Long Term Black Carbon Measurements at Two Urban Locations in New York

Oliver V. Rattigan1*, Kevin Civerolo1, Prakash Doraiswamy2+, H. Dirk Felton1, Philip K. Hopke3

1 New York State Department of Environmental Conservation, Division of Air Resources, 625 Broadway, Albany, NY, USA
2 Atmospheric Sciences Research Center, University at Albany, Albany, NY, USA
3 Center for Air Resources Engineering and Science, Clarkson University, Potsdam, NY, USA

ABSTRACT

Measurements of PM2.5 black carbon (BC) over an 8–9 year period are used to characterize temporal patterns at sites in New York City (NYC) and Rochester, NY. Annual mean BC at the NYC location ranges from 1.4 to 2.0 μg/m³, whereas mean concentrations at Rochester are approximately a factor of 2–3 lower. BC amounts to 15–20% of PM2.5 mass in NYC compared to 7–10% at Rochester. Seasonal patterns reveal the highest BC concentrations in NYC from November to February versus June to November at Rochester. At both locations, mean weekday (Monday-Friday) BC concentrations are statistically higher compared to weekends (Saturday and Sunday). Weekday BC diurnal profiles exhibit a morning peak between 6–10 AM EST followed by an afternoon minimum, with a secondary peak in the late evening. Sunday BC diurnal profiles show highest concentrations at night, from 8 PM to 2 AM. These patterns are consistent with vehicle counts on nearby roadways and boundary layer dynamics at both locations. Simultaneous measurements of BC at 370 and 880 nm show an enhancement in BC370 relative to the BC880 from October to March. This enhanced signal is most evident at Rochester during late evening and early morning hours (8 PM to 4 AM) on weekends, and is attributed to UV absorbing species (such as wood smoke markers) in the ambient particle mixture. At the NYC site, the levels of Nickel (Ni) and Cobalt (Co) in PM10 are elevated during the heating season due to residual oil combustion. The long term datasets are used to explore the seasonal relationship between BC880 and EC at both sites.

Keywords: Black carbon; Aethalometer®; Diurnal pattern; Wood smoke; Nickel.

INTRODUCTION

Black carbon (BC) generally refers to the portion of ambient particles that absorbs strongly in the visible and near infrared. This quantity is also referred to as “light absorbing carbon” or LAC (Malm et al., 1994). Elemental carbon (EC) is operationally defined by thermal optical analysis methods whereby a particulate sample is oxidized and the evolved gas quantified relative to a fixed methane concentration (Birch and Cary, 1996; Chow et al., 2007). Since BC and EC are method dependent definitions simultaneous measurements are not necessarily equivalent due to variations in optical properties of the ambient aerosol depending on the composition, source, and degree of aging (Lioussse et al., 1993; Jeong et al., 2004; Park et al., 2006; Salako et al., 2012). Interferences from organic species (Wallen et al., 2010) can impact thermo-optical measurements of EC. BC or EC are produced from fossil fuel combustion in, for example, motor vehicles including diesel engines, heating and biomass burning (Hildemann et al., 1991; Seinfeld and Pandis, 1998 and references therein; Schauer, 2003; Bond et al., 2007) and are considered indicators of primary combustion emissions. In the US, approximately 50% of BC emissions are from the transportation sector with the majority from diesel combustion sources (http://www.epa.gov/blackcarbon).

Carbonaceous particles can play an important role in climate processes (Jacobson 2002; IPCC 2007; Ramathan and Carmichael 2008) after cloud properties (Albrecht, 1989), impact air quality and human health. Residents living near major roadways and exposed to vehicle combustion particles including BC may be at increased risk for respiratory symptoms and reduced lung function (Kim et al., 2004; Cornell et al., 2012). Measurements of BC are important in order to understand the major sources and subsequent evolution in the atmosphere. Long term measurements are of key interest in the evaluation of emission control measures and to assess impacts on health and climate.

Several different optical methods are used to measure
BC. Among the time-resolved filter based methods, the Aethalometer® (Hansen et al., 1984) is one of the most extensively used instruments. Alternative filter based methods include the Multiangle Absorption Photometer, MAAP (Petzold et al., 2002), and the Particle Soot Absorption Spectrometer, PSAP (Bond et al., 1999). These methods operate by measuring a change in optical light transmission through a fiber filter as it samples ambient particles. The optical response of these instruments may depend on the mixing state of black carbon particles. For example optical saturation under heavy BC loadings leads to an underestimation in concentration and interference from particles other than BC may contribute to light absorption (Kirchstetter et al., 2004). The data presented here have been corrected for optical saturation using the Virkkula et al. (2007) algorithm. In the US BC has been measured (using the Aethalometer®) along with certain hazardous air pollutants (HAPs) at key areas (predominantly urban) designated as National Air Toxics Trends Station (NATTS) sites (listed at http://www.epa.gov/ttn/amtic/natts.html). This monitoring program was designed to characterize trends in ambient HAP levels in support of human exposure assessments and for the evaluation of emission control strategies.

The focus of this paper is to characterize the temporal patterns in PM$_{2.5}$ BC using long-term measurements from two urban locations in New York. The data were used to examine annual, seasonal, day of week and diurnal patterns and the relationship with co-pollutants in an effort to understand major source impacts. In addition, we explore the seasonal relationship between BC and EC from different measurement methods.

**METHODS**

**New York City, Bronx**

The ambient air monitoring site in New York City is at Intermediate School 52 (IS 52), 681 Kelly Street in the South Bronx, NY (Rattigan et al., 2010). The site grid reference is 40°48'57"N and 73°54'07"W and the USEPA site identification code is 36-005-0110. This site lies in a mainly residential area but is impacted by traffic from several nearby highways. These include the Bruckner Expressway (I-278) approximately 0.5 km to the southeast and Interstates 87 (Major Deegan), 895 and 95 all which lie within 1–3 km of the site. The New York State Department of Transportation (http://www.dot.ny.gov) annual average daily traffic (AADT) counts on these roadways range from 110,000–160,000. Hunts Point produce market and Oak Point freight yard are within 1–2 km of the site, leading to significant heavy duty diesel truck traffic on connecting roadways. The Bronx has a total population of approximately 1.3 million inhabitants. The site is one of two NATTS sites in New York and contains an extensive set of ambient monitoring equipment for gaseous and particle sampling (http://www.dec.ny.gov/airmon). Air sampling at the site was suspended for an approximate two year period from July 2010 to June 2012 due to roof repair work. During this period measurements of BC were carried out at a site approximately 7 km to the northwest (site identification code 36-005-0080).

**Rochester**

Rochester lies on the southern shore of Lake Ontario approximately 400 km northwest of New York City. Ambient measurements are performed at the New York State Department of Environmental Conservation (NYS DEC) site in Rochester, New York (43°08’46"N, 77°32’52"W). Equipment is housed in or positioned on the rooftop of an air conditioned shelter located on National Grid property off Yarmouth Road (Wang et al., 2010). This site is the second NATTS site in the state (USEPA site identification code 36-055-1007). The site is located in Brighton on the east side of the urban center adjacent to the intersection (~300 m) of two major highways (I-490 and I-590) with AADT counts ranging from 100,000–120,000. Recent estimates indicate the population of Rochester is approximately 210,500 (U.S. Census Bureau, http://www.census.gov/), making it the third largest city in New York State.

**BC Instrumentation**

Black carbon measurements began at the Bronx site in May 2003 and at the Rochester site in May 2004. At both sites a Magee Scientific, model AE-21, dual wavelength (880 nm and 370 nm) Aethalometer® (Hansen et al., 1984) sampled PM$_{2.5}$ particles through a BGI model 1.828 sharp cut cyclone at 5 LPM onto a quartz fiber filter tape (Pallflex®). The Aethalometer® calculates light attenuation (ATN) due to particle deposit on the filter relative to a clean part of the filter. By measuring the rate of change in light attenuation assumed due solely to absorption by BC the mass concentration is determined. A 5 min time interval and an absorption coefficient of 16.6 m$^2$/g at 880 nm were used in this study. However, the relationship between ATN and BC may become non-linear (Petzold et al., 1997; Bond et al., 1999; Weingartner et al., 2003; Arnott et al., 2005; Turner et al., 2007; Virkkula et al., 2007; Park et al., 2010). The Aethalometer® accumulates sample on a portion of the filter tape and as the BC loading increases a change in the optical properties of the particle laden filter causes the ATN to BC response to become non-linear. This manifests itself in a saw-tooth like signal, where the BC concentration increases immediately following a tape advance, i.e., ATN increases immediately following a tape advance, i.e., ATN itself in a saw-tooth like signal, where the BC concentration loading increases. The empirical algorithm developed by Virkkula et al. (2007) attempts to correct for this effect;

$$BC_{corrected} = (1 + K \times ATN)BC_0$$  \hspace{1cm} (1)

where K is the correction factor and $BC_0$ is the uncorrected BC. The correction factor was calculated for each filter spot i, using the average of the last two data points of spot i and the first two data points of spot i + 1 according to Eq. (2);

$$K_i = \frac{1}{ATN(t_{i,last})} \left( \frac{BC_0(t_{i+1,first})}{BC_0(t_{i,last})} - 1 \right)$$  \hspace{1cm} (2)
Five minute Aethalometer® raw data were corrected and processed into hourly intervals using software developed by the Air Quality Laboratory at Washington University (Turner et al., 2007).

A Thermo Scientific model 5012 MAAP was also used for measurements of BC (670 nm) over an approximately two and a half year period, 2007–2009, at the Bronx site. The MAAP sampled particles through a PM$_{2.5}$ sharp cut cyclone at 16.7 LPM onto a glass fiber filter tape (Whatman®, GF10). Visible light from a 670 nm source was directed onto the particle laden filter. Light transmitted through the filter into the forward hemisphere and reflected into the back hemisphere was measured by a series of photodetectors. A radiative transfer algorithm accounted for multiple scattering effects (Petzold et al., 2002). A mass absorption cross section of 6.6 m$^2$/g at 670 nm was used to estimate the BC concentration.

Co-Pollutant Measurements

Bronx Site: A model 3 Sunset Laboratory Inc. OCEC carbon aerosol analyzer collected particles through a BGI, model SCC 2.354, PM$_{2.5}$ sharp cut cyclone at 8 LPM which provided hourly measurements of EC and OC (Rattigan et al., 2010). Interagency for Monitoring Protected Visual Environments (IMPROVE) 24 h integrated PM$_{2.5}$ particle sample collection (Malm et al., 1994) for chemical analysis was also carried out at the Bronx site from 2004 to 2010. IMPROVE carbon samples were analyzed according to the thermal optical reflectance (TOR) method (Chow et al., 2007). Ambient measurements of nitrogen oxides, (NO, NO$_2$, and NO$_x$ = NO + NO$_2$), SO$_2$ and O$_3$ were made using Thermo Scientific gas analyzers, models 42C, 43C and 49C, respectively. Hourly PM$_{2.5}$ total mass was measured using a Thermo Scientific Tapered Element Oscillating Microbalance (TEOM) model 1400ab. Sub-hourly measurements of PM$_{2.5}$ sulfate and nitrate were made using a Thermo Scientific model 5020i Sulfate Particulate Analyzer and a Rupprecht & Patashnick Co., Inc. model 8400N Ambient Particle Nitrte Monitor, respectively.

Measurements of SO$_2$, CO, O$_3$ and PM$_{2.5}$ mass are recorded on hourly intervals at Rochester. At both sites 24 h integrated carbonaceous measurements were made every third day employing the Chemical Speciation Network (CSN) sampling protocol allowing for a comparison between BC and EC. Two different CSN sampling and analysis protocols are used for 24 h carbon sampling as described previously (Rattigan et al., 2011). CSN 24 h carbon sampling was carried out using either a Rupprecht and Patashnick Co. Inc. Partisol model 2300 Speciation Sampler (R&P 2300) or a Spiral Aerosol Speciation Sampler, Met One Instruments (MetOne). Both instruments employed prefired 47 mm diameter Whatman® QMA quartz fiber filters with analysis based on the NIOSH 5040 thermal optical transmission (TOT) protocol (Peterson and Richards, 2002). CSN 24 h carbon filter sampling was changed to the URG-3000N Sequential Particulate Speciation System, (URG) in May 2007 at the Bronx and in April 2009 at Rochester. The URG-3000N uses a similar protocol to the IMPROVE with 25 mm Pallflex® Tissuquartz filters and analysis based on the IMPROVE A thermal optical reflectance (TOR) method (Chow et al., 2007). Chemical speciation sampling at the Bronx ceased in June 2010. PM$_{2.5}$ mass was collected over 24 h periods using the Thermo Scientific Partisol Plus 2025 Sequential Ambient Particle Sampler. PM$_{10}$ samples were also collected over 24 h periods using a Partisol Plus 2025 sampler every 6$^{th}$ day and analyzed for selected trace metals. Sample inlets were positioned 2 m above rooftop (approximately 15 m above ground level at the Bronx and 5 m at Rochester).

Planetary Boundary Layer Heights

The planetary boundary layer (PBL) heights were calculated in the Nonhydrostatic Mesoscale Model of the Weather Research and Forecasting System (WRF-NMM) using the Mellor–Yamada–Janjic Parameterization (Janjic, 2001). The weather forecast product consisted of the 12:00 coordinated universal time cycle forecasts of the National Center for Environmental Prediction (NCEP) operational 12-km, WRF-NMM based North American Mesoscale model, which had already been processed by the NCEP WRF postprocessor. PBL heights were extracted for the grid cells in which the air monitoring stations were located (Doraismwamy et al., 2010).

RESULTS AND DISCUSSION

Aethalometer® Loading Correction Factor and Comparison to MAAP

BC$_{880}$ data were corrected for particle loading based on the Virkkula et al. (2007) method. Mean K factors at 880 nm from Eq. (2) varied from approximately 0.001 in summer (June, July and August) to 0.012 in winter (December, January and February) at the Bronx (Fig. 1) and 0.0015 to 0.008 at Rochester. This amounted to a correction in BC$_{880}$ ranging from approximately 50% in winter to 15% in summer. The seasonal variation in the K factor over a two year period at the Bronx is shown along with 24 h integrated sulfate and the OC/EC ratio in Fig. 1. A higher contribution of scattering components e.g., sulfate and secondary organic carbon to the total particle mass in summer offsets the loading effect as previously indicated (Weingartner et al., 2003; Virkkula et al., 2007). A negative K indicates the raw BC data are underestimated. Hourly Aethalometer® BC$_{880}$ (raw and corrected) data are compared to MAAP BC at the Bronx in Fig. 2. Mean MAAP BC was 50% higher than raw Aethalometer® BC$_{880}$ data from November to February and around 15% higher from June to September. Following correction of the Aethalometer® data the monthly differences were within the 2 σ confidence intervals as indicated in Fig. 2.

Annual Trends

Hourly BC$_{880}$ Corrected for particle loading are shown as box and whisker plots in Fig. 3; Bronx data in Fig. 3(a) and Rochester Fig. 3(b). A comparison of annual BC$_{880}$ concentrations and other co-pollutants at the Bronx and Rochester are shown in Table 1 and Table 2, respectively. Annual mean BC$_{880}$ at the Bronx was 2.0 μg/m$^3$ during
Fig. 1. Comparison of mean monthly K factors for 880 nm, 24-h integrated SO$_4$ and OC/EC ratio during 2004–2005 at the Bronx.

Fig. 2. Comparison of mean Aethalometer® BC$_{880}$ raw (black), BC$_{880}$ corrected (light grey) and the MAAP BC (dark grey) at the Bronx. The error bars denote the 95% confidence limits.

2003–2004 but concentrations were 15% lower in 2005 (significant at the 95% CI, p < 0.05). Concentrations remained relatively uniform within 3% between 2005 and 2008 with a slightly higher variation of 6% from 2008 to 2011. However, BC$_{880}$ was 20% lower in 2012. Most other pollutants show a more uniform downward trend. For example, SO$_2$ and NO$_x$ decreased by 4.6 and 8 ppb respectively, between 2005 and 2009 amounting to an approximate 40% reduction in both cases. PM$_{2.5}$ mass declined steadily throughout the period decreasing by 4 µg/m$^3$ (30%) since 2003. The majority of the PM$_{2.5}$ mass reduction is attributed to PM$_{2.5}$ sulfate which dropped by 2–3 µg/m$^3$ (if ammonium sulfate is considered) and nitrate which decreased by approximately 0.7 µg/m$^3$. These reductions are attributed to federal SO$_2$ and NO$_x$ emission controls (National Emissions Inventory (NEI) summary obtained from http://www.epa.gov/ttn/chief/trends/index.html) during the past decade that had a large impact on emissions from coal burning power utilities resulting in a decrease in PM$_{2.5}$ particle species across the eastern US. BC$_{880}$ at the Bronx amounted to approximately 14–20% of total PM$_{2.5}$ mass, Fig. S1(a).
Annual mean BC$_{880}$ at Rochester, Fig. 3(b), was approximately a factor of 2–3 lower than the Bronx varying between 0.70 to 0.76 $\mu g/m^3$ during 2004 and 2007. Thereafter BC$_{880}$ decreased reaching 0.57 $\mu g/m^3$ in 2009 with a further drop to 0.46 $\mu g/m^3$ in 2010 representing a 30% decrease in BC$_{880}$ since 2007. Mean BC$_{880}$ for 2011

![Box and whisker plot of annual BC$_{880}$ concentrations in $\mu g/m^3$ at (a) the Bronx and (b) Rochester. The boxes represent the 25th percentile (lower edge), median (solid line), mean (dashed line) and 75th percentile (upper edge). Whiskers represent the 10th and 90th percentiles and the solid circles represent the 5th and 95th percentiles.](image)

**Fig. 3.** Box and whisker plot of annual BC$_{880}$ concentrations in $\mu g/m^3$ at (a) the Bronx and (b) Rochester. The boxes represent the 25th percentile (lower edge), median (solid line), mean (dashed line) and 75th percentile (upper edge). Whiskers represent the 10th and 90th percentiles and the solid circles represent the 5th and 95th percentiles.

**Table 1.** Annual mean concentration of selected pollutants at the Bronx.

| Year | BC ($\mu g/m^3$) | SO$_2$ (ppb) | NO$_x$ (ppb) | PM$_{2.5}$ ($\mu g/m^3$) | SO$_4$ ($\mu g/m^3$) | NO$_3$ ($\mu g/m^3$) |
|------|-----------------|--------------|--------------|-------------------------|-------------------|-------------------|
| 2003 | 2.04            | 8.6          | 49.6         | 14.8                    | 4.37              | 2.37              |
| 2004 | 1.99            | 10.3         | 21.6         | 13.7                    | 3.71              | 2.09              |
| 2005 | 1.72            | 10.9         | 21.0         | 13.8                    | 4.49              | 2.20              |
| 2006 | 1.69            | 8.83         | 17.1         | 12.6                    | 3.69              | 1.90              |
| 2007 | 1.67            | 7.86         | 16.0         | 12.7                    | 4.07              | 2.15              |
| 2008 | 1.71            | 6.58         | 14.3         | 11.8                    | 3.44              | 1.90              |
| 2009 | 1.81            | 6.33         | 13.1         | 10.8                    | 2.57              | 1.67              |
| 2010 | 1.61            | 6.68         | 14.0         | 10.2                    | 2.39              | 1.68              |
| 2011 | 1.76            | a            | a            | a                       | a                 | a                 |
| 2012 | 1.39            | 2.42         | 10.5         | 10.0                    | a                 | a                 |

*No data.
BC\textsubscript{880} at the Bronx occurs from November to February as observed at some Asian sites (Baxla et al., 2009) but lower than those at India and Bangladesh (Baxla et al., 2009; Begum et al., 2012). However in Hong Kong maximum BC was found from July to September (Cheng et al., 2012) indicating sources other than nearby traffic was impacting the site during the summer period. A larger seasonal gradient in BC was observed at the Asian locations, factor of 3 to 5 change, compared to 40 to 50% in New York.

### Weekday/Weekend Pattern

BC\textsubscript{880} day of week pattern versus season at the Bronx shows concentrations are highest on weekdays, Monday-Friday and lowest on Sundays with Saturdays in between, Fig. 5. The most pronounced Sunday/weekend difference is in spring and summer with differences of 40–45%, whereas the least difference is in winter (20%). Two sample t-tests show mean weekday BC\textsubscript{880} concentrations are higher compared to those on a Saturday or Sunday, significant at the 95% CI (\(p < 0.05\)). In addition concentrations on Saturday are significantly higher than on Sunday. NO\textsubscript{3}\textsuperscript{-} concentrations show a similar pattern to BC\textsubscript{880} and are approximately 40% higher on a weekday versus a Sunday in spring, summer and 20% higher in winter, Fig. 6. Mean weekday SO\textsubscript{2} is approximately 20% higher than on Sundays in spring and summer but no discernable weekday to weekend difference are observed in autumn and winter. The different SO\textsubscript{2} day of week pattern reflects additional emissions from sources other than traffic such as fuel oil combustion (used for heating water and space heating) which has no particular day of week pattern. Mean daily vehicle counts on the nearby Bruckner Expressway during 2008 varied from 110,000 to 135,000 with Sundays the lowest, but not significant at the 95% confidence interval. Therefore, it is unlikely that the small differences in daily vehicle counts could explain the different weekday versus weekend BC\textsubscript{880} pattern at the Bronx. BC\textsubscript{880} at Rochester is also lower on Sunday compared to weekdays varying from 20% lower in winter to 37% during spring, Fig. S3. At Rochester however, there is a significant difference between weekday and weekend daily vehicle counts which could explain differences in BC\textsubscript{880} on weekdays versus weekends. Mean weekday vehicle counts on the I-590 range from 98,000 to 115,000 throughout the year with 20–30% fewer vehicles on Saturdays and 40% less on Sundays, Fig. S4. The different day of week vehicle count pattern between the two sites reflects the size in the urban centers with roadways in the

### Monthly Pattern

Monthly data presented in Fig. 4(a) show that maximum BC\textsubscript{880} at the Bronx occurs from November to February as represented by the 95 percentiles with a secondary maximum in June and July. The former is attributed to lower boundary layer heights and increased emissions from space heating use in colder periods and the latter to increased emissions from outdoor activity such as vehicles, construction work and outdoor food grilling. The fraction of BC\textsubscript{880} to PM\textsubscript{2.5} mass is lowest in July (mean 12%) reflecting the higher contribution of secondary aerosol to PM\textsubscript{2.5} mass during summer compared to other periods, Fig S2(a). In contrast BC\textsubscript{880} at Rochester is highest from June to November, with maximum concentrations in November, Fig. 4(b). This pattern reflects increased summer activity and increased emissions from space heating sources such as wood burning with the onset of colder periods in October and November (Wang et al., 2011b). With the advancement of winter and snow cover over outdoor wood storage piles there is a preference for natural gas over wood burning for space heating. The proportion of BC\textsubscript{880} to PM\textsubscript{2.5} mass is highest in autumn, Fig. S2(b), reflecting increased space heating use with the onset of colder periods. Maximum BC was also observed from November to February at urban locations in the USA (Park et al., 2006), Europe (Saha and Despiau, 2009) India and Bangladesh (Baxla et al., 2009; Begum et al., 2012; Swamy et al., 2012).

| Year | BC (\(\mu\text{g/m}^3\)) | EC (\(\mu\text{g/m}^3\)) | SO\textsubscript{2} (ppb) | CO (ppb) | PM\textsubscript{2.5} (\(\mu\text{g/m}^3\)) | SO\textsubscript{4} (\(\mu\text{g/m}^3\)) | NO\textsubscript{3} (\(\mu\text{g/m}^3\)) |
|------|-----------------|-----------------|-----------------|-------|-----------------|-----------------|-----------------|
| 2004 | 0.70            | 0.39            | 4.58            | 468.8 | 11.0            | 3.27            | 1.82            |
| 2005 | 0.76            | 0.45            | 4.20            | 461.6 | 12.5            | 3.65            | 1.91            |
| 2006 | 0.71            | 0.42            | 4.18            | 495.6 | 9.5             | 2.87            | 1.43            |
| 2007 | 0.72            | 0.51            | 4.07            | 426.0 | 9.9             | 3.29            | 1.50            |
| 2008 | 0.67            | 0.48            | 2.91            | 388.0 | 9.0             | 2.64            | 1.58            |
| 2009 | 0.57            | 0.46            | 2.54            | 420.4 | 7.6             | 1.92            | 1.42            |
| 2010 | 0.46            | 0.44            | 2.19            | 428.8 | 8.4             | 1.77            | 1.04            |
| 2011 | 0.58            | 0.47            | 1.22            | 189.6 | 9.2             | 1.76            | 1.13            |
| 2012 | 0.57            | 0.66            | 1.04            | 208.0 | 8.7             | 1.60            | 1.05            |
larger NYC urban center experiencing a more uniform total daily number of vehicles throughout the week. Elevated BC was observed during weekdays at other urban sites (Baxla et al., 2009; Saha and Despiau, 2009; Cheng et al., 2012; Swamy et al., 2012). Lough et al., (2006) found elevated EC and motor vehicle molecular markers during weekdays at Los Angeles.

Diurnal Pattern

The weekday BC$_{880}$ diurnal pattern at the Bronx reveals an increase in the morning concentration starting around 5 am with a peak between 7–9 am and a minimum in the early afternoon followed by a late evening gradual rise, Fig. 7. Although the range in concentrations varies across seasons a consistent overall weekday diurnal pattern is observed. Seasonal differences are attributed to differences in emissions, different sources coupled with variations in boundary layer height across seasons, Fig. S5. A similar diurnal pattern is observed for EC and NO$_x$ but not SO$_x$ or OC which are dominated by secondary formation process, particularly in summer, as shown in Fig. S6. The diurnal profile on Saturdays shows a reduced morning maximum whereas on Sundays the morning peak is absent with highest concentrations at night, Fig. 8(a). To understand these BC$_{880}$ patterns we need to look at both vehicle counts and boundary layer height. Fig. 8(b) shows mean hourly vehicle counts on the nearby Bruckner expressway. The weekday morning increase in BC$_{880}$ coincides with a rapid rise in vehicle counts starting around 4–5 am. There is an 8-fold increase in hourly vehicle counts from 1000 vehicles per hour at 4 am to the peak of around 8,200–8,400 vehicles between 7–8 am. Vehicle counts subsequently decrease

![Box and whisker plot of monthly BC$_{880}$ concentrations in μg/m$^3$ at (a) the Bronx and (b) Rochester, NY. Symbol key as for Fig. 3.](image-url)
before leveling off around 6,700 between 10 AM and noon (a 20% decrease compared to the peak) followed by a slow gradual afternoon rise with a broad secondary maximum between 2–4 PM. Although the total vehicle count may vary with season and also with the roadway size, there is a consistent diurnal traffic count profile throughout the year. The late morning and early afternoon decrease in BC<sub>880</sub> (and NO<sub>x</sub> shown in Fig. S6) amounts to approximately a factor of two variation with respect to the maximum. By comparison since vehicle counts only vary by approximately 20% on the nearby highway, the bulk of the morning pollutant decrease is attributed to an increase in boundary layer height (Fig. 8(c)). The variation in boundary layer height driven by solar heating is accompanied by increased wind speed and atmospheric turbulence leading to a dilution and dispersion of pollutants during the day. The boundary layer height remains elevated into the evening hours which is the reason for the absence of a secondary BC<sub>880</sub> maximum during the afternoon and evening commute. A gradual rise in BC<sub>880</sub> and NO<sub>x</sub> is observed as the boundary layer collapses and pollution dispersion decreases in the evening hours despite the drop in vehicle counts. On Saturdays, there is a slower rise in vehicle counts compared to weekdays with no early morning peak but rather a broad maximum extending from around noon to 7 PM. Therefore, by the time traffic counts reaches its peak on Saturdays, the boundary layer has already expanded, and thus the influx of fresh pollutants is dispersed. On Sundays, there is a further lag in traffic growth which starts an hour or two later compared to weekdays and there is little or no morning BC<sub>880</sub> peak.

The diurnal pattern at Rochester is similar although the late evening peak approaches the magnitude of the morning maximum particularly in summer and autumn, Fig. S7. Differences predominantly in the vehicle count profile and emission sources coupled with boundary layer dynamics are attributed to the differences in the late evening BC<sub>880</sub> patterns between the two locations. Therefore, the timing of the morning traffic growth in relation to the boundary layer dynamics explains the different BC<sub>880</sub> diurnal pattern on a weekday versus a weekend and the seasonal variation in BC<sub>880</sub> diurnal profile. This also explains the weekday to weekend differences described in the previous section.

Similar diurnal profiles were observed at other urban locations (Park et al., 2006; Baxla et al., 2009; Saha and Despiau, 2009; Cheng et al., 2012; Swamy et al., 2012) although in some instances (Swamy et al., 2012) the weekday to weekend differences may not be as large as observed in New York.
Winter Pollution Episode

As previously mentioned, on average BC$_{880}$ amounts to 14–20% of the PM$_{2.5}$ mass at the Bronx. However, during periods dominated by primary pollutants, BC$_{880}$ can account for a much higher proportion of PM$_{2.5}$ mass. A pollution event occurred on February 22, 2007 during a winter inversion characterized by temperatures near 0°C and wind speeds below 1 m/s, Fig. 9. Pollutant concentrations rose sharply during the early morning with PM$_{2.5}$ mass peaking close to 60 μg/m$^3$, BC$_{880}$ near 20 μg/m$^3$ and total NO$_x$ of 570 ppb between 6 and 7 AM. Prior to 2 AM, BC$_{880}$ was around 15% of PM$_{2.5}$ mass but this increased to 30–38% between 4 and 8 AM. Many of the pollutants were correlated typical of an event dominated by primary pollutants. For example, BC was highly correlated with EC and NO$_x$, R$^2$ above 0.97, but also with PM$_{2.5}$ mass, R$^2$ of 0.94. A linear regression of OC and EC (excluding two points around the 7 AM peak when EC was higher than OC by 15–25%) showed an R$^2$ of 0.93 and a mean OC/EC ratio of 1.1 characteristic of primary OC/EC combustion ratios. After 7 AM, the temperature and wind speed increased and pollutants dispersed. Particle nitrate, however, which was relatively stable at around 2 μg/m$^3$ prior to 8 AM, rose sharply reaching a peak of 14 μg/m$^3$ around 11 AM, likely the result of photochemical oxidation of the large amount of NO$_x$. Pollutant concentrations were low in the early afternoon from 12–2 PM but concentrations increased again as the wind speed dropped showing a secondary but lower peak, PM$_{2.5}$ mass of 20 μg/m$^3$, around 6–8 PM.

Specific Source Markers

Previous Aethalometer® measurements showed that enhanced absorption at 370 nm relative to 880 nm (BC$_{370}$ − BC$_{880}$ = Delta C) tracked wood smoke markers, levoglucosan and elemental potassium at the Rochester site (Wang et al., 2011b) although Delta C is not a quantitative measure of mass concentration. This enhanced Delta C signal is evident from October to March at both sites (Fig. S8) but particularly during the late evening hours on weekends at Rochester (Fig. S9b) when residential wood burning is prominent and Delta C can reach several μg/m$^3$. An example of an event is shown in Fig. S10 on the weekend of Nov. 1–2, 2008 when Delta C peaked near 5.5 μg/m$^3$ around 9 PM on Saturday. BC$_{880}$, PM$_{2.5}$ mass and CO showed coincidental peaks of around 4, 24 μg/m$^3$ and 1 ppm, respectively. This event is typical of winter weekend nocturnal pollution episodes when ambient temperatures drop below zero °C during the night and light winds from a southerly direction cause wood smoke emissions to advect towards the monitoring site leading to hourly Delta C of several μg/m$^3$ (Wang et al., 2012). In contrast a weaker Delta C signal is observed at the Bronx site with mean late evening Delta C

Fig. 6. Mean day of week concentration of NO$_x$ (shaded box, right axis) and SO$_2$ (filled box, left axis) in ppb versus season at the Bronx.
Fig. 7. Box and whisker plot showing the weekday diurnal BC$_{880}$ concentration versus season at the Bronx. Symbol key as for Fig. 3.

values ~0.2 μg/m$^3$ in winter, Fig. S9(a). On the other hand, measurements of PM$_{10}$ trace elements show elevated nickel (Ni) during the heating season at the Bronx site, Fig. 10(a). Mean winter Ni concentrations of 17.6 ng/m$^3$ are approximately a factor of 3 higher than in summer, 6.2 ng/m$^3$ consistent with earlier findings by Lippmann (2009). In addition a linear regression of Ni and Co (cobalt), Fig. 10(b), shows a mean Ni:Co ratio of 12.2 (for screened data with Co 2 ng/m$^3$ or below) close to the ratio of 14 observed in residual oil number 6, used for home heating in the Northeastern US (Graham, 2010). The large scatter in the Ni to Co relationship reflects different sources (for example different grade heating fuels or impact from other sources). Observations at rooftop level in the Bronx reveals frequent puffs of black smoke from building stacks, Fig. S11. These emissions which are attributed to the cycling on and off of oil boilers reveal hourly BC concentrations above 40 μg/m$^3$ (Rattigan et al., 2010). At Rochester, Ni concentrations are a factor of 5–20 lower than at the Bronx with no winter/summer concentration gradient. Natural gas and wood rather than fuel oil (number 4 and 6) are the predominant heating sources in the mainly single dwelling residences at Rochester. By contrast much of the surrounding residential area in the Bronx is high rise apartment blocks employing old style boilers burning residual oil for space and water heating as previously noted by Lippmann (2009).

Black Carbon vs. Elemental Carbon

As shown above BC$_{880}$ was correlated with EC and NO$_x$ with a diurnal pattern indicative of emissions from traffic sources. However, it is known that the relationship between BC$_{880}$ and EC can vary depending on the particle composition (Jeong et al., 2004; Kim et al., 2006; Park et al., 2006; Salako et al., 2012). Jeong et al. (2004) and Kim et al. (2006) reported that the BC/EC relationship varied depending on the source and physical and chemical properties of the aerosol. Salako et al. (2012) analyzed the BC to EC relationship at various locations across Asia and the South Pacific. BC was measured by reflectrometry whereas EC was determined using the IMPROVE method (Chow et al., 2007). Mean BC/EC ratios varied from a minimum of 0.47 for a rural site in Australia to a maximum of 3.56 for Bangkok, Thailand. Correlations were lowest for Malaysia, Bangkok and Australia and highest for Bangladesh and New Zealand. The authors attributed these results to differences in particle light absorptivity resulting from differences in
ambient source contributions (biomass burning versus vehicular emissions) and degree of atmospheric processing. Here two methods were used in the comparison of BC_{880} and EC; (1) BC/EC ratios and (2) a robust linear regression of BC_{880} and EC. The long term trends show that the annual ratio of BC_{880} to EC and BC_{880} to TC measured by the CSN TOT method (Peterson and Richards, 2002) at the Bronx varied from 1.4 to 1.8 and 0.4 to 0.5, respectively. The corresponding ratios at Rochester were 1.4 to 1.6 and 0.2 to 0.3, respectively. Monthly ratios of BC_{880} to EC measured by the CSN TOT and IMPROVE TOR (Chow et al., 2007) methods are shown in Figs. 11(a) and 11(b), respectively. At both sites the ratio of BC_{880} to EC analyzed using the TOT method, Fig. 11(a), is elevated in summer (June to August) compared to winter. The mean ratio is 1.3 with a standard deviation of 0.3 from October to March increasing to a ratio of 1.8 with a standard deviation of 0.2 from June to August. In contrast the ratio of BC_{880} to IMPROVE EC, Fig. 11(b), shows little seasonal gradient at the Bronx site with a mean annual ratio of 1.3 close to the ratio of 1.4 for the EC CSN TOT method during winter. The ratio of BC to IMPROVE EC at Rochester is similar except for the month of June when it is below 1. Note that in the case of Rochester, the IMPROVE carbon protocol was introduced in April 2009 and therefore does not extend over the same sampling period as the CSN TOT data.
Results of the robust linear regressions of BC_{880} with the various EC methods are shown in Table 3 and 4. Seasonal regression coefficients and the combined results are provided. At the Bronx the regression slopes reflect the pattern observed in the BC_{880}/EC ratios. Thus slopes are above unity with generally higher values in summer for the comparison with the EC TOT methods (MetOne, R&P 2300). For example the slope in summer is 64% higher compared to winter for the MetOne comparison versus 30% higher for the R&P 2300. However, except for the MetOne, the seasonal differences overlap within the uncertainties. Regression slopes for the comparison of BC_{880} with EC analyzed by the IMPROVE TOR (URG-3000N) vary between 1.31 to 1.44 showing seasonal differences of 10%. In all cases correlation coefficients, R, are above 0.8 and intercepts are zero or close to zero within the uncertainties. Regression slopes for comparison of BC_{880} and EC analyzed using TOT (MetOne, R&P 2300) at Rochester are similar in magnitude to the Bronx. However, slopes show less of a summer/winter contrast, 22–24% at Rochester compared to 30–64% the Bronx. The regression slopes for BC_{880} and TOR EC at Rochester are unity within the uncertainties and approximately 30–40% lower than observed at the Bronx. The variation in the BC_{880}/EC relationship with location and season is consistent with a change in optical properties of the ambient particle which impact optical measurement of BC as indicated in previous findings (Petzold et al., 1997; Jeong et al., 2004; Park et al., 2006; Salako et al., 2012). Thus assuming a fixed absorption coefficient may be incorrect (Petzold et al., 1997). The results here also show that the BC_{880}/EC relationship depends on the EC measurement method used consistent with their method dependent definitions.

CONCLUSIONS

Measurements of BC_{880} over multiple years were used to characterize temporal patterns on an annual, seasonal, day
of week and diurnal basis at two urban locations in New York. Annual mean $\text{BC}_{380}$ concentrations in New York City ranged from 1.4 to 2.0 $\mu g/m^3$ and were a factor of 2–3 lower at Rochester. Similar day of week and diurnal patterns are attributed to traffic emissions from nearby roadways at both sites. Higher weekday (Monday–Friday) $\text{BC}_{380}$ concentrations compared to weekends (Saturday and Sunday) results from increased vehicle emissions prior to the expansion of the boundary layer during the morning workday commute. Weekend vehicle emissions lag those on weekdays and evolve as the boundary expands resulting in their dilution and dispersion. The effect is most evident on Sundays as indicated in the lack of a morning $\text{BC}_{380}$ peak. Dual wavelength BC measurements showed an enhanced $\Delta C = \text{BC}_{370} - \text{BC}_{880}$ signal from October to March predominantly during late evening weekend hours at Rochester. In contrast a weaker $\Delta C$ signal was observed at the Bronx location. However, Ni and Co were elevated in $\text{PM}_{10}$ samples during the heating season at the Bronx site since residual oil rather than wood or natural gas is used as a heating fuel source. In addition the ratio of Ni to Co was close to the ratio found in residual oil number 6. Comparison of $\text{BC}_{380}$ with EC showed an enhancement in the $\text{BC}_{380}/\text{EC}$ ratio during summer compared to winter for thermal optical transmission methods, particularly at the Bronx where summer $\text{BC}_{380}/\text{EC}$ ratios were 30–60% higher compared to winter. By contrast the $\text{BC}_{380}/\text{EC}$ ratio showed little seasonal variation for the EC IMPROVE thermal optical reflectance method.

**ACKNOWLEDGMENTS**

The authors wish to thank Richardson Colas, and Michael Abbott for help with instrument operation at the South Bronx and Tom Everts for assistance at the Rochester site.
Table 3. Robust linear regression coefficients and 1σ uncertainties for comparison of BC$_{880}$ and EC from various methods at the Bronx site.

(a) Comparison of BC$_{880}$ and MetOne TOT EC at the Bronx site.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | (1.46 ± 0.14) | (–0.23 ± 0.18) | 0.90 | 89 |
| Summer | (1.93 ± 0.31) | (–0.39 ± 0.34) | 0.82 | 71 |
| Autumn | (1.38 ± 0.16) | (–0.09 ± 0.20) | 0.98 | 61 |
| Winter | (1.18 ± 0.13) | (–0.15 ± 0.23) | 0.96 | 67 |
| All    | (1.37 ± 0.08) | (–0.11 ± 0.10) | 0.91 | 288 |

(b) Comparison of BC$_{880}$ and R&P2300 TOT EC at the Bronx site.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | (1.72 ± 0.21) | (–0.45 ± 0.27) | 0.94 | 60 |
| Summer | (2.03 ± 0.30) | (–0.16 ± 0.33) | 0.88 | 66 |
| Autumn | (1.67 ± 0.13) | (–0.30 ± 0.25) | 0.81 | 73 |
| Winter | (1.57 ± 0.28) | (–0.29 ± 0.39) | 0.89 | 50 |
| All    | (1.77 ± 0.12) | (–0.31 ± 0.10) | 0.86 | 249 |

(c) Comparison of BC$_{880}$ and IMPROVE TOR EC at the Bronx site.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | (1.40 ± 0.10) | (–0.25 ± 0.13) | 0.92 | 177 |
| Summer | (1.37 ± 0.12) | (–0.15 ± 0.17) | 0.88 | 161 |
| Autumn | (1.44 ± 0.11) | (–0.15 ± 0.13) | 0.89 | 172 |
| Winter | (1.31 ± 0.10) | (–0.26 ± 0.16) | 0.86 | 169 |
| All    | (1.35 ± 0.05) | (–0.17 ± 0.07) | 0.88 | 679 |

Table 4. Robust linear regression coefficients and 1σ uncertainties for comparison of BC$_{880}$ and EC from various methods at Rochester.

(a) Comparison of BC$_{880}$ and MetOne TOT EC at Rochester.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | 1.78 ± 1.06 | (–0.16 ± 0.45) | 0.89 | 25 |
| Summer | 1.69 ± 1.14 | (–0.08 ± 0.63) | 0.83 | 21 |
| Autumn | 1.70 ± 0.64 | (–0.20 ± 0.39) | 0.79 | 40 |
| Winter | 1.36 ± 0.65 | (–0.07 ± 0.32) | 0.64 | 52 |
| All    | 1.61 ± 0.38 | (–0.14 ± 0.20) | 0.78 | 138 |

(b) Comparison of BC$_{880}$ and R&P2300 TOT EC at Rochester.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | 1.96 ± 0.57 | (–0.16 ± 0.24) | 0.90 | 80 |
| Summer | 2.10 ± 0.65 | (–0.11 ± 0.29) | 0.74 | 98 |
| Autumn | 1.91 ± 0.41 | (–0.17 ± 0.23) | 0.89 | 96 |
| Winter | 1.71 ± 0.49 | (–0.10 ± 0.22) | 0.83 | 78 |
| All    | 1.93 ± 0.25 | (–0.14 ± 0.12) | 0.84 | 352 |

(c) Comparison of BC$_{880}$ and IMPROVE TOR EC at Rochester.

| Season | Slope (±σ) | Intercept (±σ) | R    | N  |
|--------|------------|----------------|------|----|
| Spring | 1.03 ± 0.25 | (–0.03 ± 0.11) | 0.90 | 100 |
| Summer | 1.14 ± 0.32 | (–0.03 ± 0.18) | 0.87 | 75 |
| Autumn | 1.15 ± 0.23 | (–0.00 ± 0.13) | 0.94 | 83 |
| Winter | 1.27 ± 0.29 | (–0.03 ± 0.13) | 0.99 | 77 |
| All    | 1.14 ± 0.13 | (–0.01 ± 0.06) | 0.93 | 335 |

DISCLAIMER

The opinions expressed in this paper do not necessarily reflect the views of New York State Department of Environmental Conservation.

SUPPLEMENTARY MATERIALS

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

Albrecht, B.A. (1989). Aerosols, Cloud Microphysics and Fractional Cloudiness. Science 245: 1227–1230.

Arnott, W.P., Hamasha, K., Moosmuller, H., Sheridan, P.J. and Ogren, J.A. (2005). Towards Aerosol Light-Absorption Measurements with a 7-Wavelength Aethalometer: Evaluation with a Photoacoustic Instrument and 3-Wavelength Nephelometer. Aerosol Sci. Technol. 39: 17–29.

Baxla, S.P., Roy, A.A., Gupta, T., Tripathi, S.N. and Bandyopadhyaya, R. (2009). Analysis of Diurnal and Seasonal Variation of Submicron Outdoor Aerosol Mass and Size Distribution in a Northern Indian City and Its Correlation to Black Carbon. Aerosol Air Qual. Res. 9: 458–469.

Begum, B.A., Hossain, A., Nahar, N., Markwitz, A. and Hopke, P.K. (2012). Organic and Black Carbon in PM_{2.5} at an Urban Site at Dhaka, Bangladesh. Aerosol Air Qual. Res. 12: 1062–1072.

Birch, M.E. and Cary, R.A. (1996). Elemental Carbon-based Method for Monitoring Occupational Exposures to Particulate Diesel Exhaust. Aerosol Sci. Technol. 25: 221–241.

Bond, T.C., Anderson, T.L. and Campbell, D. (1999). Calibration and Intercomparison of Filter-based Measurements of Visible Light Absorption by Aerosols. Aerosol Sci. Technol. 30: 582–600.

Bond, T.C., Bhardwaj, E., Dong, R., Jogani, R., Jung, S., Bond, T.C., Anderson, T.L. and Campbell, D. (1999). Calibration and Intercomparison of Filter-based Measurements of Visible Light Absorption by Aerosols. Aerosol Sci. Technol. 30: 582–600.

Cheng, Y., Ho, K.F., Wu, W.J., Ho, S.S.H., Lee, S.C., Huang, Y., Zhang, Y.W., Yao, P.S., Gao, Y. and Chan, C.S. (2012). Real-Time Characterization of Particle-Bound Polycyclic Aromatic Hydrocarbons at a Heavily Trafficked Roadside Site. Aerosol Air Qual. Res. 12: 1181–1188.

Chow, J.C., Watson, J.G., Chen, L.W.A., Chang, M.C.O., Robinson, N.F., Trimble, D. and Kohl, S. (2007). The IMPROVE_A Temperature Protocol for Thermal Optical Carbon Analysis: Maintaining Consistency with a Long-Term Database. J. Air Waste Manage. Assoc. 57: 1014–1023.

Cornell, A.G., Chillrud, S.N., Mellins, R.B., Acosta, L.M., Miller, R.L., Quinn, J.W., Yan, B., Divjan, A., Olmedo, O.E., Lopez-Pintado, S., Kinney, P.L., Perera, F.P., Jacobson, J.S., Goldstein, I.F., Rundle, A.G., and Perzanowski, M.S. (2012). Domestic Airborne Black Carbon and Exhaled Nitric Oxide in Children in NYC. J. Exposure Sci. Environ. Epidemiol. 22: 258–266.

Doraiswamy, P., Hogrefe, C., Hao, W., Civerolo, K., Ku, J.Y. and Gopal Sistla, G. (2010). A Retrospective Comparison of Model-Based Forecasted PM_{2.5} Concentrations with Measurements. J. Air Waste Manage. Assoc. 60: 1293–1308.

Graham, J. (2010). Select Trace Elemental Composition of Fuel Oil used in the Northeastern United States, EM Magazine, Air & Waste Management Association, p. 16–22.

Hansen, A.D.A., Rosen, H. and Novakov, T. (1984). The Aethalometer -An Instrument for the Real-time Measurement of Optical Absorption by Aerosol Particles. Sci. Total Environ. 36: 191–196.

Hildemann, L.M., Markowski, G.R., Jones, M.C., Cass, G.R. (1991). Submicrometer Aerosol Mass Distribution of Emissions from Boilers, Fireplaces, Automobiles, Diesel Trucks, and Meat Cooking Operations. Aerosol Sci. Technol. 14: 138–152.

Intergovernmental Panel on Climate Change (2007). Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment, Report of the Intergovernmental Panel on Climate Change, Solomon, S. et al., (Eds.), Cambridge Univ. Press, Cambridge, U.K.

Jacobson, M.Z. (2002). Control of Fossil-Fuel Particulate Black Carbon and Organic Matter, Possibly the Most Effective Method of Slowing Global Warming. J. Geophys. Res. 107: 4410, doi: 10.1029/2001JD001376.

Janjic, Z.I. (2001). Nonsingular Implementation of the Mellor-Yamada Level 2.5 Scheme in the NCEP Meso Mode, NCEP Office Note No. 437, p. 1–61.

Jeong, C.H., Hopke, P.K., Kim, E. and Lee, D.W. (2004). The Comparison between Thermal-optical Transmittance Elemental Carbon and Aethalometer Black Carbon Measured at Multiple Monitoring Sites. Atmos. Environ. 38: 5193–5204.

Kim, J.J., Smorodinsky, S., Lipsett, M., Singer, B.C., Hodgson, A.T. and Ostro B. (2004). Traffic-related Air Pollution near Busy Roads The East Bay Children’s Respiratory Health Study. Am. J. Respir. Crit. Care Med. 170: 520–526.

Kim, Y.J., Kim, M.J., Lee, K.H. and Park, S.S. (2006). Investigation of Carbon Pollution Episodes Using Semi-continuous Instrument in Incheon, Korea. Atmos. Environ. 40: 4064–4075.

Kirschstawter, T., Novakov, T. and Hobbs, P. (2004). Evidence that Spectral Light Absorption by Aerosols Emitted by Biomass Burning and Motor Vehicles is Different due to Organic Carbon. J. Geophys. Res. 109: D21208.

Lioussse, C, Cachier, H. and Jennings, S.G. (1993). Optical and Thermal Measurements of Black Carbon Aerosol Content in Different Environments: Variations of the Specific Attenuation Cross-section, Sigma(s). Atmos. Environ. 27A: 1203–1211.

Lippmann, M. (2009). Semi-continuous Speciation Analysis for Ambient Air Particulate Matter: An Urgent Need for Health Effects Studies. J. Exposure Sci. Environ. Epidemiol. 19: 235–247.

Lough, G.C., Schauer, J.J. and Lawson, D.R. (2006). Day of Week Trends in Carbonaceous Aerosol Composition in the Urban Atmosphere. Atmos. Environ. 40: 4137–4149.

Malm, W.C., Sisler, J.F., Huffman, D., Eldred, R.A. and Cahill, T.A. (1994). Spatial and Seasonal Trends in Particle Concentration and Optical Extinction in the US. J. Geophys. Res. 99: 1347–1370.

Park, K., Chow, J.C., Watson, J.G., Trimble, D.L.,
Doraiswamy, P., Park, K., Arnott, W.P., Stroud, K.R., Bowers, K., Bode, R., Petzold, A. and Hansen, A.D.A. (2006). Comparison of Continuous and Filter-Based Carbon Measurements at the Fresno Supersite. *J. Air Waste Manage. Assoc.*, 56: 474–491.

Park, S.S., Hansen, A.D.A., Sung, Y. and Cho, S.Y. (2010). Measurement of Real Time Black Carbon for Investigating Spot Loading Effects of Aethalometer Data. *J. Air Waste Manage. Assoc.*, 56: 474–491.

Peterson, M.R. and Richards, M. H. (2002). In Winegar, E.D. and Tropp, R.J. (Eds.), Thermal-Optical Transmittance Analysis for Organic, Elemental, Carbonate, Total Carbon, and OCS2 in PM2.5, by the EPA/NIOSH Method, Proceedings, Symposium on Air Quality Measurement Methods and Technology, 83-1–83-19.

Petzold, A., Kopp, C. and Niessner, R. (1997). The Dependence of the Specific Attenuation Cross-Section On Black Carbon Mass Fraction and Particle Size. *Atmos. Environ.*, 31: 661–672.

Petzold, A., Kramer, H. and Schönlinner, M. (2002). Continuous Measurement of Atmospheric Black Carbon Using a Multi-Angle Absorption Photometer. *Environ. Sci. Pollut. Res.*, 4: 78–82.

Ramanathan, V. and Carmichael, G. (2008). Global and Regional Climate Changes due to Black Carbon. *Nat. Geosci.*, 1: 221–227.

Rattigan, O.V., Felton, H.D., Bae, M.S., Schwab, J.J. and Demerjian, K.L. (2010). Multi-year Hourly PM2.5 Carbon Measurements in New York: Diurnal, Day of Week and Seasonal Patterns. *Atmos. Environ.*, 2043–2053.

Rattigan, O.V., Felton, H.D., Bae, M.S., Schwab, J.J. and Demerjian, K.L. (2011). Comparison of Long-term PM2.5 Carbon Measurements at an Urban and Rural Location in New York. *Atmos. Environ.*, 45: 3228–3236.

Saha, A. and Despiau, S. (2009). Seasonal and Diurnal Variations of Black Carbon Aerosols over a Mediterranean Coastal Zone. *Atmos. Res.*, 92: 27–41.

Salako, G.O., Hopke, P.K., Cohen, D.D., Begum, B.A., Biswas, S.K., Pandit, G.G., Chung, Y.S., Rahman, S.A., Hamzah, M.S., Davy, P., Markwitz, A., Shagijamba, D., Lodosyamba, S., Wimolwattanapun, W. and Bunprapob, S. (2012). Exploring the Variation between EC and BC in a Variety of Locations. *Aerosol Air Qual. Res.*, 12: 1–7.

Schauer, J.J. (2003). Evaluation of Elemental Carbon as a Marker for Diesel Particulate Matter. *J. Exposure Sci. Environ. Epidemiol.*, 13: 433–453.

Seinfeld, J.H. and Pandis, S.N. (1998). *Atmospheric Chemistry and Physics: From Air Pollution to Climate Change*, Wiley, New York, p. 700–763.

Received for review, February 25, 2013

Accepted, May 22, 2013