonstrated adverse effect.

Second, the accessible size range was limited to particles larger than approximately 0.8 μm , below which particles could not be identified by Raman microscopy. Conclusions regarding number concentrations, size distributions, and source contributions therefore apply only to this range. This limit is not fundamental, however, and could be lowered in future work through improved μ -Raman instrumentation, stimulated Raman scattering, and the use of collection substrates with smaller pore sizes, extending the analysis into the submicron regime.

Third, the deployment of three parallel aerosol samplers per site was originally intended to provide analytical replicates. Theoretically, this would allow for the pooling of counts and the extrapolation of total particle concentrations assuming Poisson-distributed counting statistics. However, the count of coarse particles varied substantially between the filters, showing a high degree of sample heterogeneity. This observed variance is a known phenomenon in indoor air sampling, largely attributable to the spatial heterogeneity of coarse aerosols, localized air currents, and the incomplete mixing of mechanically generated particle plumes. Consequently, uncertainties in this study are reported as the standard error of the mean across replicates to accurately reflect this inherent environmental variability.

The most significant challenge, however, remains the spectroscopic overlap between infill-derived particles and compositionally similar background particles. In the hall with crumb rubber infill, the infill and a major outdoor source — tyre wear from road traffic — share the same base material and are therefore indistinguishable by their Raman spectra alone. One promising avenue for resolving this specific case is particle size: we hypothesize that tyre wear particles from outdoor sources are generally smaller than indoor-released crumb-rubber-derived particles, owing to different abrasion mechanics and the size-selective sedimentation that occurred during outdoor transport. This distinction warrants dedicated investigation.

687 More generally, this overlap problem arises wherever particles of interest and co-occurring background particles are
688 compositionally similar, e.g., when characterising microplastic fibres from synthetic textiles indoors, where shedding from
689 clothing, furnishings, and carpets may involve the same polymer (e.g., PET). Several complementary strategies could improve
690 source attribution in such cases. For sufficiently large particles, spatially resolved Raman mapping can reveal distinct material
691 components within a single particle rather than a superimposed bulk spectrum, though this depends on the spatial resolution of
692 our fibre-coupled μ -Raman system, i.e., the confocal laser excitation and Raman detection volume lateral diameter of about
693 300 nm and vertical extension of about 1000 nm at 532 nm. EDS mapping provides chemical information at considerably higher
694 lateral and comparable vertical resolution, and clustering algorithms that group spectra by similarity can delineate regions of
695 uniform composition within a particle. The combined morphological, Raman, and EDS information could ultimately be used
696 to train an artificial neural network capable of classifying both particle type and origin, which is an approach that would require
697 a reference library of particles with known type and provenance.

698 Despite these constraints, this study differentiates the particle exposure in indoor football halls by size and emission source.
699 This way it establishes a methodological basis and provides a reference dataset that provide a foundation for translating
700 exposure into risk.

701 **5. Conclusion**

702 This study provides an exposure-relevant characterization of airborne particles released during real football play on two
703 indoor artificial turf pitches, comparing a conventional crumb rubber infill with an olive sand infill. Using a correlative SEM–
704 μ -Raman workflow with AI-assisted segmentation, we identified single particles chemically and quantified the inhalable
705 fraction per particle class, including tyre-wear particles, which we distinguished by their characteristic carbon-black Raman
706 signature.

707 Our central finding is that inhalable particles were released during play in both environments. While our study cannot isolate
708 the infill as the sole driver of indoor air quality, the data from these case studies suggest that replacing crumb rubber with a bio-
709 based infill does not eliminate airborne particle exposure—rather, it alters the exposure profile. For example, the presence of
710 fungal spores (*Aspergillus glaucus*, *Penicillium crustosum*) alongside olive stone-related particles in the olive sand hall
711 indicates that organic bio-based infills can support microbial growth, adding an exposure pathway that synthetic infills typically
712 lack. Therefore, the assumption that a "greener" material is inherently more hygienic to breathe is not supported by our
713 observations.

714 Two considerations shape how these results should be read. First, the comparison is confounded and based on a small sample:
715 the halls differed in location, ventilation, and usage as well as infill, so observed differences cannot be attributed to the infill
716 alone. We therefore present these findings as pioneering case studies rather than a general comparison of infill types. Second,

717 we characterized exposure rather than hazard; measuring particle concentration, size, and chemistry, but not inhaled dose or
718 biological response. Our results should inform health risk assessment accordingly, not pre-empt it, by flagging which particle
719 classes warrant priority in future study.

720 **Declarations**

721 **Availability of data and materials**

722 The datasets used and/or analysed during the current study are available from the corresponding author on reasonable request.

723 **Competing interests**

724 The authors declare no conflict of interest.

725 **Funding**

726 This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant
727 agreement No 964766; Understanding human exposure and health hazard of micro- and nanoplastic contaminants in our
728 environment.

729 **Authors' contributions**

730 M.A., A.M.-P., I.S., H.D., R.H.H.P, B.G. and D.B. conceptualized the measurement campaign. M.A., I.S., H.D. and B.G.
731 secured access to the football halls for measurement and obtained consent from players. M.A., I.S., T.K.E., H.D. and B.G.
732 conducted the aerosol sampling. C.T. and N.D. performed the scanning electron microscopy of aerosol samples. T.P., B.L. and
733 A.M.-P. developed and trained the ANN model for particle analysis. M.S. performed the Raman microscopy of aerosol samples.
734 M.S., B.L. and D.B. analysed and interpreted the datasets acquired by correlative SEM/Raman microscopy. M.S. and D.B.
735 drafted the original manuscript, which was reviewed by B.L., M.A., I.S., H.D. and B.G and subsequently edited by M.S. and
736 D.B.. Supervision was provided by H.D., R.H.H.P, B.G. and D.B.. All authors read and approved the final manuscript.

737 **Ethics declaration**

738 The Football Field study was approved by the Regional Committees for Medical and Health Research Ethics Ref. 268504.

739 **Acknowledgements**

740 We would like to thank the students and employees of Lambertseter High School, Oslo, for their dedication and cooperation,
741 which made this study possible.

742 **List of abbreviations**

<u>Abbreviation</u>	<u>Full term</u>
ANN	Artificial neural network
Binary semantic segmentation	An image analysis method that classifies each pixel in one of two classes
CFU	Colony-forming units
CM	Correlative microscopy
D-peak	Defect peak of carbon structures observed in Raman spectroscopy
EDS	Energy dispersive X-ray spectroscopy
EMCCD camera	Electron multiplying charge-coupled device camera

G-peak	Graphitic peak of carbon structures observed in Raman spectroscopy
IOM	Institute of Occupational Medicine located in Edinburgh, Scotland
JRC	Joint Research Centre: European Commission's official science and knowledge service
MNPs	Micro- and nanoplastic particles
PC	Polycarbonate
PET	Polyethylene terephthalate
PM	Particulate matter
PM ₁₀	Particulate matter with aerodynamic diameter ≤ 10 micrometres
PM _{2.5}	Particulate matter with aerodynamic diameter ≤ 2.5 micrometres
POI	Particle of interest
RM	Raman microscopy
SD	Standard deviation
SE	Secondary electron
SEM	Scanning electron microscopy
SEM-Raman	Scanning electron microscopy combined with Raman spectroscopy
TRBA	German Technical Rules for Biological Agents
TWP	Tyre wear particle (usually originating from urban road traffic)
WHO	World Health Organization

743 References

- 744 1. Haan WP de, Quintana R, Vilas C, Cózar A, Canals M, Uviedo O, et al. The dark side of artificial greening: Plastic turfs
745 as widespread pollutants of aquatic environments. *Environ Pollut.* 2023;334:122094. doi:10.1016/j.envpol.2023.122094.
- 746 2. Duque-Villaverde A, Armada D, Dagnac T, Llompарт M. Recycled tire rubber materials in the spotlight. Determination
747 of hazardous and lethal substances. *Sci Total Environ.* 2024;929:172674. doi:10.1016/j.scitotenv.2024.172674.
- 748 3. Pronk MEJ, Woutersen M, Herremans JMM. Synthetic turf pitches with rubber granulate infill: are there health risks for
749 people playing sports on such pitches? *Journal of Exposure Science & Environmental Epidemiology.* 2020;30:567–84.
750 doi:10.1038/s41370-018-0106-1.
- 751 4. Ryan-Ndegwa S, Zamani R, Martins T. Exploring the Human Health Impact of Artificial Turf Worldwide: A Systematic
752 Review. *Environ Health Insights.* 2024;18:11786302241306291. doi:10.1177/11786302241306291.
- 753 5. Harusato A, Kato M. Mechanisms of Generation and Ecological Impacts of Nano- and Microplastics from Artificial
754 Turf Systems in Sports Facilities. *Environments.* 2025;12:109. doi:10.3390/environments12040109.
- 755 6. Foitzik M-J, Unrau H-J, Gauterin F, Dörnhöfer J, Koch T. Investigation of ultra fine particulate matter emission of
756 rubber tires. *Wear.* 2018;394-395:87–95. doi:10.1016/j.wear.2017.09.023.
- 757 7. Giechaskiel B, Grigoratos T, Mathissen M, Quik J, Tromp P, Gustafsson M, et al. Contribution of Road Vehicle Tyre
758 Wear to Microplastics and Ambient Air Pollution. *Sustainability.* 2024;16:522. doi:10.3390/su16020522.
- 759 8. Mayer PM, Moran KD, Miller EL, Brander SM, Harper S, Garcia-Jaramillo M, et al. Where the rubber meets the road:
760 Emerging environmental impacts of tire wear particles and their chemical cocktails. *Sci Total Environ.*
761 2024;927:171153. doi:10.1016/j.scitotenv.2024.171153.
- 762 9. Christou A, Giechaskiel B, Olofsson U, Grigoratos T. Review of Health Effects of Automotive Brake and Tyre Wear
763 Particles. *Toxics.* 2025;13:301. doi:10.3390/toxics13040301.
- 764 10. Panko JM, Chu J, Kreider ML, Unice KM. Measurement of airborne concentrations of tire and road wear particles in
765 urban and rural areas of France, Japan, and the United States. *Atmospheric Environment.* 2013;72:192–9.
766 doi:10.1016/j.atmosenv.2013.01.040.
- 767 11. Osman AI, Hosny M, Eltaweil AS, Omar S, Elgarahy AM, Farghali M, et al. Microplastic sources, formation, toxicity
768 and remediation: a review. *Environmental Chemistry Letters.* 2023;21:1–41. doi:10.1007/s10311-023-01593-3.
- 769 12. Oliveira CRS de, Da Silva Júnior AH, Mulinari J, Ferreira AJS, Da Silva A. Fibrous microplastics released from textiles:
770 Occurrence, fate, and remediation strategies. *Journal of Contaminant Hydrology.* 2023;256:104169.
771 doi:10.1016/j.jconhyd.2023.104169.
- 772 13. Zhang Y, Lu J, Wu J, Wang J, Luo Y. Potential risks of microplastics combined with superbugs: Enrichment of
773 antibiotic resistant bacteria on the surface of microplastics in mariculture system. *Ecotoxicology and Environmental*
774 *Safety.* 2020;187:109852. doi:10.1016/j.ecoenv.2019.109852.

14. Rai PK, Lee J, Brown RJ, Kim K-H. Micro- and nano-plastic pollution: Behavior, microbial ecology, and remediation technologies. *Journal of Cleaner Production*. 2021;291:125240. doi:10.1016/j.jclepro.2020.125240.
15. Prata JC. Microplastics and human health: Integrating pharmacokinetics. *Critical Reviews in Environmental Science and Technology*. 2023;53:1489–511. doi:10.1080/10643389.2023.2195798.
16. Geiser M, Kreyling WG. Deposition and biokinetics of inhaled nanoparticles. *Part Fibre Toxicol*. 2010;7:2. doi:10.1186/1743-8977-7-2.
17. Lazaridis M. Modelling approaches to particle deposition and clearance in the human respiratory tract. *Air Qual Atmos Health*. 2023;16:1989–2002. doi:10.1007/s11869-023-01386-1.
18. Kittner M, Coesfeld B, Werischak T, Schlau S, Altmann K, Kalbe U. Simultaneous sampling for microplastics and environmental contaminants from artificial turf: development of a new integrated microplastics eluate lysimeter. *Environ Sci Eur*. 2025;37:178. doi:10.1186/s12302-025-01235-1.
19. Xie Y, Li Y, Feng Y, Cheng W, Wang Y. Inhalable microplastics prevails in air: Exploring the size detection limit. *Environment International*. 2022;162:107151. doi:10.1016/j.envint.2022.107151.
20. Yang S, Brasseur G, Walters S, Lichtig P, Li CWY. Global atmospheric distribution of microplastics with evidence of low oceanic emissions. *npj Clim Atmos Sci*. 2025;8:81. doi:10.1038/s41612-025-00914-3.
21. Yang Z, Wang M, Feng Z, Wang Z, Lv M, Chang J, et al. Human Microplastics Exposure and Potential Health Risks to Target Organs by Different Routes: A Review. *Curr Pollution Rep*. 2023;9:468–85. doi:10.1007/s40726-023-00273-8.
22. Cumberbatch IS, Richardson L, Grant-Bier E, Kayali M, Mbithi M, Riviere RF, et al. Artificial Turf Versus Natural Grass: A Case Study of Environmental Effects, Health Risks, Safety, and Cost. *Sustainability*. 2025;17:6292. doi:10.3390/su17146292.
23. Salonen H, Salthammer T, Morawska L. Human exposure to air contaminants in sports environments. *Indoor Air*. 2020;30:1109–29. doi:10.1111/ina.12718.
24. Breuhauer K, Reese G. A problem to tackle: soccer players' awareness of microplastic infill in soccer fields predicts support for pro-environmental interventions. *Ger J Exerc Sport Res*. 2024;54:64–75. doi:10.1007/s12662-023-00920-8.
25. Zuccaro P, Thompson DC, Boer J de, Llompert M, Watterson A, Bilott R, et al. The European Union Ban on Microplastics Includes Artificial Turf Crumb Rubber Infill: Other Nations Should Follow Suit. *Environmental Science & Technology*. 2024;58:2591–4. doi:10.1021/acs.est.4c00047.
26. Massey R, Pollard L, Jacobs M, Onasch J, Harari H. Artificial Turf Infill: A Comparative Assessment of Chemical Contents. *New Solut*. 2020;30:10–26. doi:10.1177/1048291120906206.
27. Al Hallak M, Verdier T, Bertron A, Roques C, Bailly J-D. Fungal Contamination of Building Materials and the Aerosolization of Particles and Toxins in Indoor Air and Their Associated Risks to Health: A Review. *Toxins (Basel)*. 2023;15:175. doi:10.3390/toxins15030175.
28. Andersen B, Rasmussen TV. Biobased building materials: Moisture characteristics and fungal susceptibility. *Building and Environment*. 2025;275:112720. doi:10.1016/j.buildenv.2025.112720.
29. Wildman J, Shea A, Cascione V, Henk D. Mould susceptibility of bio-based insulation materials in modern construction. *npj Mater Degrad*. 2026;10:29. doi:10.1038/s41529-026-00742-7.
30. Meyer-Plath A, Bäger D, Dziurawitz N, Perseke D, Simonow BK, Thim C, et al. A Practicable Measurement Strategy for Compliance Checking Number Concentrations of Airborne Nano- and Microscale Fibers. *Atmosphere*. 2020;11:1254. doi:10.3390/atmos11111254.
31. Wright SL, Levermore JM, Kelly FJ. Raman Spectral Imaging for the Detection of Inhalable Microplastics in Ambient Particulate Matter Samples. *Environmental Science & Technology*. 2019;53:8947–56. doi:10.1021/acs.est.8b06663.
32. Schmidt R, Nachtnebel M, Dienstleder M, Mertschnigg S, Schroettner H, Zankel A, et al. Correlative SEM-Raman microscopy to reveal nanoplastics in complex environments. *Micron*. 2021;144:103034. doi:10.1016/j.micron.2021.103034.
33. Caldwell J, Loussert-Fonta C, Toullec G, Heidelberg Lyndby N, Haenni B, Taladriz-Blanco P, et al. Correlative Light, Electron Microscopy and Raman Spectroscopy Workflow To Detect and Observe Microplastic Interactions with Whole Jellyfish. *Environmental Science & Technology*. 2023;57:6664–72. doi:10.1021/acs.est.2c09233.
34. Schymanski D, Goldbeck C, Humpf H-U, Fürst P. Analysis of microplastics in water by micro-Raman spectroscopy: Release of plastic particles from different packaging into mineral water. *Water Research*. 2018;129:154–62. doi:10.1016/j.watres.2017.11.011.
35. Marina-Montes C, Pérez-Arribas LV, Anzano J, Vallejuelo SF-O de, Aramendia J, Gómez-Nubla L, et al. Characterization of atmospheric aerosols in the Antarctic region using Raman Spectroscopy and Scanning Electron Microscopy. *Spectrochim Acta A Mol Biomol Spectrosc*. 2022;266:120452. doi:10.1016/j.saa.2021.120452.
36. Broßell D, Visileanu E, Grosu C, Meyer-Plath A, Stange M. Correlating Scanning Electron Microscopy and Raman Microscopy to Quantify Occupational Exposure to Micro- and Nanoscale Plastics in Textile Manufacturing. *Pollutants*. 2026;6:6. doi:10.3390/pollutants6010006.
37. Statens vegvesen. Vegkart. 2026. <https://vegkart.atlas.vegvesen.no/>. Accessed 2 Sep 2026.

38. Kassu A, Farley C, Sharma A, Kim W, Guo J. Effect of Pore Size and Film Thickness on Gold-Coated Nanoporous Anodic Aluminum Oxide Substrates for Surface-Enhanced Raman Scattering Sensor. *Sensors (Basel)*. 2015;15:29924–37. doi:10.3390/s151229778.
39. Peters T, Schumann J, Kämpf K, Meyer-Plath A. AI-assisted recognition of fibre instances in scanning electron microscope images for aerosol characterization and exposure assessment. *Powder Technology*. 2026;469:121909. doi:10.1016/j.powtec.2025.121909.
40. Schumann J, Peters T, Schiweg H, Dziurawicz N, Thim C, Bäger D, et al. Automatisierung der Bestimmung der Asbestfaserkonzentration/Automation of the determination of the asbestos fibre concentration. *Gefahrstoffe Reinhaltung der Luft*. 2025;85:133–44. doi:10.37544/0949-8036-2025-07-08-7.
41. S.T. Japan GmbH. S.T. Japan Database. Deutschland: S.T. Japan-Europe GmbH; 2023.
42. Lafuente B, Downs RT, Yang H, Stone N. 1. The power of databases: The RRUFF project. In: Armbruster T, Danisi RM, editors. *Highlights in Mineralogical Crystallography*. Berlin, München, Boston: DE GRUYTER; 2015. p. 1–30. doi:10.1515/9783110417104-003.
43. CEN. DIN CEN/TS 18117:2025-04, Exposition am Arbeitsplatz - Nachweis und Charakterisierung von luftgetragenen NOAA durch Elektronenmikroskopie - Regeln für Probenahme und Analyse; Englische Fassung CEN/TS_18117:2025. Berlin: DIN Media GmbH. doi:10.31030/3545288.
44. Freedman D, Diaconis P. On the histogram as a density estimator: L_2 theory. *Z. Wahrscheinlichkeitstheorie verw Gebiete*. 1981;57:453–76. doi:10.1007/BF01025868.
45. Hinds WC. *Aerosol Technology: Properties, Behavior, and Measurement of Airborne Particles*. 2nd ed. Hoboken: Wiley-Interscience; 1999; 206–315.
46. Lunter D, Klang V, Kocsis D, Varga-Medveczky Z, Berkó S, Erdő F. Novel aspects of Raman spectroscopy in skin research. *Exp Dermatol*. 2022;31:1311–29. doi:10.1111/exd.14645.
47. Donaldson L. Autofluorescence in Plants. *Molecules*. 2020;25:2393. doi:10.3390/molecules25102393.
48. Zhang M, Feng Y, Li L, Zhang X, Xu F. Reducing fluorescence interference for improved Raman spectroscopic analysis of plant cell walls. *Wood Sci Technol*. 2024;58:1697–710. doi:10.1007/s00226-024-01587-6.
49. Yuniati Y, Elim PE, Alfanaar R, Kusuma HS, Mahfud. Extraction of anthocyanin pigment from *hibiscus sabdariffa* l. by ultrasonic-assisted extraction. *IOP Conf. Ser.: Mater. Sci. Eng*. 2021;1010:12032. doi:10.1088/1757-899X/1010/1/012032.
50. Qi J-S, Duan Y, Li Z-C, Gao J-M, Qi J, Liu C. The alkynyl-containing compounds from mushrooms and their biological activities. *Nat Prod Bioprospect*. 2023;13:50. doi:10.1007/s13659-023-00416-w.
51. Lamberth C. Alkyne chemistry in crop protection. *Bioorg Med Chem*. 2009;17:4047–63. doi:10.1016/j.bmc.2008.11.037.
52. Weinbruch S, Matthies J, Zou L, Kandler K, Ebert M, Bigalke M. Deposition rates and air concentrations of tire and road wear particles near a motorway in Germany. *Atmospheric Environment*. 2025;352:121228. doi:10.1016/j.atmosenv.2025.121228.
53. Elagami H, Ahmadi P, Fleckenstein JH, Frei S, Obst M, Agarwal S, et al. Measurement of microplastic settling velocities and implications for residence times in thermally stratified lakes. *Limnology & Oceanography*. 2022;67:934–45. doi:10.1002/lno.12046.
54. Gao Z, Cizdziel JV, Wontor K, Clisham C, Focia K, Rausch J, et al. On airborne tire wear particles along roads with different traffic characteristics using passive sampling and optical microscopy, single particle SEM/EDX, and μ -ATR-FTIR analyses. *Front. Environ. Sci*. 2022;10:1022697. doi:10.3389/fenvs.2022.1022697.
55. Park I, Kim H, Lee S. Characteristics of tire wear particles generated in a laboratory simulation of tire/road contact conditions. *Journal of Aerosol Science*. 2018;124:30–40. doi:10.1016/j.jaerosci.2018.07.005.
56. Rausch J, Jaramillo-Vogel D, Perseguers S, Schnidrig N, Grobety B, Yajan P. Automated identification and quantification of tire wear particles (TWP) in airborne dust: SEM/EDX single particle analysis coupled to a machine learning classifier. *Sci Total Environ*. 2022;803:149832. doi:10.1016/j.scitotenv.2021.149832.
57. Järnlskog I, Jaramillo-Vogel D, Rausch J, Gustafsson M, Strömvall A-M, Andersson-Sköld Y. Concentrations of tire wear microplastics and other traffic-derived non-exhaust particles in the road environment. *Environment International*. 2022;170:107618. doi:10.1016/j.envint.2022.107618.
58. Dye C, Bjerke A, Schmidbauer N, Manø S. Measurement of air pollutants in indoor artificial turf areas. NILU. 2006. <https://nilu.com/publication/9504/>.
59. Samland M, Badeke R, Grawe D, Matthias V. Variability of aerosol particle concentrations from tyre and brake wear emissions in an urban area. *Atmospheric Environment: X*. 2024;24:100304. doi:10.1016/j.aeaoa.2024.100304.
60. Qian J, Peccia J, Ferro AR. Walking-induced particle resuspension in indoor environments. *Atmospheric Environment*. 2014;89:464–81. doi:10.1016/j.atmosenv.2014.02.035.
61. Cao J, Huang H, Jiao R, Pei J, Xu Y, RuiRen, et al. The study of wear particle emissions of soft rubber on rolling contact under braking conditions. *Wear*. 2022;506–507:204431. doi:10.1016/j.wear.2022.204431.

62. Chen S, Zhang Q, Wu X, Lu Y, Wang F, Wei S, et al. Characterization of the abrasion behavior, particle morphology, and size distribution of rubber composites with various plasticizer structures. *Wear*. 2025;568-569:205904. doi:10.1016/j.wear.2025.205904.
63. Zhai X, Zheng H, Xu Y, Zhao R, Wang W, Guo H. Characterization and quantification of microplastics in indoor environments. *Heliyon*. 2023;9:e15901. doi:10.1016/j.heliyon.2023.e15901.
64. Perera K, Ziajahromi S, Nash SB, Leusch FDL. Microplastics in Australian indoor air: Abundance, characteristics, and implications for human exposure. *Sci Total Environ*. 2023;889:164292. doi:10.1016/j.scitotenv.2023.164292.
65. Jahandari A. Microplastics in the urban atmosphere: Sources, occurrences, distribution, and potential health implications. *Journal of Hazardous Materials Advances*. 2023;12:100346. doi:10.1016/j.hazadv.2023.100346.
66. Ministerio JRB, Amer NB, Romarate II RA, Bacosa HP. Atmospheric Microplastics in Indoor and Outdoor Air: Abundance and their Potential as Vectors for Heavy Metals. *Environ. Nat. Resour. J*. 2026;24:1–12. doi:10.32526/ennrj/24/20250103.
67. World Health Organization. WHO global air quality guidelines: particulate matter (PM_{2.5} and PM₁₀), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. World Health Organization. 2021.
68. IQAir. Air quality in Norway. IQAir. 2026. <https://www.iqair.com/qa/norway>. Accessed 2 Sep 2026.
69. Srikanth M, Sandeep TSRS, Sucharitha K, Godi S. Biodegradation of plastic polymers by fungi: a brief review. *Bioresour Bioprocess*. 2022;9:42. doi:10.1186/s40643-022-00532-4.
70. Luhung I, Bakker A, Angel DM, Sheu R, Gomez O, Siegel JA, et al. Fungal emissions from air conditioning cooling coils. *Indoor Environments*. 2026;3:100162. doi:10.1016/j.indenv.2026.100162.
71. Mumcu A, Deliboran A. Potential of food by-products. *Nor. Afr. J. Food Nutr. Res*. 2025;9:S1-S17. doi:10.51745/najfnr.9.S1.S1-S17.
72. Li X, Liu D, Yao J. Aerosolization of fungal spores in indoor environments. *Sci Total Environ*. 2022;820:153003. doi:10.1016/j.scitotenv.2022.153003.
73. Ausschuss für Biologische Arbeitsstoffe (ABAS). Technische Regeln für Biologische Arbeitsstoffe: TRBA 400: Handlungsanleitung zur Gefährdungsbeurteilung und für die Unterrichtung der Beschäftigten bei Tätigkeiten mit biologischen Arbeitsstoffen. 2017. <https://www.baua.de/DE/Angebote/Regelwerk/TRBA/TRBA-400>. Accessed 2 Sep 2026.
74. Ausschuss für Gefahrstoffe (AGS). Technische Regeln für Gefahrstoffe 900: Arbeitsplatzgrenzwerte. 2006. <https://www.baua.de/DE/Angebote/Regelwerk/TRGS/TRGS-900>. Accessed 2 Sep 2026.
75. Blachere FM, Green BJ, Jensen PA, Law BF, Lindsley WG, Martin Jr SB, et al. Sampling and characterization of bioaerosols. 2017/03/01. <https://stacks.cdc.gov/view/cdc/210966>.
76. Hughes KM, Price D, Torriero AAJ, Symonds MRE, Suphioglu C. Impact of Fungal Spores on Asthma Prevalence and Hospitalization. *Int J Mol Sci*. 2022;23:4313. doi:10.3390/ijms23084313.
77. Jeong E, Lee J-Y, Redwan M. Animal exposure to microplastics and health effects: A review. *Emerging Contaminants*. 2024;10:100369. doi:10.1016/j.emcon.2024.100369.
78. Li Y, Le Tao, Wang Q, Wang F, Li G, Song M. Potential Health Impact of Microplastics: A Review of Environmental Distribution, Human Exposure, and Toxic Effects. *Environ Health (Wash)*. 2023;1:249–57. doi:10.1021/envhealth.3c00052.
79. Prata JC, Da Costa JP, Lopes I, Duarte AC, Rocha-Santos T. Environmental exposure to microplastics: An overview on possible human health effects. *Sci Total Environ*. 2020;702:134455. doi:10.1016/j.scitotenv.2019.134455.

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- [SupplementalInformation.docx](#)