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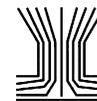
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# Efficient Collection of Atmospheric Aerosols with a Particle Concentrator—Electrostatic Precipitator Sampler

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A novel particle sampling methodology developed recently by our group (Han et al. 2008) has been extended in this article to collect atmospheric particles in electrostatic precipitators (ESPs) for chemical and biological–toxicological analysis. Particles are grown to super-micron droplets via condensation of ultrapure deionized water, and concentrated by virtual impaction in the versatile aerosol concentration enrichment system (VACES). The grown droplets are charged in a carbon fiber charger with negligible ozone generation, and diffusion-dried to their original particle size, while preserving their acquired charges. The charged particles are subsequently collected on suitable substrates in two different ESP prototypes, which can then be used for further chemical (e.g., Inductively Coupled Plasma Mass Spectrometry, Ion Chromatography, organic analysis by means of either gas chromatography-mass spectroscopy (GC-MS) or high performance liquid chromatography (HPLC)), as well as toxicological analyses using cellular or non-cellular assays. To minimize possible chemical reactions between sampled particles and ions generated in the corona region, the previously developed carbon fiber charger was modified, by separating the charging zone from the ionization zone. By combining this novel charger with the VACES, we achieved a higher number of elementary charges per particle (i.e., more than 50) and high particle removal efficiency (i.e., more than 90%) in the ESP, while preserving the chemical composition of the sampled atmospheric aerosols. Uniform particle deposition, which is an essential

feature for cell exposures to particulate matter (PM), was accomplished on the ESP substrate designed for biological PM analysis.

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## INTRODUCTION

Numerous epidemiological studies have shown a relationship between ambient particulate pollution and adverse health effects on humans (Schwartz 2001; Samoli et al. 2005). Recently, ultrafine particles of less than approximately 100 nm in diameter have been of particular concern, because of their greater potential risk to human health, manifested by their ability to penetrate the lower areas of human lung, coupled with their increased per mass content of toxic organic compounds (Delfino et al. 2005). Nonetheless, our understanding of how particle properties such as particle size, surface area, and chemistry affect their toxic properties remains rather poor.

To determine the toxicological properties of ambient aerosols, particles have usually been collected on filters and removed using solvent extraction. Despite its simplicity and widespread use, filtration has several shortcomings related to sample pretreatment, such as extraction, lyophilization, and sonication. Filtration is also susceptible to both physical and chemical artifacts occurring during sampling, including evaporation of semi-volatile compounds, adsorption of gases on the filter material, and possible reactions between collected particles and gaseous compounds, because of the large effective surface area of the filter matrix (Eatough et al. 2003; Schauer et al. 2003; Turpin et al. 2000).

Electrostatic precipitators (ESPs) have been proposed as an alternative for collection of ambient aerosols (Volckens and Leith 2002; Sillanpää et al. 2008). Sampling artifacts, such as vapor adsorption and particle evaporation, can be reduced in an ESP compared to a filter because the particle collection surface area in the ESP is significantly smaller than the effective surface area of a filter. However, ESP samplers are also subject to potential chemical artifacts. Ozone is a byproduct of corona discharge in air and may alter the chemical composition of particles

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sampled in ESPs. According to previous studies (Arnold et al. 1997; Volckens and Leith 2002), high-voltage electrical fields and corona discharges in ESPs generate ozone and oxidant ions, such as  $O_2^+$ ,  $O^+$ ,  $N_2^+$ ,  $N^+$ ,  $NO^+$ , and  $H_3O^+$ . These species have the potential to react with both particles and vapors that enter the plasma region (Seinfeld 2006). Degradation of particle-bound compounds by these reactions may limit or even preclude the use of ESPs as a sampler for atmospheric particulate matter.

To overcome some of the aforementioned shortcomings of particle collection methodologies, we developed a new unipolar charging technique in combination with a particle concentrator using water-based condensational growth (Han et al. 2008a). Particles are grown to super-micron droplets via condensation of ultrapure deionized water, and subsequently concentrated by virtual impaction in a versatile aerosol concentration enrichment system (VACES) (Kim et al. 2001a; Kim et al. 2001b; Misra et al. 2004). The grown droplets are charged in a carbon fiber charger, and dried to the original particle size distribution using a diffusion dryer, while preserving their acquired charges. The concentrated aerosol delivered by the VACES allows sampling for short time intervals, which favors cell viability and exposure characterization. Furthermore, the carbon fiber charger used in this technique charges effectively fine and ultrafine particles with very little or no ozone generation (Han et al. 2008b).

In this study, our previously developed system has been expanded to include collection of ambient particles on the ESP collectors for chemical and toxicological applications. Mobility distributions, average charge numbers and overall particle collection efficiency were investigated for particles charged by the carbon fiber charger with and without the use of the VACES upstream of the charger. Uniformity of particle deposition was also investigated for the ESP designed for cell exposure applications. In field experiments, the chemical components of ambient particles deposited on the ESP were measured and compared to those of reference filters sampling in parallel.

## EXPERIMENTAL SETUP

The particle charging system used in this study was slightly modified from the one developed in our previous investigations (Han et al. 2008a). In our earlier design, particles were charged via direct charging in the ionization region, which included the carbon fiber ionizers, on which a high positive electric field (of 2.5 kV/cm) was applied. By contrast, in the current design, the charging region was separated from the ionization region, thus particles were charged in an electric field-free zone (i.e., by indirect charging) to minimize chemical reactions between particles and ions, and to enhance particle penetration more than 90% by reducing losses in the charger. Figure 1a shows the carbon fiber charger used in this study, which consists of a carbon fiber ionizer and a mixing chamber. The carbon fiber ionizer consists of a carbon fiber electrode placed inside a grounded stainless steel (SS) cylinder (Model 304L-HDF4-150, Swagelok, Solon,

OH). The carbon fiber electrode (Fu Fong Enterprises Co., Ltd, Taiwan) is composed of a bundle of approximately 300 carbon fibers, each about 5–10  $\mu\text{m}$  in diameter and 5 mm in length. The bundle of carbon fibers is connected to a SS rod (3.18 mm O.D.) with a crimp socket connector, and then covered with a heat shrinkable tube. The SS rod is covered with a Teflon tube inside a SS cylinder for electrical insulation. Compressed HEPA-filtered air is introduced at 3 l/min into the SS cylinder via a SS tube (6.35 mm O.D.) to drive the ions generated in the ionizer into the mixing chamber. The end of the SS tube is placed near the tip of carbon fiber to create a jet that flushes out the generated ions to the mixing chamber, thus minimizing the loss of ions in the SS cylinder where a high electric field is formed. Positive ions, created by the carbon fiber ionizer operating with D.C. positive voltage of 2.5–5.0 kV are introduced into the mixing chamber. The aerosol stream is also introduced into the mixing chamber at 3 l/min and is mixed with the incoming ion stream. The particle residence time in the 188  $\text{cm}^3$  mixing chamber is about 0.5 s.

Figures 1b and c show the ESPs designed for sampling and collection of atmospheric aerosols, and for *in vitro* cell exposure, respectively. The ESP shown in Figure 1b consists of two parallel circular plates made of copper and cylindrical Delrin housings for electrical insulation. The diameter of each plate was 47 mm and the distance between the plates was 10 mm. The lower plate was made to fit commercially available filters of 47 mm in diameter. Two male connectors of 9.53 mm O.D. were connected to the inlet and outlet of the ESP. A positive voltage of 10 kV was applied to the upper plate, and the resulting field forced incoming charged particles towards the lower grounded substrate of the ESP. The ESP for cell exposure, shown in Figure 1c, was designed to deposit charged particles onto cell cultures grown on a commercial cell culture insert (FALCON 35-3090, Becton Dickinson Labware, NJ). This design is similar to that described in an elegant system recently developed by Savi et al. (2008) using an alternating electrical field to deposit PM on cell cultures. A particle inlet tube of 19.05 mm O.D. was placed at the center of the upper cylindrical Delrin housing and at a distance of 6 mm from the bottom of the cell culture insert. At the end of the tube, a fine metal mesh (20  $\times$  20, 0.4 mm wire), acting as a grounded electrode, was placed to produce a parallel electric field between the tube-end and the bottom of the insert. The bottom of the lower Delrin housing on which the culture medium is placed was made of stainless steel (SS). A negative voltage of 3 kV was applied to the SS plate to collect positively charged particles on the bottom of the insert. To distinguish these two designs, we will refer to the first ESP for sampling of atmospheric aerosols as “PM-ESP” and to the second ESP for cell exposures as “Cell-ESP,” respectively. Other than the type of the collection electrode, the two designs are identical in all other aspects.

Figure 2 shows the experimental setup, consisting of five parts, i.e., particle generation, particle concentration (i.e., VACES), particle charging, particle collection, and particle

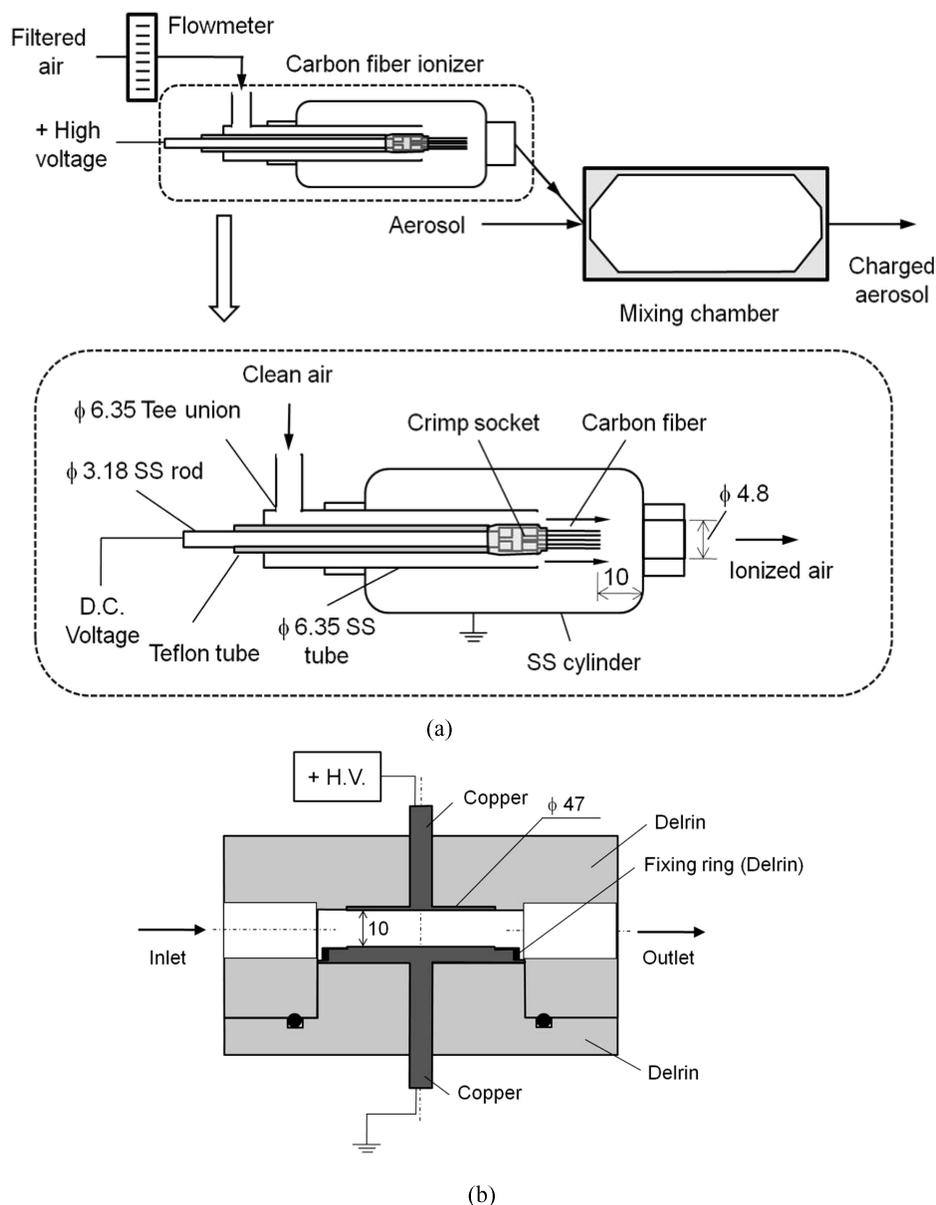


FIG. 1. Schematics of (a) carbon fiber charger, (b) PM-ESP, and (c) Cell-ESP developed in this study. All dimensions are in mm.

measurement systems. Sodium chloride (NaCl) particles were generated by nebulizing NaCl solutions with a nebulizer (VORTRAN Medical Technology, Inc., Sacramento, CA), to which filtered compressed air with a pressure of about 1.5–2 bar and a flow rate of about 5 l/min was introduced and the generated particles were mixed with particle-free filtered air. The particles passed through the VACES system at 100 l/min, in which the sampled aerosol was introduced over a pool of ultrapure (Milli-Q) deionized water (resistivity 18.2 megaohm; total organic compounds <10 ppb; particle-free; bacteria <1 colony forming unit/ml) maintained at 28–30°C, and mixed with saturated water vapor, and then passed through a cooling condenser

maintained at  $-5$  to  $-6^{\circ}\text{C}$ . The aerosol is cooled in the condensers to roughly  $21$ – $22^{\circ}\text{C}$ ; the produced supersaturation allows particles to grow to micrometer-sized droplets (typically with a geometric mean size of  $3\ \mu\text{m}$ ;  $\text{GSD} = 1.8$ ) by means of condensation of water vapor onto the incoming particles. The grown particles are drawn through a custom made virtual impactor (Sioutas et al. 1999), with a designed 50% cut-point of  $1.5\ \mu\text{m}$  in aerodynamic diameter, with major and minor flows of 94 l/min and 6 l/min, respectively. Inertial forces concentrate the particle-containing droplets into the minor flow of the virtual impactor, while the major flow is drawn away with a vacuum pump. The concentration factor in this case is ideally 16.7-fold

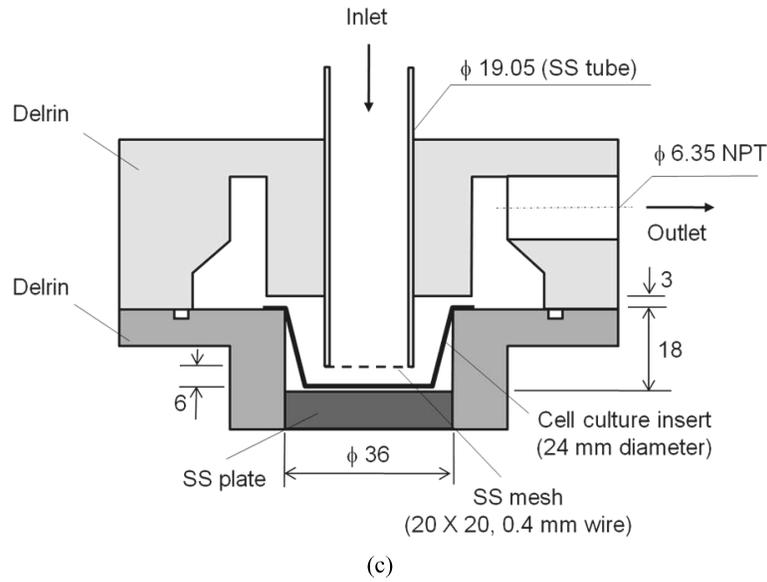


FIG. 1. Continued

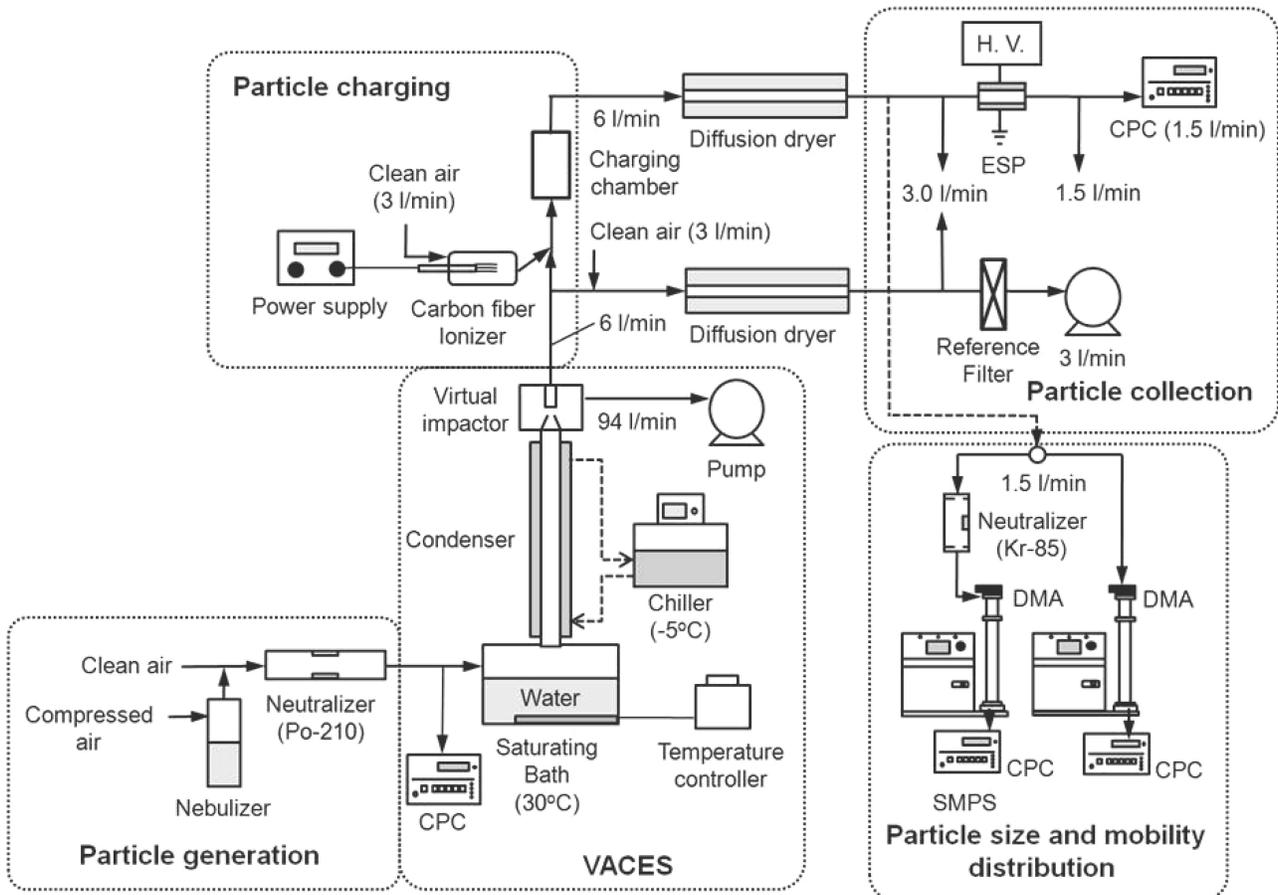


FIG. 2. Experimental setup for this study, showing the particle concentrator–electrostatic precipitator assembly.

(i.e., the ratio of 100 l/min to 6 l/min). The concentrated and enlarged particle stream from the minor flow of the impactor passes through a particle charger equipped with a carbon fiber ionizer as described in Figure 1a with a flow rate of 3 l/min. The charged particles pass through a diffusion dryer (Model 3062, TSI Inc., Shoreview, MN) that removes the excess water and returns the particles to their original size while preserving their acquired charges. The charged particles are then introduced in the ESPs as described in Figure 1b and c at a flow rate of 3 l/min for PM-ESP and 0.3–1.5 l/min for Cell-ESP. A scanning mobility particle sizer (SMPS; Model 3936, TSI Inc.) consisting of a Kr-85 neutralizer (Model 3077, TSI Inc.), a differential mobility analyzer (DMA, Model 3081, TSI Inc.), and a condensation particle counter (CPC; Model 3022A, TSI Inc.) measured the particle size distributions. Another SMPS system without a neutralizer was used to measure the particle mobility distributions after charging. Sheath flow and aerosol flow of the two DMAs were 15 l/min and 1.5 l/min, respectively. The remaining flow of 3 l/min is drawn to another diffusion dryer followed by a reference filter.

Following the laboratory experiments, the particle concentrator–electrostatic precipitator system was deployed inside the particle instrumentation unit (PIU) trailer of the University of Southern California to conduct a field evaluation, using atmospheric particles as the test aerosols. The PIU is located in an urban area about 150 m downwind of a major freeway and about 3 km south of downtown Los Angeles, CA. A total of 6 sets of ambient particle samples were collected on pre-cleaned 47 mm Teflon filters (PTFE, Pall Life Sciences, Teflon w/ring, PTFE membrane, porosity 2.0  $\mu\text{m}$ ), placed on the PC-ESP substrate, and on identical filters, sampling in parallel to the PC-ESP, placed downstream of the diffusion dryer of the VACES and upstream of the PC-ESP. The reference filters sampled at a flow of 3 l/min. Sampling for each experimental set was restricted to 2–3 h to reflect sampling conditions that are necessary to preserve cell viability in future toxicity studies (see also our discussion on this topic in our paper by Sillanpaa et al. 2008). Similarly to our earlier work discussed in Ning et al. (2008), the PTFE filters of PC-ESP and reference sampler were weighed with a microbalance (Model MT 5, Mettler-Toledo Inc., Hightstown, NJ, USA) before and after sampling in a weighing room with controlled relative humidity (RH) and temperature. The samples were stored in the freezer at 20°C and were submitted for chemical analysis at the end of the sampling campaign. Off-line chemical analyses on the filters/substrates included ion chromatography (IC) for the analysis of inorganic ions (chloride, nitrate, phosphate, ammonium, and sulfate), selected trace elements measured via inductively coupled plasma-mass spectroscopy (ICP-MS), and water soluble organic carbon (WSOC) using a Shimadzu TOC-5000A liquid analyzer (Decesari et al. 2001). The details of the procedures used for handling and processing the samples during IC and ICP-MS analyses are described in detail in Lough et al. (2005). Filters from the 6 field runs were combined into two composited sets of samples. Each filter was cut in four equal sec-

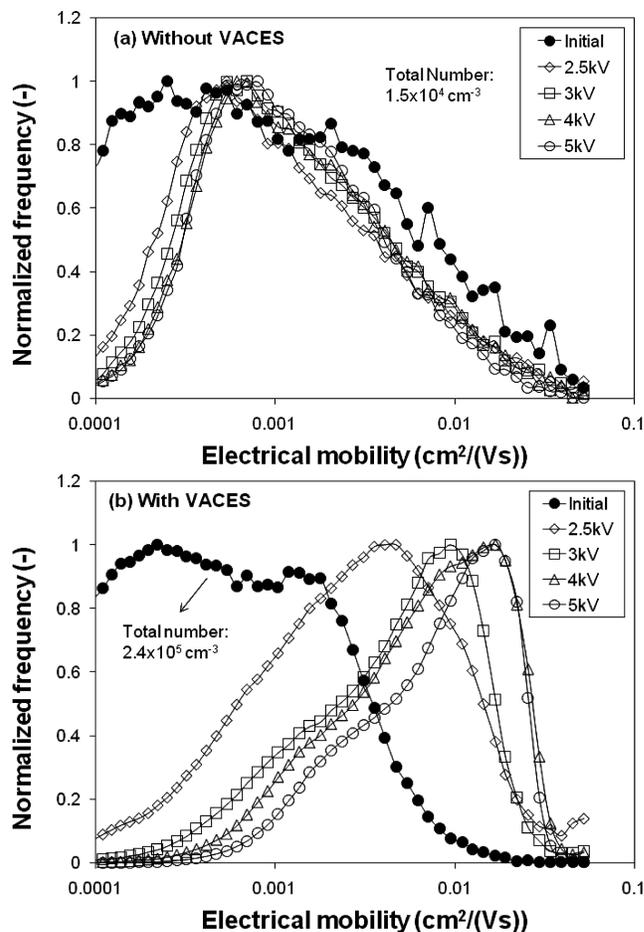


FIG. 3. Mobility distributions of charged particles (a) without and (b) with the VACES.

tions;  $1/4$  was analyzed for IC,  $1/4$  for WSOC, and  $1/2$  for ICP-MS, respectively.

## RESULTS AND DISCUSSION

### Particle Mobility Distribution

Mobility distributions for particles charged by the carbon fiber charger with and without the VACES are shown in Figure 3a and b, respectively. Total particle number concentration was  $1.5 \times 10^4$  particles/cm<sup>3</sup> for the pre-enriched particles without VACES and  $2.4 \times 10^5$  particles/cm<sup>3</sup> for the post-enriched particles with VACES, and thus the enrichment factor was about 16, which was quite similar to the ideal enrichment of 16.7. A peak mobility value was observed at about  $2\text{--}4 \times 10^{-4}$  cm<sup>2</sup>/(Vs) for initial NaCl particles with the mean diameter of about 85 nm. The mobility peaks shifted to larger values ( $5\text{--}9 \times 10^{-4}$  cm<sup>2</sup>/(Vs)) when higher voltages of 2.5–5 kV were applied to the charger without the VACES. By comparison, when the charger was preceded by the VACES, mobility peaks shifted to markedly higher values (up to  $1.5 \times 10^{-2}$  cm<sup>2</sup>/(Vs)) for an applied voltage of 5 kV to the charger. A higher applied voltage

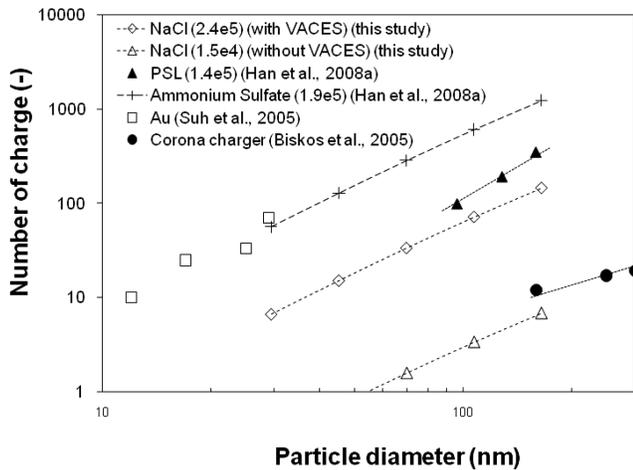


FIG. 4. Number of charges for particles charged with and without the VACES before the charger, and comparison with previous studies. Number in parentheses of the explanatory notes in the figure indicates particle number concentration.

led to a higher number of charges acquired by particles, and thus to an increased electrical mobility. The difference in the number of charges per particles with and without the VACES indicates that, for the same charging voltage, the number of charges is significantly increased when charging occurs after these particles are enlarged to super micrometer droplets by condensational growth in the VACES.

#### Number of Elementary Charges Per Particle

The number of charges acquired by particles is shown in Figure 4 as a function of particle size. The number of charges per particle was obtained from the mobility distributions of the charged aerosol, according to the methodology that we have discussed in detail in our previous work (Han et al. 2008a). Results from previous studies, including our own earlier work, were also plotted in Figure 4 for comparison. When the carbon fiber charger was used with VACES, the number of acquired charges was about one order of magnitude higher compared to those obtained without the VACES. This is because the incoming particles grow to larger, micron-sized droplets in the VACES, and thus acquire a higher number of charges than they would at their original (smaller) size. As the concentrated droplets are dried by diffusion drying, they are returned to the original particle size distribution while preserving the charges acquired as a droplet. However, the numbers of charges acquired with the VACES in this work was kept intentionally to a lower level (by almost 10-fold) than our previous study using the same charger (Han et al. 2008a). In that study, particles were charged in the ionization zone via direct charging; an approach similar to the work of Suh et al. (2005) utilizing the principle of particle condensational growth. The lower number of charges in our present configuration is attributed to the modifications in the charger configuration, i.e., separation of the charging region from the ionization region. Ion concentrations in the charging

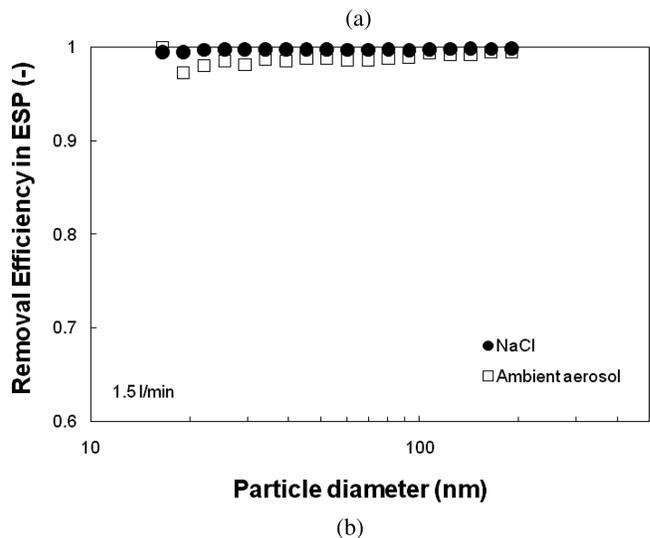
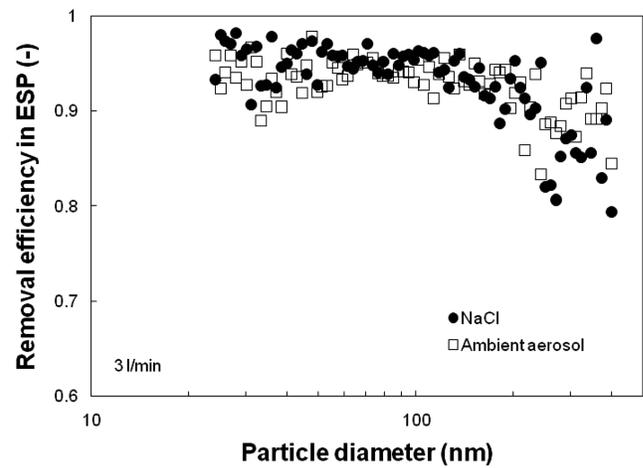


FIG. 5. Particle removal efficiency in (a) PM-ESP and (b) Cell-ESP as a function of particle size.

region are substantially lower than previous studies, since a significant ion loss occurs during the transport of ions from the ionization to the charging zone. Nonetheless, the number of acquired charges per particle in the present study is still considerably higher than that obtained from conventional corona chargers (Biskos et al. 2005).

#### Removal Efficiency in ESP

The particle removal efficiency of the two ESPs as a function of particle size is shown in Figure 5a and b for particles charged after exiting the VACES. The removal efficiency was measured by comparing the size distributions obtained by the SMPS (Model 3936, TSI Inc., St. Paul, MN) upstream and downstream of the ESPs. Removal efficiencies greater than 90% were accomplished for particles smaller than 200 nm in the PM-ESP at 3 l/min, whereas more than 95% removal was achieved for particles smaller than 200 nm in Cell-ESP at

1.5 l/min. These results imply that ultrafine particles (smaller than 100 nm), which have been traditionally difficult to charge by means of conventional charging techniques, can be effectively charged and collected in both concentrator-ESP systems. The loss in efficiency at higher sizes in Figure 5a is due to the relatively lower electrical mobility of larger particles (0.2–0.4  $\mu\text{m}$ ) as they require higher number of charges per particle to acquire comparable electrical mobility. However, the collection efficiency does not decrease below 80%, since both gravimetric and chemical analyses, based on a variety of chemical species in PM<sub>2.5</sub>, indicated an average collection efficiency of about 80–85% for the PM-ESP. The mass median diameter of several of these species in PM<sub>2.5</sub> in Los Angeles is typically in the 0.4–0.8  $\mu\text{m}$  range (Sardar et al. 2005).

### Collection Efficiency on the ESP Substrates

Particle collection efficiency was measured by comparing either the particle mass, or fluorescent intensity of particles collected on the ESP substrates, to that of a reference filter in a parallel line. The collection efficiency refers to the fraction of particles introduced in the ESPs that actually deposit on the substrates, whereas the removal efficiency discussed in the previous section refers to the particle fraction that does not penetrate the ESPs. In the case of zero or negligible internal losses in the ESPs, the two efficiencies are identical. These tests were conducted to confirm that particles are collected on the targeted substrates and are not lost elsewhere in the ESPs. Figure 6a compares the particle mass and fluorescent intensity for aluminum foil or Teflon filter substrates used in the PM-ESP to reference filter. For the fluorescence test, monodisperse fluorescent polystyrene latex (PSL) particles (Polyscience Inc., Warrington, PA, USA) of 100 nm in diameter were used as a test aerosol, and a UV spectrometer (Model FD-500, GTI, Concord, MA, USA) was used to analyze the fluorescent intensity of the particles. Ethyl acetate (99.5%, Fischer Scientific) was used for the PSL extraction solution. The fluorescence spectrometer has a capacity to detect 1  $\mu\text{g}$  of PSL particles per ml solution and it is linear up to about 1.5 mg of particles per ml of ethyl acetate solution (Sioutas et al. 1994). Resultant fluorescent intensities were compared and after each measurement the system was flushed with pure ethyl acetate. The collection efficiency of fluorescent PSL particles was similar to that of NaCl particles based on gravimetric measurements; about 80–85% of total particles were captured on the PM-ESP substrate. This value is slightly lower than the removal efficiency based on particle number measurements, which, for particles in that size range was about 80–95%, as shown in Figure 5a. The difference may be a result of particle losses on the insulating parts of the PM-ESP and possibly on other surfaces (i.e., tubing) during transport. Overall, however, these losses are not substantial, as they represent less than 10% of difference between the collection and removal efficiencies of the PM-ESP.

Figure 6b shows the collection efficiency of fluorescent PSL particles of different size for the Cell-ESP at a flow rate of 0.3

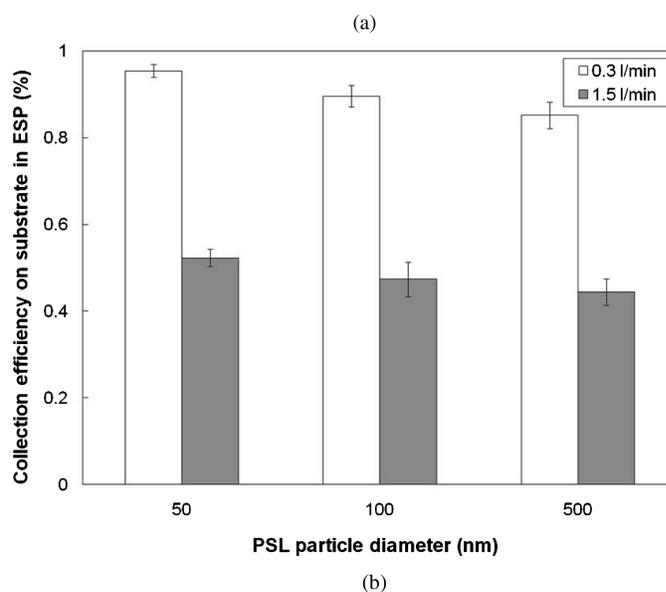
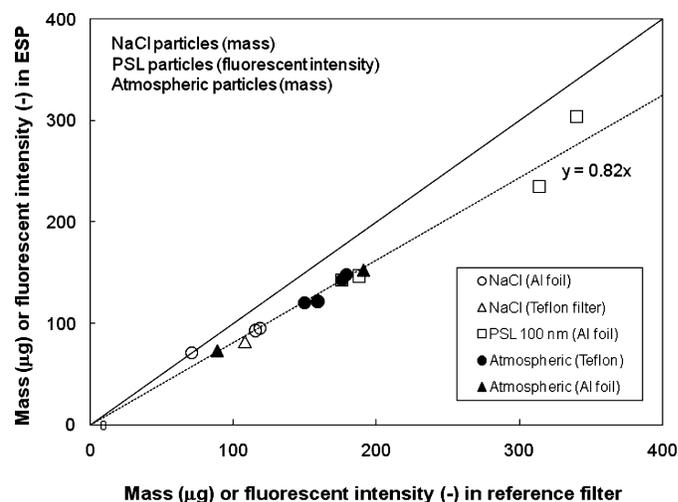


FIG. 6. (a) Comparison of mass or fluorescent intensity of particles collected on the PM-ESP substrate with those on a reference filter in laboratory and field tests, and (b) collection efficiency of fluorescent PSL particles on the substrate of the Cell-ESP.

and 1.5 l/min. About 85–95% of the particles were collected on the 4.52  $\text{cm}^2$  substrate at the bottom of the insert in the Cell-ESP at 0.3 l/min. However, the efficiency decreased to 45–55% at higher flow rate of 1.5 l/min. These values are lower than the removal efficiency (>95%) based on particle number concentration, as shown in Figure 5b. We found that more than 40% of the particles entering the Cell-ESP were collected on the wall of the cell culture insert at 1.5 l/min because of the short residence time in the collection region. Nonetheless, even at 1.5 l/min, these collection efficiencies are still considerably higher compared to those (15–30%) obtained in the recently developed ESP system for cell exposures by Savi et al. (2008), even though the flow rate of our system (1.5 l/min) is 30 times higher than that (0.05 l/min) of Savi et al. (2008).

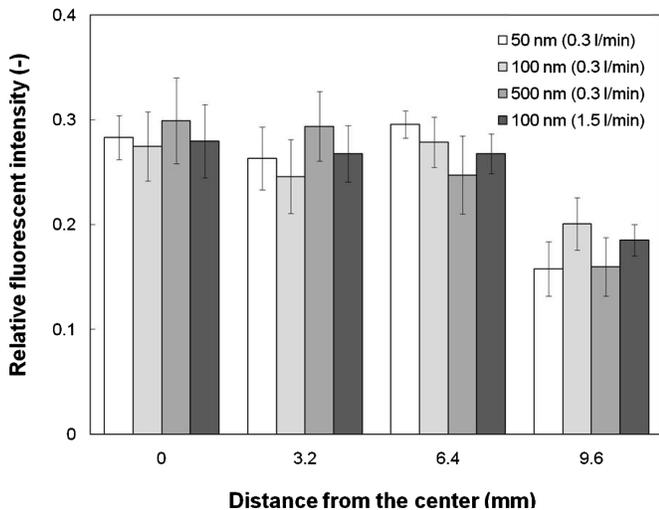


FIG. 7. Uniformity of deposited particles on the substrate of the Cell-ESP.

### Uniformity of Particle Deposition in the Cell-ESP

For the Cell-ESP system, we investigated the uniformity of particle deposition on the ESP substrate, because this is an important consideration in developing particle collectors for cell exposures to PM, as it assures that all cells are exposed to similar PM loadings. Figure 7 shows the uniformity of the PSL particles deposited on the cell culture insert of the Cell-ESP. For this test, the insert was covered with a thin layer of Al foil, on which fluorescent particles were deposited. For subsequent analysis the foil was later cut into. Four square pieces, each  $3.2 \times 3.2 \text{ mm}^2$  were cut at 0, 3.2, 6.4, and 9.6 mm from the center of the substrate, and the fluorescent intensity of the deposited particles on each square was compared. As indicated in Figure 7, the deposited particles were evenly distributed over the substrate's surface in the cell culture insert, irrespective of particle size and gas flow rate, except for the outer edges of the insert. The differences were within 5% for the inner three positions from the center, and about 6–12% lower than the average at the inner three positions at the outer edge. This decrease at the edge is probably due to the higher gas velocity in that area. Acknowledging the requirement for comparably uniform deposition of particles on the cell culture to ascertain uniform dose, the authors suggest seeding of cells uniformly over the tissue culture dish instead of a seeding in the middle to accommodate for the reduced particle deposition at the edges (Savi et al. 2008).

### Ozone Emission

Ozone concentrations during our tests were measured by means of a UV photometric ozone monitor (Model 1003-AH, Dasibi Environmental Corp., Glendale, CA) at the outlet of the charger. The ozone concentration at charger voltages less than +4.0 kV was below the detection limit (2 ppb) of the  $\text{O}_3$  monitor. Detectable ozone concentrations were measured at +4.5 kV, and reached a level of 14 ppb at +5.0 kV, but quite negligible compared to the National Ambient Air Quality Standards

(NAAQS) of United States for ozone (80 ppb). The measured concentrations indicate exceptionally low ozone production in the ESP, even when such high voltages are applied to the charger.

In conventional corona discharge ESP, ozone concentrations were reported to be in the range of 30–90 ppb in a previous study by our group (Sillanpää et al. 2008), which had also used the VACES with a conventional charger to sample and collect concentrated ambient particles. Ozone concentrations were about 200–300 ppb and 125 ppb, respectively, in previous studies by Kaupp and Umlauff (1992) and Volckens and Leith (2002), all using ESPs as an alternative to filters for sampling and collecting of atmospheric particles. As we discussed earlier, such high ozone concentrations and the associated generation of free radicals may degrade organic compounds, such as PAHs, alkenes and other organic species, via chemical reactions, which may limit or even preclude the use of ESPs as a sampler of atmospheric particulate matter.

### Collection of Atmospheric Aerosols Using the PM-ESP

The amounts in  $\mu\text{g}$  of atmospheric particles collected on the PM-ESP substrate as well as on the reference filter in the field tests are shown in Figure 6a as well. Similarly to the laboratory tests, about 80–85% of atmospheric particles entering the PM-ESP were collected on the targeted substrate of ESP, regardless of whether the substrate was aluminum foil or Teflon filter.

Figure 8 shows results from the ICP-MS analysis for trace elements, using Teflon filters as substrates on the PM-ESP and reference filters. The plotted data in the figure indicate very good overall agreement between the PM-ESP and the reference filter sampler, with PM-ESP values being on average  $0.81 (\pm 0.12)$  of the reference values for the 24 elements shown in the figure. For most elements, the ratios varied in the 0.8–1.1 range. Slightly lower concentration ratios between PM-ESP and reference samplers were obtained for Ni, Cu, and Zn, with values

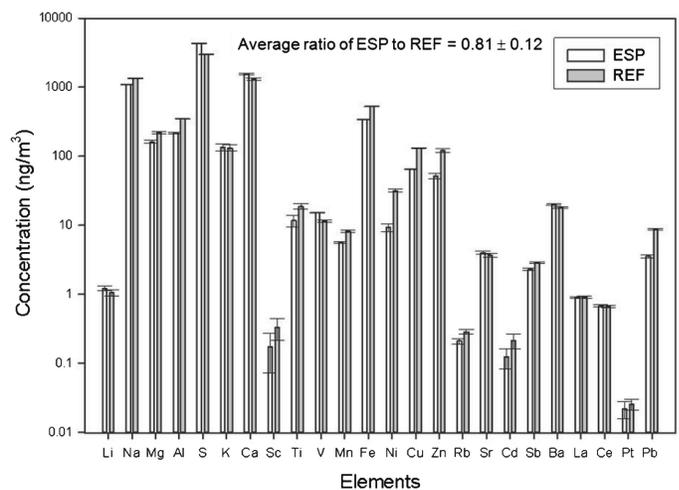


FIG. 8. Comparison of the concentrations of trace elements collected on the substrate in PM-ESP with those on reference filter in field tests.

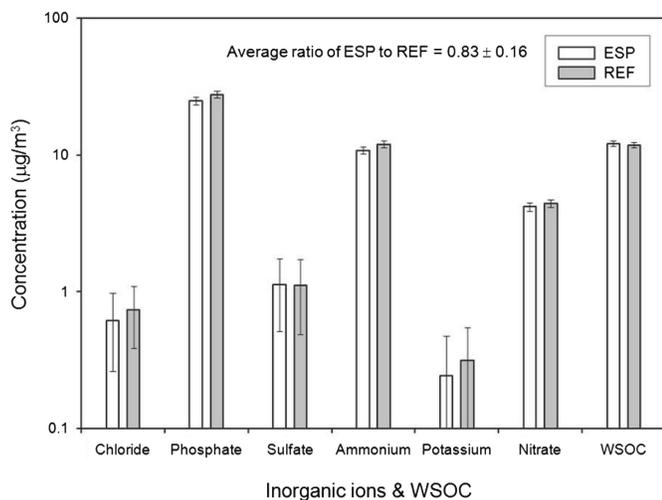


FIG. 9. Comparison of the concentrations of inorganic ions and water soluble organic carbon (WSOC) collected on the substrate in PM-ESP with those on reference filter in field tests.

ranging from 0.58–0.75. We do not have an obvious explanation for these lower concentrations in the PM-ESP. The concentration ratio values are consistent with the PM-ESP collection efficiency estimated from gravimetric measurements, discussed earlier. The analytical uncertainty is indicated on the graph as error bars. However, it is possible that the differences in part are due to the possible reactions (oxidation/reduction) caused by ions generated in corona discharge and interference caused by products in the ICP-MS analysis. Further, considering the analytical uncertainties associated with the low ambient values of some of these elements, the obtained agreement within the experimental data should be considered very good.

Figure 9 presents results from the comparison between the PM-ESP and reference filter sampler based on inorganic PM ions as well as water-soluble organic carbon (WSOC). The ions analyzed were chloride, nitrate, phosphate, sulfate, sodium, ammonium, and potassium. Similarly to the results based on trace elements, the data in Figure 9 are on a log scale to accommodate the differences in magnitude of the ambient concentration of these species. The linear regression between the ESP concentration and the reference filter gives a slope of 0.83 and a high regression coefficient ( $R^2$ ) value of 0.94. The average PM-ESP to reference filter sampler ratio was 0.83 ( $\pm 0.16$ ) for these species. The results shown in Figures 8 and 9 confirm that the concentration ratios between the PM-ESP and reference sampler for inorganic ions, water-soluble organic carbon and trace elements agree very well with the gravimetrically obtained concentration ratios (80–85%). The consistent agreement across a variety of PM species indicates that the particle concentrator–electrostatic precipitator system is efficient in collecting ambient aerosols while preserving their chemical composition.

## SUMMARY AND CONCLUSIONS

We have developed a novel particle collection methodology, the particle concentrator–electrostatic precipitator, by combining an ozone-free carbon fiber charger with a particle concentrator, (VACES). The number of charges acquired by particles and their collection efficiency has been evaluated experimentally, and was compared to previous studies. We found that our methodology can collect ambient atmospheric particles on the substrates of the ESPs with high efficiency and uniformity, and with little generation of ozone, thus with negligible chemical artifacts. Our experimental results suggest that this technology is a promising tool for applications involving sampling and collection of atmospheric aerosols for chemical and toxicological analysis, including the possibility of conducting direct in vitro cell exposures. Moreover, the concentration enrichment of ambient aerosols by the VACES allows for short-term exposure studies, which preserves cell viability and enables studies to PM generated from specific sources and/or formation mechanisms in the atmosphere, which occur within short time scales.

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