The potential of commercial sensors for short-term spatially dense air quality monitoring: based on multiple short-term evaluations of 30 sensor nodes in urban areas in Korea

Ho-Seon Park¹, Ra-Eun Kim¹, Yong-Mi Park¹, Kyu-Cheol Hwang¹, Seung-Hyun Lee¹, Jae-Jin Kim², Jin-Young Choi³, Dae-Gyun Lee³, Lim-Seok Chang³ and Wonsik Choi²*

¹Pukyong National University, Department of Earth Environmental System Science, Busan, Korea.
²Pukyong National University, Department of Environmental Atmospheric Science, Busan, Korea.
³National Institute of Environmental Research, Climate and Air Quality Research Department, Incheon, Korea.

Abstract

Recently, highly spatially dense air quality monitoring networks using low-cost sensors have been attempted worldwide. However, the data quality from these sensor networks remains to be validated. This study attempted to assess the potential of low-cost sensors for spatially dense air quality monitoring. Thirty sets of air quality sensor nodes for CO, NO₂, O₃, PM₂.₅, and PM₁₀ were custom-built to evaluate consistency, both among 30 sensor nodes and between the sensor node and instruments that use federal reference/equivalent methods (FRM/FEM) in the real atmosphere under distinctly differing meteorological conditions (summer and winter) in Seoul and Busan, Korea. We found that commercially available low-cost sensors have great potential for short-term air quality monitoring studies in urban areas, at least for one-month period, given that (1) the self-consistency among 30 sensors was great (R²>0.93), (2) the consistency between the sensors and FRM/FEM instruments was good (R²>0.87 for overall comparison periods), and (3) the consistency among sensors and between the sensors and FRM/FEM instruments was maintained in the summer and winter. However, vigorous data post-processing is needed to obtain reliable air quality data. For longer-term or temporally discontinuous monitoring, several issues should be addressed, including sensor lifetime, degradation of sensor performance over time, long warm-up times for gaseous pollutants sensors. The O₃ sensors required minimal post-processing correction. The PM₂.₅ and CO sensors agreed well with FEM instruments after appropriate scale correction. The NO₂ sensors required additional efforts to correct for effects of meteorological conditions and interfering materials. Our results suggest that for investigating spatiotemporally heterogeneous distributions of air pollutants in various urban environments, a three-dimensional sensor network can be a useful tool for short-term monitoring, as long as data are corrected properly.

Keywords: air quality; high-resolution pollution monitoring; low-cost sensors; sensor network; spatiotemporal variations.

* Correspondence to: Wonsik Choi (wschoi@pknu.ac.kr)
INTRODUCTION

Air pollution has become one of the major global threats to public health, particularly in low- and middle-income countries (WHO, 2018). About 4.2 million annual deaths could be attributed to ambient air pollution based on the updated global burden of disease in 2015 (Ostro et al., 2018). In addition, about 90% of the global population is exposed to high levels of air pollutants (WHO, 2018). In this respect, public concerns about air pollution have increased rapidly worldwide, leading to changing paradigms in air quality monitoring, from official monitoring stations operated by governments to low-cost air quality sensors carried by citizen scientists to monitor the air quality at the location where they are breathing (Kumar et al., 2015).

In developed countries, one air quality monitoring station (AQMS) operated by a government generally represents 100,000 people (Lewis and Edwards, 2016). Seoul, South Korea, which is one of the most populous cities in the world, operates 25 urban and 18 roadside AQMS, each of which covers an area of roughly 4 × 4 km and represents 227,442 people (Bae et al., 2013; MOIS, 2019). However, air pollutants, particularly those emitted from vehicles in cities, are highly spatially heterogeneous due to the rapid dilution of vehicle emissions and the complex wind fields and turbulence created by built environments (Morawska et al., 2008; Karner et al., 2010; Choi et al., 2016). For example, elevated concentrations of pollutants from vehicular emissions are diluted to ambient levels within a few hundred meters in daytime (Karner et al., 2010). At night when the atmosphere is stable, roadway plumes can reach up to 2 km, but elevated concentrations decrease sharply within the first several hundred meters (Hu et al., 2009; Choi et al., 2012). Wind fields and turbulence intensities are modified by the surrounding built environment (Kim and Baik,
making ventilation more (or less) effective below urban canopy levels, even on a block scale (Pirjola et al., 2012; Choi et al., 2016; Choi et al., 2018).

The high spatiotemporal heterogeneity of pollutant distributions in urban areas requires spatially dense air quality monitoring in near real time to control exposure (Kumar et al., 2015). In addition, as public concern over air pollution has grown recently, the market for low-cost air quality sensors has also grown rapidly. The reasons for using low-cost air pollution sensors are diverse, from warning systems for gas leakage and occurrence of high pollution events to high-density air quality monitoring in cities (Lewis and Edwards, 2016). This study focused on assessing the potential of low-cost sensors for spatially dense air quality monitoring.

Numerous recent studies have evaluated the performance of low-cost sensors for their specific purposes under both laboratory and field conditions (Morawska et al., 2018). Bart et al. (2014) compared semiconductor O₃ sensors with a ratified reference instrument over 48 h and showed strong agreement between them, with a standard error of 3 ppb in the ambient O₃ range (0–50 ppb). Moltchanov et al. (2015) compared two pairs of electrochemical NO₂ and O₃ sensors at four different sites, including an AQMS. They reported good correlations between the two sensors at all sites, with $r > 0.92$ and 0.78 for O₃ and NO₂, respectively. They also found good correlations between the sensors and reference instrument at the AQMS site. However, the relationship between the sensor and reference instrument differs among individual sensors. Masson et al. (2015) conducted a long-term comparison between an electrochemical NO sensor and the federal reference method (FRM) instrument and reported good agreement, with a root mean square error of 14.6 ppb after correcting for sensitivity and offset, which was improved to 13.6 ppb with additional corrections using multivariate analysis, although they did not discuss signal attenuation effects over time. Mead et al. (2013) carried out comprehensive testing of the performance of
NO, NO\textsubscript{2}, and CO sensors and quantified the effects of meteorological conditions and interfering materials on sensor outputs, suggesting the necessity for sophisticated data correction. They deployed 46 sensor nodes in Cambridge, United Kingdom, and suggested that low-cost sensors have great potential for high-density air quality monitoring. In addition, they reported that the degradation of NO sensor performance was negligible over one year. Nonetheless, their sensor reproducibility tests were limited to two pairs of sensor nodes over short time periods. Zicoba et al. (2017) carried out intercomparison tests of 66 infra-red LED-based dust sensors for PM\textsubscript{2.5} and the Grimm instrument in both indoor and outdoor settings. The correlations among the sensors were $R^2 > 0.9$, showing excellent sensor reproducibility. However, the correlations between the sensors and the Grimm instrument were not satisfactory, with $R^2 = 0.53$ and 0.22 in indoor and outdoor environments, respectively.

Although many studies have evaluated sensor performance, most have used a limited number of sensors for comparison with a ratified reference instrument, and results from intercomparison tests among large numbers of sensors, which are essential for quantifying the spatial heterogeneity of air pollutants at a fine scale, are lacking. In addition, despite their promising applications, the quality of data obtained from low-cost sensors remains in doubt (Cross et al., 2017). This data quality issue should be carefully addressed before low-cost sensor networks are used in practice, as the spread of unreliable information may cause further confusion among the public, irrational anxiety or relief about air quality, and the establishment of inappropriate strategies for exposure reduction (Masiol et al., 2018).

In this study, we investigated the performance of low-cost sensors under different meteorological conditions, comparing ambient concentrations of air pollutants among a large number of sensors (30 sets of sensor assemblies to examine inter-sensor variations within a sensor network) and between the sensors and the instruments used in federal reference methods (to determine the accuracy and sensitivity of the sensor as well as meteorological or
temporal effects on sensor characteristics). Piedrahita et al. (2014) suggested that correcting sensor data from collocation calibrations produced more reliable concentrations than laboratory calibrations. Based on experimental results, we discuss the potential of sensor networks for actual air quality monitoring as well as the questions and problems that should be addressed for practical applications of air quality monitoring sensor networks.

During the intercomparison tests between the sensor assembly and FRM or federal equivalent method (FEM) instruments, 29 sensor assemblies were deployed in the center of Seoul, South Korea, in an area of about 800 × 800 m to investigate the heterogeneous distributions of air pollutants in the city. The detailed results of the sensor network study will be discussed in subsequent paper.

METHODS

Air quality sensor node

In this study, the focus was on criteria air pollutants: CO, NO2, O3, PM2.5, PM10, temperature, and relative humidity. Temperature and humidity data were used for further correction of meteorological effects on sensor performance. Because the goal of the study was not to evaluate the performance of numerous commercially available sensors, but rather to apply these sensors to the construction of a highly spatially resolved air quality monitoring network, we chose a sensor for each pollutant from the literature based mainly on evaluation results published by the United States Environmental Protection Agency (US EPA) and the South Coast Air Quality Management District (SCAQMD; California, USA). The US EPA and SCAQMD (Air Quality Sensor Performance Evaluation Center) have evaluated commercially available low-cost air pollutant sensors, providing comparisons with federal reference and equivalent methods in the real atmosphere and laboratory since 2013 (EPA,
2017; SCAQMD, 2017). Additionally, we reviewed published articles containing sensor
evaluations (e.g., Williams et al., 2013; Jiao et al., 2016; Cross et al., 2017; Ly et al., 2018).
Thus, although we cannot say that the selected sensors are the best performing sensors
available, they represent sensors with high sensitivity and accuracy on the market.

The sensors selected are listed in Table 1. A particulate matter (PM) sensor
(PMS5003, Plantower) employs a laser scattering technique with laser diode for a light
source and photodiode detector for a receptor (Zheng et al., 2018). CO (CO-B4, Alphasense)
and O₃ (SM50, Aeroqual) sensors use an electrochemical sensing technique, which measures
gas concentrations by measuring the current generated by the potential difference between
working and counter electrodes when the target gas undergoes an oxidation or reduction
reaction on working electrode (Mead et al., 2013). NO₂ concentrations were measured with a
metal oxide sensor (MiCS-2714, SGX), which measures the change in the electrical
conductivity occurring when the target gas contacts the semiconductor surface (Urasinska-
Wojcik et al., 2017).

Among these sensors, the CO and O₃ sensors were factory calibrated, but NO₂
sensor was not calibrated initially. Although we can obtain PM concentrations directly from
sensor outputs, the manufacturer did not provide information on a calibration algorithm
(Zheng et al., 2018). However, SCAQMD mentioned that PMS5003 was factory calibrated
(SCAQMD, 2019). We did not apply any additional pre-adjustment to sensor readings before
storing the data. All the adjustments were made in the data post-processing stage based on
intercomparison test results in real atmospheric conditions. The air quality data were
averaged for 10 s and stored on a microSD card at a 0.1-Hz temporal resolution in a data-
logger (ATmega2560, Atmel), which could be simultaneously transferred via a wireless
network to a smartphone (with PHPoC Shield for arduino). However, due to a limitation of
wireless network access for 30 sensor nodes, we used only the stored data in SD card for data
analyses, and the wireless connection was used only for checking the mal-function of sensor node with a direct connection between the sensor node and smartphone.

The sensors were assembled in a platform made of acrylonitrile butadiene styrene plastic (dimensions 200 × 100 × 300 mm; Fig. 1). An intake for ambient air and an exhaust for in-box air were located at the top and bottom of the front side of the platform, with a fan in the exhaust portion. The calculated ventilation rate was more than 20 times the platform volume per minute. We acknowledge that there could be a wall-loss issue for some pollutants (e.g., ozone). In this study, we could not quantitatively determine the effects of wall loss on ozone concentrations in a box. However, Ainsworth et al. (1981) suggested that ozone wall loss can lead to 2% uncertainty at the surface level, and Itoh et al. (2012) discussed the effective ozone lifetime due to wall loss in an acrylic resin tube is about 10^4 s at the surface level. Considering the ventilation rate in our sensor node box, we expect the wall loss effect would be minor. Even in case that we cannot ignore ozone wall loss effect (and for other pollutants), we think this effect could implicitly be compensated when we corrected pollutants concentrations corresponding to the concentrations from the reference instruments.

**Comparison of air quality data among sensors by sampling site, period, and meteorological conditions**

To obtain reliable spatial distributions of air pollutants, the comparability of data among sensors should be ensured, so that a value obtained at one site can be compared directly with values from other sites. To date, most sensor evaluation studies have focused on the accuracy of sensor performance, testing a small number of sensors experimentally (e.g., EPA, 2017; SCAQMD, 2017). To investigate the consistency of data among numerous sensors, we placed 30 sensor platforms together (ten sensors for O\textsubscript{3}) on the rooftop of a three-story building on the Pukyong National University campus. Pukyong National University is
located 650 m west of the coast and is surrounded by heavily trafficked roads in all directions (about 400 m from all streets). Thus, the campus represents the ambient urban air without direct influences from emission sources, considering that daytime roadway plumes dissipate within about 300 m (Karner et al., 2010).

The comparison tests were carried out four times, twice in summer and twice in winter, under distinct meteorological conditions. Within each season, the temporal gap between the two experiments was about four weeks. Each experiment was conducted for 3–5 days, depending on weather conditions. During this temporal gap, experiments using a highly spatially resolved sensor network were conducted in Seoul, the most populated megacity in Korea. The results from this air quality sensor network will be presented separately and are beyond the scope of the present study.

**Intercomparison between the sensor platform and ratified reference instruments by sampling site, AQMS, and meteorological conditions**

A sensor platform was co-located (<1 m) with the sample inlet of the AQMS operated by Seoul Institute of Health Environment for 7 days in summer (August 25–31, 2017) and 10 days in winter (January 11–20, 2018). The sample inlet was located at 16 m a.g.l. The instruments used for FRM/FEM at the AQMS are CA-751 (non-dispersion infrared absorption) for CO, NA-721 (chemiluminescence) for NO₂, OA-781 (non-dispersive ultraviolet absorption) for O₃, and PM711 (beta ray attenuation) for particulate matter (PM) (Ambient Air Monitor 700 Series; KIMOTO Electric Co., LTD, Japan).

The AQMS is located in the center of Seoul (Jongno-gu, 37.572°N, 127.005°E) and is surrounded densely with small buildings (mostly three- to four-story buildings). In this area, vehicular emissions are the major source of air pollutants, but there are no prominent emission sources in close proximity to the AQMS. Possible emission sources that may affect
the AQMS are the eight-lane road (Jong-ro) located about 100 m south, the six-lane road (Yulgok-ro) about 300 m east, and the four-lane road (Dongho-ro) about 270 m west of the AQMS. The purpose of this intercomparison study was not to investigate air quality affected by vehicular emission sources; thus, we did not analyze the traffic rates on the surrounding roads. To investigate meteorological effects on variations in the sensor characteristics, we obtained meteorological data from a nearby automatic weather station operated by the Korea Meteorological Administration, as well as temperature and relative humidity measured inside the sensor platform.

RESULTS AND DISCUSSION

Meteorological conditions

Many sensor evaluation studies have been conducted under specific meteorological conditions (e.g., Williams et al., 2013; Masson et al., 2015; Jiao et al., 2016). The meteorological conditions in Korea, however, vary significantly from season to season (roughly from −20°C to +30°C and from dry to humid). In this study, an intercomparison among sensors and between sensors and FRM instruments was conducted under two distinct weather conditions (hot and humid in summer and cold and dry in winter). In addition, for each season, two intercomparison experiments (among sensors and between each sensor and the corresponding FRM instrument) were conducted in close temporal proximity, within one month. Thus, the meteorological conditions of the two intercomparison experiments were similar in both summer and winter (Table 2).

Intercomparison among sensors

To investigate the consistency of sensor readings among a massive number of
sensors, the comparisons of data were made with unadjusted raw signals from sensor nodes. In these comparisons, we used 15-minute averaged data to examine if the sensors have the potential to capture high temporal variations in pollutants concentrations in cities. All 30 sensors agreed well with each other for all tested pollutants in both summer and winter. Here, we present the statistical results obtained from the comparison of 29 sensor platforms with one reference sensor platform (the reference sensor platform was installed at AQMS to conduct further intercomparison tests with reference instruments). The mean $R^2$ values of the 29 sensors were greater than $0.933 \pm 0.012$ in summer and $0.965 \pm 0.020$ in winter for all pollutants (Table 3). Slightly poorer linearity in summer for CO, PM2.5, and PM10 was likely caused by relatively low ambient concentrations in summer compared to those in winter (O3 showed the opposite trend, with higher summer concentrations). Thus, the consistency among sensors appears to be sufficient to determine the relative spatial distributions of pollutants obtained by the sensor network, at least under similar weather conditions. Accuracy issues are discussed in the next section.

Although the linearity among sensors was excellent for all pollutants, the slopes of 1:1 plots between the 29 sensors and the reference sensor were more widely scattered for NO2 and CO compared with PM and O3 sensors. Assuming the zero values of all sensors are identical, if no slope correction is carried out, the concentrations among sensors may differ by up to 40% for CO (slopes ranged from 0.74–1.13 in summer), 73% for NO2 (0.93–1.66 in winter), 17% for O3 (0.87–1.05 in winter), and 28% for PM2.5 (0.84–1.12 in winter). Because the PM sensor is based on a light scattering technique, PM2.5 and PM10 showed almost the same temporal variations (discussed in the next section in more detail). Thus, PM10 is not discussed here.

Intercomparison tests among sensors were conducted twice during each season (before and after two-week intercomparison tests with the FRM instruments). The two
datasets obtained from two separate experiments agreed well (falling onto one regression line) for all pollutants in both summer and winter (Supplementary Information ((SI) Figs. S1–S5)). During the two experimental periods in each season, meteorological conditions were similar (Table 2). Thus, we conclude that sensor characteristics do not change over time, at least for a one-month period under similar weather conditions.

The consistency among sensors in different seasons (thus, under different meteorological conditions and with a six-month time interval) agreed well, except for that of the CO sensors (SI Fig. S6), which showed variable consistency. Most CO sensors showed good agreement between seasons (SI Fig. S6a), but some showed different relationships (SI Fig. S6e). Even when the sensors showed inconsistent seasonal relationships, the slopes between the sensors in the two seasons were similar. Thus, these differences were likely caused by zero drift due to differing meteorological conditions or the time interval. Only three sensors out of 29 showed seasonal inconsistency, with a maximum zero-shift of 137 ppb (SI Fig. S6e). Nonetheless, most sensors (26 out of 29, $R^2 > 0.9$) agreed with each other reasonably well. Thus, we conclude that the consistency among sensors is reasonably good regardless of meteorological conditions, and that this consistency is maintained for a period of at least 6 months (sensors were stored in plastic bags filled with N$_2$ gas when they were not operated). Although the consistency among sensors was excellent, the correspondence with air pollution data obtained from FRM/FEM instruments under different meteorological conditions should also be examined, and is discussed in the next section.

**Intercomparison between sensors and reference instruments and meteorological effects**

The reference sensor platform described in **Intercomparison among sensors** was co-located with the AQMS inlet to determine the accuracy and consistency of the sensors compared to FRM/FEM instruments for summer and winter. In these comparisons, we used
1-hour averaged sensor readings because the AQMS provides 1-hour averaged air quality data. Generally, the consistency between sensor readings and the concentrations recorded with FRM instruments was reasonably good for CO, O₃, and PM₂.₅ (Table 4 and Fig. 2).

- **Gas sensors (CO, O₃, and NO₂)**

  The CO sensor agreed well with the corresponding FRM instrument in both summer ($R^2 = 0.814$) and winter ($R^2 = 0.930$). Although the slopes differed significantly (summer: 1.03; winter: 1.63) (Fig. 2a), the data ranged much more widely in winter (0.4–2 ppm) than in summer (0.3–0.8 ppm). Thus, the winter slope likely represents the actual sensor characteristics. Indeed, both the summer and winter data appear to lie along the line fitted for the entire season, and the slope of that line is similar to that for winter. However, considering the possibility of zero shift in the CO sensor, as discussed in *Intercomparison among sensors* (although this shift was observed in a minority of CO sensors), we cannot exclude the possibility that the slope is affected by meteorological conditions. Overall, we conclude that CO sensors have the potential for application in high-density stationary and mobile sensor networks, but their data should be properly calibrated before being used or shared.

  The O₃ sensor showed great consistency with the FRM instrument in both summer (hot and dry conditions) and winter (cold and dry conditions) ($R^2 = 0.894$ and 0.944 in summer and winter, respectively). In addition, the data from the two seasons showed almost identical relationships with the FRM instrument (summer and winter relationships agreed each other within 4% over the 20–100 ppb range). Thus, meteorological factors (temperature and relative humidity) in two different seasons did not affect sensor responses within this ambient concentration range. This finding indicates that O₃ sensors have great potential to be employed in high-resolution sensor networks with minimal post-processing or correction of data. When O₃ concentrations were below 20 ppb, sensor readings were lower than the actual
concentrations. In particular, at levels less than 10 ppb, the O$_3$ sensor did not detect O$_3$.

However, these low concentrations are not of interest from an air quality perspective; therefore, we excluded these data from our analysis (indicated by the brown rectangle in Fig. 2c).

Although the NO$_2$ sensor and FRM instrument have positive correlations in each season, the 1:1 plots were inconsistent between summer and winter, with more widely scattered data compared to other pollutant sensors ($R^2 = 0.41$ in summer and $0.79$ in winter). Because the signals from all NO$_2$ sensors were consistent with each other (Figs. S1 and S2 in SI), their seasonal inconsistency with the FRM instrument and wider scatter appear to have been caused by meteorological conditions and other interfering factors (e.g., O$_3$; Mead et al., 2013). However, after correcting for these factors, NO$_2$ concentrations from the sensor agreed much better with FRM concentrations, as discussed in Post-processing of sensor data and uncertainties. Thus, to employ this NO$_2$ sensor in a sensor network, intensive post-processing of data is required.

- **PM sensor**

The PM sensor showed excellent consistency with the FEM instrument ($R^2 = 0.955$ for PM$_{2.5}$ and $0.917$ for PM$_{10}$). In addition, the 1:1 relationship in summer and winter had the same trend (Fig. 2d and e). The summer and winter relationships with the FEM instrument differed slightly, but this was likely caused by differing ambient levels of PM$_{2.5}$ between the two seasons (summer range: $<10–23$ μg·m$^{-3}$; winter range: 10–100 μg·m$^{-3}$). However, the sensor read concentrations 2.6 and 2.2 times those measured with the FEM instrument in summer and winter, respectively. PM$_{10}$ data were scattered more widely compared to PM$_{2.5}$, particularly in summer ($R^2 = 0.34$; Table 5). This scatter was prominent when PM$_{2.5}$ was low and the PM$_{10}$/PM$_{2.5}$ ratio was high (see a green rectangle in Fig. 2e). The same trend was also present in the time-series figure of the SCAQMD report (SCAQMD, 2017; SI Fig. S7),
although the report does not discuss this trend and its possible causes. Thus, we conclude that although PM sensors can be applied to high-resolution sensor networks, the data should be calibrated carefully before deployment and PM$_{10}$ is less reliable under conditions of low PM$_{2.5}$ and high PM$_{10}$/PM$_{2.5}$ ratios. We do not discuss this issue in more detail because it is beyond the scope of this study to elucidate the causes of this feature.

SCAQMD (2017) showed that commercially available PM sensors have a wide range of performance, and Morawska (2018) noted that pre-testing or calibration under realistic use conditions is necessary. Our results also suggest that the characteristics and performance of the selected sensor should be carefully determined and the data should be corrected properly before deployment.

- **Post-processing of sensor data and uncertainties**

Data post-processing was conducted separately for each season to avoid additional complexities, such as sensor aging and long-term zero drift. All data post-processing was conducted with MATLAB 2017b (MathWorks®) due to the massive dataset from 30 sensor nodes. For practical application in massive mobile or stationary sensor networks, only zero and scale corrections were used based on linear regression results for CO, O$_3$, and PM sensors. For NO$_2$ sensors, linear fitting did not produce satisfactory results (Table 4). Thus, an additional multivariate regression method was applied to improve the reliability and quality of sensor data. Because semi-conductor gas sensors (e.g., NO$_2$ sensor) are known to be influenced by temperature, humidity, and interfering gases (Barsan et al., 2007; Kleffman et al., 2013; Mead et al., 2013), we used temperature, humidity, and O$_3$ as explanatory variables (NO$_2$ concentrations could be grouped by O$_3$ level; data not shown). The linearity with AQMS NO$_2$ improved significantly after correction with multivariate regression method (equations are shown in SI Fig. S8), especially for summer data ($R^2 =$ 0.37 to 0.80 for summer and 0.77 to 0.82 for winter; SI Fig. S8). The data post-processing (for both
normalization to a reference and multivariate regression) were based on the entire observation data because these corrections were for diagnostic evaluation, not for prognostic purposes.

The time-series of post-adjusted air quality data from the sensors corresponded to those from the FRM/FEM instruments in both the magnitude and temporal variation (SI Fig. S9). The results of several statistical tests to evaluate the uncertainty of post-adjusted sensor data compared to AQMS data are also summarized in Table 5. The larger mean absolute error (MAE) of the CO sensor was caused by higher ambient CO concentrations (the mean relative error (MRE) of the CO sensor was about 10%, lower than those of other sensors). In contrast, O₃ and PM₂.₅ sensors had larger MRE values (25–30%), but their MAE levels were acceptable (O₃: 3–4 ppb; PM₂.₅: 2–5 μgꞏm⁻³).

- Application of post-processing to the other 29 sensor nodes data in a sensor network

During the periods for which one representative sensor node was co-located with reference instruments for intercomparison tests, the other 29 sensor nodes were deployed in a nearby urban area with a spatial domain size of 800 m × 800 m to investigate spatiotemporal heterogeneity of air pollutant distributions in complex urban environments (SI Fig. S10). To compare directly air pollutant concentrations read by each sensor node deployed in a sensor network, we applied a two-stage post-adjustment to the other 29 sensor nodes. At the first stage, we corrected the readings from 29 sensor nodes corresponding to the representative sensor node that was located with reference instruments based on the comparison results among sensors (Table 3; see the section of Intercomparison among sensors). At the second stage, because all 30 sensor nodes agreed well with each other (Table 3), we applied the post-processing of sensor data that was derived from the intercomparison results between the sensor node and reference instruments to the other 29 sensor nodes (corrected at the first stage) to adjust sensor readings corresponding to those of reference instruments.

Figure 3 shows the time-series of pollutants concentrations from the sensor node
after being corrected with the two-stage post-processing method mentioned above as well as the concentrations measured at nearby AQMS (37.565°N, 126.976°E). This AQMS is a different place from the site where the intercomparison tests were conducted between the sensor node and reference instruments. Thus, these AQMS data were independent of the post-processing of sensor readings. The sensor was located in the ground-level (about 3 m a.g.l.), and AQMS is located about 40 m from the sensor horizontally, at 15 m a.g.l. (SI Fig. S10). Considering the different positions of the sensor node and AQMS (horizontally and vertically), the time-series of the sensor node and reference instruments agreed well, demonstrating the potential of sensor measurements for air quality monitoring. The sensor node recorded high CO concentrations at mid-nights in the early period of monitoring in summer (tinted areas in Fig. 3a). At that time, construction equipment was operated beside the sensor.

The detailed results and discussion of the actual deployment of sensors in a network in complex urban micro-built environments will be discussed in the separate paper.

**Temporal variations in sensor characteristics and stabilization time**

Electrochemical gas sensors and semi-conductor sensors require time for stabilization prior to reading ambient pollution levels (Futata and Ogino, 1998; Burgues and Marco, 2018). For example, electrochemical sensors measure the current generated by reduction–oxidation (RedOx) reactions that occur on the electrode of the sensor and then convert these currents into pollutant concentrations. These RedOx reactions continuously occur, even when the sensor is not powered, and currents accumulate on the electrode. When the electrochemical sensor is powered, release of these currents to ambient levels takes time (Futata and Ogino, 1998). This means that when most cost-effective gas sensors (excluding non-dispersive infrared gas sensors; Dinh et al., 2016) are powered, we had to wait for the
sensors to stabilize before recording the real signals (SI Fig. S11). However, we found that when power was momentarily disconnected (< 1 min), all sensors stabilized within 1 min (data not shown).

As expected, the stabilization time for CO and NO₂ sensors were significant and increased with the period during which sensors were not operated (resting time) (Fig. 4). When the resting time was less than 5 days, the stabilization time increased sharply, up to 1.7 h for the CO sensor and ~3 h for the NO₂ sensor. With more than 5 days of resting time, the stabilization time did not increase significantly (up to ~3 h for the CO sensor and no change in the NO₂ sensor). However, when the resting time was longer than 3 months, the stabilization time again increased sharply, up to ~5 h for CO and >13 h for the NO₂ sensor. Of note, the stabilization time is a characteristic of the individual sensor, and each sensor may have a different stabilization time (SI Fig. S11b and c). Thus, we consider the stabilization times presented here to be lower limits.

Unlike the CO and NO₂ sensors, the unstabilized O₃ sensor gave zero values (SI Fig. S11a). The O₃ sensor also required time for stabilization, but not as much as the CO and NO₂ sensors. Regardless of resting time, the stabilization time of the O₃ sensor appeared to be no longer than 10 min (based on limited experiments, data not shown). The PM sensor is based on a light scattering technique and did not require any stabilization time.

Although we do not need to consider the stabilization time once the sensor network has been operated continuously for a long time, it is important to be aware of this characteristic for discontinuous monitoring of air pollutants (e.g., episodic monitoring of air pollutants by smartphone users) to avoid recording and sharing inaccurate air pollution data.

**CONCLUSION (POTENTIAL OF LOW-COST SENSORS FOR HIGHLY SPATIALLY RESOLVED AIR QUALITY MONITORING)**
The purposes for using low-cost air pollution sensors are diverse, from warning of gas leakage or occurrence of high pollution events to high-density air quality monitoring in cities. This study focused on assessing the potential of low-cost sensors for spatially dense air quality monitoring.

The sensors currently available on the market have great potential to be used in high-density air quality monitoring networks in large cities containing various built environments given that (1) the self-consistency among 30 sensors was reasonably good, with R² values larger than 0.93 for all tested pollutants; (2) the consistency between the sensors and FRM/FEM instruments was also reasonably good, with R² values larger than 0.87 except for the NO₂ sensor (for overall periods in Table 4); (3) the consistency among sensors and between the sensors and FRM/FEM instruments (except the NO₂ sensor) was maintained under two distinct meteorological conditions (hot and humid in summer and cold and dry in winter), which approximate the extremes experienced in most cities.

However, some issues must be appropriately addressed for the use of low-cost sensors for spatially dense air quality monitoring networks. First, the differences in unadjusted concentrations obtained by the sensors for direct comparisons were non-negligible: up to 17%, 28%, 40%, and 73% for O₃, PM₂.₅, CO, and NO₂, respectively. Thus, the sensor data must be corrected appropriately for use in investigating the qualitative spatial distributions of air pollutants at a fine scale.

Second, the unadjusted concentrations recorded by sensors differed significantly from those of FRM instruments (60~70% lower for CO, about 2.5-fold higher for PM), although O₃ sensors agreed quite well with the FRM instrument (within 20% for 15 – 70 ppb range). In addition, the NO₂ sensor required additional correction (e.g., the multivariate regression method used in this study). Thus, to obtain reliable air quality data from the sensor
networks, sensor data should be carefully corrected through data post-processing. Nonetheless, a simple two-step calibration process (i.e., linearity correction among sensors followed by scale correction between the sensors and FRM instruments) can produce reliable air quality data from a spatially dense sensor network.

Third, for some sensors (particularly the NO₂ sensor in this study), the effects of meteorological conditions and interfering materials may be significant. Thus, we must correct for these effects before using the sensor data. This correction could be carried out through multivariate regression analysis, and neural network statistical analysis was similarly applicable (Jianlin Hu at Nanjing University of Information Science and Technology, personal communication, February 2, 2018).

Fourth, stabilization time is needed for some sensors (i.e., the CO and NO₂ sensors in this study). This time requirement increased significantly after unpowered periods from a few hours to more than half a day, and the sensor produced unrealistic data during the warm-up period. Thus, warm-up time should be considered, especially for discontinuous monitoring efforts, such as mobile monitoring by citizen scientists with hand-held sensors.

Unfortunately, this study could not quantify the degradation of sensor performance caused by long-term operation. However, considering the relatively short lifetimes of low-cost sensors (particularly electrochemical and metal oxide sensors), we should quantify the degradation of sensor performance over time and correct for this effect. Cross et al. (2017) noted their short lifetime (24–36 months) as one of the problems with low-cost sensors, and Bai et al. (2018) suggested the performance of low-cost PM sensor gradually degraded for the 18-month operation period. Although some issues remain for fine-scale air quality monitoring with low-cost sensors, those issues could be properly addressed with data post-processing.

Thus, we conclude that a massive spatially dense air quality monitoring network using currently available low-cost sensors is feasible for short-term experiments (shorter than a few
months, which could be substantially extended for PM and O₃ based on the longer-term experimental results in literature (e.g., SCAQMD, 2017; Jiao et al., 2016). To obtain reliable data, the experiments must be designed properly. For long-term or discontinuous monitoring, sensor lifetime and the degradation of sensor performance over time should be quantified. In addition, more systematic and generally applicable algorithms must be developed to correct for meteorological effects, so that near-real-time air quality information can be provided at a fine scale. Statistical and/or machine-learning methods with large datasets might allow these issues to be addressed.

ACKNOWLEDGMENTS

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REFERENCES


MOIS (2019). At the end of 2018, the number of residents registered 51,226,000 people, increased by about 50,000 a year, Ministry of the Interior and Safety (South Korea), http://www.korea.kr/news/pressReleaseView.do?newsId=156313022, last access: 01/14 2019.


Table 1. Specifications of the sensors selected for this study.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Manufacturer / model</th>
<th>Specifications</th>
</tr>
</thead>
</table>
| CO        | Alphasense / CO-B4   | * Linearity test with ref. method: $R^2 = 0.88$, slope = 0.94 (5-min avg.)  
* Noise level: 4 ppb  
* Range: up to 1000 ppm  
* Operating conditions: $-30$–$50^\circ C$ / 15–90% RH  
* Method: electrochemical sensor  
* Reference: Cross et al. (2017) |
| NO$_2$    | SGX / MiCS-2714      | * Detection limit: 11.6 ppb  
* Linearity test with ref. method: $R^2 = 0.98$  
* Accuracy: 3 ppb  
* Operating conditions: $-30$–$85^\circ C$ / 5–95% RH  
* Method: metal oxide sensor  
| O$_3$     | Aeroqual / SM50      | * Linearity test with reference method: $R^2 = 0.91$–0.97  
* Accuracy: < 10 ppb  
* Resolution: 1 ppb  
* Operating conditions: $-20$–$50^\circ C$ / 5–95% RH  
* Method: electrochemical sensor  
* Reference: Jiao et al. (2016) |
| PM$_{2.5}$/PM$_{10}$ | Plantower / PMS5003 | * Linearity test with reference method: $R^2 = 0.92$ (1-h avg.)  
* Range of concentrations: 0–500 μg m$^{-3}$  
* Resolution: 1 μg m$^{-3}$  
* Operating conditions: $-10$–$60^\circ C$ / 0–99% RH  
* Counting efficiency: 50% at 0.3 μm and 98% at $\geq$ 0.5 in  
* Method: light scattering  
* Reference: Wang et al. (2017) |
| Temperature / Humidity | Sensirion / SHT21   | * Resolution: 0.01°C / 0.04%  
* Accuracy: $\pm 0.3^\circ C$ / $\pm 2\%$  
* Operating conditions: $-40$–$125^\circ C$ / 0–100%  
* Reference: manufacturer’s specification document |
| Data logger | Atmel / ATmega2560 | * Power: DCV 7–12, Max 300mA  
* Digital I/O Pins: 54 (of which 15 provide PWM output)  
* Analog Input Pins: 16bit  
* Clock Speed: 16 MHz  
* Micro SD memory: 16GB  
* RTC: Rechargeable Battery (5.8mAh), 30day |
Table 2. Temperature and relative humidity during the intercomparison experiments.

<table>
<thead>
<tr>
<th>Season</th>
<th>Avg. (std)</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Among sensors</td>
<td>With FRM instruments</td>
</tr>
<tr>
<td></td>
<td>Temp. (℃)</td>
<td>Humid. (%)</td>
</tr>
<tr>
<td>Summer</td>
<td>29.9 (4.2)</td>
<td>60.7 (10.8)</td>
</tr>
<tr>
<td>(Aug. and Sep.)</td>
<td>26.1 (3.3)</td>
<td>53.8 (15.7)</td>
</tr>
<tr>
<td>Winter</td>
<td>4.1 (5.2)</td>
<td>24.7 (10.1)</td>
</tr>
<tr>
<td>(Dec. and Jan.)</td>
<td>2.5 (5.5)</td>
<td>47.3 (12.6)</td>
</tr>
</tbody>
</table>
**Table 3.** Results of linear regression between 29 sensors and the reference sensor in summer and winter (using 15-minute averaged unadjusted data). The reference sensor represents the sensor node that was co-located with FRM/FEM instruments at AQMS.

<table>
<thead>
<tr>
<th></th>
<th>CO</th>
<th>NO2</th>
<th>O3</th>
<th>PM2.5</th>
<th>PM10</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Summer</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Aug. and Sep.)</td>
<td>Mean (std)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CO</td>
<td>0.87 (0.08)</td>
<td>1.03 (0.08)</td>
<td>1.00 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.93 (0.04)</td>
</tr>
<tr>
<td>NO2</td>
<td>1.03 (0.08)</td>
<td>1.03 (0.08)</td>
<td>1.00 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.93 (0.04)</td>
</tr>
<tr>
<td>O3</td>
<td>1.00 (0.04)</td>
<td>1.00 (0.04)</td>
<td>1.00 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.93 (0.04)</td>
</tr>
<tr>
<td>PM2.5</td>
<td>0.94 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.94 (0.04)</td>
<td>0.93 (0.04)</td>
</tr>
<tr>
<td>PM10</td>
<td>0.93 (0.04)</td>
<td>0.93 (0.04)</td>
<td>0.93 (0.04)</td>
<td>0.93 (0.04)</td>
<td>0.93 (0.04)</td>
</tr>
<tr>
<td><strong>Range</strong></td>
<td>0.74–1.13</td>
<td>0.88–1.23</td>
<td>0.96–1.06</td>
<td>0.87–1.01</td>
<td>0.95–1.06</td>
</tr>
<tr>
<td><strong>R²</strong></td>
<td>Mean (std)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CO</td>
<td>0.94 (0.055)</td>
<td>0.982 (0.012)</td>
<td>0.997 (0.001)</td>
<td>0.945 (0.012)</td>
<td>0.933 (0.012)</td>
</tr>
<tr>
<td>NO2</td>
<td>0.982 (0.012)</td>
<td>0.982 (0.012)</td>
<td>0.997 (0.001)</td>
<td>0.945 (0.012)</td>
<td>0.933 (0.012)</td>
</tr>
<tr>
<td>O3</td>
<td>0.997 (0.001)</td>
<td>0.997 (0.001)</td>
<td>0.997 (0.001)</td>
<td>0.945 (0.012)</td>
<td>0.933 (0.012)</td>
</tr>
<tr>
<td>PM2.5</td>
<td>0.945 (0.012)</td>
<td>0.945 (0.012)</td>
<td>0.945 (0.012)</td>
<td>0.945 (0.012)</td>
<td>0.933 (0.012)</td>
</tr>
<tr>
<td>PM10</td>
<td>0.933 (0.012)</td>
<td>0.933 (0.012)</td>
<td>0.933 (0.012)</td>
<td>0.933 (0.012)</td>
<td>0.933 (0.012)</td>
</tr>
<tr>
<td><strong>Range</strong></td>
<td>0.739–0.979</td>
<td>0.953–0.997</td>
<td>0.994–0.998</td>
<td>0.911–0.967</td>
<td>0.905–0.951</td>
</tr>
<tr>
<td><strong># of data (N)</strong></td>
<td>312</td>
<td>284</td>
<td>325</td>
<td>42</td>
<td>342</td>
</tr>
</tbody>
</table>

| **Winter** |        |        |        |        |        |
| (Dec. and Jan.) | Mean (std) | | | | |
| CO         | 0.90 (0.06) | 1.17 (0.30) | 0.96 (0.06) | 1.03 (0.06) | 1.01 (0.07) |
| NO2        | 1.17 (0.30) | 1.17 (0.30) | 0.96 (0.06) | 1.03 (0.06) | 1.01 (0.07) |
| O3         | 0.96 (0.06) | 0.96 (0.06) | 0.96 (0.06) | 1.03 (0.06) | 1.01 (0.07) |
| PM2.5      | 1.03 (0.06) | 1.03 (0.06) | 1.03 (0.06) | 1.03 (0.06) | 1.01 (0.07) |
| PM10       | 1.01 (0.07) | 1.01 (0.07) | 1.01 (0.07) | 1.01 (0.07) | 1.01 (0.07) |
| **Range**  | 0.79–1.07 | 0.83–1.66 | 0.84–1.05 | 0.84–1.12 | 0.80–1.12 |
| **R²**     | Mean (std) | | | | |
| CO         | 0.972 (0.039) | 0.965 (0.020) | 0.975 (0.004) | 0.991 (0.008) | 0.988 (0.006) |
| NO2        | 0.965 (0.020) | 0.965 (0.020) | 0.975 (0.004) | 0.991 (0.008) | 0.988 (0.006) |
| O3         | 0.975 (0.004) | 0.975 (0.004) | 0.975 (0.004) | 0.991 (0.008) | 0.988 (0.006) |
| PM2.5      | 0.991 (0.008) | 0.991 (0.008) | 0.991 (0.008) | 0.991 (0.008) | 0.988 (0.006) |
| PM10       | 0.988 (0.006) | 0.988 (0.006) | 0.988 (0.006) | 0.988 (0.006) | 0.988 (0.006) |
| **Range**  | 0.960–0.988 | 0.93–0.994 | 0.970–0.981 | 0.955–0.995 | 0.961–0.993 |
| **# of data (N)** | 484 | 377 | 463 | 574 | 574 |
Table 4. Summary of the intercomparison results between sensors and FRM instruments (using 1-hour averaged unadjusted sensor data). Because the NO$_2$ readings were raw signals from the sensor (not factory-calibrated signals), the slope and y-intercept are not very meaningful.

<table>
<thead>
<tr>
<th></th>
<th>CO</th>
<th>NO$_2$</th>
<th>O$_3$</th>
<th>PM$_{2.5}$</th>
<th>PM$_{10}$</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Summer</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Aug. and Sep.)</td>
<td>Slope</td>
<td>1.03</td>
<td>-</td>
<td>0.88</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td>y-intercept</td>
<td>224.04</td>
<td>-</td>
<td>7.04</td>
<td>3.59</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.814</td>
<td>0.407</td>
<td>0.894</td>
<td>0.738</td>
</tr>
<tr>
<td></td>
<td># of data</td>
<td>141</td>
<td>135</td>
<td>147</td>
<td>145</td>
</tr>
<tr>
<td><strong>Winter</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Dec. and Jan.)</td>
<td>Slope</td>
<td>1.63</td>
<td>-</td>
<td>0.83</td>
<td>0.40</td>
</tr>
<tr>
<td></td>
<td>y-intercept</td>
<td>219.73</td>
<td>-</td>
<td>8.26</td>
<td>4.98</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.930</td>
<td>0.785</td>
<td>0.944</td>
<td>0.880</td>
</tr>
<tr>
<td></td>
<td># of data</td>
<td>170</td>
<td>134</td>
<td>175</td>
<td>165</td>
</tr>
<tr>
<td><strong>Overall</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Summer &amp; Winter)</td>
<td>Slope</td>
<td>1.7286</td>
<td>-</td>
<td>0.86</td>
<td>0.42</td>
</tr>
<tr>
<td></td>
<td>y-intercept</td>
<td>-</td>
<td>7.79</td>
<td>2.27</td>
<td>14.99</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.925</td>
<td>0.005</td>
<td>0.877</td>
<td>0.955</td>
</tr>
<tr>
<td></td>
<td># of data</td>
<td>311</td>
<td>269</td>
<td>322</td>
<td>310</td>
</tr>
</tbody>
</table>
Table 5. Summary of several statistical tests for uncertainty in corrected (post-adjusted) sensor data (numbers of data used in the tests are identical with those in Table 4).

<table>
<thead>
<tr>
<th>Substance</th>
<th>Bias(^1)</th>
<th>RMSE(^2)</th>
<th>IOA(^3)</th>
<th>MAE(^4)</th>
<th>MRE(^5)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CO</td>
<td>-1.4</td>
<td>67.8</td>
<td>0.92</td>
<td>51.8</td>
<td>12.2</td>
</tr>
<tr>
<td>NO(_2)</td>
<td>0.04</td>
<td>10.0</td>
<td>0.61</td>
<td>8.0</td>
<td>34.5</td>
</tr>
<tr>
<td>O(_3)</td>
<td>-1.64</td>
<td>7.7</td>
<td>0.93</td>
<td>4.3</td>
<td>28.9</td>
</tr>
<tr>
<td>PM(_{2.5})</td>
<td>0.07</td>
<td>2.7</td>
<td>0.90</td>
<td>2.2</td>
<td>30.2</td>
</tr>
<tr>
<td>PM(_{10})</td>
<td>0.23</td>
<td>14.3</td>
<td>0.46</td>
<td>4.2</td>
<td>20.8</td>
</tr>
<tr>
<td>Winter</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CO</td>
<td>-111.5</td>
<td>142.0</td>
<td>0.95</td>
<td>116.8</td>
<td>0.1</td>
</tr>
<tr>
<td>NO(_2)</td>
<td>-0.75</td>
<td>8.6</td>
<td>0.94</td>
<td>6.8</td>
<td>34.1</td>
</tr>
<tr>
<td>O(_3)</td>
<td>-2.63</td>
<td>4.2</td>
<td>0.96</td>
<td>3.2</td>
<td>24.3</td>
</tr>
<tr>
<td>PM(_{2.5})</td>
<td>0.50</td>
<td>7.2</td>
<td>0.98</td>
<td>4.5</td>
<td>26.6</td>
</tr>
<tr>
<td>PM(_{10})</td>
<td>-0.92</td>
<td>11.6</td>
<td>0.97</td>
<td>8.8</td>
<td>17.3</td>
</tr>
</tbody>
</table>

1. Bias (mean error): \(\frac{1}{N} \sum (M_i - O_i)\)
2. RMSE (Root mean square error): \(\sqrt{\frac{\sum (M_i - O_i)^2}{N}}\)
3. IOA (Index of agreement): \(1 - \frac{\sum |M_i - O_i|}{\sqrt{\sum (|M_i - M| + |M_i - O|)^2}}\)
4. MAE (Mean absolute error): \(\frac{1}{N} \sum |M_i - O_i|\)
5. MRE (Mean relative error): \(\frac{1}{N} \sum \frac{|M_i - O_i|}{|O_i|} \times 100\) (%)
Figure Captions

**Fig. 1.** Image of the sensor assembly. The space at the bottom holds the battery.

**Fig. 2.** Results of the comparison between AQMS and unadjusted sensor data for summer and winter (hourly averaged data): (a) CO, (b) NO₂, (c) O₃, (d) PM₂.₅, and (e) PM₁₀. Red stars represent summer data and blue circles indicate winter data. The black dotted lines denote the linear fit for winter, black dashed lines those for summer, and green solid lines those for both seasons combined.

**Fig. 3.** Time-series of adjusted air quality data obtained from sensors (black line) and data from AQMS (red dashed line with squares) for the summer (left panel) and winter periods (right panel) (1-hr averaged). The sensor node is not the one co-located with FRM/FEM instruments, but one of the sensor nodes in a network (SI Fig. S10). The AQMS in this figure is the nearby station from this sensor node location, not the one where intercomparison tests were conducted.

**Fig. 4.** Stabilization time for the (a) CO and (b) NO₂ sensors. Resting time on the x-axis refers to the period for which the sensors were unpowered.
Fig. 1.
Fig. 2.
Fig. 3.
Fig. 4.