Impact of Varying Wind Patterns on PM₁₀ Concentrations in the Seoul Metropolitan Area in South Korea from 2012 to 2016

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ABSTRACT

In this study, we classify wind patterns that impacted PM₁₀ concentrations in the Seoul Metropolitan Area (SMA), South Korea, from 2012 to 2016 and analyze their contributions to annual variability in particulate matter smaller than 10 μm in diameter (PM₁₀). Using a *k*-means clustering analysis, we identify major wind patterns affecting PM₁₀ concentrations from 2002 to 2016. We confirm that the impact of wind pattern changes on PM₁₀ variability in the SMA from 2012 to 2016 was relatively greater than the impact from 2002 to 2011. We find that PM₁₀ from 2012 to 2016 was mainly affected by wind patterns that were 1) associated with the transport of foreign emissions (our clusters H2, H4, and H5) and 2) favorable for ventilation (our clusters L1 and L2). This finding shows that PM₁₀ variability was determined by overall variations in the respective wind patterns particularly associated with high (over 80 $\mu\text{g m}^{-3}$) and low (below 30 $\mu\text{g m}^{-3}$) PM₁₀ concentrations. The results from 2012 to 2016 CMAQ simulations indicate that the effects of meteorological conditions (e.g., wind, temperature, humidity, and so on) on PM₁₀ vary from year to year. The calculated PM₁₀ anomalies from 2012 to 2016 were -4.97 , 3.55 , 1.73 , 0.15 , and $-0.46 \mu\text{g m}^{-3}$, suggesting that the wind patterns in 2012 produced the least PM₁₀ and those in 2013 produced the most.

1. Introduction

High levels of particulate matter (PM) is one of the most serious social problems in South Korea. Studies have found an adverse effect of PM on human health (Lee and Lee 2014; Wyzga and Rohr 2015; Khan and Strand 2018), and several more recent studies have reported that high PM₁₀ could be associated with an increase in the number of suicides (Kim et al. 2015; Kim et al. 2018; Lin et al. 2016).

The South Korean government has devoted considerable effort to improving air quality over South Korea (Kim and Shon 2011) and through legislation enacted in 2005 (Ministry of Environment 2005), which prompted a downward trend in the number of domestic emissions. In response to this trend, PM₁₀ concentrations over South Korea decreased gradually (Sharma et al. 2014; Yoo et al. 2015). Nevertheless, we have witnessed a reversal of PM₁₀ trend in the 2010s (Ahmed et al. 2015; Yang et al. 2016). Kim et al. (2017a) found that since 2012, PM₁₀ concentrations over the Seoul Metropolitan Area (SMA) of South Korea have been trending upward, the result of reduced wind speed, leading to unfavorable atmospheric conditions for the dispersion of air pollutants. This finding suggests that meteorological conditions (e.g., changes in

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wind patterns) has had a greater impact on PM₁₀ changes in South Korea during the 2010s than it had during the early 2000s. Thus, an understanding of changes in wind patterns on interannual PM₁₀ variability in South Korea could help the government determine effective policy for improving air quality in recent days.

A number of studies have examined the relationship between PM concentrations and wind patterns in South Korea (Kim et al. 2006; S. Lee et al. 2011; Kim et al. 2017a; Lee and Kim 2018). While most studies have focused on classifying and investigating predominant wind patterns associated with highly polluted air, none has examined long-term variability in classified wind patterns affecting PM₁₀ concentrations in South Korea so far. Since predominant wind patterns impacting PM₁₀ can vary by year, knowledge about their interannual variability would lead to a more comprehensive understanding of the trends of PM₁₀ concentrations that cannot be solely explained by changes in emissions in South Korea.

This study investigates the impact of varying wind patterns on PM₁₀ variability in South Korea. We classify predominant wind patterns impacting PM₁₀ concentrations in the SMA in the 15 years (2002–16) and analyze their annual variability. We will begin by comparing emissions and observed PM₁₀ concentrations in the SMA to investigate their relationship and annual variation, then conduct wind pattern clustering to classify predominant wind patterns impacting annual PM₁₀ changes in the SMA, and finally compare their relative contributions. Using the Community Multiscale Air Quality Modeling System (CMAQ, version 5.0.2) (Byun and Schere 2006), we will also simulate a multiyear pattern of PM₁₀ to quantify the impact of meteorological conditions on PM₁₀ in the SMA in each year.

2. Method

a. Clustering of wind patterns

In this study, we conducted a cluster analysis to classify wind patterns impacting PM₁₀ concentrations in the SMA of South Korea. To properly classify the respective clusters associated with variable wind patterns, we implemented a *k*-means cluster analysis (Darby 2005), a commonly used and reliable non-hierarchical method of classifying clusters based on the distance between data (Souri et al. 2016). Before conducting the clustering analysis, we determined the number of clusters (*k*) to avoid over or under-interpretation of the classified clusters. A large *k* might yield similar patterns among different clusters, but a small *k* would not sufficiently capture the diversity of

clustered products (Carro-Calvo et al. 2017). For a proper determination of *k*, we used a methodology suggested by Jin et al. (2011) that calculates the within-group sum of squares (WSS) as follows:

$$\text{WSS} = \sum_1 (x_{i1} - \bar{x}_1)^2 + \sum_2 (x_{i2} - \bar{x}_2)^2 + \cdots + \sum_k (x_{ik} - \bar{x}_k)^2.$$

WSS is the sum of squared distances between cluster centroids ($\bar{x}_k$) and corresponding data (x_{ik}). As *k* increases, the WSS decreases and converges to a particular constant value, indicating that the increase in *k* is no longer effective for analysis. We calculated the WSS values for each *k* and determined the optimal *k* to be the point at which the WSS converges to a particular constant value. After the optimal *k* is determined, each datum was assigned to the nearest cluster centroid, which is updated as the mean of points assigned to the cluster. This clustering process underwent 1000 iterations until it produces an optimal cluster allocation for this study. We set the domain for the clustering to cover northeastern Asia, the Korean Peninsula, eastern China, and Japan (Fig. 1). For the cluster analysis, we used Final (FNL) Operational Global Analysis data (1° by 1°) from the National Centers for Environmental Prediction. We obtained 15-yr (2002–16) gridded 900 hPa horizontal wind speed (*U* and *V*) over the domain (31 × 41 grid points) from the FNL dataset and used them for the *k*-means clustering to classify wind patterns affecting PM₁₀ concentrations in the SMA.

b. Surface measurements

We used surface observational data from the air quality monitoring station (AQMS) network operated by the National Institute of Environmental Research of South Korea. The network measures the concentrations of real-time air pollutants and provides hourly concentrations for CO, NO₂, O₃, PM_{2.5} (only available since 2015), PM₁₀, and SO₂. We gathered measured PM₁₀ concentration data from 2002 to 2016 at 86 AQMS sites in the SMA (Fig. 1) to analyze the variabilities in PM₁₀.

c. The modeling system

We used the CMAQ model to quantify the impact of meteorological conditions on PM₁₀ simulations and configured the modeling domain with a grid resolution of 36 km (96 × 86) covering northeastern Asia (Fig. 1). We acquired emission data from the mosaic Asian anthropogenic emission (MIX) inventory of 2010 (Li et al. 2017) and meteorological data for the CMAQ simulation from a Weather Research and Forecasting (WRF) Model (Skamarock et al. 2008).

FIG. 1. (left) The domain used for wind pattern clustering and CMAQ modeling in this study. (right) The location of the SMA and the distributions of the 86 AQMS sites used for the k -means clustering.

simulation, the initial and boundary conditions of which came from the $1^\circ \times 1^\circ$ FNL data. The detailed physical and chemical options for the WRF and CMAQ simulations were the same as those in Jeon et al. (2018). To quantify the impact of meteorological conditions on annual PM_{10} simulations, we used a fixed emissions dataset for the CMAQ runs for 2012–16.

3. Results and discussion

a. PM_{10} variability in the SMA in 2002–16

Figure 2a shows annual PM_{10} concentrations at 86 AQMS sites in the SMA during the 15 years (2002–16). PM_{10} concentrations exhibited a downward trend ($-2.123 \mu\text{g m}^{-3} \text{ yr}^{-1}$; $p = 0.000$) from 2002 to 2011, sharply declined to their lowest level ($46.31 \mu\text{g m}^{-3}$) in 2012, began to increase in 2013, and then varied marginally after 2013 ($+1.006 \mu\text{g m}^{-3} \text{ yr}^{-1}$; $p = 0.124$), consistent with the results in Kim et al. (2017a).

To investigate the major contributors affecting interannual PM_{10} variability in the SMA, we first compared annual mean concentrations of PM_{10} with emissions of PM_{10} and its precursors (i.e., NO_x and SO_x) from 2002 to 2016. According to Son et al. (2012), organic carbon (OC), nitrate, and sulfate were the major constituents of PM_{10} in the SMA. Although all of the emissions of these constituents should have been used for the comparison, we could not obtain OC emissions because the Clean Air Policy Support System inventory

does not provide it. As a result, we excluded OC from the analysis. As seen in Fig. 2, the annual variation of mean PM_{10} concentrations and PM_{10} emissions in the SMA indicated close agreement from 2002 to 2011 with a correlation coefficient R of 0.88 ($p = 0.001$). During this period, the annual variability in NO_x and SO_x emissions, with R values of 0.83 ($p = 0.000$) and 0.90 ($p = 0.003$), respectively (Table 1), closely agreed with that in SMA PM_{10} . Since 2012, however, PM_{10} and NO_x and SO_x emissions showed large discrepancies in their variability. In particular, SO_x emissions and PM_{10} concentrations indicated a negative correlation with an R value of -0.39 ($p = 0.611$) in 2012–16. These results, however, were not statistically significant.

Despite no consideration of OC emissions in this work, the results in Fig. 2 can demonstrate that the annual PM_{10} variation in the SMA in 2002–11 could be reasonably explained by annual changes in local emissions but maybe not those in 2012–16. Moreover, the amount of Chinese emissions, which could have strongly affected the air quality in South Korea (Kim et al. 2017b), continued to decline after 2012 (Zheng et al. 2018), supporting the assumption that the changes that occurred in PM_{10} in the SMA after 2012 were more strongly affected by meteorological conditions than those in 2002–11.

For a more detailed analysis, we divided the original PM_{10} data into three groups according to concentration

FIG. 2. Annual variations in (top) PM_{10} concentrations (average of 86 sites) and (bottom) the amount of PM_{10} emissions (aggregation of all districts) in the SMA from 2002 to 2016. Emissions data for 2016 are not shown because they have yet to be released.

levels as reported by previous studies (K. J. Lee et al. 2011; Hur et al. 2016). Group 1 included days of high PM_{10} concentrations exceeding the South Korean standard for “bad” air quality ($\text{PM}_{10} > 80 \mu\text{g m}^{-3}$; high- PM_{10} days); group 2 included days of low PM_{10} concentrations below the South Korean standard for “good” air quality ($\text{PM}_{10} \leq 30 \mu\text{g m}^{-3}$; low- PM_{10} days); group 3 included days of “moderate” air quality ($30 \mu\text{g m}^{-3} < \text{PM}_{10} \leq 80 \mu\text{g m}^{-3}$; moderate- PM_{10} days). To determine whether the three groups indicated

comparable PM_{10} variabilities in 2002–16, we counted the frequencies of each group separately.

As shown in Fig. 3, the number of high- PM_{10} days in 2002–11 indicated a downward trend (-6.6 days yr^{-1}), but that of low- PM_{10} days showed the opposite ($+5.7$ days yr^{-1}). This finding indicates that the decline in PM_{10} concentrations in the SMA in 2002–11 (Fig. 2) could have resulted from the decrease of high- PM_{10} days and the increase of low- PM_{10} days during the period. One explanation for the variation of high- and low- PM_{10}

TABLE 1. Correlation coefficients R of PM_{10} concentrations and the amount of emissions (PM_{10} , NO_x , or SO_x) in the SMA. The values for two time periods (2002–11 and 2012–16) were separately calculated. Asterisks indicate that the value is statistically insignificant ($p > 0.05$).

	PM_{10}	NO_x	SO_x
R for 2002–11	0.88 ($p = 0.001$)	0.83 ($p = 0.003$)	0.90 ($p = 0.000$)
R for 2012–16	0.57* ($p = 0.434$)	0.33* ($p = 0.673$)	−0.39* ($p = 0.611$)

FIG. 3. Annual variability of the counts of high (over $80 \mu\text{g m}^{-3}$), low (below $30 \mu\text{g m}^{-3}$), and moderate (between 30 and $80 \mu\text{g m}^{-3}$) PM_{10} days during the 2002–16 period.

days in 2002–11 was the gradual decline in PM_{10} emissions during these years (Fig. 2). While the number of high- PM_{10} days in 2002–11 was highly correlated with the amount of PM_{10} emissions ($R = 0.97$; $p = 0.000$), the number of low- PM_{10} days was inversely correlated ($R = -0.85$; $p = 0.002$). The number of moderate- PM_{10} days did not indicate a clear trend in 2002–11. These results demonstrate that the annual variability of high and low PM_{10} concentrations in the SMA in 2002–11 could be explained well by changes in emissions.

In contrast to the 2002–11 results, the 2012–14 results could not attribute the variability in PM_{10} concentrations in the SMA to emissions. Although emissions in the SMA did not noticeably increase (Fig. 2) and foreign emissions even decreased during the years (Zheng et al. 2018), the number of high- PM_{10} days increased from 2012 to 2014. Despite this increase, the PM_{10} concentration in the SMA exhibited a slight decrease in 2015 and the number of low- PM_{10} days indicated a declining trend after 2012. These findings reveal the possibility of another influence on PM_{10} variation in the SMA from 2012 to 2014, different from the case from 2002 to 2011.

From the above results, one could assume that the PM_{10} variability in the SMA in 2012–16 was affected by annual changes in meteorological conditions, leading to considerable air pollution. However, as a number of meteorological factors (e.g., wind, temperature, humidity, and so on) affect PM_{10} concentrations, examining their individual contributions to PM_{10} variability in the SMA is not easy. Thus, this study focused only on the impact of wind pattern changes on PM_{10} variability, referring to previous studies (e.g., Kim et al. 2017a). We intensively analyzed annual variations of

wind patterns associated with high, low, and moderate PM_{10} concentrations in the SMA to verify their impact on PM_{10} variation.

b. Cluster analysis

We used k -means clustering to classify wind patterns impacting PM_{10} concentrations in the SMA during the 15 years (2002–16). The number of clusters was determined to be five (i.e., the calculated optimized $k = 5$) on the basis of the method by Jin et al. (2011). We conducted k -means clustering for group 1 (high- PM_{10} days), group 2 (moderate- PM_{10} days), and group 3 (low- PM_{10} days) separately, and analyzed the respective results.

1) WIND PATTERNS AFFECTING HIGH PM_{10}

Figure 4 displays the five wind pattern clusters (clusters H1–H5; hereinafter C_H1–C_H5) affecting high PM_{10} concentrations (over $80 \mu\text{g m}^{-3}$) in the SMA in 2002–16. Wind patterns C_H1 (19.2%) and C_H3 (15.4%) consisted of days with weak synoptic winds during the spring/early summer (C_H1) and the spring/early winter (C_H3), which were favorable for air pollution. C_H2 (21.2%) consisted of days with prevailing northwesterly winds during the spring, which coincided with the transport of Asian dust from source regions (e.g., the Gobi Desert and Inner Mongolia) to South Korea. C_H4 (19.3%) consisted of days with prevailing westerly winds during the winter/early spring influenced by the transport of foreign emissions from the Shandong Peninsula and northeastern China. C_H5 (24.9%) included days with weak northwesterly winds during the late autumn/winter/early spring influenced by the transport of foreign emissions from northeastern China and North Korea. Among all of the clusters, the

FIG. 4. Five wind patterns (clusters) associated with high PM_{10} (over $80 \mu\text{g m}^{-3}$) in the SMA during the 2002–16 period, classified by k -means clustering. The percentage values denote the fractions of each wind pattern to the total number of high- PM_{10} days in 2002–16 [i.e., $[\text{Number of days of each wind pattern (C_H1–C_H5)} \text{ in } 2002\text{--}16 / \text{Total number of high-PM}_{10} \text{ days in } 2002\text{--}16] \times 100$].

frequency of C_H5 (233 days) was the highest, followed by C_H2 (198 days), C_H4 (181 days), C_H1 (180 days), and C_H3 (144 days). Their relative contributions to PM_{10} concentrations in the SMA are analyzed in detail in section 3b(4).

Figure 5 shows that as the total number of high- PM_{10} days continued to decrease in response to the decreasing emissions from 2002 to 2011 (Fig. 2), the counts of each cluster indicated declining trends. In 2012–16, however, the counts of the respective clusters indicated annual variations that differed from those in 2002–11. C_H1 ($+0.500 \text{ days yr}^{-1}$), C_H2 ($+0.800 \text{ days yr}^{-1}$), and C_H3 ($+0.500 \text{ days yr}^{-1}$) indicated upward trends while C_H4 ($-0.600 \text{ days yr}^{-1}$) and C_H5 ($-1.900 \text{ days yr}^{-1}$) showed the opposite, canceling out the upward trends. The annual variations of C_H2 and C_H3, which are the wind patterns associated with Asian dust and air stagnation, in 2012–16 closely agreed with that of high counts of PM_{10} . This finding suggests that PM_{10} concentrations in the five years from 2012 to 2016 may have been related to variations in C_H2 and C_H3. In particular, the lowest number of Asian dust events in the SMA in 2012 (Fig. 6) (Korea Meteorological Administration 2015) was likely the primary reason for the minimum PM_{10} concentration in 2012. The increase in PM_{10} in 2013 and 2014 could also be

partially attributable to the increased occurrence of Asian dust.

The frequency of Asian dust transport, however, was not the only factor determining high PM_{10} in the SMA. For instance, the PM_{10} concentration in 2015 was slightly lower than it was in 2014 despite even higher numbers of Asian dust events and PM_{10} emissions in 2015 than in 2014. The low PM_{10} concentration in 2015 was due to the significantly decreased count of C_H5 in 2015, which was 66.4% lower than that in 2014. In addition, the noticeable increase in PM_{10} in 2013 over that in 2012 (Fig. 2) could have been affected by the significant increase in the count of C_H4 in 2013, which was 320.0% higher than that in 2012.

The above results demonstrate that PM_{10} concentrations in the SMA from 2012 to 2016 were not solely affected by a single wind pattern. They were exacerbated by the annual counts of wind patterns associated with high PM_{10} , particularly C_H2, C_H4, and C_H5. The PM_{10} in 2016, however, could not be reasonably explained by their variability, which will be further discussed in section 3b(3). Although those results identified the major wind patterns affecting PM_{10} variability in the SMA, the potential causes of the annual variabilities of each wind pattern are unclear. Therefore, a follow-up study could analyze the impact

FIG. 5. Annual counts (i.e., number of days in each year) of the five wind patterns (clusters) associated with high PM₁₀ (over 80 $\mu\text{g m}^{-3}$) in the SMA during the 2002–16 period. Dashed lines denote the trends of each cluster during the two periods of 2002–11 and 2012–16.

of meteorological conditions on interannual PM₁₀ variability in the SMA.

2) WIND PATTERNS AFFECTING LOW PM₁₀

To further explain annual PM₁₀ variability in the SMA in 2012–16, we also classified wind patterns impacting low PM₁₀ concentrations (below 30 $\mu\text{g m}^{-3}$) (Fig. 7). C_L1 (26.7%) consisted of days with the strong southwesterly winds during the summer caused by low pressure centered northwest of the Korean Peninsula. C_L2 (26.3%) included days with prevailing easterly winds during the summer/autumn caused by high pressure centered northeast of the Korean Peninsula. C_L3 (17.1%) comprised days with frequent precipitation caused by seasonal cold fronts located in southern South Korea. C_L4 (16.4%) consisted of days with strong northeasterly winds during the late summer/autumn

influenced by typhoons south/southeast of South Korea. C_L5 (13.5%) were days with strong northwesterly winds during the autumn/early winter.

C_L1, C_L2, and C_L5 wind patterns favorable for ventilation that can lead to low PM₁₀ accounted for 66.5% of the total number of low-PM₁₀ days. As presented in Fig. 8, their annual counts indicated an upward trend in 2002–11 and a downward trend in 2012–16, which was inversely proportional to the trend in PM₁₀ emissions in the SMA (Fig. 2). Declining counts of C_L1, C_L2, and C_L5 in 2012–16 indicate that the weather conditions around South Korea became unfavorable for the ventilation and dispersion of air pollutants during these years, consistent with findings by Kim et al. (2017a). Thus, the increase in PM₁₀ from 2012 to 2014 might be attributable to the decreased counts of C_L1, C_L2, and C_L5 in 2012–14, which

FIG. 6. Annual variation of Asian dust counts during the 2002–16 period observed at four sites (Seoul, Incheon, Suwon, and Baekryongdo) in the SMA.

FIG. 7. As in Fig. 4, but for low-PM₁₀ days.

corresponded with the increased count of C_H2 during these years. The counts of C_L1, C_L2, and C_L5 continued to decrease in 2015–16, but they did not lead to a noticeable increase in PM₁₀ concentrations. Their contributions might have been mitigated by a low count of

C_H5 and decreased amounts of foreign (i.e., Chinese) emissions in 2015–16 (Zheng et al. 2018).

It is worthwhile to note that the wind pattern associated with rainfall (C_L3) in the summer did not exhibit a clear trend in 2012–16 (≈ 0.000 days yr⁻¹), demonstrating that

FIG. 8. As in Fig. 5, but for low-PM₁₀ days.

annual PM₁₀ variability in the SMA was only slightly affected by changes in the frequency of precipitation, particularly during the summer. The wind pattern associated with an oncoming typhoon near South Korea (C_L4) in 2012–16 indicated a clear upward trend (+1.500 days yr⁻¹) but marginally impacted PM₁₀ variation in the SMA during the years because C_L4 did not constitute a large portion of the total number of low-PM₁₀ days.

In summary, variations in wind patterns C_L1 and C_L2 mainly contributed to low PM₁₀ concentrations (below 30 $\mu\text{g m}^{-3}$) in the SMA. Their annual counts gradually decreased after 2012, demonstrating that the weather conditions during the 5-yr period (2012–16) were more favorable to air stagnation.

3) WIND PATTERNS AFFECTING MODERATE PM₁₀

The wind patterns associated with moderate-PM₁₀ days (between 30 and 80 $\mu\text{g m}^{-3}$) in the SMA (Fig. S1 in the online supplemental material), as compared with high and low-PM₁₀ days, did not indicate noticeable annual variability (Fig. S2 in the online supplemental material). C_M4 (i.e., weak clockwise synoptic wind under high pressure over the Korean Peninsula in the autumn/early winter) was the predominant wind pattern impacting moderate PM₁₀ concentrations in the SMA in 2002–16. Its annual counts peaked in 2016, implying that weather conditions in 2016 were relatively more stagnant than other years and favorable to the accumulation of air pollutants. As described in the previous section, the annual counts of C_L1, C_L2, and C_L5 in the 2012–16 period were the lowest in 2016. These results demonstrate that the wind patterns associated with moderate and low-PM₁₀ days in 2016 were noticeably favorable to stagnant air, but PM₁₀ concentrations did not show a dramatic increase in 2016 because during this year, the counts of wind patterns associated with high PM₁₀ concentrations (i.e., C_H1–C_H5) were not high and the number of Chinese emissions was at their lowest level. However, because the number of domestic PM₁₀ emissions in 2016 is as yet unknown, identifying the major driver of PM₁₀ concentrations in 2016 is challenging.

Overall, we were unable to attribute annual PM₁₀ variability in the SMA in 2012–16 solely to changes in emissions, but we show the contributions of annual variability in wind patterns associated with high (C_H2, C_H4, and C_H5), low (C_L1, C_L2), and moderate (C_M4) PM₁₀ concentrations. We wish to note, however, that the cluster analysis in this study was performed for 2002–16. Thus, the major wind patterns that affected PM₁₀ variability in the SMA from 2012 to 2016 could have differed from those in this study. Thus, we could enhance our understanding of the impact of wind pattern changes on more recent PM₁₀

TABLE 2. The calculated contributions (%) of each wind pattern to annual mean PM₁₀ concentrations in the SMA in 2012–16. The wind patterns that affected PM₁₀ the most in 2012–16 are boldfaced.

	2012	2013	2014	2015	2016	2012–16
C_H1	2.05	1.81	2.11	2.45	3.00	2.28
C_H2	0.66	1.48	2.81	4.94	1.85	2.35
C_H3	1.38	1.31	3.21	2.69	1.29	1.98
C_H4	1.38	5.31	1.60	2.60	1.19	2.42
C_H5	3.99	2.65	3.94	0.95	0.87	2.48
Avg	1.89	2.51	2.73	2.73	1.64	2.30
C_M1	0.20	0.11	−0.78	0.30	0.08	−0.02
C_M2	−0.10	−2.08	−1.94	−1.41	−1.50	−1.41
C_M3	0.83	−0.04	−0.13	−1.77	−1.22	−0.47
C_M4	1.39	0.58	0.56	1.14	0.54	0.84
C_M5	0.81	−0.41	−0.06	−0.55	−0.13	−0.07
Avg	0.62	−0.37	−0.47	−0.46	−0.45	−0.22
C_L1	−4.35	−3.69	−3.96	−2.73	−2.05	−3.36
C_L2	−4.13	−3.27	−2.82	−2.90	−1.18	−2.86
C_L3	−1.50	−0.95	−1.53	−2.07	−1.29	−1.47
C_L4	−0.98	−1.10	−1.79	−2.83	−1.77	−1.69
C_L5	−2.25	−2.17	−1.56	−1.35	−0.43	−1.55
Avg	−2.64	−2.23	−2.33	−2.38	−1.35	−2.19

variability in the SMA by conducting an additional cluster analysis focusing on the years after 2012 and comparing it with the analysis in this study.

4) CONTRIBUTIONS OF CHANGES IN WIND PATTERNS TO PM₁₀ VARIATIONS

This section presents a comparison of the relative contributions of each wind pattern (C_H1–C_H5, C_M1–C_M5, and C_L1–C_L5) to PM₁₀ concentrations in the SMA for 2012–16. For the comparison, we calculated changes (increases and decreases) in PM₁₀ concentrations (%) when the days of each cluster (i.e., wind pattern) were excluded in the calculations of annual mean PM₁₀ concentrations, the results of which appear in Table 2. A positive/negative value (%) indicates that a wind pattern contributed to increased/decreased PM₁₀ concentrations, and higher values denote relatively large contributions.

In 2012, the average contribution of C_L1–C_L5 (2.6%, negative) was relatively larger than contributions of C_H1–C_H5 (1.9%, positive) and C_M1–C_M5 (0.6%, positive). This finding shows that the lowest PM₁₀ in 2012 was relatively more affected by high counts of low-PM₁₀-related wind patterns, particularly C_L1 (4.4%, negative). In 2013–16, the contributions of C_H1–C_H5 were larger than those of C_L1–C_L5 and C_M1–C_M5. This finding suggests that high-PM₁₀-related wind patterns played a greater role in determining PM₁₀ concentrations in the SMA in those years than low and moderate-PM₁₀-related wind patterns. The wind patterns that contributed the most PM₁₀ from 2012 to 2016

FIG. 9. Annual mean PM_{10} concentrations in the SMA during the 5-yr period of 2012–16, simulated by CMAQ with fixed emissions. The values are averages of 86 AQMS sites in the SMA.

were C_L1 (4.4%, negative), C_H4 (5.3%, positive), C_L1 (4.0%, negative), C_H2 (4.9%, positive), and C_H1 (3.0%, positive).

Overall, the contributions of high- PM_{10} -related wind patterns to PM_{10} concentrations in the SMA during the 5-yr period (2012–16) were relatively larger than those of low and moderate- PM_{10} -related wind patterns. Furthermore, annual PM_{10} variability in 2012–16 was determined by overall variations in the respective wind patterns particularly associated with high and low PM_{10} concentrations.

c. Quantified impact of meteorological conditions on PM_{10} variation in the SMA

In the previous sections, we identified the predominant wind patterns affecting PM_{10} changes in the SMA and compared their relative contributions. In this section, we further quantified the contribution of meteorological conditions to annually PM_{10} concentrations from 2012 to 2016 using the WRF-CMAQ modeling system. WRF and CMAQ simulations for this time period were conducted with a fixed emissions dataset (i.e., MIX 2010 inventory) to quantify the impact of meteorological conditions on PM_{10} simulations in each year.

Figure 9 shows CMAQ-simulated annual mean PM_{10} concentrations in the SMA from 2012 to 2016. Because the five years (2012–16) were characterized by unique meteorological conditions, simulated annual PM_{10} concentrations exhibited clear differences despite the fixed emissions. The PM_{10} differences were the largest at the surface level (≈ 25.15 m) (Fig. S3 in the online supplemental material) because most of the emission sources are distributed near the surface, resulting in a relatively large impact by varying meteorological conditions in comparison with those at the upper levels. During the 5-yr period, simulated surface PM_{10} was the

TABLE 3. Simulated PM_{10} anomalies ($\mu\text{g m}^{-3}$) in the SMA during the 5-yr period from 2012 to 2016, derived from the CMAQ model with fixed emissions. The values are the averages of 86 AQMS sites in the SMA.

	2012	2013	2014	2015	2016
Annual	−4.97	3.55	1.73	0.15	−0.46
Spring	−4.43	2.92	3.99	−1.56	−0.90
Summer	−8.35	0.18	2.87	2.86	2.45
Autumn	−1.88	−2.35	0.84	−1.34	4.74
Winter	−5.23	13.46	−0.78	0.67	−8.12

lowest in 2012 and the highest in 2013, indicating that the meteorological conditions in 2012 and 2013 were respectively the most unfavorable and favorable to high concentrations of PM_{10} . Thus, the dramatic increase in observed PM_{10} concentrations in 2013, relative to those in 2012 (Fig. 2), is likely attributable to their differing atmospheric conditions.

To further investigate the annual varying impact of meteorological conditions on PM_{10} concentrations in the SMA, we calculated the PM_{10} anomaly [$\text{PM}_{10}(\text{each_year}) - \text{PM}_{10}(5\text{-yr_mean})$] of each year of the 5-yr period (2012–16). As listed in Table 3, the PM_{10} anomaly in 2012 had the largest negative value ($−4.97 \mu\text{g m}^{-3}$), suggesting that the atmospheric conditions in 2012 were the most unfavorable to high PM_{10} because of low counts of high- PM_{10} -related wind patterns such as C_H2, C_H3, and C_H4 (Fig. 4 and Table 2) and high counts of low- PM_{10} -related wind patterns such as C_L1, C_L2, and C_L5 (Fig. 8 and Table 2) in 2012. In contrast to 2012, 2013 exhibited the highest positive anomaly value ($3.55 \mu\text{g m}^{-3}$), revealing the most favorable meteorological conditions for high PM_{10} in 2013. The anomaly in 2013 was particularly large during the winter ($13.46 \mu\text{g m}^{-3}$) when high- PM_{10} episodes frequently occur in South Korea. This finding indicates that the weather conditions during the winter of 2013 were particularly favorable to high PM_{10} and likely attributable to the highest count of C_H4 (Fig. 4 and Table 2), which is associated with high PM_{10} in the winter.

The PM_{10} anomaly in 2014 was lower than that in 2013, indicating relatively unfavorable meteorological conditions for high PM_{10} in 2014 in comparison with that in 2013. In particular, the winter of 2014 exhibited a negative PM_{10} anomaly ($−0.78 \mu\text{g m}^{-3}$), which was significantly lower than that of 2013. An explanation for this finding is that the count of C_H4 was significantly lower in 2014 than it was in 2013 (Fig. 4 and Table 2). The lower number of Chinese emissions in 2014 than in 2013 (Zheng et al. 2018) could also explain the smaller increase in PM_{10} in 2014 than in 2013 (1.42%). PM_{10} anomalies in 2015 and 2016 were close to zero, suggesting that meteorological conditions in the two years

did not significantly contribute to changes in the concentrations of PM_{10} .

The above results demonstrate that the impact of meteorological conditions on PM_{10} in the SMA varied from year to year and that the variable meteorological conditions was closely related to annually varying frequencies of wind patterns associated with high and low PM_{10} . Therefore, with a greater understanding of the annually varying contributions of meteorological conditions to PM_{10} concentrations, policymakers in South Korea will be able to more effectively evaluate and establish PM reduction policy. It would be more speculation, however, to conclude that annual differences in meteorological conditions are caused only by wind pattern changes. Since the changes in other meteorological factors (e.g., temperature and humidity) could also affect PM_{10} variation, an analysis of their relative contributions could help us identify the major causes of annual variations in meteorological conditions. We plan to follow this direction of research in the future.

4. Summary and conclusions

This study classified the wind patterns impacting PM_{10} concentrations in the Seoul Metropolitan Area of South Korea and analyzed their annual variability in 2012–16. The annual mean PM_{10} in the SMA indicated a clear downward trend in 2002–11 ($-2.123 \mu\text{g m}^{-3} \text{yr}^{-1}$) that corresponded to a gradual decrease in the number of local emissions. The trend from 2012 to 2016 ($0.006 \mu\text{g m}^{-3} \text{yr}^{-1}$), however, was not reasonably attributable to emissions but implied a greater impact of meteorological conditions on PM_{10} variability from 2012 to 2016 than from 2002 to 2011.

For a further investigation of the impact of wind pattern changes on PM_{10} variability in the SMA in 2002–16, we used *k*-means clustering to analyze wind patterns. We classified the patterns that impacted high (over $80 \mu\text{g m}^{-3}$), low (below $30 \mu\text{g m}^{-3}$), and moderate (between 30 and $80 \mu\text{g m}^{-3}$) PM_{10} concentrations and intensively analyzed their annual changes during the 5-yr period (2012–16). C_H2, C_H4, and C_H5 (wind patterns associated with the transport of foreign emissions, including Asian dust) were the major wind patterns affecting high PM_{10} in the SMA and their annual variabilities largely affected PM_{10} concentrations in 2012–15. C_L1 and C_L2 (wind patterns inducing low PM_{10} in South Korea) were the predominant wind patterns affecting low PM_{10} concentrations. Their annual counts indicated a clear downward trend in 2012–16, demonstrating that the weather conditions in South Korea became unfavorable for the ventilation and the dispersion of air pollutants during this period. Their

decreased counts might have contributed to increased concentrations of PM_{10} from 2012 to 2014, which corresponded to the increased count of C_H2 during these years. The variability in the wind patterns associated with natural meteorological phenomena (C_L3 and C_L4) such as summer monsoons and typhoons only slightly affected annual PM_{10} changes in the SMA. Although C_M4 (the wind pattern inducing stagnant air over South Korea) was the primary wind pattern impacting moderate PM_{10} concentrations (but not high and low PM_{10} concentrations), it did not exhibit noticeable annual variability.

The relative contributions of the wind patterns associated with high PM_{10} (C_H1–C_H5) were larger than those associated with low and moderate PM_{10} (C_L1–C_L5 and C_M1–C_M5, respectively). In addition, PM_{10} variation in 2012–16 was determined by the overall variations in the respective wind patterns particularly associated with high and low PM_{10} concentrations.

The results from CMAQ simulations for the 5-yr period (2012–16) with a fixed emissions dataset (i.e., MIX 2010 inventory) demonstrated that the impact of meteorological conditions on PM_{10} concentrations in the SMA varied from year to year, with the lowest in 2012 and the highest in 2013. They also demonstrated that annual changes in meteorological conditions were closely related to annually varying counts of wind patterns associated with high and low concentrations of PM_{10} . The quantified impacts of meteorological conditions on PM_{10} concentrations from 2012 to 2016) were -4.97 , 3.55 , 1.73 , 0.15 , and $-0.46 \mu\text{g m}^{-3}$, respectively. More case studies that 1) investigate and compare the effects of other meteorological factors (e.g., temperature, humidity, and so on) on PM_{10} variability before and after 2012 and 2) identify the regionally different impact of wind pattern changes should enable the South Korean government to enact effective PM reduction policy.

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