

1       **A novel method to collect and chemically characterize milligram**  
2                   **quantities of airborne indoor particulate matter**

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11  
12       **Abstract**

13  
14       Because people spend the majority of the day indoors, indoor air quality, especially  
15       airborne particulate matter (PM), is important to evaluate for its potential health effects.  
16       Collection of milligram-sized samples of indoor PM is necessary for detailed chemical and  
17       biological assays, but is challenging because of the device noise levels, power requirements, and  
18       size of traditional PM samplers. To overcome this need, a novel method for the collection of  
19       indoor PM was developed using an electrostatic precipitator (ESP). Laboratory experiments were  
20       conducted to characterize ESP collection efficiency (41-65%) and PM recovery (50-95%) for  
21       three aerosol types. After characterization, the ESPs were deployed in 21 homes in eastern Iowa  
22       over 30 days and collected 6-87 mg of indoor PM. Samples were acid digested and subsequently  
23       analyzed by inductively coupled plasma mass spectrometry for magnesium, aluminum, vanadium,  
24       manganese, iron, nickel, copper, zinc, arsenic, and lead. Crustal metals (magnesium, iron, and  
25       aluminum) had the largest PM mass fractions, ranging from 3,000-25,000 ng mg<sup>-1</sup>. The relative  
26       abundance of metals was similar between homes, although the PM mass fractions were highly  
27       variable. The ESP sampling method can be applied in future studies to collect milligram-sized  
28       quantities of indoor PM for detailed studies of its composition and potential health effects.

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## 35 INTRODUCTION

36

37 The adverse health effects of exposure to ambient particulate matter (PM) depend on its  
38 chemical composition and physical characteristics (Kim et al., 2015). Essential elements like iron  
39 can support the growth of bacteria in the lung, contributing to bacterial respiratory infections due  
40 to reduced antimicrobial function that hinder airway innate immunity (Bullen et al., 2005; Ghio,  
41 2009). Metals such as nickel, lead, and copper cause inflammatory stress, while vanadium and  
42 manganese are toxic and have adverse effects on the cardiovascular system (Duvall et al., 2008;  
43 Antonini et al., 2010; Cortijo et al., 2010). Indoor PM is particularly important to study because  
44 humans spend the majority of their time indoors; Americans, for example, spend an average of  
45 80% of their time indoors (Wallace, 2012). Indoor PM has been attributed to a multitude of  
46 sources, including cooking, home heating, deterioration of building furnishings, and particles  
47 entrained from outdoors (Jones, 1999; Zhao et al., 2006; Cortez-Lugo et al., 2008; Nazaroff,  
48 2018). However, evaluating the chemical, physical, and biological properties of PM that can  
49 affect human health generally requires at least 4 mg of material (McGee et al., 2003). Finding a  
50 means of affordable, inconspicuous collection of milligram-quantities of indoor PM is the focus  
51 of this study.

52 To collect and analyze suspended indoor PM, filter-based samplers or exposure monitors  
53 are typically used, because they provide particle collection with high efficiency and can  
54 accommodate large air flows, enabling collection of large sample masses (Kamens et al., 1991;  
55 Chow and Watson, 1998; Yanosky et al., 2002; Kalogerakis et al., 2005). However, strong air  
56 pumps are needed to overcome the pressure drops across filters, making these samplers noisy and  
57 requiring large amounts of electricity. The use of smaller pumps reduces noise and electricity  
58 consumption but decreases airflow rates and collects smaller quantities of PM (typically less than  
59 1 mg). Vacuum cleaner methods have been used to collect settled dust with particle size

60 resolution, but vacuums are particularly noisy and settled dust may not best represent inhalable  
61 particles (Cox et al., 2017).

62 An attractive option is the use of electrostatic precipitators (ESPs), which have been used  
63 in industry as large-scale air purification devices. Power plants use ESPs as pollution control  
64 devices to reduce particle emissions (Qi and Yuan, 2011). Due to their effectiveness at collecting  
65 aerosols, ESPs have been adapted for airborne particle sampling. The success in PM collection  
66 has prompted NIOSH (The National Institute for Occupational Safety and Health) to develop a  
67 handheld ESP to collect PM onto substrates for microscopy (Miller et al., 2010). Sample  
68 collection times in this application are very short (~1 minute) to avoid overloading of the  
69 substrate (Miller et al., 2010). ESPs used for aerosol sampling in current literature do not use its  
70 high flow sampling capabilities, and thus do not collect large quantities of PM. Since ESP  
71 devices adapted for indoor PM sampling produce little noise, long-term sampling (30 days per  
72 the manufacturer's recommend length of use for optimal particle collection) can be achieved in a  
73 non-invasive way (Bacchiega et al., 2006).

74 The aims of this study are to 1) develop a method of collecting milligram quantities of  
75 indoor PM using an ESP that is not disruptive to the indoor environment; 2) deploy such ESPs  
76 for collection of indoor PM in a cohort study; and 3) evaluate the metal content of the collected  
77 PM. Our overarching goal is to collect sufficiently large samples of indoor PM to enable  
78 characterization by chemical and biological assays, and thereby better understand the role of  
79 indoor PM exposures on respiratory health.

80

## 81 **MATERIALS AND METHODS**

82

### 83 *Collection of indoor PM via electrostatic precipitation*

84

85 The O-Ion B-1000 ESP was selected from 4 ESPs (Table S1) as a means of collecting  
86 total suspended particles (TSP) because of the simplicity of harvesting PM from the collection  
87 electrode. The O-Ion B-1000 is also inexpensive, small (0.43 m × 0.14 m × 0.17 m) and portable,  
88 making it easily deployable in environments with limited space. The interior portion of this ESP  
89 is shown schematically in **Fig 1**. The collection electrode is an elongated rectangle with concave  
90 sides. A wire electrode is held 7 mm from the center of the face of the collection electrode. Air is  
91 pulled through the unit with a squirrel cage blower located in the top of the ESP. It passes  
92 through slots in the lower portion of the outside of the housing, through the area with the wires  
93 and collection electrode, and finally through a small gap at the top of the ESP.

94 The carbon filter and the UV-light at the exit of the commercial device were removed  
95 prior to sampling. For low and high settings on the ESP, the velocity of the air was measured  
96 with a thermal anemometer (VelociCalc 8360, TSI Inc., Shoreview, MN, USA) in 6 equally-  
97 spaced locations in the plane of the ESP outlet. Airflow rate was calculated as the mean air  
98 velocity multiplied by the outlet area (low setting = 128 L min<sup>-1</sup>; high setting = 170 L min<sup>-1</sup>). At 0,  
99 5, and 10 cm from the outlet of ESP, ozone concentration was measured with an ozone sensor  
100 (Portasense 2 gas leak detector, Analytical Technology Inc., Collegeville, PA, USA) and noise  
101 was measured at the outlet with a sound level meter (SoundTrack LxT, Larson Davis, Depew,  
102 NY, USA), each in triplicate.

103

#### 104 ***Determination of particle collection efficiency***

105 The O-Ion B-1000 collection efficiency was characterized as a function of particle size for  
106 three polydisperse aerosols: sodium chloride (NaCl), Arizona road dust (ARD), and diesel fumes.  
107 NaCl was selected as a common inorganic salt that has high water solubility (Sandu et al., 2010).  
108 NaCl aerosol was generated at fine and ultrafine particle sizes (< 1 μm) by spraying 0.9% NaCl

109 irrigation solution (2F7123, Baxter Healthcare Co., Deerfield, IL) with a vibrating mesh  
110 nebulizer (Aeroneb Solo System, Aerogen, Ireland) followed by drying with a silica dryer. Diesel  
111 fumes were chosen to represent combustion particles that have low water solubility (Figler et al.,  
112 1996). The diesel fume particles were generated at fine and ultrafine particle sizes by a diesel  
113 generator (DG6LE, Red Hawk Equipment LLC, Akron, NY, USA). ARD consisted primarily of  
114 coarse particles ( $> 1\mu\text{m}$ ) and was selected to represent a crustal dust that may enter homes  
115 through resuspension of soil tracked in from outdoors or from intrusion of windblown dust  
116 (Curtis et al., 2008). A fluidized bed aerosol generator (3400A, TSI Inc., Shoreview, MN, USA)  
117 was used to aerosolize ARD (Fine Grade, Part No. 1543094., Powder Technology Inc., Arden  
118 Hills, MN, USA). Together, these three aerosol types span a broad range of aerosol sizes,  
119 chemical composition, and degree of water-solubility.

120 Collection efficiency was determined experimentally (Fig. 2). Aerosols were injected into  
121 the mixing chamber ( $0.64\text{ m} \times 0.64\text{ m} \times 0.66\text{ m}$ ) and diluted with clean air that had been passed  
122 through two high efficiency particulate air filters. A condensation particle counter (CPC) (Model  
123 3007, TSI Inc, Shoreview, MN, USA) was used to monitor the particle concentration in the  
124 chamber. Size-resolved aerosol number concentrations were measured with a scanning mobility  
125 particle sizer (SMPS, composed of an Electrostatic Classifier Model-3082 and a Nano Water-  
126 Based Condensation Particle Counter Model-3788, TSI Inc., Shoreview, MN, USA) for particles  
127 smaller than  $1\mu\text{m}$  and an aerodynamic particle sizer (APS, Model3321, TSI Inc. Shoreview, MN,  
128 USA) for particles larger than  $1\mu\text{m}$ . Size-resolved aerosol number concentrations were measured  
129 upstream of the ESP with the electric field off, downstream of the ESP with the electric field off,  
130 and downstream of the ESP with the electric field on.

131 Collection efficiency was calculated from the number concentration downstream of the  
132 ESP with electric field turned off ( $C_{\text{down, off}}$ ) and turned on ( $C_{\text{down, on}}$ ) following Eq. (1):

133

$$\text{Collection Efficiency} = 1 - \frac{C_{down,on}}{C_{down,off}} \quad (1)$$

134

### 135 ***Recovery of PM from the ESP electrode***

136 PM was collected from the ESP electrode using a filter wiping method described by  
137 Afshar-Mohajer et al. (2017) in which polyvinyl chloride (PVC) filters wetted with isopropanol  
138 were used to wipe PM from an ESP collection electrode (Afshar-Mohajer et al., 2017). Because  
139 isopropanol oxidizes the surface of the electrode (O-Ion Technologies 2016) and reduces its  
140 collection efficiency, isopropanol was replaced with deionized (DI) water in this study. Filters  
141 were pre- and post-weighed using a microbalance (MT-5, Mettler-Toledo, Columbus, OH, USA)  
142 in a temperature (21-24 °C) and humidity-controlled (40-41% RH) room to determine the mass of  
143 PM collected from the ESP.

144 Filters wetted with one mL of DI water were used to wipe the electrode in an up-and-  
145 down motion for a total of four cycles on each of the four sides or until no observable PM  
146 remained. A minimum of two filters were used on each side of the electrode. After drying for 48  
147 hours, filters were re-weighed. Eight laboratory blanks were collected by wiping a clean ESP  
148 (four before deployment, four after deployment).

149 The efficiency of particle recovery from the electrode was determined in the laboratory  
150 experiments described in section 2.1b, each in triplicate. Particles were collected onto a pre-  
151 cleaned ESP electrode for 30 minutes, while PM mass was simultaneously collected on 37-mm  
152 PVC filters (FPVC537, Zefon International, Inc., Ocala, FL, USA) in a cassette sampler  
153 operating at an airflow of 10 L/min (Gilian 12, Sensidyne, LP, St. Petersburg, FL, USA). PM from  
154 the ESP electrode was collected by the wetted filter method. Particle recovery (R) was calculated

155 as the ratio of the mass of PM recovered from the ESP electrode ( $m_{ESP}$ ) and the expected PM  
156 mass measured in the chamber following Eq (2):

157

$$R = \frac{m_{ESP} \times Q_{37}}{Q_{ESP} \times m_{37} \times CE_{ESP}} \quad (2)$$

158

159 where  $m_{37}$  is the mass collected from the chamber on a 37 mm PVC filter,  $Q_{37}$  is the flow rate for  
160 the cassette sampler,  $Q_{ESP}$  is the flow rate of ESP device, and  $CE_{ESP}$  is the collection efficiency of  
161 the ESP device. The collection efficiency of the cassette sampler is assumed to be 1.

162

### 163 ***Field deployment***

164 ESPs were deployed in the homes of 21 former and current smokers with and without  
165 chronic obstructive pulmonary disease (COPD) for 30 days. The ESP collection device was  
166 transferred to patients at a clinic visit during which they received training on its installation and  
167 operation. Surveys were used to collect data from study participants about frequency and  
168 methods of cleaning and cooking, cigarette smoking, candle use, home heating methods, pets,  
169 home renovation projects, and square footage of the home. Patients installed the device in their  
170 home in the room where they spend the most time, generally in the living room or bedroom.  
171 Patients then returned the ESP to the clinic after the 30 day collection period. Recovery of  
172 collected material required eight filters, except in the case of one heavily loaded ESP that  
173 required nine. Two filters were used on each of the four sides of each collection electrode. The  
174 amount of PM collected during the field sampling was measured gravimetrically. PM mass  
175 measurements were not corrected for particle collection efficiency or recovery from the electrode.

176

### 177 ***Sample digestion and metals analysis***

178

179 To evaluate metals in indoor PM, PM samples recovered from the ESPs were digested in  
180 acid and analyzed by inductively coupled plasma mass spectrometry (ICP-MS). Filters were  
181 placed in a modified polytetrafluoroethylene (PTFE-TFM) vessel with 6 mL of concentrated  
182 nitric acid (68-70 % w/w Fisher), 1 mL of hydrogen peroxide (30% w/w Sigma-Aldrich), and a  
183 50  $\mu\text{gL}^{-1}$  indium (Perkin Elmer) internal standard spike. Once submerged in the acid matrix, the  
184 filters were digested with microwave assistance (Multiwave Go, Anton Parr). The microwave  
185 program began with a 30-minute ramp to 200°C followed by a 30-minute hold, following a  
186 modified version of EPA method 3051 that provides greater recoveries of metal analytes (Hewitt  
187 and Cragin, 1991; Hassan et al., 2006). A 50% v/v nitric acid solution was used to quantitatively  
188 transfer solutions, and a 2% nitric acid solution was used to reconstitute samples to 10 mL. A  
189 1:10 dilution in 2% nitric acid was performed prior to instrumental analysis. The accuracy and  
190 precision of the extraction method were determined using a standard reference material of indoor  
191 PM (NIST 2584 trace elements in indoor dust).

192 Metals were analyzed using an ICP-MS (Agilent 7900) equipped with a collision cell to  
193 reduce interference from other ions in the solution. An 11-point calibration curve was prepared  
194 from a multi-element standard (QCP-QCS-3 Inorganic ventures) spanning 0.1-500  $\mu\text{g/L}$ , with an  
195 indium internal standard concentration of 50  $\mu\text{gL}^{-1}$ . Analyte masses and the order of samples for  
196 analysis followed National Environmental Methods Index (NEMI) standard method 3125 to  
197 account for memory effects, interferences, and matrix effects (NEMI, 2018). Before each batch of  
198 sample analysis, the instrument was stabilized and tuned to achieve <10% RSD for all analytes of  
199 interest.

200 Accuracy and precision of digestion methods were evaluated using spike recovery  
201 experiments. Reference material (NIST 2584 “Trace elements in indoor dust”) was spiked onto



202 three blank PVC filters and digested alongside three blank PVC filters. Spike recoveries were  
203 calculated using Eq (3).

204

$$\text{Recovery \%} = \frac{(C_{\text{recovered}} - C_{\text{blank}})}{C_{\text{spiked}}} * 100 \quad (3)$$

205

## 206 **Results and Discussion**

207

### 208 *Characterization of ESP Performance*

209

210

211 From the four ESPs considered for the collection of indoor PM (Table S1), the O-Ion B-  
212 1000 was the selected because of its small size, low-cost, removable carbon filter and UV-C light,  
213 quiet fans, and easily removable collection electrode. The noise level generated by the ESP on  
214 the high setting was  $45.8 \pm 0.2$  dB and on the low setting was  $43.1 \pm 0.4$  dB (Table S2),  
215 comparable to a small box-fan. The noise levels were near the EPA suggested indoor-  
216 environment noise levels ( $\sim 45$  dB) that permit spoken conversation and allow for sleeping and  
217 recreational activities (Fink, 2017). The noise levels were well below the 70 dB limit that when  
218 exceeded over 24 hours, is likely to produce measurable hearing loss over a lifetime.

219 A problem with using an ESP for sampling is that any use of electrostatics will produce ozone, a  
220 chemical species known to cause airway inflammation, from surrounding oxygen in the air  
221 (Cardello et al., 2002; Lippmann, 2012). It was thus important to determine whether the O-Ion-  
222 B-1000 would generate harmful levels of ozone. Ozone production on the low setting of the ESP  
223 device was 0.04 ppm at the outlet, 0.04 ppm 5 cm from the outlet, and nil at 10 cm, meaning at  
224 all distances ozone concentration was below the EPA standard of 0.07 ppm (Table S3). When  
225 switched to the high setting 0.12 ppm ozone was produced at the outlet, 0.088 ppm 5 cm away,  
226 and reduced to 0.047 ppm 10 cm from the outlet (Table S3). The greater airflow rates on the  
higher ESP setting resulted in three times the amount of ozone produced at the outlet and twice as

227 much ozone produced 5 cm from the outlet. The removal of the carbon filter provided with the O-  
228 Ion B device slightly increased the amount of ozone produced, though each setting and distance  
229 produced ozone concentrations within 0.01 ppm of that measured with the carbon filter (Table  
230 S4). Since flow rate has been shown to be important with ozone production, removal of the  
231 carbon filter likely provides a small increase in flow rate which can be attributed to the  
232 consistently increased ozone concentration. The limit of ozone exposure set from National  
233 Ambient Air Quality Standards (NAAQS) is 0.070 ppm over 8 hours, and exceedance of this  
234 concentration can cause various health effects: coughing, deep pain in the respiratory tract,  
235 shortness of breath, and increased susceptibility to lung disease (Planning, 1982; Lippmann,  
236 2012). With sufficient and typical household ventilation, the ESP will not generate enough ozone  
237 to exceed the NAAQS limit and should not be a problem in field deployment of the device.

#### 238 ***Collection efficiency of the ESP***

239 The collection efficiency of the ESP was characterized by using three aerosol types  
240 characteristic of soluble salts, combustion particles, and airborne crustal matter, encompassing a  
241 range of chemical species likely to be present in the indoor environment (Fig. 3). From 40-400  
242 nm, collection efficiencies of NaCl, diesel, and ARD were similar at 45%. This finding is  
243 consistent of typical ESP efficiencies, where the efficiency is commonly lowest between about  
244 0.1  $\mu\text{m}$  and 1  $\mu\text{m}$  (Mizuno, 2000). Neither diffusion charging (random collisions that induce  
245 charge) nor field charging (ions travel along electric field lines and intersect with particles)  
246 dominate for particles in this size range. Consequently, the system produces fewer charged  
247 particles and collection efficiency is reduced (Hinds, 1999; Hinds, 2004; Miller et al., 2010). For  
248 particles larger than 400 nm, the efficiency of the ARD increased with particle size as field  
249 charging became the predominant charging mechanism. With field charging, increasingly large  
250 particles achieve greater electrical mobility toward the electrode (Hinds, 2004).

251 For particles smaller than 100 nm, the predominant mode of particle deposition is  
252 diffusion charging (Miller et al., 2010). With diffusion charging, the migration velocity in an ESP  
253 is proportional to the Cunningham correction factor, a correction to predict drag force for smaller  
254 particles (Mizuno, 2000). The collection efficiency for NaCl particles smaller than 40 nm  
255 increased with decreasing particle size, due to diffusion charging being the predominant mode of  
256 particle charging resulting in migration to the collection electrode. The difference in efficiency  
257 for ARD and NaCl particles larger than 400 nm may be due to NaCl having a higher electrical  
258 resistivity than ARD, although their respective electrical resistivities are not known. The higher  
259 resistivity would make it more difficult to collect particles, resulting in a lower collection  
260 efficiency.

261 Over the full range of particle sizes generated, the average collection efficiencies were  
262 43% for NaCl, 41% for diesel fumes, and 59% for ARD (Table S5). Collection efficiencies in  
263 this range are typical for an ESP and suggest a slight preference for the collection of large  
264 particles (> 400 nm) by field charging over small particles (< 100 nm) by diffusion charging.

265

### 266 ***PM Recovery***

267 Particle recovery from the collection electrode was evaluated for the three aerosol types.  
268 Wetting the PVC filter with DI water resulted in a 21% increase of ARD recovered from the  
269 collection electrode compared to wiping with a dry PVC filter (Table S6). In order to remove all  
270 visible PM from one collection electrode, three wet filters or four dry filters were required. The  
271 first filter used to clean each side of the electrode generally had a higher mass, with subsequent  
272 filters collecting less material.

273 Using this method, NaCl had the greatest recovery of  $95\pm 5.8\%$ , followed by ARD at  
274  $63\pm 1.9\%$ , and then diesel fumes at  $48\pm 11.2\%$  (Table S5). The high recovery of NaCl is attributed

275 to its high water solubility, which made it easier to remove the particles from the collection  
276 electrode. ARD is partially water soluble, making the particles easier to remove. ARD is also  
277 composed of primarily large particles, which can be easily dislodged from a surface. Thus, the  
278 lower than anticipated particle recovery for ARD may be attributed to particles falling off the  
279 electrode during the filter wipe process. ARD was also the most consistent recovery, with a  
280 standard deviation of 1.9%. The lower recovery of diesel PM is likely attributed to diesel fumes'  
281 being insoluble in water, which made it difficult to recover as the particles became viscous.  
282 Overall, the recovery data suggest that wiping the electrode with filters wetted with water  
283 preferentially recovers water-soluble species and is less effective at removing low-solubility  
284 combustion particles.

285

#### 286 ***Limitations of the ESP collection method for indoor PM***

287 While capable of collecting milligram quantities of airborne PM indoors, there are notable  
288 limitations of the ESP method. There is a preference towards the collection of coarse particles  
289 (2.5-10 microns) like crustal materials relative to smaller, potentially more toxic, fine particles (<  
290 2.5 microns) that arise from combustion. Meanwhile, the collection efficiency for particles >10  
291 microns is not known because it was not accessible by the particle measurement devices used for  
292 ESP characterization. Because there is no particle sizing upstream of the collection electrode,  
293 total suspended particles (TSP) are collected. Consequently, there is potential for a small number  
294 of large particles to have a substantial impact on the measured particle mass. Also, there is the  
295 preference towards the collection of recovery of water-soluble species from the electrode surface  
296 relative to low-solubility combustion particles, resulting in chemical differences in the collected  
297 PM relative to the recovered PM. The characterization studies conducted herein provide some

298 bounds on the extent to which collection efficiency and particle recovery vary across different  
299 particle sizes and types.

300

### 301 ***Field results***

302 The average mass of indoor PM collected was 20.8 mg with a standard deviation of 17.3  
303 mg, a range of 6.36-86.7 mg, and propagated error ranging from 0.008-0.013 mg (Fig. 4). There  
304 was no significant relationship between the square footage of the home and the mass of dust  
305 collected ( $r = -0.27$ ,  $p = 0.30$ ). High PM masses generally corresponded to homes in which  
306 tobacco is smoked and with a high frequency of vacuuming, although the significance of these  
307 trends could not be established due to the low sample numbers in this study. The variability in  
308 PM collected is attributed to indoor PM loadings that vary with building furnishings, airflow  
309 pathways, ventilation, number of occupants, number of pets, frequency of cleaning, open or shut  
310 windows, and household activities such as cleaning or combustion processes (cooking, heating,  
311 smoking, candle burning, etc.).

312

### 313 ***Determination of Metals in Indoor PM***

314 Three methods of digesting PVC filters for analysis of total metals were examined,  
315 including use of hot plates with two acid matrices and microwave-assisted digestion (Fig. S1).  
316 Hot plate digestion with an acid matrix of 5 mL concentrated nitric acid (68-70% w/w) and 1 mL  
317 concentrated hydrofluoric acid (49% w/w) yielded recoveries ranging 16-93%. Recoveries were  
318 improved to 82-98% (except for arsenic at 8%) when hydrogen peroxide was added to aid in the  
319 oxidation of organic material and more complete digestion of the sample (Mesko et al., 2011).  
320 With microwave digestion, recoveries ranged from 80-98%. The microwave-assisted method  
321 generally provided more accurate and precise (RSD= 6.9%) spike recoveries (Fig. S2) and a

322 much shorter digestion time (2 hours as opposed to 72 hours with a hot plate). Overall, the  
323 microwave digestion system was preferred for its efficiency, consistency, and capability to digest  
324 material without the use of hydrofluoric acid.

325 Metals were analyzed in the indoor PM collected from 21 homes of smoker subjects with  
326 and without COPD. Two of the eight PVC filters were digested from each household, along with  
327 eight field blanks. Weighted average mass concentrations were calculated to account for the  
328 differences in PM loadings across filters (Table S7). The most abundant metals observed were  
329 magnesium, aluminum, and iron; these metals were almost an entire order of magnitude more  
330 concentrated than were the other species (Fig. 5). This trend was also observed in prior studies  
331 (Rasmussen et al., 2001) and attributed to these being crustal metals commonly found in soil that  
332 can penetrate the indoor environment via windblown dust or adhering to materials entering the  
333 home (Curtis et al., 2008). Zinc, manganese, lead, and copper were also detected in each  
334 household sample at lower mass fractions. Zinc and copper are frequently found in building  
335 furnishings, and in cooking tools (See and Balasubramanian, 2008), while arsenic is associated  
336 with pesticides, but likely has additional sources. The mass fractions of metals in indoor PM were  
337 highly variable across homes, though their relative abundances were similar (Fig. 5).

338 The NIST 2584 standard reference material (SRM) can be used to compare the metal  
339 mass fraction values of these studies findings to a well-characterized sample of collected indoor  
340 PM. In the case of lead the 21 households in eastern Iowa averaged  $91.2 \text{ ng mg}^{-1}$ , while the NIST  
341 reference material had was  $9761 \text{ ng mg}^{-1}$ . (In this study, none of the homes had lead-based paint,  
342 while all of those used to collect the NIST reference material did.) While the most abundant  
343 metals observed in this study were  $\text{Al} > \text{Mg} > \text{Fe}$ , they were  $\text{Al} > \text{Fe} > \text{Mg}$  in the NIST reference  
344 standard; this difference is likely due to variations in crustal composition from different  
345 geological locations. Nickel and zinc, for instance, have much lower abundance in eastern Iowa

346 than do the reference material mass fractions, while arsenic, copper, manganese, and vanadium  
347 levels are all comparable to the corresponding NIST 2584 values.

348 The relative abundance of metal species observed in this study is consistent with prior  
349 research on indoor PM. Indoor dust ranging 100-250  $\mu\text{m}$  was collected from 48 homes in Ottawa,  
350 Ontario by vacuum cleaner method followed by sieving, and the reported metals included  
351 analytes analyzed in Eastern Iowa. The median mass fractions of metals in this study (Table S8)  
352 agreed with those from Ottawa: aluminum was the most abundant (22900  $\text{ng mg}^{-1}$ ), followed by  
353 iron (13150  $\text{ng mg}^{-1}$ ), magnesium (9285  $\text{ng mg}^{-1}$ ), zinc (633.1  $\text{ng mg}^{-1}$ ), manganese 266.5 ( $\text{ng mg}^{-1}$ )  
354  $^1$ ), lead (222.22  $\text{ng mg}^{-1}$ ), copper (157.30  $\text{ng mg}^{-1}$ ), nickel (51.5  $\text{ng mg}^{-1}$ ), vanadium (22.0  $\text{ng}$   
355  $\text{mg}^{-1}$ ), and arsenic (4.1  $\text{ng mg}^{-1}$ ) (Rasmussen et al., 2001). Most of the Ottawa mass fractions are  
356 within the 95% confidence interval of the mean in this study (Table S8), although magnesium  
357 was more abundant in eastern Iowa (15000  $\text{ng mg}^{-1}$ ). In another study, indoor  $\text{PM}_{2.5}$  and  $\text{PM}_{10}$   
358 samples were collected in Windsor, Ontario and mass fractions of metals were reported from 121  
359 homes (Rasmussen et al., 2018). While total suspended dust ( $\text{PM}_{10-2.5}$ ) reported in Windsor,  
360 Ontario was in good agreement with eastern Iowa ESP results, the Windsor  $\text{PM}_{2.5}$  results reported  
361 much lower mass fractions of aluminum, magnesium, and iron, suggesting crustal metals are  
362 primarily found in settled dust (Rasmussen et al., 2018). Altogether, these results indicate that the  
363 ESP collection of TSP in this study agrees with vacuum cleaner  $\text{PM}_{10}$  sampling methods in that  
364 particle chemistry is dominated by crustal elements.

365

## 366 **Conclusions**

367 The O-Ion B ESP device is suitable for collection of indoor particulate matter (TSP) of  
368 milligram quantities of PM for further characterization. In our field application, the ESP collected  
369 6-86 mg of indoor PM over 30 days in 21 homes. Metals analysis required only 0.6-3.5 mg of

370 sample, with plentiful sample remaining for further characterization studies. In addition to being  
371 able to collect large masses of indoor PM, the ESP has the advantages of being quiet and  
372 deployable for up to 30 days at a time. While the ESP produces low quantities of ozone, these  
373 levels are not expected to be problematic in homes with suitable ventilation. As demonstrated in  
374 our characterization studies, the ESP is more efficient at collecting particles > 400 nm,  
375 particularly those that are conductive. Crustal material and soluble salts were most efficiently  
376 recovered from the electrode. Agreement of the metals mass fractions and relative abundance  
377 with prior data indicates that samples collected with the ESP have comparable composition to  
378 those collected by more traditional methods. Thus, the ESP is an excellent choice for long term  
379 sampling to acquire a large mass of indoor PM.

380

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388



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490

491

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## Figure Captions

492

493 **Fig. 1** Schematic diagram of the electrostatic precipitator portion of the O-Ion B-1000.

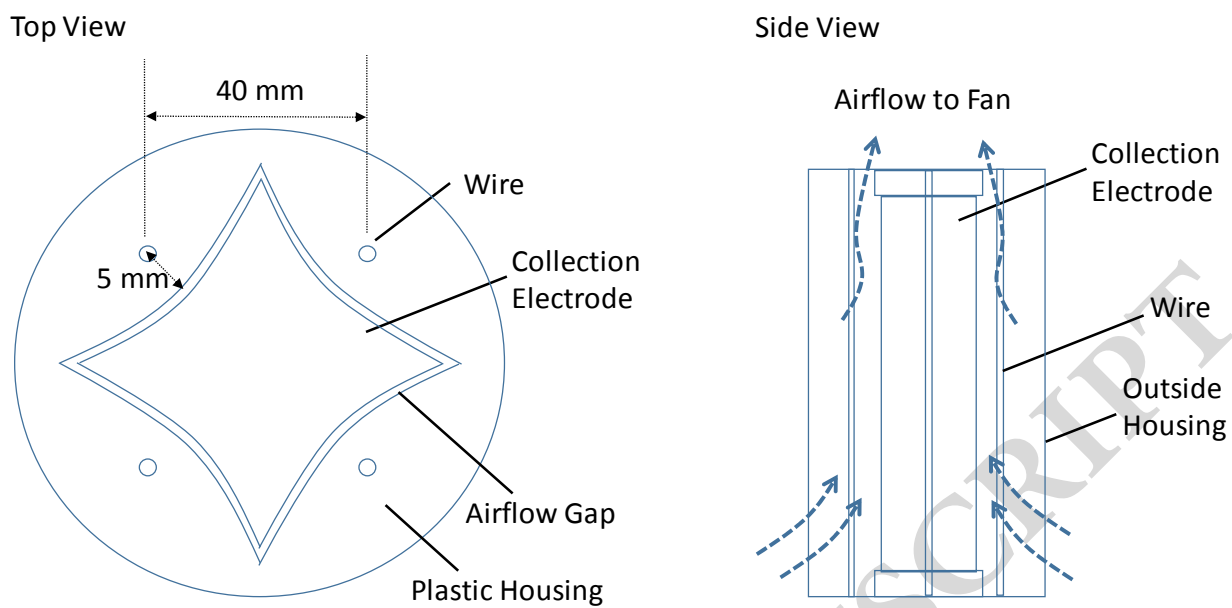
494 **Fig. 2.** Experimental setup used to measure collection efficiency by particle size. Each aerosol  
495 generation system was designed for the specific aerosol type: Sodium chloride, Arizona road dust,  
496 or diesel fumes.

497 **Fig. 3.** Collection efficiency of ESP for ARD, NaCl, and diesel fumes in relation to particle  
498 geometric diameter. Error bars correspond to one standard deviation.

499 **Fig. 4.** Mass of indoor PM wiped from ESPs deployed in homes for 30 days over winter and  
500 measured using a Mettler Toledo MT-5 microbalance.

501 **Fig. 5.** Base 10 log scale of metal mass fractions (ng/mg) from 21 homes in Eastern Iowa. The  
502 whiskers represent the minimum and maximum values for each species. The box is split into two  
503 pieces for the top and bottom half of the data with the median represented as a line in the middle.  
504 The average is shown with an x.

505



506

507 **Fig. 1**

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508

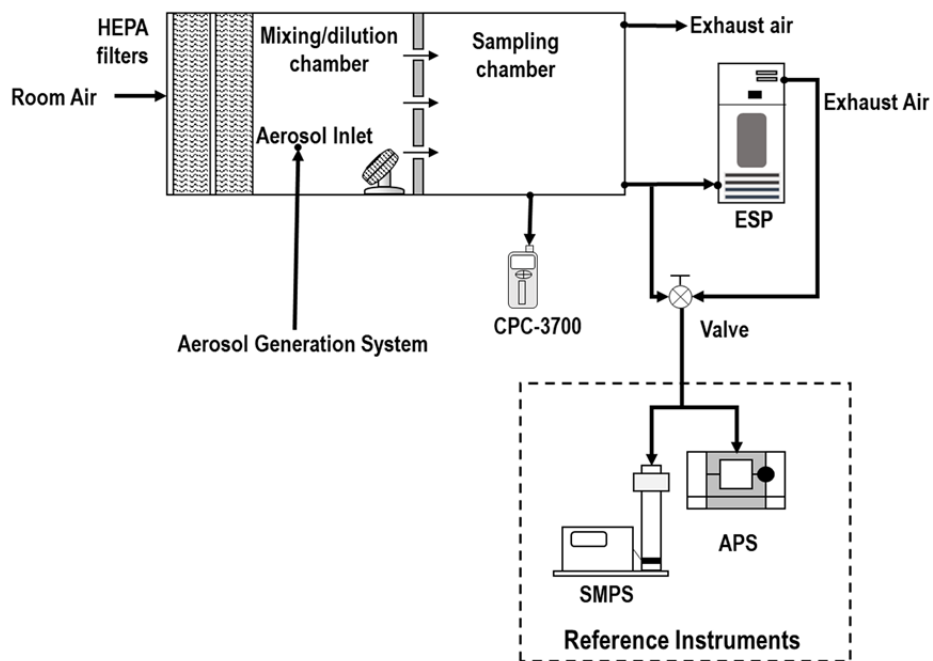


Fig. 2.

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Collection Efficiency

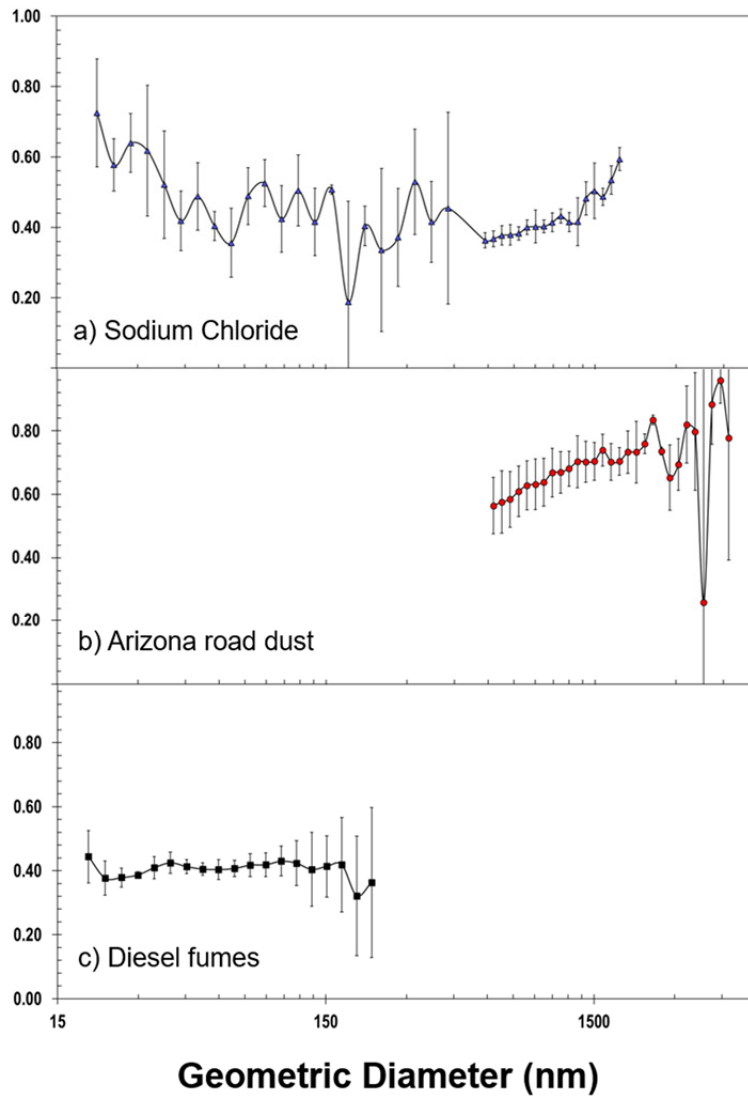


Fig. 3.

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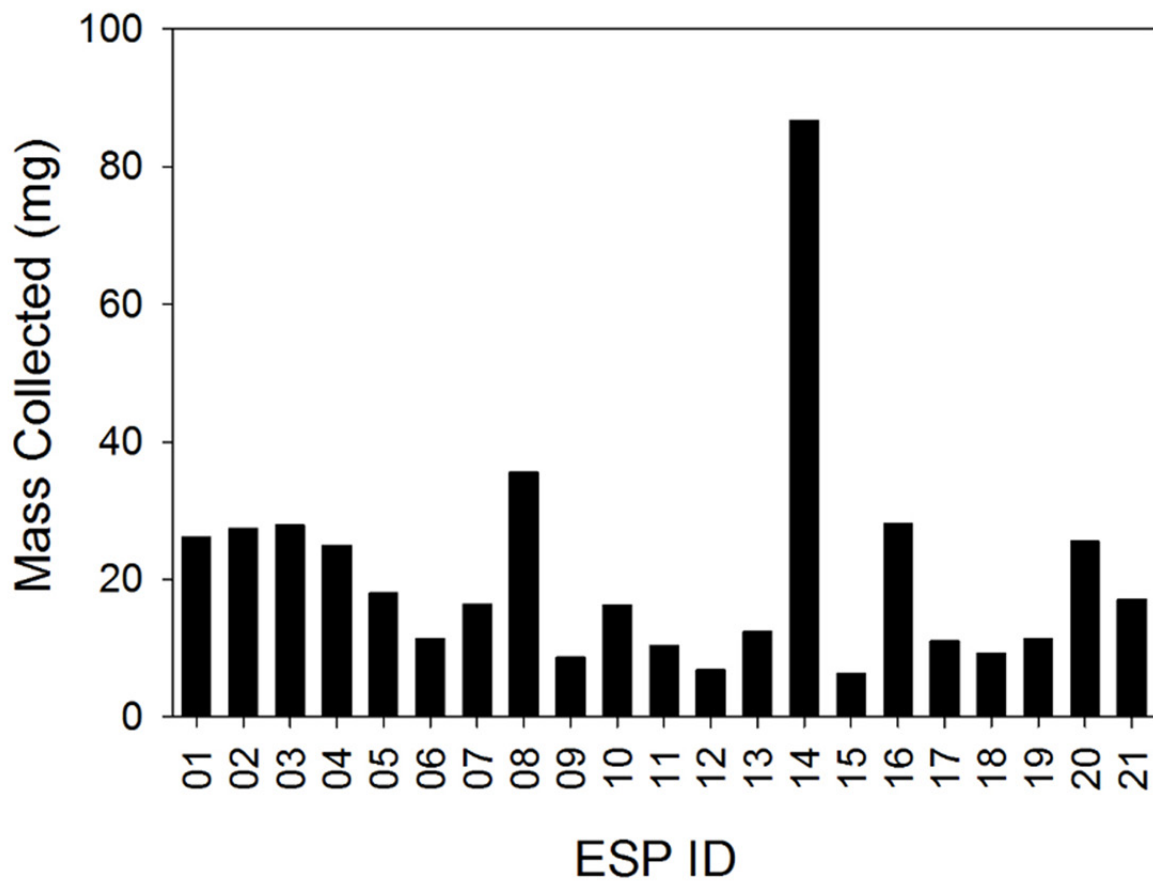


Fig 4.



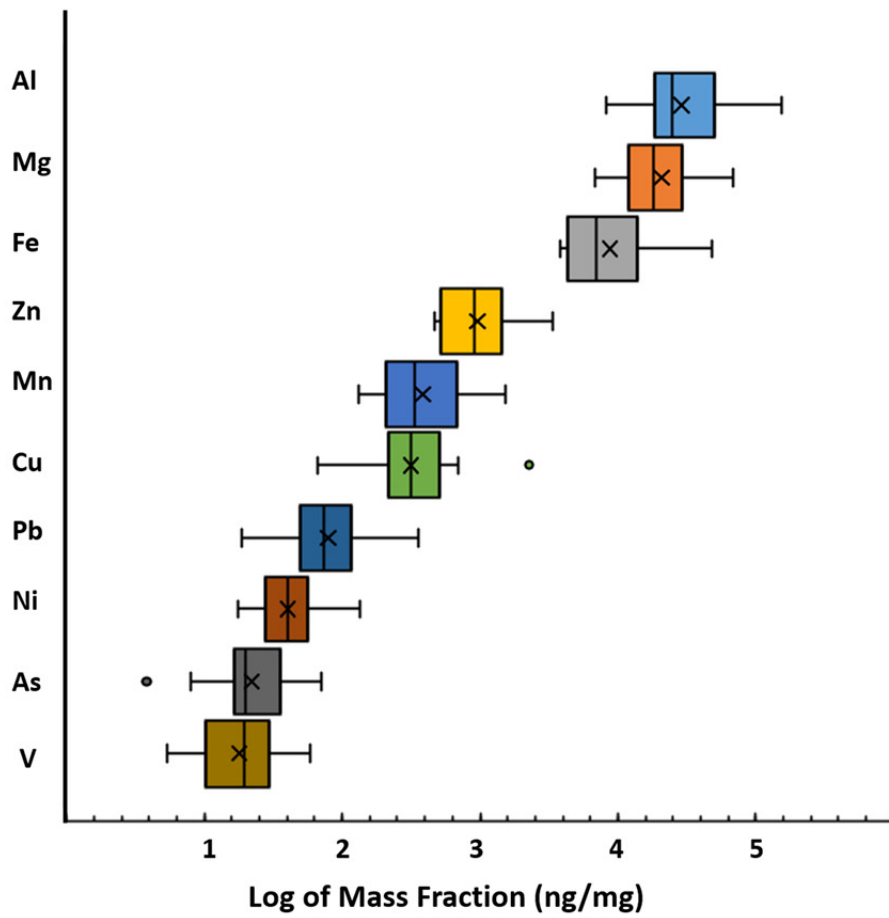


Fig. 5.

