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Original article

Influence of wind direction and speed on the transport of particle-bound PAHs in a roadway environment

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ARTICLE INFO

Article history:

Received 13 February 2015

Received in revised form

24 May 2015

Accepted 25 May 2015

Available online 18 October 2015

Keywords:

Wind speed

Wind direction

Urban air pollution

On-road measurement

Particle-bound PAHs

ABSTRACT

The influence of wind direction and speed on the transport of vehicular air pollutants was investigated using a mobile laboratory in an urban area. We considered three spatial scales covering typical transport phenomena observed in urban areas: micro-scale (a few tens of meters), middle-scale (a few hundred meters) and neighborhood-scale (a few kilometers). Particle-bound polycyclic aromatic hydrocarbons (PAHs) were selected as an indicator of vehicle emissions in the roadway environment. From the micro-scale measurements, we found a significant difference in the concentration of particle-bound PAHs within a small spatial area, on a two-way road, due to the effect of wind direction when the prevailing wind direction was constant. In the middle-scale experiments, an exponential decrease in the concentration of particle-bound PAHs was observed with distance from the intersection in the direction of the prevailing wind, which is consistent with the results of previous studies. The concentration of particle-bound PAHs near the road was ~10 times higher than that at 200 m away from the road due to the significantly low background concentration, indicating that particle-bound PAHs are a good indicator of vehicle emissions in a roadway environment. From long-distance (~15 km) traveling measurements made over 14 days, we found that the average concentration of particle-bound PAHs on the road decreased as the prevailing wind speed increased at the neighborhood-scale, when the wind speed was higher than 2.0 m/s. This analysis, over three spatial scales, will be helpful in understanding the effect of wind on air pollution in a roadway environment in urban areas.

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1. Introduction

The use of diesel-powered vehicles has been targeted to control the emission of particulate matter with a diameter <10 µm (PM₁₀) and NO_x in urban areas, including the Seoul Metropolitan Area in Korea. However, the reduction of ambient PM₁₀ concentrations from such environmental strategies has not always been observed, because of the influence from other air pollution sources and by meteorological conditions. Furthermore, unlike point and area sources, mobile sources such as motor vehicles emit air pollutants

during their movement along roads. This can create difficulties in understanding the emission, transport, and dilution processes and in accurately determining the effect of vehicle emissions on urban air quality. Air pollution from vehicle emissions often occurs in local hot spots, because of the great spatial and temporal variations (Kim et al., 2014).

The greater the distance from the pollution source, the greater the effect of meteorological conditions (Zhou and Levy, 2007), such as wind direction and wind speed, on horizontal dilution and the mixing layer height, which controls vertical dilution. Air pollution levels measured at roadside air quality monitoring stations, located next to the intersections of busy arterial roads, are used as an indicator of the impact of vehicle emissions on urban air quality. However, the effect of meteorological conditions, in particular wind direction and speed, on pollutant concentrations should also be considered at the roadside and even on roads. Therefore, the influence of wind on the transport and dilution of traffic-related air

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Peer review under responsibility of Turkish National Committee for Air Pollution Research and Control.

pollutants on roads, at the roadside and along the roads, needs to be systematically characterized to explain the variations in air pollution measured at these locations.

Here, we used three spatial scales to study the transport of vehicular air pollutants in a roadway environment in an urban area. Wind conditions affecting air pollution distribution were considered at the micro-scale (a few tens of meters) within the width of road for both directions, at the middle-scale (a few hundred meters) beyond the road according to the direction perpendicular to the road length, and at the neighborhood-scale (a few kilometers) along the road, as shown in Fig. 1.

Many previous studies of the transport and dispersion of vehicle emissions have identified an exponential decrease of air pollutant concentrations with distance from roads in the downwind middle-scale area. This has been demonstrated for traffic-related gases, ultrafine particles, black carbon (BC), and size-resolved PM mass concentrations (Hitchins et al., 2000; Zhu et al., 2002a,b; Bowker et al., 2007; Baldauf et al., 2008; Hagler et al., 2009; Hu et al., 2009; Kozawa et al., 2012). Recently, Hu et al. (2009) and Choi et al. (2012) made highly time resolved measurements up to 2500 m downwind and about 1000 m upwind of major freeways and measured ultrafine particles, particle-bound PAHs, BC, and

other gaseous and particulate pollutants using a mobile laboratory under nocturnal surface inversion conditions. However, these studies are mainly focused on the effect of atmospheric stability (before and after sunrise) and the relationship of air pollutants with traffic.

Wind direction and speed in relation to neighborhood-scale urban air quality have been studied for PM_{10} , $PM_{2.5-10}$, and $PM_{2.5}$ on a mass basis, together with related gaseous pollutants (Harrison et al., 1997, 2001; Keary et al., 1998; Chaloulakou et al., 2003; Aldrin and Haff, 2005; Akyüz and Çabuk, 2009; Pateraki et al., 2012). In general, these studies have found weak or moderate correlations of PM mass with wind speed and have considered wind direction to have a significant but local effect on air pollution levels, with higher wind speeds reducing air pollutant concentrations (Keary et al., 1998; Chaloulakou et al., 2003; Aldrin and Haff, 2005; Akyüz and Çabuk, 2009; Pateraki et al., 2012).

With the growing importance of ultrafine particles in terms of toxicity, there has been more interest in determining particle number concentrations (Kim et al., 2015). Several studies have reported a decrease in ultrafine particle number concentrations as a function of wind speed, while concentrations of larger particles displayed a U-shape relationship with wind speed (Ruuskanen

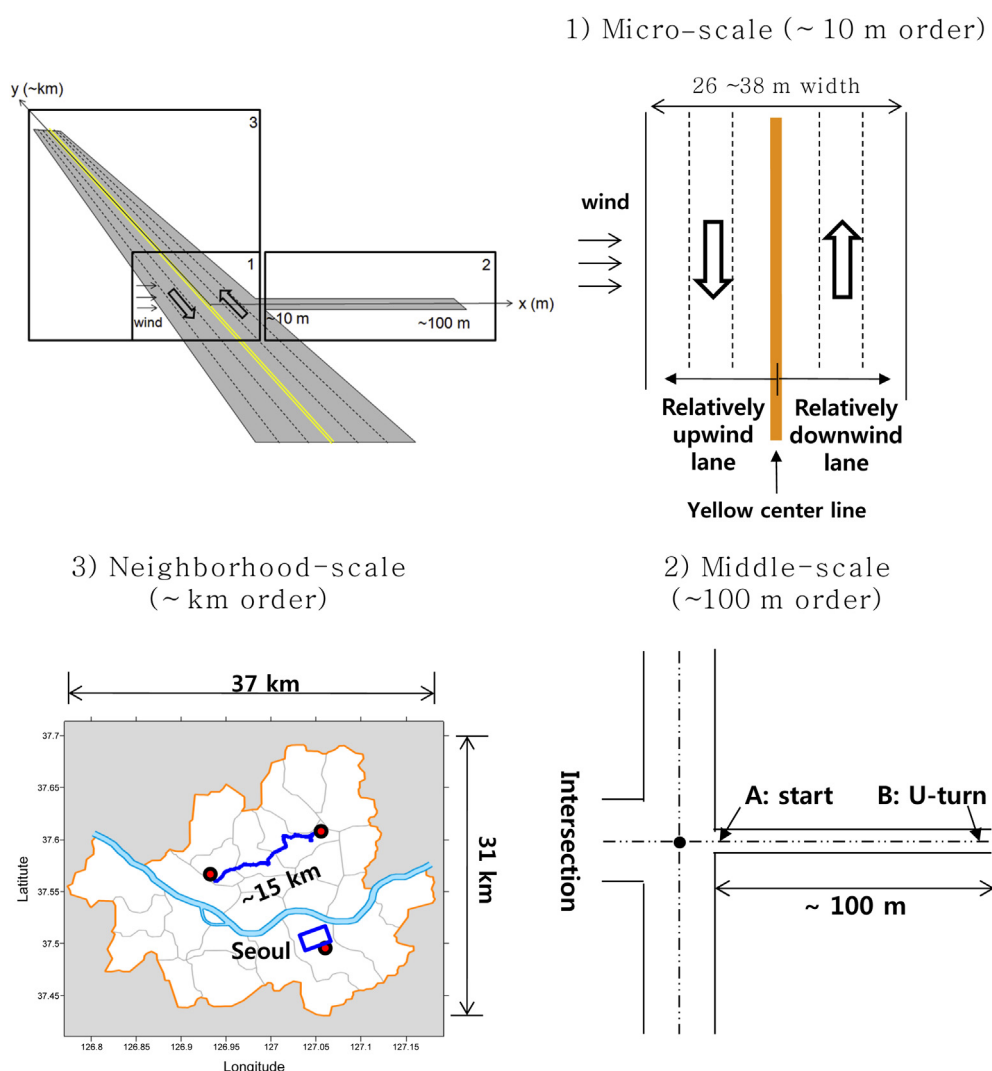


Fig. 1. Classification of wind effects on air pollution in urban areas.

et al., 2001; Molnár et al., 2002; Zhu et al., 2002a,b; Charron and Harrison, 2003; Hussein et al., 2005, 2006; Levy et al., 2003). Namely, relatively larger particles, which can be easily resuspended by wind, could display high concentrations for low wind speeds, a minimum from medium wind speeds, and then concentrations increase again for wind speeds above a certain level. Recently, Kozawa et al. (2012) focused on the wind direction effects on near-freeway ultrafine particle size distributions and gradients. They mainly compared the effects of wind direction when the wind blows perpendicularly or parallel from freeways. They could not observe the relationship of air pollutant concentrations with wind speed although they mentioned that lower wind speeds reduce dilution. Therefore, they concluded that wind direction rather than wind speed was a better measure of dilution and dilution speed for wind speeds in the range of 0.9–3.3 m/s. Because the number concentration of ultrafine particles, even at the roadside, can be affected by ambient regional nucleation events (Pirjola et al., 2006), data regarding their chemical composition, such as particle-bound PAHs, are useful to trace the impact of vehicle emissions.

There is currently no published data reporting trends of particle-bound PAHs at the micro- and middle-scale, which are considered useful for identifying vehicle exhaust emissions because of the low urban background concentrations. In this study, particle-bound PAHs were chosen as a representative air pollutant because of the important health implications, i.e., being possible human carcinogens (DFG, 2001; IARC, 2002; US EPA, 2002). They are also a good indicator of incomplete combustion sources (Mansurov, 2005) and fast resolution of PAH concentrations with real-time instruments is possible (Polidori et al., 2008).

2. Methods

The three most common approaches used in experimental studies of the spatial distribution of urban air pollution include: 1) measurements at a major road or in its immediate vicinity, 2) measurements as a function of distance from a major road, and 3) measurements at several point locations within the city (Morawska et al., 2008). The present study covers the first two experimental approaches by conducting micro- and middle-scale measurements using a mobile laboratory (ML) to examine the effect of wind on the transport and dilution processes of vehicle emission, which are considered a line source in an urban area. Instead of the third approach, we conducted neighborhood-scale measurements on roads over a long distance, using an ML. The term for wind effect includes wind direction and wind speed.

2.1. Micro-scale measurement

The influence of wind direction at the micro-scale area within a road was examined on the main roadways in the Gangnam-gu district of Seoul, Korea, as indicated by the measurement area (1) in Fig. 2. The experiment was performed by changing the driving direction of the ML when the prevailing wind was constantly blowing from a west southwest (WSW) direction. This measurement area was intentionally selected for its geographical features (i.e., flat ground), which allowed observation of the influence of wind direction. Observations of the effect of wind direction were made on upwind and downwind lanes of the same roadways, (width 26–39 m) by driving clockwise or counterclockwise through the measurement sections [1] to [4].

The micro-scale measurement was conducted twice in each direction using an ML and by driving mainly in the second or third lane from the outside on 20 February 2012, as shown in Table 1. The lower mixing layer height and moderate wind speed in the winter season were expected to assist with observing the effects of wind at

the micro-scale. Meteorological data for the period during the micro-scale measurement were obtained from the nearest automatic weather station (AWS), Gangnam-gu AWS, denoted as W1 in Fig. 2, and are also shown in Table 1.

The width of one lane was approximately 3.25 m for all roads in our measurement sections. Sections [1], [2], [3], and [4] consisted of 10, 10, 12, and 8 lanes, respectively, and the length of each section ranged from 1.4 to 3.6 km, as listed in Table 2. The traveling distance was calculated from longitude and latitude data obtained with a GPS installed in the mobile laboratory. The driving route was divided into four sections, denoted as S[1], S[2], S[3], and S[4] according to the relative location of the driving lane to the prevailing wind direction.

2.2. Middle-scale measurement

Hitchins et al. (2000) reported that the concentrations of ultrafine and fine particles decreased to about half of their maximum at a distance of 100–150 m from the road. Furthermore, Zhu et al. (2002a,b) also observed an exponential decrease of CO, BC, and particle number concentration as a function of distance from a freeway, indicating the importance of wind speed and direction for atmospheric dispersion and coagulation.

The differences between our middle-scale measurements near to a roadside compared to previous studies are: 1) an ML was utilized as a rapid measurement method, as also used by many investigators (Bowker et al., 2007; Baldauf et al., 2008; Hu et al., 2009; Kozawa et al., 2012), but measurements included the number concentration of ultrafine particles, and high spatial resolution (10 m) was achieved with reproducibility; 2) a roadway intersection was selected as the main measurement area; and 3) particle-bound PAHs were used as a tracer of vehicle emission.

The middle-scale measurement was undertaken on the access way to Yonsei University at Sinchon-dong on the afternoon of 18 June 2012, when the prevailing winds blew from southwest to northeast, as shown in Table 1. Yonsei University campus was selected as the middle-scale measurement area because there were no other major source in the vicinity and the access road to the campus extended almost perpendicularly to the main roads. Meteorological data during the middle-scale measurement were obtained from the Seodaemun-gu AWS, denoted as W2 in Fig. 2, and are also shown in Table 1.

2.3. Neighborhood-scale measurement

To better understand the effect of wind speed in an urban area, we considered a roadway measurement to represent a line pollution source in Seoul. Fourteen days of measurements were carried out along approximately 14.8 km of arterial roadways, from the northeast to southwest of northern Seoul. Measurements were made on weekday mornings in January and February 2012. The lower mixing layer height in the winter season could enhance the role of horizontal wind speed in the transport and dispersion processes from line emission sources (e.g., roads) as well as the other emission sources. The average traveling speed was 29.1 ± 15.7 km/h, and the traveling time ranged from 40 to 56 min.

Meteorological data during the neighborhood-scale measurement were calculated as an average from three AWSs at Seodaemun-gu, Songwol-dong in Jongno-gu, and Seongsbuk-gu, denoted as W2, W3, and W4, respectively, in Fig. 2, and are also shown in Table 1. Background concentrations of particle-bound PAHs were monitored at a playground of high school located at the center of micro-scale measurement route indicated as background 1, Yonsei University indicated as background 2, and KIST indicated as background 3 in Fig. 2.

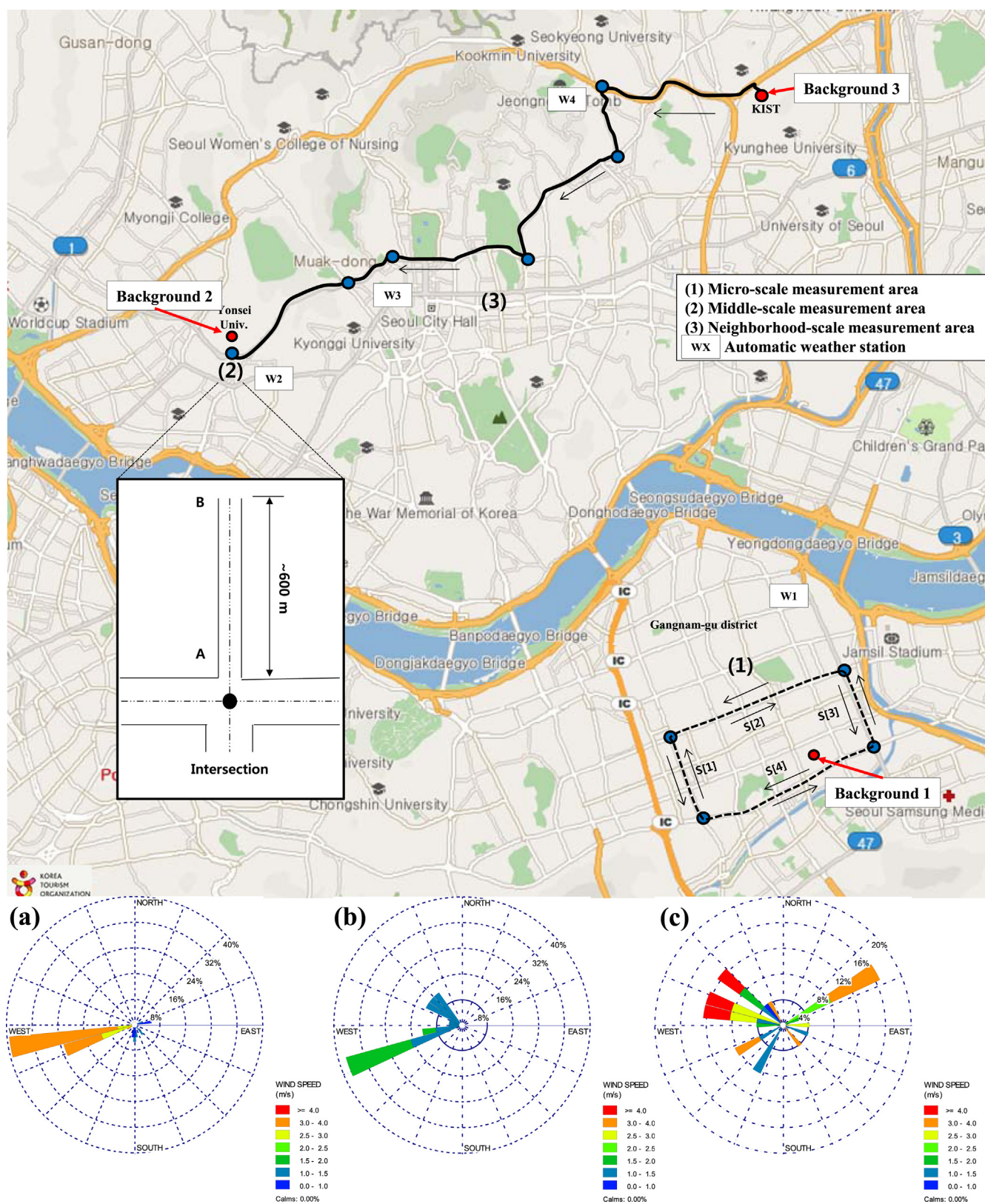


Fig. 2. The areas selected for measurements at the micro-scale (1), middle-scale (2), and neighborhood-scale (3) and windroses in Seoul, Korea. (a), (b), and (c) show windrose of micro-, middle-, and neighborhood-scale, respectively.

Table 1

Outlines of the measurements at three different scales and the associated meteorological data.

Scale	Date	Number of repetition	Time	Traveling speed (km/h)	Temperature (°C)/RH (%)	Wind direction	Wind speed (m/s)
Micro	20/02/12	2 for each direction	10:30–18:30	14.9 ± 4.9	4/46	WSW	2.4 ± 1.1
Middle	18/06/12	3	13:00–17:00	8.4 ± 3.9	30/38	WSW	1.5 ± 0.3
Neighborhood	16/01/12–10/02/12	14	10:30–12:00	17.9 ± 2.0	–0.9/37	WNW	2.1 ± 0.9

± indicates 1 σ of measured values.**Table 2**

The number of road lanes, width, and distance for each measurement section for the measurements at the micro-scale.

Section	Road lane (both directions)	Road width (m)	Distance (km)	Traveling speed (km/h)
Section [1]	10	33	1.7 ± 0.1	9.2 ± 1.5
Section [2]	8–10	26–33	3.4 ± 0.0	16.5 ± 6.1
Section [3]	12	39	1.4 ± 0.1	17.7 ± 3.6
Section [4]	8–10	26–33	3.6 ± 0.1	16.1 ± 2.4

± indicates 1 σ of measured values.

2.4. Mobile laboratory

An ML was used to measure air pollutant concentrations on the roads during movement, especially for the micro-scale measurement within the road (Kim et al., 2015). This allowed the collection of detailed pollutant distribution data with a high spatial and temporal resolution, without altering the traffic flow. The sampling inlets of the ML used in this study consisted of a stainless steel pipe and Teflon tubing for particulate and gaseous pollutants, respectively. The sampling height was approximately 2 m above ground level. A PAH monitor with a photoelectric aerosol sensor (PAS 2000, EcoChem, League City, TX, USA) was installed in the mobile laboratory to determine representative particulate PAHs (Westerdahl et al., 2005; Zhu et al., 2008). Particle-bound PAHs with three or more benzene rings were detected through the measurement of electrons emitted by organic molecules on particles irradiated by UV light. Thus, we could only quantify the total concentrations of particle-bound PAHs that have a photo-electric activity. No information about the PAH species could be obtained through the PAHs monitor. Further information regarding the principle of measurement by this technique is described in detail elsewhere (Burtcher, 1992; Dunbar et al., 2001). The sampling air flow rate for the PAH monitor was set at 2.0 L/min, and a lamp with a wavelength of 220 nm was set to turn off for 2 s for the background measurement and then turn on for 4 s for the sample measurement.

Additional measurements were also simultaneously undertaken as follows. A fast mobility particle sizer (FMPS Model 3091, TSI Inc., Shoreview, MN, USA) was used to measure the particle number size distribution based on the particle mobility diameter. The surface area of nanoparticles that deposit in the lung, the alveolar (A) regions of the human respiratory tract, was monitored by a nanoparticle aerosol monitor (NAM Aerotrak 9000, TSI Inc.) equipped with a PM_{1.0} cyclone. Other measured pollutants included BC, NO_x, CO, and CO₂. The BC sample line was equipped with a PM_{2.5} cyclone, before utilizing a portable aethalometer (AE42 with dual-wavelength, Magee Scientific, Berkeley, CA, USA). Nitrogen oxides were measured with a NO_x analyzer (AC32M, Environmental S.A., Poissy, France). Carbon monoxide and carbon dioxide were measured with a non-dispersive infra-red (NDIR) analyzer (CO12M, Environmental S.A.). The FMPS and NAM had a 1-s time resolution, while CO/CO₂, NO_x, and the aethalometer had 3-, 10-, and 60-s time resolutions, respectively. GPS information was collected every second, and the times on all instruments were synchronized before starting each measurement. During the measurement periods, calibrations for gaseous instruments and zeroing of all instruments were regularly carried out before and after all measurements for

quality control and assurance. Further information about the ML is described in Kim et al. (2015).

3. Results

Fig. 2(a)–(c) show the wind direction and speed as wind rose from the nearest (or average of three nearest) AWSs during the micro-, middle-, and neighborhood-scale measurement periods, respectively. Prevailing winds with high speeds blew from the WSW, WSW, and west northwest (WNW) during the micro-, middle-, and neighborhood-scale measurements, respectively. Average wind speeds were 2.4 ± 1.1, 1.5 ± 0.3, and 2.1 ± 0.9 m/s for the micro-, middle-, and neighborhood-scale measurements, respectively, as listed in Table 1. Average roughness height for Seoul Metropolitan area is found to be 2.17 m while the average roughness height for neighborhood-scale measurements (Seodaemun-gu, Jongno-gu, and Seongbuk-gu) is found to be 1.78 m (Choi et al., 2007). The plot shown in Fig. 2(c) and the average calculation were made using wind data from the three nearest AWSs. Wind rose and frequency were obtained using a WRPLOT program (1998–2008, Lakes Environmental Software, USA). Measured particle-bound PAHs concentrations at background sites 1 for micro-scale, 2 for middle-scale, and 3 for neighborhood-scale were 9.9 ± 4.9, 8.8 ± 3.2, and 29.2 ± 14.2 ng/m³, respectively.

3.1. The effect of wind direction at the micro-scale

During the measurement, the prevailing wind blew from the WSW direction and wind speeds ranged from 0.6 to 4.1 m/s. A wind class frequency distribution shows that moderate wind speeds below 2.0 m/s occurred for 32.4% of the period, while strong wind speeds over 2.0 m/s accounted for the remaining 67.6%. As shown in Fig. 2(a), a strong wind blew from the WSW or west direction. For the micro-scale measurement, we divided the measurement area into four sections, as shown in Fig. 2(1), to classify the driving lanes into upwind and downwind areas by section. For example, the lanes in sections [1] and [2] were located in the downwind area for clockwise directional driving and in the upwind area for counter-clockwise directional driving, as long as the prevailing wind direction was from the WSW. Note that Korea is a right-hand drive country. Fortunately, the prevailing wind direction did not change significantly and the strong prevailing wind blew relatively constantly during the short measurement period. On-road measurements were undertaken twice for each driving direction.

Fig. 3 shows the measured concentration of particle-bound PAHs in both the upwind and downwind areas for each

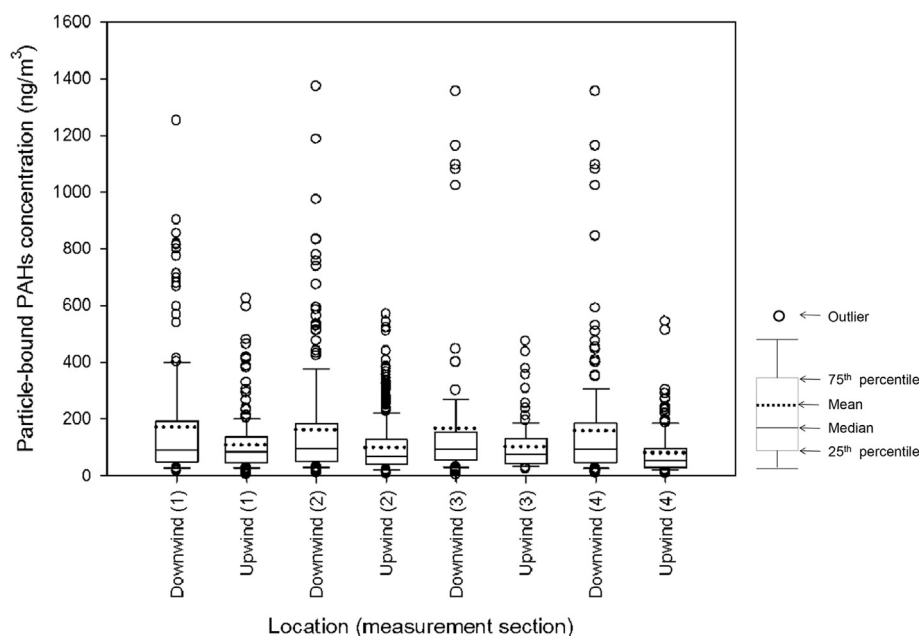


Fig. 3. Comparison of the concentrations of particle-bound PAHs measured in upwind lanes and downwind lanes at the micro-scale (solid lines in the box plots indicate median values while the dotted lines in the box plots indicate mean values of the observations).

measurement section. In all measurement sections, the average concentrations of particle-bound PAHs measured in the downwind lanes were found to be higher than those measured in the upwind lanes of the same road. The concentrations in the downwind lanes were almost double those of the upwind lanes. The outliers for “downwind” and “upwind” measurements also exhibit the importance of downwind and upwind lanes in the same roadways. The outliers, maximum observation values, and 75th percentile values for downwind lanes also showed high particle-bound-PAHs and variations compared to those observed for upwind lanes in the same roadways as shown in Fig. 3, indicating the significant influence of motor vehicles running the opposite roadways. This big difference in air pollution levels within the width of the road strongly suggests the importance of prevailing wind direction.

3.2. The effect of wind direction at the middle-scale

Fig. 4(a) shows the measured exponential decrease of average concentrations of particle-bound PAHs at an interval of 10 m for three replicated measurements on the Yonsei University campus access road as a function of the distance from the roadway intersection when the prevailing wind was from WSW to east northeast (ENE). As shown in Table 1 and Fig. 2, the average wind speed during the measurement was 1.5 ± 0.3 m/s.

The concentrations 18 m from the intersection had decreased to approximately half that of the maximum measured 2 m from the intersection, which is similar to results presented by Zhu et al. (2002b) who reported ~60% decrease in CO, BC, and particle number concentration from 30 m to 90 m in the downwind from a major roadway. The exponential decrease was found to be a good estimator for predicting total particle number, BC, and CO at different locations (Zhu et al., 2002b). Thus, an exponential decrease curve for particle-bound PAHs was determined using the SigmaPlot nonlinear curve fitting procedure and was compared with those for particle number and BC concentrations observed by Zhu et al. (2002b), as shown in Fig. 4(b). The average wind speeds in these two studies were all close to 1.5 m/s. Concentrations of particle-bound PAHs observed in the present

study decreased at a similar rate as those previously reported for particle number and BC (Zhu et al., 2002b) close to the sources, implying that atmospheric dilution had a comparable role in the present study with the previous study (Zhu et al., 2002b). Approximately 200 m from the intersection, the measured pollutants were present at concentrations approaching the local urban background, which agrees with the results of other studies (Morawska et al., 1998; Shi et al., 1999; Hitchins et al., 2000). The asset of using the ML for the observation of exponential decrease in air pollutants from the roadway is that high temporal and spatial resolution can be achieved with simplicity of the measurement. Spatial resolution of the measurements for particle-bound PAHs was about 10 m in our study. This study also demonstrates the feasibility of using the ML as a fast measurement method for traffic-related air pollutants in the vicinity of roadway environment, which can be practically used for human exposure or modeling studies.

3.3. The effect of wind speed at the neighborhood-scale

To clarify the effect of wind speed at the neighborhood-scale, measurements were divided into low (wind speed <2.0 m/s) and high (wind speed >2.0 m/s) wind speed days. Overall, high pollutant concentrations were observed on low wind speed days ($n = 7$), while low pollutant concentrations were observed on high wind speed days ($n = 7$), indicating a remarkable dispersion and dilution of air pollutants from the line source on windy days and an accumulation of air pollutants on less windy days. The average wind speeds on low and high wind speed days were 1.4 ± 0.3 and 2.8 ± 0.8 m/s, respectively. The average traffic volumes for low and high wind speed days were 3508 and 3408 vehicles/day, respectively, indicating that wind speed is the primary factor determining air pollution levels at the neighborhood-scale in an urban area. Particle-bound PAHs concentrations on a low wind speed day (17 January 2012) and high wind speed day (7 February 2012) as a function of the distance that the ML traveled showed that air pollutant concentrations are strongly dependent on the wind speed of the measurement day over the whole measurement area. The

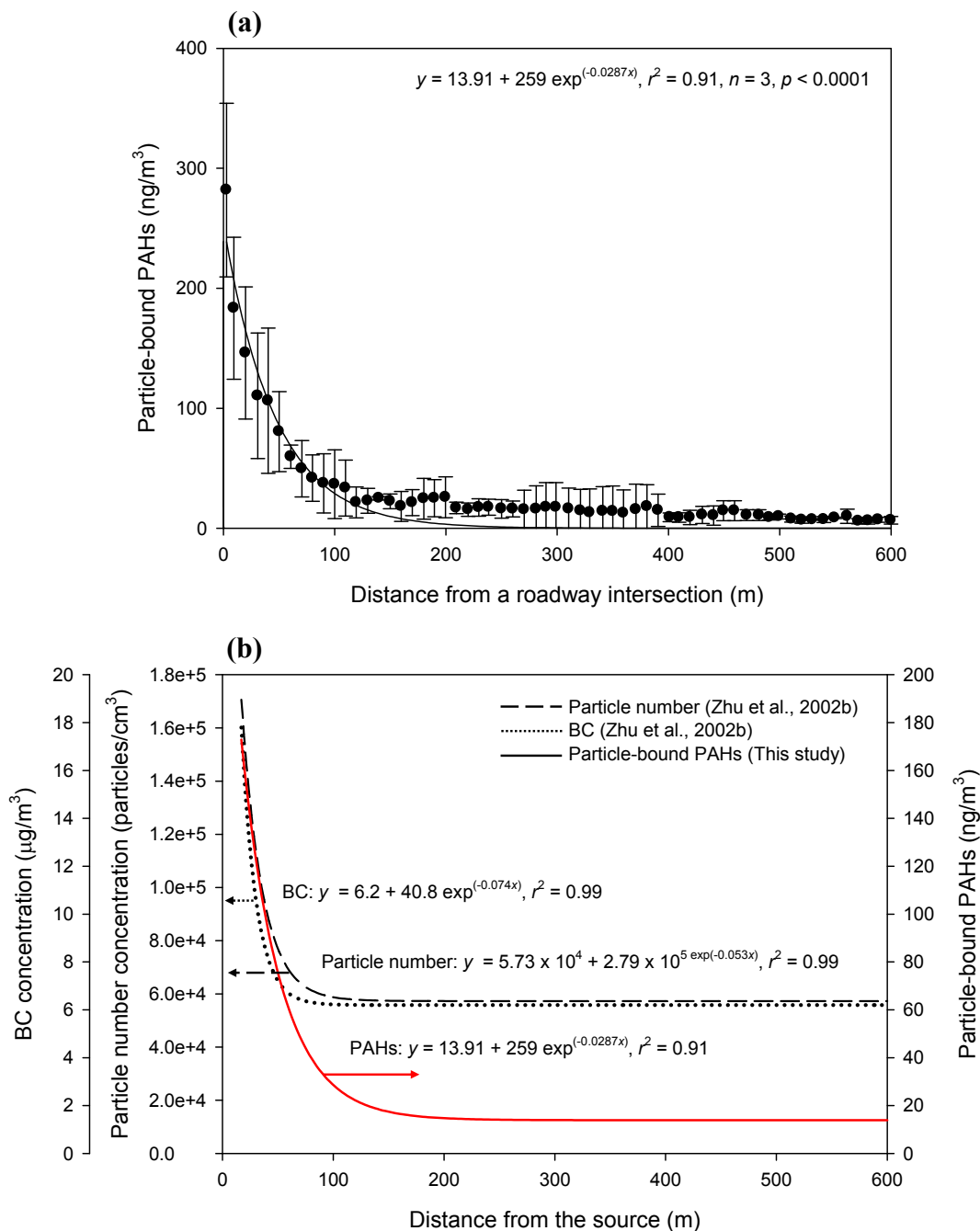


Fig. 4. Exponential decrease in the concentration of particle-bound PAHs as a function of distance from a roadway intersection (a) and exponential decrease of particle number and BC mass as a function of distance from a roadway observed by [Zhu et al. \(2002b\)](#) compared with the results of this study (b).

average concentration of particle-bound PAHs on low wind speed days was 297 ± 271 ng/m³, while on high wind speed days it was 89 ± 88 ng/m³. 25, 50, and 75 percentile of particle-bound PAHs on a low wind speed day were 139, 213, and 365 ng/m³ whereas those on a high wind speed day were 35, 61, and 116 ng/m³, respectively, indicating about three folds higher concentrations for low wind speed day as shown in [Fig. 5](#).

The average concentration of particle-bound PAHs with wind speed for each measurement day was plotted together to show the relationship of air pollutants with wind speed ([Fig. 6](#)). The plots were also divided into two groups for low and high wind speed days to clarify the distinct wind speed effect when the wind speed was higher than 2.0 m/s. The measured concentration of particle-

bound PAHs displayed an inverse relationship with wind speed. This inverse relationship was stronger when only high wind speed days were considered (total days: $r = 0.73$, high wind speed days: $r = 0.88$).

4. Discussion

The concentrations of particle-bound PAHs were almost double within the road for all sections of the downwind lanes compared to the upwind lanes, indicating that pollutants emitted from the opposite lanes were transported by wind moving into a downwind location. Interestingly, gaseous pollutants such as NO_x did not show a significant difference, while concentrations of BC displayed

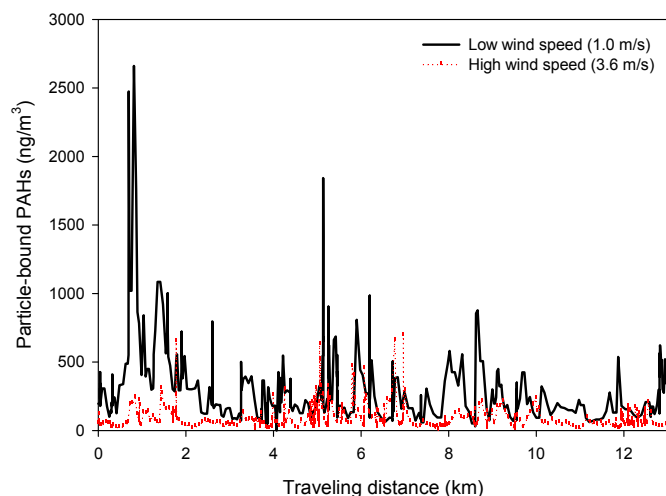


Fig. 5. Comparison of the concentrations of particle-bound PAHs for the measurements on low (17 January 2012) and high (7 February 2012) wind speed days.

significant differences between upwind and downwind measurement locations. The average BC concentrations in upwind and downwind locations were 8.3 and 14.9 $\mu\text{g}/\text{m}^3$, respectively, indicating concentrations almost a factor of 1.8 times higher in downwind locations compared to upwind locations. For the total number concentrations of particles ranged from 5.6 to 560 nm in diameter, there was not a large difference in concentration between upwind and downwind locations. Number concentrations of particles larger than 100 nm in diameter displayed a small increase in downwind locations.

The increase in the number concentrations of large particles in downwind locations was likely due to larger particles being more affected by wind direction and speed compared to smaller particles. The increased concentrations of BC and particle-bound PAHs observed in downwind roadways also support this explanation because a large portion of BC and particle-bound PAHs are associated with micron or submicron particles (Kim et al., 2015). Furthermore, it is likely that condensation and coagulation of smaller particles emitted from motor vehicles may influence on the increase in relatively larger particles at downwind locations under given wind speed in the present study. Small particles ($D_p < 100$ nm) behave like gas molecules, with diffusion being the

primary transport mechanism. Gaseous pollutants in the micro-scale environment (within a few tens of meters) also did not exhibit significant downwind and upwind effects in our study, indicating that diffusion was the primary transport mechanism. From our observations, it can be concluded that wind does not significantly influence gaseous pollutants and smaller particles ($D_p < 100$ nm) within the scale of a few tens of meters, because these pollutants are mainly governed by diffusion as their primary transport mechanism.

Particle number, BC mass, and the surface area of lung-depositing particles decreased to approximately half those of the maximum recorded (2 m from the intersection) at a distance of about 150 m from the intersection, which agreed with the results of Hitchins et al. (2000), while the concentration of particle-bound PAHs displayed a more rapid exponential decrease occurring at about 18 m from the intersection, as shown in Fig. 4. Approximately 300 m from the intersection, the concentration of measured pollutants had decreased to levels approaching the local urban background, which agreed with the results of previous studies (Morawska et al., 1998; Shi et al., 1999; Hitchins et al., 2000). When the exponential decrease of total particle number and BC mass as a function of distance from the roadway was compared with studies by Zhu et al. (2002a,b), a relatively slower decrease of pollutant concentrations was observed in this study due to lower pollutant concentrations. Note that freeways were investigated by Zhu et al. (2002a,b), while an intersection of arterial roads was investigated in our study. The high particle number concentration with a rapid decrease as a function of distance from the roadway was directly related to the freeway environment where motor vehicles operate at high speeds, and the background site experienced no pollutant source other than the freeway. Particle number concentrations on a roadway increase with vehicle speed and decrease when engines are idling (Sioutas et al., 2005). However, our study provides information relevant to the health risk of an urban roadway environment, to which many people are potentially exposed (Kim et al., 2014).

Particle-bound PAHs displayed an inverse relationship with wind speed. The inverse relationship was strong when only high wind speed days were plotted against the concentration of particle-bound PAHs concentrations, implying an insignificant wind speed effects when the wind speed was below 2.0 m/s. This relationship suggests that wind speed has an important role in atmospheric dilution when it exceeds 2.0 m/s in urban areas. Therefore, pollution levels due to vehicle emissions in a roadway environment are likely to be higher on low wind speed days (wind speed < 2.0 m/s). Other studies focusing on the carbonaceous components of ultra-fine and fine particles in the roadside environment have also observed an increased influence from motor vehicle emissions when the wind speed was lower than 2.5 m/s (Kim et al., 2011). Other measured pollutants also exhibited an inverse relationship with wind speed, as shown in Fig. 7. On-road measurement characterizes neither the vehicle emissions nor the ambient levels of pollutant concentrations. The on-road measurement data shows the transient level of air pollutant concentration from the motor vehicle exhausts to on-road environment. Although the measurements were taken inside of the mixing zone of the roads, our on-road measurement data shows that the concentrations of air pollutant are closely related to the wind effect, indicating the importance of wind effect on pollutant dispersion of on-road environment.

5. Conclusions

The wind effect for air pollution levels of on-road environment that covers the transport and dilution of traffic-related air

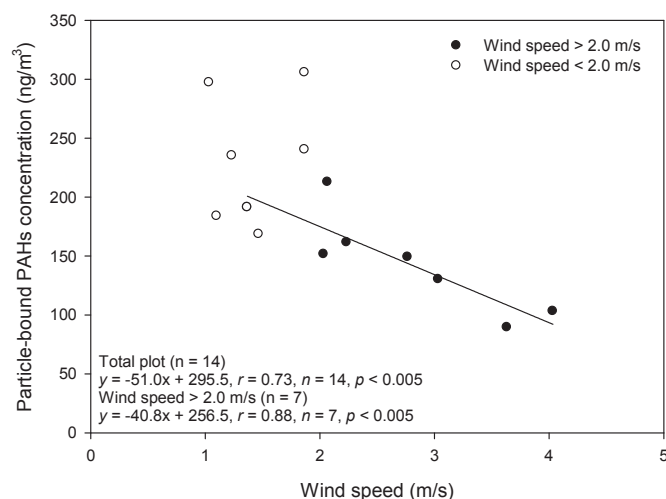


Fig. 6. Inverse relationships of the concentration of particle-bound PAHs with wind speed.

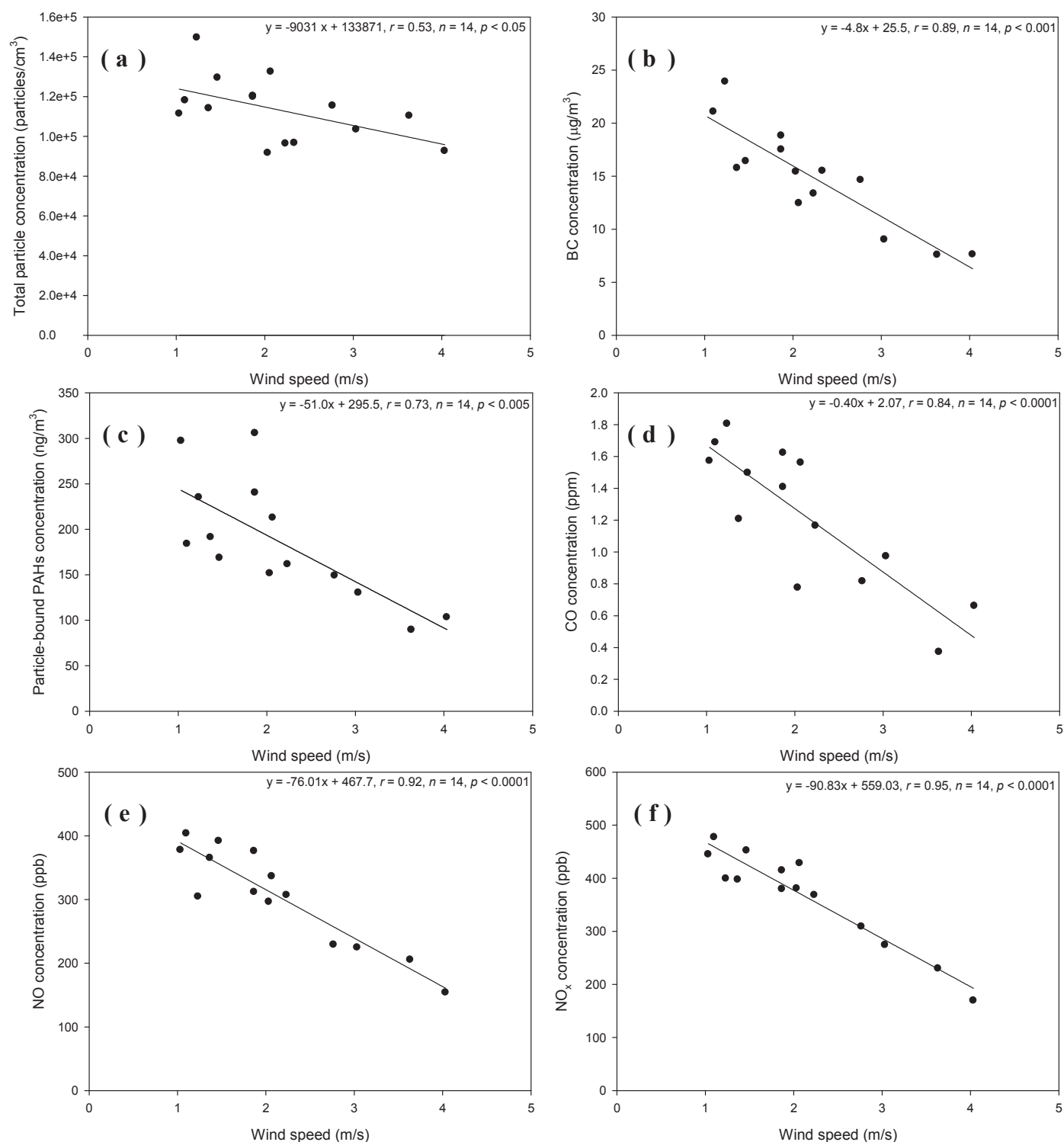


Fig. 7. Inverse relationships of air pollutants with wind speeds ($n = 14$).

pollutants on roads, at the roadsides and along the roads was studied systematically in order to explain the variations in urban air pollution measured at these locations.

The micro-scale measurement emphasizes that pollutant concentration can be significantly influenced by the prevailing wind direction showing higher concentration in the downwind lanes than in the upwind lanes even within a small scale of roadway

environment. The middle-scale measurement for the exponential decrease of particle-bound PAHs showed how traffic-related air pollutants can be transported from a roadway to our living environment under given wind direction and speed. Particle-bound PAHs showed very high concentration (376 ± 72 ng/m³) at roadside and low concentration (6 ± 3 ng/m³) at background site, reflecting that the particle-bound PAHs can be used as a useful

indicator of vehicle emissions around the roadway environment. Similarly, Kozawa et al. (2009) also reported that particle-bound PAHs showed relatively higher concentrations closed to a highway while it showed lower concentrations about 150 m away from the highway due to relatively low background levels of particle-bound PAHs away from line sources. We have also demonstrated the feasibility that the exponential decrease in air pollutants from the roadway can also be observed with high temporal and spatial resolution when using the ML as a fast measurement method.

The on-road concentration of particle-bound PAHs measured along the arterial roads displayed a pronounced negative linear relationship with prevailing wind speed at the neighborhood-scale, which implies that the effect of dilution processes caused by wind speed is important not only at the ambient location but also at the road, or adjacent to the line emission source. It is crucial to understand how the wind affects to the pollution levels of on-road environment in order to interpret the obtained data between measurements for different days appropriately. Otherwise, it would be very difficult to find out the causes for the large concentration differences between measurements for the on-road pollution levels. The rapid on-road measurements at three spatial scales undertaken in this work will contribute to our understanding of the effects of wind on air pollution in the urban roadway environment.

Conflict of interest

The author states that there is no conflict of interest.

Acknowledgments

This study was supported in part by the Center for Environmentally Friendly Vehicles and the Institutional Program (2E25290) of the Korea Institute of Science and Technology.

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