A Low-Cost Device for Bulk Sampling of Airborne Particulate Matter: Evaluation of an Ionic Charging Device

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ABSTRACT

Bulk sampling of aerosols is often needed for the determination of physical properties, chemical composition and toxicity assessments of airborne particulate matter. Conventional aerosol samplers have several limitations for use as bulk aerosol collectors including cost, noise levels, power requirements associated with the use of a pump, limited flow rate, and a relatively long sampling time needed to collect sufficient mass to achieve gravimetric or other method limits of detection. In this study, a low-cost ionic charging device (ICD) was evaluated that addresses many of the drawbacks of conventional aerosol samplers. Different types of particles including incense fume, Arizona Road Dust (ARD) powders and Polystyrene Latex (PSL) spheres of different sizes were aerosolized then sampled using three ICDs and compared to conventional inhalable and $PM_{2.5}$ (particulate matter with aerodynamic diameter less than 100 µm and 2.5 µm, respectively) aerosol samplers in a controlled laboratory chamber at varying concentrations. The device was also evaluated in indoor environments. ICDs operate at almost 18.5 times higher flow rate than conventional personal samplers and provided up to 9 times greater total collected mass compared to the conventional samplers over the same time frame. Using a regression analysis, aerosol-specific linear equations with slopes ($C_{PM2.5}/C_{ICD}$) from 1.21 to 7.10 and R^2 from 0.74 to 0.99 for estimating the inhalable and $PM_{2.5}$ mass concentrations using the ICD were derived. This study suggests that the ICD provides a less accurate estimate of size-selective PM mass concentrations than conventional personal aerosol samplers; however, it collects coarse particles efficiently and increases total sampled mass per time at a lower cost and without noise associated with traditional sampling methods. Therefore, the ICD can be used as a bulk aerosol collector for composition analyses and in-vitro toxicology tests of coarse PM.

Keywords: Bulk aerosol sampling; Inhalable aerosol sampler; $PM_{2.5}$ aerosol sampler; Coarse PM; Indoor air.

INTRODUCTION

Aerosol samplers are widely used in air quality studies to estimate the mass concentration of particulate matter (PM) and to assess the degree of its health hazard. Bulk sampling of aerosol (sampling airborne particles of all sizes) is commonly used for identifying the physical properties, chemical composition and toxicity of PM in both indoor and outdoor environments (Brauer *et al.*, 1991; Camuffo *et al.*, 1999; Lioy *et al.*, 2002; Gysels *et al.*, 2004; Says and Warheit, 2009; Mischler *et al.*, 2013). To collect sufficient mass of particles for the types of analyses described above, either a

long sampling duration or high flow rate is required. Current samplers capable of collecting high PM masses use either a filter (e.g., the bulk filter system utilized by Li-Jones *et al.*, 1998), inertial impaction (e.g., the virtual impactor developed by Sioutas *et al.*, 1997 or the high volume cascade impactor developed by Demokritou *et al.*, 2002) or centrifugal motion (e.g., the sequential cyclone system developed by Rule *et al.*, 2010). However, these samplers are relatively expensive, require large and noisy pumps, and many sample particles onto collection substrates that require a liquid-based extraction to remove particles (Chow, 1995; Koziel *et al.*, 2001; Weber *et al.*, 2001). Thus their application for indoor air sampling has been limited.

Personal samplers such as the Button Aerosol Sampler for collection of inhalable aerosols (particles penetrating through nose/mouth with 50% of penetrated particles less than 100 micrometers) and the Personal Environmental Monitors (PEM) for concentration measurement of PM_{10} or

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PM$_{2.5}$ (particulate matter with aerodynamic diameter less than 10 or 2.5 micrometers, respectively), are used to evaluate exposures to aerosols in occupational (Li *et al.*, 2000) and indoor settings (Ward and Noonan, 2008; McCormack *et al.*, 2009). Such samplers have the advantage of being portable and can be used with battery-powered personal sampling pumps. However, aerosol sampling using these two samplers is costly ($250–650 for the sampling inlet and $1000+ for a personal sampling pump) and long sampling times are required to collect substantial masses of particles when aerosol concentrations are low.

Inspired by the corona discharge mechanism often applied in electrostatic precipitators (ESPs) (Mizuno 2000), an ionic charging device (ICD, Inspirotec Inc., Chicago, IL, USA) has been evaluated as a potential platform for bulk aerosol sampling, which can address some of the challenges associated with filter-based and impaction-based PM sampling. An ICD supplies a fixed, high voltage to a conductive wire (diameter of 0.2 mm) with respect to two grounded strip metal plates (125 mm × 5 mm × 0.2 mm), oriented as shown in Fig. 1. The electrostatic field strength in the neighborhood of the wire is extraordinarily high and non-uniform. This leads to frequent collisions of the positive ions generated from the charging wire electrode with electrically neutral air molecules. The movement of the ions and air molecules induces a conductive pressure to the air and creates a momentum transfer from the ionizing wire to the surrounding air (Zhao and Adamiak, 2005). Subsequently, the airborne particles obtain a positive charge because of the ionic collisions and drift toward the grounded strip electrodes (Gordon *et al.*, 2011, 2015). Therefore, the ICD actively moves the air and particles by converting electrical energy into air movement without the aid of moving parts and facilitates bulk aerosol sampling of residential indoor environments. The ICD outlet is on its back and it is designed with the same shape as the inlet. In order to minimize the ozone often formed as a byproduct of any ESP device, a catalyst block of honeycomb ceramic coated with MnO$_2$ was placed in front of the charging electrode (Gordon, 2015).

ICDs have been evaluated as efficient and low-cost aerosol samplers for allergen airborne particles (Parvaneh *et al.*, 2000; Custis *et al.*, 2003; Platts-Mills *et al.*, 2005; Fox *et al.*, 2008; Gordon *et al.*, 2016). However, there is no knowledge regarding the adequacy of an ICD for bulk collection of indoor aerosol. The advantages of using an ICD for PM sampling include: 1) portability and low weight, 2) long duration sampling on wall power rather than using a battery, 3) silent operation, 4) high sampling flow rates, 5) easy removal of particles from the substrate for analyses and 6) low cost.

The goals of this study were: 1) to evaluate the efficacy of an ICD device for bulk aerosol sampling to provide an adequate amount of airborne PM for chemical composition determination, surface analyses, or in-vitro toxicological assessments and 2) to correlate the estimated mass concentration of different types of aerosols collected using ICD with the airborne mass concentrations of inhalable and PM$_{2.5}$. For these purposes, ICD performance was tested against different types of airborne particles and the collection
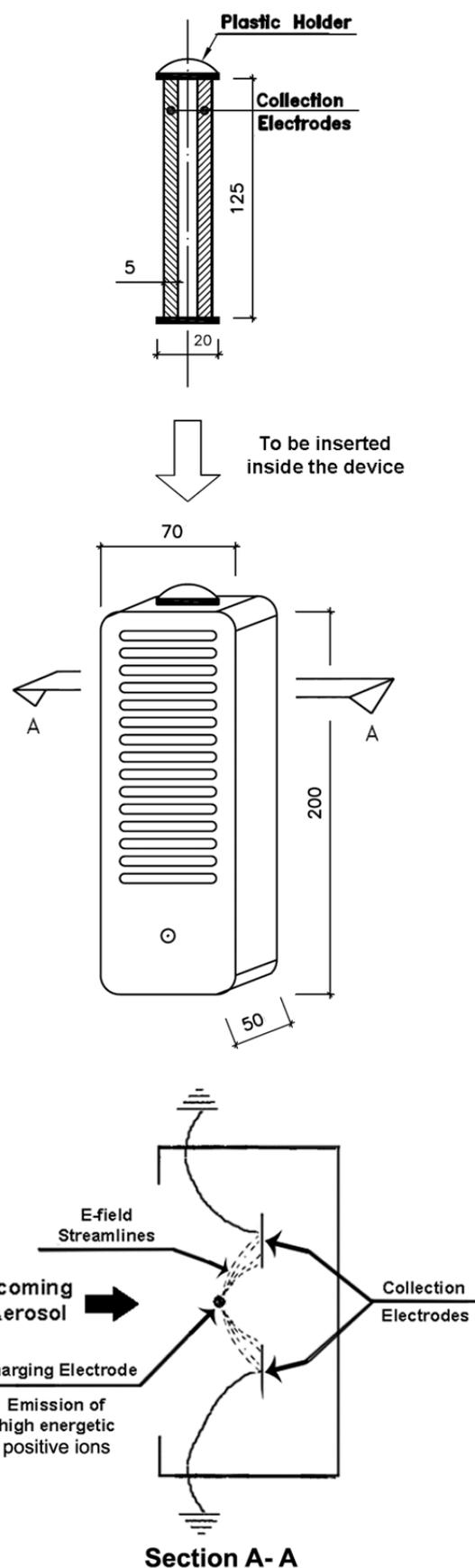
Fig. 1. Schematic of the ICD used in this study. Displayed dimensions are in mm.

efficiency was assessed using monodisperse submicron and supermicron particles.

METHODS AND MATERIALS

The ICD tested in this study was developed by Inspirotec Inc., Chicago, IL, USA (see Fig. 1) and was equipped with the mentioned honeycomb catalyst to minimize generated ozone. For all experiments, a total of three ICDs with ozone-removing catalyst were co-located with three inhalable Button samplers (Cat. No. 225-336, SKC Inc., Eighty four, PA, USA) and three personal $PM_{2.5}$ samplers (PEM Model 200, MSP Co., Shoreview, MN, USA) (Fig. 2). All samplers were vertically mounted on the chamber wall opposite to the aerosol generating source. The inhalable button sampler consists of a hemispherical metal porous inlet containing 381-µm diameter openings to screen non-inhalable particles and collect onto a 25-mm diameter filter (Li *et al.*, 2000). PEM samplers, as referenced in USEPA method IP-10A, consist of an inertial impaction sampler to screen particles larger than 2.5 µm and collect $PM_{2.5}$ particles onto a 37-mm diameter filter. Except for the ICDs, which are pumpless, the inhalable and $PM_{2.5}$ aerosol samplers were connected to a personal sampling pump at a flow rate of 4.0 ± 0.05 L min^{-1} (Model 400S, BGI Inc., Waltham, MA, USA) and fluorocarbon coated glass fiber filters (FiberFilm T60A20 pore size 1 µm, Pall Inc., Ann Arbor, MI, USA) were used for sampling.

Filters from both the inhalable and $PM_{2.5}$ aerosol samplers were gravimetrically analyzed using National Institute for Occupational Safety and Health (NIOSH) method 0500 to obtain the total aerosol mass (Eller and Cassinelli, 1994; Hauck *et al.*, 1997). An analytical microbalance (MX5 Microbalance, Mettler Toledo Inc., Columbus, OH, USA) of 1-µg precision was used to weigh the filters before and after aerosol sampling. Before weighing, the filters were equilibrated in a room under controlled temperature and relative humidity conditions (T = 21 $\pm$ 3°C and RH = 30 $\pm$ 2%) for at least 4 h. All filters were electrostatically discharged by passing over a u-shape high-voltage ionizer (Anti-Static Solutions, Mettler Toledo Inc., Columbus, OH, USA) to minimize errors caused by electrostatic forces.

We adopted a wiping method from Liang and Xu (2014) for collecting phthalate particles from the surface of stainless steel rods and by Brown *et al.* (2007) for collecting spores from stainless steel surfaces to remove the layer of particles built up on the collection electrodes of the ICD devices. Both sides of each electrode were wiped by two (total of 4 filters for each ICD sample) pre-weighed 37-mm filters (and made of the same material as the filters used in the inhalable and $PM_{2.5}$ aerosol samplers) each wetted with a 70% (by volume with 30% deionized water) isopropyl alcohol solution. Gentle back and forth movements of the filter held by tweezers enabled effective capture of the sampled particles. To ensure complete extraction of the collected particles, the number of needed filters and back and forth movements were determined (4 times) by trial (number of filters and movements increased until no further increase in
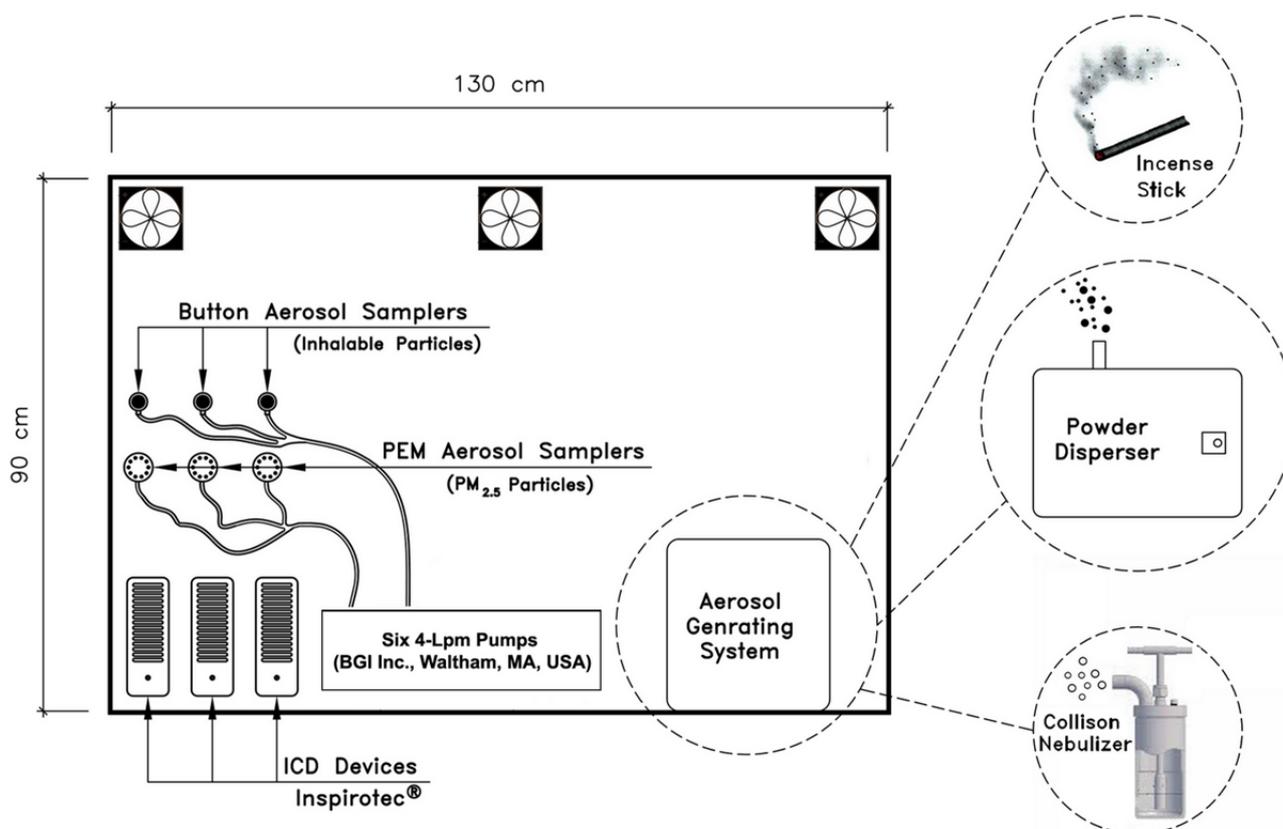

Fig. 2. Schematic of the chamber set-up.

total mass of collected particles from collection electrodes was observed). Then, filters were stored inside a desiccating chamber for 8 hr to ensure evaporation of the isopropyl solution before final weighing. Proper evaporation of the alcohol was verified by pre- and post-weight of a blank filter wetted by isopropyl solution. The sum of the weight increase measured from all 4 filters used for each ICD device was considered as the total mass of sampled particles by that ICD device. One blank filter per 10 sample filters was pre-weighed and maintained inside a weighing chamber to be post-weighed with the sampled filters at controlled temperature and relative humidity to account for the effects of environmental factors in weight change of the filters. To account for the mass loss of fibers due to the wiping in estimating the weight change, one blank filter per 10 filters was pre-weighed, wiped a clean ICD electrode as explained above, and post-weighed following the same procedures as samples. Results obtained from blank samplers were used for adjusting the mass values from the gravimetric analyses.

Measurements of Airflow

To measure the effective volumetric flow rate of the ICD device, a hot-wire anemometer (Alnor® Model AVM410, TSI Inc., Shoreview, MN, USA) was used to calculate the air velocity. The total sampling flow rate (Q) of the ICD was calculated using the following equation:

$$Q = \sum_{i=1}^{i=15} a \left(\frac{v_{1i} + v_{2i}}{2} \right) \tag{1}$$

where α is the cross sectional area of each of 15 vent slots (2.5 cm^2 each), v_{1i} is the air velocity measured at the inlet of the ith sampling vent slot in the beginning of the aerosol sampling and v_{2i} is the air velocity measured at the inlet of the ith sampling vent slot at the end of the aerosol sampling, all on the air inlet side of the sampler. The sampling flow rate was measured at the beginning and end of the sampling period and the average of the two measurements was used to estimate the sampled volume and aerosol mass concentration. For all airflow measurements, the anemometer was placed as close as possible to the vent slots (1 cm in this study).

To evaluate the significance of changes in the flow rate at the beginning and at the end of an experiment, we conducted a paired t-test, which is a preferred statistical analysis for correlated before-and-after group designs. T-tests were run separately for the flow rates of non-PSL and PSL particles. The analysis was completed using a commercial software package (SPSS Ver. 21, IBM Inc., Endicott, NY, USA). The confidence level of 95% was used to define significance. The conefficient of variations (CVs), the ratio of the standard deviation to the mean, across the samplers for flow rates were calculated to present the average variability of the ICD.

Aerosol Sampling of Laboratory Particles

Aerosol was generated in a stainless steel laboratory chamber (130 cm × 90 cm × 70 cm) (Fig. 2). Three fans were located inside the chamber to uniformly mix the generated aerosol inside the chamber. Uniformity of the aerosolized particles at three different points of the chamber was verified by two aerosol sizer instruments: an aerodynamic particle sizer (APS) (Model 3321, TSI Inc., Shoreview, MN, USA) for supermicron particles and a scanning mobility particle sizer (SMPS) (Model 3938 with Electrostatic Classifier Model 3082 and Condensation Particle Counter Model 3787, TSI Inc., Shoreview, MN, USA) for submicron particles. In addition, the aerosol mass concentration was estimated at five different points across the chamber using additional inhalable aerosol samplers when Arizona Road Dust (ARD) powder was aerosolized; less than 5% variability within the collected masses was observed using gravimetric analysis. All co-located samplers were mounted on the same wall of the chamber, furthest from the aerosol generator to ensure each sampler was exposed to similar mass concentrations.

To evaluate the size-specific collection efficiency of the ICD devices, three different types of particles were tested: 1) Incense fume, 2) ARD and 3) ARD and talc powder mixture. Measured by an APS and a SMPS, size distributions for most aerosol types were not lognormally distributed (different geometric standard deviation (GSD) for number-based and mass-based size distributions). The average median sizes and GSD of these three types of particles are summarized in Table 1. In estimation of count median diameters (CMDs) and mass median diameters (MMDs), the shape factors of all particles were assumed to be 1 (spherical particles) and calculations were completed using a software package (Aerosol Instrument Manager V.10.1.0.6, TSI Inc., Shoreview, MN, USA).

Incense fume particles were generated by burning 25-cm incense sticks (Dragons Blood, HEM Co, Mumbai, India). Sampling times for incense particles ranged from 20 to

Table 1. Count median diameter (CMD) and mass median diameter (MMD) of the generated polydisperse aerosols. Values in parentheses indicate standard errors.

Type of generated particles*	Count median diameter (CMD) (µm)	Number-based GSD	Mass median diameter (MMD) (µm)	Mass-based GSD
Incense fume	0.104 (± 0.003)	1.59 (± 0.02)	0.173 (± 0.047)	1.45 (± 0.001)
ARD-3 powder**	0.754 (± 0.17)	1.61 (± 0.54)	2.08 (± 1.48)	1.22 (± 0.15)
ARD-70 powder***	0.867 (± 0.007)	1.56 (± 0.02)	4.18 (± 0.19)	2.15 (± 0.04)
ARD and talc powder	0.929 (± 0.001)	2.06 (± 0.04)	6.86 (± 0.11)	2.34 (± 0.017)

* CMD and MMD of PSL particles are identical, known and thus were excluded from this table.
** ARD-3 is defend as ARD powder sample sieved to include particle diameters smaller than 3 µm.
*** ARD-70 is defined as ARD powder sample sieved to include particle diameters smaller than 70 µm.

80 min. The incense sticks were located at the opposite end of the chamber away from the aerosol samplers.

Sieved in two size ranges, 0 to 3 µm (ARD-3) and 0 to 70 µm (ARD-70), the ARD powders (ARD, Powder Technology Inc., Arden Hills, MN, USA) were aerosolized using a small-scale powder disperser (SSPD, Model 3343, TSI Inc., Shoreview, MN, USA). A third powder, which was a mixture of 50% (w/w) fine talc powder and 50% fine ARD (< 2.5 µm) was also examined. Due to the lower concentrations of the particles generated from the powder disperser compared to incense burning, sampling times of 1, 1.5, 2 and 2.5 hr were used for this set of experiments.

Fluorescent PSL particles and subsequent fluorometric analysis were used to evaluate the collection efficiency of monodisperse aerosols. Fluorometric analysis is advantageous when the total mass of collected particles is less than 10 ng (Stöber and Flachsbart 1973). Aqueous suspensions of PSL aerosols (Fluoro-Max™ Green Aqueous Fluorescent Particles, Thermo Fisher Scientific Inc., Waltham, MA, USA) were aerosolized by introducing filtered compressed air at a flow rate of 5 L min^{-1} into a 1-jet Collision Nebulizer (BGI, Mesa Labs Inc., Butler, NJ, USA). The PSL suspension was diluted with nano-pure deionized (DI) water with a resistivity of 18.2 MΩ and sonicated by an ultra-sonic bath for 20 min, to break apart any agglomerates prior to nebulization. Following Qi *et al.* (2010), 0.5 g L^{-1} methanol was added to the suspension to reduce the surface tension and prevent agglomeration of particles while dispersing. Sampling times for PSL aerosol were 6 hr for d_p = 0.27, 3.5 hr for d_p = 0.94, 3 hr for d_p = 1.66 and 2 hr for d_p = 3 µm.

To extract collected fluorescent PSL particles from filter fibers (from the wipes of the ICD, and filters from the inhalable and PM$_{2.5}$ aerosol samplers), filters were placed in 15 mL centrifuge tubes filled with a base solution composed of 5% (by volume) ammonium hydroxide (NH$_4$OH) and 10% ethyl acetate (C$_4$H$_8$O$_2$) and the rest with DI water. One percent (0.15 mL) of Pyridine (C$_5$H$_5$N) was added to the extract solution of each tube to prevent ammonium salt formation. Then, all centrifuge tubes were hand-shaken for 10 s and sonicated for 20 min (Tolocka *et al.*, 2001). For fluorescence analysis, 125 µL from each tube was pipetted in triplicate into wells of a 96-well microplate, and a fluorescent detector (GloMax®Multi detection system, Promega Inc., Madison, WI, USA) with a cartridge appropriate for excitation peaks of 460 nm and emission peaks of 515 to 570 nm measured the relative fluorescence units (RFUs) of each sample. Based on a calibration plot, the RFU is linearly proportional to fluorescent mass. Thus, RFU was used as the metric for relating mass concentration by the ICD to those by the inhalable and PM$_{2.5}$ aerosol samplers.

Indoor Air Sampling

Measurement of the total mass of collected particles and estimation of indoor inhalable and PM$_{2.5}$ concentrations of airborne particles in a real-world setting followed the same protocol as for the laboratory chamber experiments. Indoor spaces included an idle ventilated laboratory (sampling over a 1-day period), outside the kitchen of a pet-free apartment (sampling from a 5 m × 5 m × 2.3 m living room over 1-

day, 3-day and 5-day periods) and at the entrance of a restaurant kitchen (sampling over a 1-day period). Similar to the arrangement and orientation of aerosol samplers inside the laboratory chamber, 3 ICDs were each co-located with an inhalable and a PM$_{2.5}$ sampler. All samplers were at least 2 m away from the nearest furniture. The only potential source of airborne particles in the laboratory was expected to be resuspension of the surface deposited particles due to the ventilation system. Samplers in the living room were exposed to more mechanisms emitting aerosol due to human activities (e.g., cooking, walking on the carpet). The restaurant kitchen had continuous cooking activities and a higher number of occupants (5 cooks working within 2 m of the samplers) as aerosol emitting sources.

Measurement of Ozone

Potential ozone generation is a general drawback of conventional ESPs (Hinds 1999). To investigate levels of ozone concentration emission while the ICD was operating, a personal ozone monitor (POM) (Model POM 202, 2B Technologies Inc., Boulder, CO, USA) was placed inside the laboratory aerosol chamber containg 3 running ICDs equipped with the ozone removal catalyst to monitor real-time changes in ozone concentration. The mechanism of ozone detection by the POM is based on the UV absorption radiation wherein absorption of the sampled gas is determined at the wavelength of 254 nm when exposed to a mercury lamp (Andersen *et al.*, 2010).

RESULTS

Measurements of Airflow

In contrast to conventional aerosol samplers, flow rates of the ICDs are not controllable and they were calculated using the procedure described in Secton 2.1. The initial sampling flow rate of the 3 ICDs averaged over all experimental runs (across samplers) for different aerosol types was 74 ± (4.54 as standard error). This is about 18.5 times higher than the sampling flow rate of the inhalable and PM$_{2.5}$ aerosol samplers). A decrease in post sampling flow rates was observed when non-PSL particles were sampled. The greatest decrease was related to ARD with average CV of 6% (across samplers decrease at the end of the experiment). In contrast, for PSL particles the final flow rates of the ICDs increased 8% from their initial values. Results of the paired t-test for the abovementioned observations revealed that the *p*-values were 0.016 and 0.064, respectively for non-PSL and PSL particles. Therefore, the decrease in the flow rate of an ICD when collecting non-PSL particles was significant but was only marginally significant for the flow rate reduction when sampling PSL particles.

ICD Performance for Aerosol Sampling of Laboratory Particles
Incense Fume and ARD Powder

The ratios of total mass of particles collected by the ICD to the mass collected by the inhalable and PM$_{2.5}$ aerosol samplers (M$_{ICD}$/M$_{PM2.5}$), as a function of MMD, are shown in Fig. 3. Except for ARD-70 (MMD = 4.18 µm), total mass

and mass concentration results obtained from the inhalable and $PM_{2.5}$ aerosol samplers were similar and comparisons with the inhalable sampler are shown in the supplementary data (Fig. S1). Generally, the mass ratios increased with increasing MMD for the polydiseperse aerosols (see Fig. 3(a) and Fig. S1(a)). The $M_{ICD}/M_{PM2.5}$ ratio for incense fume with MMD of 0.173 μm was 3.31 ± 0.53 and it increased to 5.15 ± 0.23 for ARD-3 powder with MMD of 2.08 μm. Both means and standard errors are averaged over 3 ICDs for the pertinent sampling period. This trend of increase in $M_{ICD}/M_{PM2.5}$ ratios when the particle size was increased continued and reached the greatest values of 9.05 ± 0.13 for larger sizes of ARD particles (ARD-70). However, a small decrease was observed as for the mixture of ARD and talc powder. Thus, the higher sampling flow rate of the ICD resulted in 3 to 9 times more mass of collected particles compared to the masses collected by either inhalable or $PM_{2.5}$ samplers. The wiping method for extraction of the particles from metallic electrodes of the ICD was also evaluated as the four sequential wiping filters of ARD-3 samples yielded 78, 11, 5 and 3% of the total collected particle mass, on average.

In order to compare the mass concentration ratios of different aerosols (total mass of collected particles per unit air volume), the total air volume pulled through each type of aerosol sampler should be taken into account. Mass concentration ratios estimated using different aerosol samplers are displayed in Fig. 3(b) for ICD comparison with $PM_{2.5}$ mass concentration and comparison with inahalable mass concentration in Fig. S1(b). Aerosol concentration estimated by the ICD (C_{ICD}) was 19% to 62% of estimates from both inhalable and $PM_{2.5}$ aerosol samplers ($C_{Inhalable}$ and $C_{PM2.5}$). Mass concentration ratios increased when the median size of the particles increased (except for the ARD and talc powder aerosol), but was always underestimated.

Although standard errors of concentration values for ICDs were generally larger than those for the inhalable and $PM_{2.5}$ aerosol samplers, the relationship between the ICD and conventional samplers was highly linear (Fig. 4). However, the slope was a strong function of the particle size distribution. All linear relationships versus inhalable aerosol sampler are similar to those for $PM_{2.5}$ and are shown in the supplementary data (Fig. S2).

Fluorescent PSL Particles

Since fluorescence (measured in RFU) of the PSL samples is proportional to aerosol mass, $M_{ICD}/M_{inhalable}$ was used to represent collection efficiency of the ICD. These ratios for $d_p = 0.27$, 0.94, 1.66 and 3 μm were 1.37, 1.53, 1.70 and 3.67, respectively (see Fig. S3). Collection efficiency was low but increased as the particle size increased.

### *ICD Performance for Indoor Air Sampling*

The total mass of collected particles by the ICD was considerably greater than the $PM_{2.5}$ aerosol sampler in indoor sampling experiments. After 24 hr of bulk aerosol sampling, the average mass of collected particles by ICDs in the ventilated laboratory, pet-free apartment and restaurant kitchen were 0.075, 0.29 and 0.62 mg, respectively. For comparison, the total masses sampled by the $PM_{2.5}$ aerosol sampler were 0.025, 0.11 and 0.15 mg (Fig. 5(a)) indicating a 2.64 to 4.13 times increase when the ICDs were used for aerosol sampling. Aerosol concentrations estimated by the $PM_{2.5}$ aerosol samplers for different averaging time periods ranged from 4 to 24 μg m^{-3} and the concentration range estimated by ICDs was 0.47 to 4.02 μg m^{-3} (Fig. 5(b)). Relative to the laboratory tested particles, the R^2 value was somewhat lower at 0.74.

Measurement of Ozone

Over 2 hours with three ICDs running in the aerosol chamber, the ozone concentration only increased about 3 ppb (about 10%) compared to the pre-experiment concentration. Thus the ozone catalyst seems to be efficient for minimizing ozone production. According to unpublished data reported by the manufacturer, even when five ICDs with the ozone removal catalyst were operated in a small room (2.3 m × 2.7 m × 2.5 m), the ozone concentration was below 25 ppb, suggesting each ICD contributes less than 5 ppb of ozone.
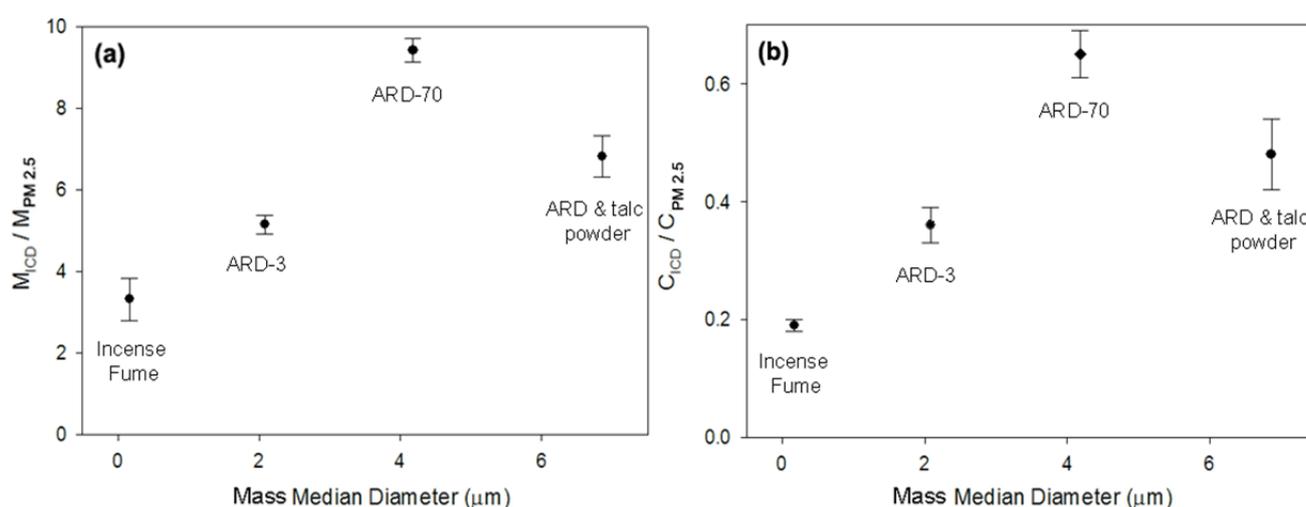
Fig. 3. The ratio of total masses and mass concentrations resulting from the ICDs to those of the $PM_{2.5}$ aerosol samplers: a) total mass ratios and b) mass concentration ratios. Error bars indicate standard errors.
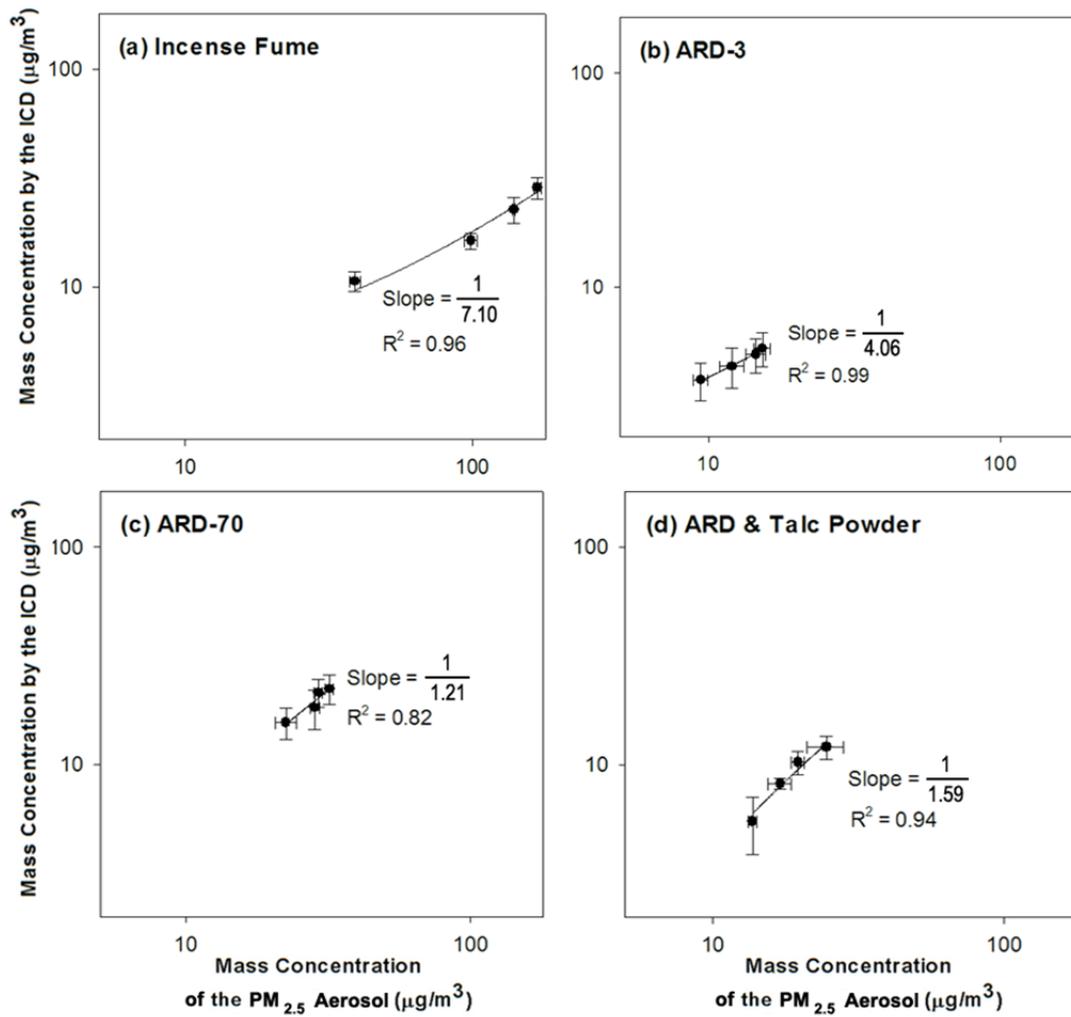
Fig. 4. Relationship between aerosol mass concentrations estimated by the ICD and the $PM_{2.5}$ aerosol samplers for the collection of: a) incense fume, b) ARD-3, c) ARD-70 and d) ARD & talc powder. Error bars indicate standard errors.
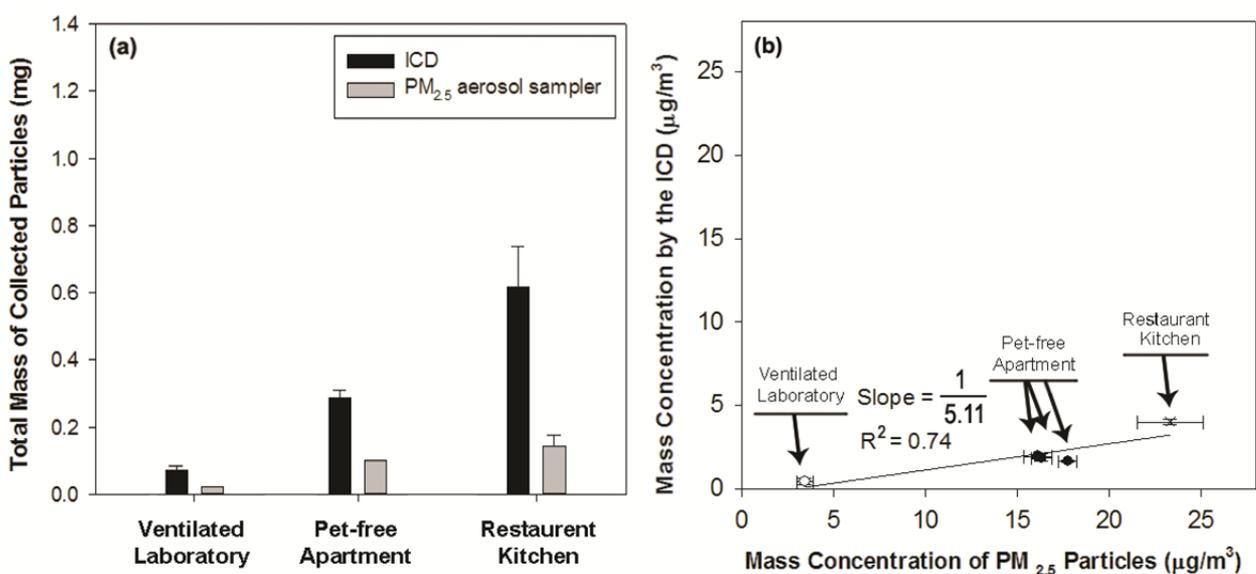
Fig. 5. Comparative results of the indoor aerosol sampling: a) Total mass of particles collected by ICDs compared to $PM_{2.5}$ aerosol samplers for 1-day indoor air sampling periods and b) Linear relationship between aerosol mass concentration by the ICDs and $PM_{2.5}$ aerosol samplers. Error bars indicate standard error.

DISCUSSION

The accuracy of the ICD for estimating PM mass concentrations appears to depend on particle size, the surface charge of particles in the ambient air, and particle composition, which all influence the charging efficiency of the particles. In general, larger particles were collected more efficiently due to their larger surface area, which increases the number of electric charges on the surface and increases the resulting electrostatic force on the particles (Hinds, 1999; Kim and Lee, 1999). The particle size is proportional to the charge limit of positively charged particles: $q_L \propto (d_p)^{1.5}$ (Hinds, 1999). The aerosol generation used for the lab-based tests likely result in preexisting charges on the particles prior to sampling. In particular, Qi *et al.* (2009) demonstrated that nanoparticles (50–200 nm) aerosolized by a nebulizer may have up to 20% higher or lower surface charges compared to what is predicted to accumulate only due to the electrostatic field. The powder disperser for aerosolizing ARD powder may also generate highly-charged particles due to the tribocharging. While particle charges could not be measured in this study, the relatively higher collection efficiency of the ICD for ARD powder and talc and ARD powder mixture may be due to pre-existing charges on the airborne particles. The pre-existing charges are likely smaller for the finer particles, which have a smaller surface area. There is also the potential for size-dependent extraction efficiency of the wiping method because the surface forces (e.g., van der Waals forces) are higher for smaller particles. However, on a mass-based measurement, the contribution of these particles would be negligible.

For particle sizes larger than 300 nm, field charging by the charging wire in the ICD is the dominant mechanism for charging the particle (Hinds 1999; Kaminski *et al.*, 2013) and particle composition also influences the collection performance of an ICD (Cooper and Alley, 2010; Zhuang *et al.*, 2000). The charging efficiency of particles depends on the dielectric constant of the particles, which is a material property and defined as the ratio of the permittivity of the substance to the permittivity of free space. The dielectric constant is a measure for expressing the ease with which particles are charged. As the dielectric constant increases, a given size of particles can hold larger quantities of charge, but they also break down more easily when subjected to intense electric fields (Sadiku, 2001).

The PSL particles showed a low and similar collection efficiency by the ICD for submicron particle sizes. PSL particles might have negative pre-existing charges before ionization by the positive ions, reducing the net positive charges on the PSL particles and consequently their collection efficiencies compared to the trend other particle types suggest. The second reason for this reduced collection efficiency may be due to the material dependence of the charging efficiency (i.e., lower charging efficiency for PSL suspension compared to ARD powders and incense fume). Forsyth *et al.* (1998) showed high charging efficiency of ARD particles compared to the nebulized saline solution. ARD and talc & ARD powder mixture showed higher collection efficiencies than the PSL likely due to pre-existing charges, better charging efficiency of the ARD material, and the larger size distribution of the particles. Despite having the highest MMD, the talc & ARD mixture did not have the highest collection efficiency. This might be due to the larger GSD and thus a larger proportion of smaller sized particles (< 0.5 µm) compared to ARD-3 and ARD-70 or due to lower charging efficiency of talc compared to ARD. Higher capture efficiencies than reported here have been achieved for mold spores and allergens (Gordon *et al.*, 2015) likely due to the relatively large particle sizes and possibly due to compositional differences.

The mass concentrations estimated by the ICD had a strong linear relationship with those of the inhalable and PM$_{2.5}$ aerosol samplers. The linear relationships facilitate derivation of a predictive set of equations to estimate the aerosol concentration in real-world settings when inhalable or PM$_{2.5}$ aerosol samplers are not available (Table 2).

The U.S. Environmental Protection Agency (EPA) has defined criteria for performance of a sampler compared to a reference sampler (EPA, 2016). CV values for laboratory tested aerosols were within the 10% acceptance CV value by the EPA for test instruments (except for talc and ARD powder with a CV of 13%). EPA also mandates a set of criteria including a slope of 1 ± 0.1, a y-intercept of 0 ± 5 µg m^{-3} along with an R > 0.97 (40 CFR part 53, subpart C, Table C-4) to consider measurements by a test instrument to be acceptable. As presented in Table 2, all slopes of the regression lines and the R-value of some of the aerosol types violate the above-mentioned criteria. Thus, the ICD cannot be used as a reference sampler for PM$_{2.5}$ mass or inhalable mass concentrations.

However, the total mass of collected particles by the ICD was 2 to 4 times greater than the inhalable and/or PM$_{2.5}$ aerosol samplers over the same time frame. Furthermore, the ratio of the $C_{ICD}/C_{PM2.5}$ for indoor aerosol sampling did not change as a function of sampling duration, suggesting that the collection efficiency of the ICD devices does not deteriorate even after five days of sampling time in indoor

Table 2. Linear equations for estimating inhalable and PM$_{2.5}$ mass concentrations using an ICD.

Type of generated particles	Inhalable mass concentration equations (µg m^{-3})		PM$_{2.5}$ mass concentration equations (µg m^{-3})	
Incense fume	C = 7.26 C$_{ICD}$ − 25.69	R^2 = 0.94	C = 7.10 C$_{ICD}$ − 25.54	R^2 = 0.96
Indoor aerosol	C = 5.08 C$_{ICD}$ + 5.60	R^2 = 0.71	C = 5.11 C$_{ICD}$ + 5.20	R^2 = 0.74
ARD-3 powder	C = 4.86 C$_{ICD}$ − 8.90	R^2 = 0.96	C = 4.06 C$_{ICD}$ − 5.30	R^2 = 0.99
ARD-70 powder	C = 1.84 C$_{ICD}$ + 30.33	R^2 = 0.96	C = 1.21 C$_{ICD}$ + 4.41	R^2 = 0.82
ARD and talc powder	C = 1.82 C$_{ICD}$ + 4.57	R^2 = 0.90	C = 1.59 C$_{ICD}$ + 4.41	R^2 = 0.94

environments, although the between-sampler variability in the mass concentration increased. The R^2 and slope between the ICD and inhalable or $PM_{2.5}$ sampler were lower in the real-world settings than in the controlled chamber experiments, but strong linear correlations between the samplers were observed (Fig. 5(b)). The reduction in R^2 values might be due to the wide variety of sources in indoor environments, which results in a diverse combination of particles with different charging efficiency (i.e., size distribution and particle composition).

In general, the main limitation of the ICD is that it cannot sample fine particles efficiently, leading to a negative bias in the estimation of inhalable or $PM_{2.5}$ mass. Since the collection efficiency of the ICD relies on the charging properties of the airborne particles, collection of fine particles and those with a high proportion of metallic constituents (extremely high dielectric constant) may be challenging. One should note that the wiping method applied in this study may have a lower extraction efficiency than what we reported for earlier ARD powder when extracting finer particles (e.g., 0.3 μm PSL spheres and incense fume). This is because the surface forces between collected particles and collecting electrodes increase as particle size decreases. The ICD electrodes are too long to fit in the weighing chamber of many microbalances (including the one used in this study), necessitating a wiping method for mass determination and elution of the PM from electrodes for extraction. Depending on the particle size and composition, the wiping method may lead to some inefficiencies. However, results of the study conducted by Custis *et al.* (2013) revealed that three times wiping (one wiping filter per trial) over the ICD plates remove 98% of the allergens particles, consistent with our findings for PM mass recovery. The temporal variability of the ICD airflow due to the change in exerting electrostatic field is another limitation. This increases the aerosol mass concentration uncertainty especially when sampling times are longer than 2 hr. As reported in the literature (Nashimoto, 1988; Martinez and Brandvold, 1996; Asbach *et al.* 2005), generation of nitrogen oxides (NO_x) is another undesirable byproduct of high-voltage corona discharge incidents. Although no significant concentration of NO_x was observed during ICD operations of the present study, recorded values were measured using electro-chemical sensors. Rigorous measurement of the NO_x using reference instruments is recommended.

Notwithstanding the above mentioned limitations, the increase in total mass of particles collected by the ICD compared to other indoor samplers make the ICD a good candidate to supply bulk PM for different types of analyses. The ICDs electrodes possess a high particle collection capacity and we did not observe overloading of the plates in up to five days of indoor sampling. Applications may include X-ray photoelectron spectroscopy (XPS), X-ray diffraction (XRD), scanning electron microscopy (SEM) and transmission electron microscopy (TEM), which are all commonly used in research to discover morphology and identify the chemical composition of the particles. Another possible application of the ICD could be the sampling of PM for in-vitro toxicological studies (need of a few mg of

particulate masses).

Other bulk PM samplers suffer from similar limitations as the ICD, including reduced collection efficiency for submicron particles and the need for extraction from collection substrates (Koziel *et al.*, 2001; Cooper and Alley, 2010; Rule *et al.*, 2010). However, the ICD's main advantage is a high sampling flow rate without an air pump resulting in silent operation for indoor environments; paired with its low cost, ICDs make an appealing option for bulk PM sampling.

CONCLUSIONS

The advantages of a low-cost ICD, including bulk sampling of aerosol, with minimal noise and easy-to-extract collection substrates, make it an attractive alternative to other bulk aerosol samplers. Evaluation of the particle collection efficiency by ICDs compared to the inhalable and $PM_{2.5}$ aerosol samplers revealed that it can collect a large mass in a short period of time. However, the particle mass concentrations yielded by ICDs (C_{ICD}) are biased low compared to conventional samplers ($C_{Inhalable}$ or $C_{PM2.5}$). This bias can be explained by low collection efficiency for submicron particle sizes. Although aerosol concentrations estimated from an ICD are lower than the inhalable and $PM_{2.5}$ aerosol samplers, the strong linear relationships observed suggest that site-specific calibration factors can be determined to estimate airborne mass concentrations.

The ICDs are capable of 15 to 21 times higher in sampling flow rate of the air and consequently up to 9 times increase in total mass of collected airborne particles in comparison to conventional personal aerosol samplers often used in indoor environments. The ICDs are inexpensive, run directly on AC power (no pump required), and are noiseless making them an ideal aerosol sampler for indoor environments.

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CONFLICT OF INTEREST

Julian Gordon is co-founder and head of science of the Inspirotec Inc. at the time of this research.

SUPPLEMENTARY MATERIAL

Supplementary data associated with this article can be found in the online version at http://www.aaqr.org.

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