

Cross-Border Spillover Effect of Particulate Matter Pollution between China and Korea*

Hyemin Park** · Wonhyuk Lim*** · Hyungna Oh****

Given its adverse health effects, particulate matter (PM) pollution has become a critical public policy issue in Northeast Asia. As concerns about PM pollution increase, so does the interest in identifying its origins, including transboundary pollutant sources. Employing the daily average PM10 concentration level data from Beijing, Shanghai, and Seoul in 2014–2016, we estimate the direction and extent of the spillover effect of the PM10 density between China and Korea. Estimation outcomes suggest that the PM10 density levels in Beijing and Shanghai are Granger causes of the PM density in Seoul, but not vice versa. That is, the PM 10 density in Seoul increased by 0.13 and 0.133 ppm in response to the 1 ppm increase in the PM10 density in Beijing and Shanghai on the previous day, respectively. The cross-border spillover effect from Beijing decreased by 0.076 ppm from May to October when the air flow hindered the PM10 sources generated in Beijing from reaching Seoul.

JEL Classification: Q53, Q58

Keywords: Particulate Matter, Air Pollution, Spillover, Transboundary Environmental Problem

I. Introduction

Given its adverse health effects, particulate matter (PM) has emerged in recent years as the most important target of air pollution regulation in Northeast Asia. In the 2018 Environmental Performance Index Report, the three largest economies in the region fared poorly in PM2.5 exceedance, a measure of acute exposure based on the proportion of the population that is exposed to ambient PM2.5 concentrations that exceed the critical thresholds set by the World Health Organization (WHO).¹

Received: Dec. 10, 2018. Revised: May 5, 2019. Accepted: July 5, 2019.

* Disclaimer: The findings and conclusions in this paper are those of the authors and do not represent the views of the Ministry of Environment.

** Lead Author, Deputy Director, Ministry of Environment, email: bomnamool@gmail.com

*** Author, Professor, KDI School of Public Policy and Management, email: wlim@kdis.ac.kr

**** Corresponding Author, Professor, Kyung Hee University, email: h.oh@khu.ac.kr

¹ According to this measure, the People's Republic of China (hereafter, China) ranked 177th out of

Owing to its great mobility, PM is also regarded as a regional public good (or “public bad” in this case), resulting in transboundary debates among neighboring countries. For example, a joint Korea–US air quality study in 2017 estimated that China’s contribution to PM density in Korea was as much as 34% (Korea’s National Institute of Environmental Research (NIER), 2017). In response, the Chinese press quoted Wang Gengchen, a researcher at the Chinese Academy of Social Sciences, as saying that PM spillover occurs not only from China to Korea but also from Korea and Japan to China (“Huanqiu, July 24, 2017,” 2017).

Despite the growing interest in the transboundary impact of air pollution between neighboring countries, few relevant studies exist. As Mosteller (2016) pointed out, assigning pollution origination is “an extraordinarily complex and evolving science that relies on sophisticated chemistry, complex remote sensing, statistics, and modeling.” A recent scientific article in *Nature* echoed this difficulty in understanding the cause and effect of air pollution as it is affected by numerous factors, including local and regional sources, weather conditions, and interactions among determinants (H. C. Kim et al., 2017). In the absence of reliable scientific and empirical evidence, transboundary claims have only increased international tensions instead of promoting collective efforts to improve air quality.

This study aims to identify and measure the direction and magnitude of PM spillover between Korea and China while controlling for meteorological factors and domestic economic activities, such as thermal power generation and transportation. Furthermore, this work aims to expand the research methodology in this field by employing an econometric approach over a multiyear period. This work is expected to address the limitations of previous satellite-based observations and modeling studies, which covered limited time periods and sites. We use the daily PM10 density² from 2014–2016 in three of the largest cities in Korea and China: Seoul, Beijing, and Shanghai. These three cities were chosen because they are the most populous and representative cities in both countries.³

The rest of this paper is organized as follows. Section 2 discusses the major characteristics of PM and the control measures employed to combat air pollution, including international cooperation. Section 3 briefly reviews the existing literature. Section 4 discusses the estimation strategy and data. Section 5 sets up the estimation

180 countries, the Republic of Korea (hereafter, Korea) ranked 169th, and Japan ranked 108th (Yale Center for Environmental Law & Policy and The Center for International Earth Science Information Network at Columbia University’s Earth Institute, 2018). Especially in the winter and spring, when PM density frequently exceeds the WHO air quality standards, public concern about its origins rises to high levels in Northeast Asia.

² It is well known that the risk to health of PM_{2.5} is severer than PM₁₀ due to its smaller size. This paper uses PM₁₀ concentration data since they are available and PM₁₀ is highly correlated with PM_{2.5}.

³ Beijing and Seoul are the capital of China and Korea, respectively. The geographic coordinates of the three cities are as follows: Seoul (37.34N, 126.58E), Beijing (39.55N, 116.23E), and Shanghai (31.13N, 121.28E).

model. Section 6 presents the empirical findings. Section 7 concludes this paper.

II. Overview of PM Pollution

1. Major Characteristics of PM

PM is a mixture of solid particles and liquid droplets found in the air and features an aerodynamic diameter of 10 μm (micron, or 10^{-6} m) or less. PM10 refers to particles with diameters that are 10 μm or less. PM2.5 refers to particles with diameters less than 2.5 μm . In comparison, the human hair is approximately 50–70 μm in diameter.

PM has three major characteristics. First, it is lightweight and does not easily sink in the air. Thus, it stays for a long time and travels a long distance to cause damage to a wide area. This characteristic makes PM an international problem. Second, its minuscule size enables penetration into the respiratory tract and causes cardiovascular diseases. Third, the production process of PM is diverse and complicated. It is either directly generated from the combustion of fossil fuels or secondarily produced by the physical and chemical reactions among gaseous substances (e.g., SO_x and NO_x) in the atmosphere. According to the fifth assessment report by the Intergovernmental Panel on Climate Change, the PM floating in air (i.e., aerosols) is difficult to estimate and understand because its properties largely depend on its origin and vary optically, physically, and chemically (IPCC, 2014).

In controlling PM, constituting particles, such as nitrogen oxides, sulfur oxides, and ammonia reaction; and primary particles, such as chimney smoke and scattered dust, must be managed simultaneously. PM reduction is not guaranteed even if the atmospheric environmental standard for each constituent pollutant is satisfied. Hence, PM is different from the pipe-end type of air pollution.

2. Air Pollution Control Measure and Their Limitations

Globally, in the early days of environmental regulation, air quality standards were set on the basis of the total suspended particles (TSPs). However, when particles affecting the human body were found to be small in size, the target for environmental regulation shifted to PM10 and PM2.5.

Korea provides a useful example in this regard. In the 1990s, TSPs, sulfur dioxide (SO₂), and lead (Pb) were the main concerns regarding air pollution control in Korea. Continuous efforts to combat these pollutants led to their steady decline. These efforts included the prohibition of the sale of leaded gasoline, regulation of the sulfur content of diesel and heavy oil, and tightening of site emission control.

By contrast, volatile organic compounds (VOCs) and PM (PM10, PM2.5) showed a relatively slow improvement despite the efforts. For instance, Korea's total air pollutant emissions in 2012 decreased by 16.7%, 11.8%, and 14.4% for carbon monoxide, nitrogen oxides, and sulfur oxides, respectively; whereas PM and VOCs increased by 78.1% and 24.4%, respectively, over the same period (Korean Ministry of Environment, 2015).

Korea established its air quality standards for PM10 in 1995. For PM2.5, Korea set the standards only in 2015. In addition to setting the air quality standards, Korea has also made significant efforts to combat air pollution. For example, in 2005, the Korean government legislated a special act on the improvement of air quality in the capital region (Republic of Korea, 2006), with emphasis on the reduction of fugitive dust (by street wash-out and control of industrial and construction site sources), change in transportation fuel (from diesel to compressed natural gas for buses and trucks), and change in residential heating source (from coal to liquefied natural gas). However, the effectiveness of this act is sometimes questioned by the public because they still experience severe pollutant episodes sometimes.

Other countries in Northeast Asia have also made extensive efforts. China set up special funds in December 2013 to combat air pollution and implemented the Air Pollution Prevention and Control Action Plan (2013–2017), which set emission standards as a top priority during the 12th Five-Year Economic Plan (2011–2015). In addition, the Environmental Protection Law was amended for the first time in 25 years (enacted in 1989, amended in 2014, and enforced on January 1, 2015). Similarly, the Prevention and Control of Air Pollution was strengthened (enacted in 1987, amended in 2015, and enforced on January 1, 2016). China also introduced air quality standards to regulate PM.

Although many countries have set air quality standards to regulate PM, the standards are different across countries, as shown in Table 1. For example, for the PM2.5 annual mean, Japan and Korea have stricter air quality standards at 15 $\mu\text{g}/\text{m}^3$ than China at 35 $\mu\text{g}/\text{m}^3$. In comparison, the U.S. and EU standards are 12 and 25 $\mu\text{g}/\text{m}^3$, respectively. However, as PM travels over the border, only strict control in certain countries would limit its effectiveness (International Council on Clean Transportation and DieselNet, 2018).

[Table 1] Comparison of PM quality standards in several countries (unit: $\mu\text{g}/\text{m}^3$)

Item	Standards	WHO Air Quality Guideline	Northeast Asia			Others	
			Korea	Japan	China	EU	US
PM10	Annual	20	50	-	70	40	-
	24 hour	50	100	100	150	50	150
PM2.5	Annual	10	15	15	35	25	12
	24 hour	25	35	35	75	N/A	35

Given the transboundary characteristics of air pollution, a proper policy response may be international environmental cooperation to improve regional air quality. In Northeast Asia, cooperative frameworks for improving air quality include the Acid Deposition Monitoring Network in East Asia, the Joint Research Project on Long-range Transboundary Air Pollutants in North-East Asia, the North-East Asian Sub-regional Programme for Environmental Cooperation, and the Tripartite Environment Minister Meeting among China, Japan, and Korea. However, regional cooperation has not progressed in Northeast Asia as much as it has in other regions, such as the EU (I. Kim, 2014). Shim (2017) analyzed the performance of these cooperative frameworks and concluded that none of the frameworks have developed a shared understanding of the cause and status of the problem (Shim, 2017). Drifte (2015) asserted that the biggest obstacle to cooperation in Northeast Asia lies in the fact that transboundary pollution in the region mostly originates from China and that China faces considerable domestic conflicts in priority setting in the political, economic, and environmental spheres (Drifte, 2005). He also pointed out that China's effort for the region largely depends on how much environmental aid it can extract from its regional partners. However, if the presence of a spillover is real, then strengthening international cooperation based on scientific evidence would be a reasonable policy recommendation, at least in the normative sense.

III. Literature Review

The World Bank, along with the Institute for Health Metrics and Evaluation, reported that approximately five million people die prematurely every year due to air pollution, and this value accounts for approximately one in every ten deaths annually (World Bank and Institute for Health Metrics and Evaluation, 2016). According to an article published in last March's issue of *Nature*, South Korea, North Korea, Japan, and Mongolia have a collective yearly death rate of 30,900 due to the dust originating from China (Zhang et al., 2017).

Among many air pollution sources, PM penetration is actively studied. Owing to the minuscule size of PM, it not only causes cardiovascular and respiratory diseases but also damages the entire body's metabolism (Goldberg, 2008). However, in contrast to the clear health risks associated with PM, assigning the origination of PM pollution in Northeast Asia is complicated because it is affected by numerous factors, including local and regional emissions and meteorological and chemical interactions (Park and Kim, 2014).

In recent decades, satellite-based or airborne observations covering broad areas have become increasingly available, thus improving the identification of the

temporal and spatial distributions of air pollution (Tao et al., 2012).⁴ Moreover, studies show that the aerosol optical depth (AOD) from satellite observations and the PM₁₀/PM_{2.5} density from ground monitoring stations are highly correlated (Shi et al., 2018) (Chudnovsky et al., 2012)

The PM_{2.5} density data measured from the AOD extracted from the moderate resolution imaging spectroradiometer and the multiangle imaging spectro radiometer satellite image data from 2001 to 2006 show that the PM_{2.5} density was dense in the regions of rapid economic growth and urbanization (van Donkelaar et al., 2010). In addition, the AOD annual variation observed by the Sea-Viewing Wide Field-of-View satellite from 1998 to 2010 showed a large increase in Asian countries, such as India and China, where the population density is high (Hsu et al., 2012).

These studies looked into the single-country or continent-level spatial and temporal patterns of PM density while some studies analyzed the PM spillover effects. In 2016, a field study was conducted by the NIER in Korea and the National Aeronautics and Space Administration in the US (“KORUS-AQ”, Korea–US Air Quality Study) to measure the transboundary effect of PM in Northeast Asia. On the basis of the data observed by satellites, China’s PM contribution to Korea was estimated at 34% at a specific cite (Seoul Olympic Park) during a specific period (from May 1, 2016 to June 10, 2016). However, as acknowledged by the authors, this study had substantial drawbacks, such as the limited observation period and sites and the presence of uncontrolled factors in estimating the cross-border effect of PM. In addition, the satellite data on cloudy days were not easy to work with and raised concerns about accuracy (NIER, 2017). Scientists also pointed out that the nature and origin of PM routes are impossible to identify using only satellite images because no geosynchronous satellites over the skies of Asia conduct monitoring 24 h a day (Sumitomo Mitsui Advanced Finance for Ecology, 2013).

Using the city-level panel data on the daily air pollution index (API) for Chinese cities from 2009 to 2013, Chen and Ye measured the spillover effects of air pollution across cities and published the following findings: (i) the air pollution in China has spatial spillover effects, that is, a city’s average API is expected to increase by 0.40–0.51 in response to a one unit increase in the average API of its surrounding cities; (ii) an increase in gasoline price can improve urban air quality; (iii) strong winds can mitigate air pollution (Chen and Ye, 2015). Their findings suggest that pollution control policies must be coordinated among cities and provinces to effectively abate urban air pollution.

Jia and Ku (2015) assessed the impact of cross-border air pollution from China to

⁴ In addition to ground and airborne monitoring, remote sensing and chemical transport models have also been used to characterize spatiotemporal patterns and simulate the emergence, expansion, and dissipation of air pollution (Cuchiara et al., 2014).

Korea by exploiting the within-Korea variations in the incidence of Asian dust⁵ and the temporal variations in China's air quality index (Jia and Ku, 2015). This work concluded that a one standard deviation increase in China's pollution leads to approximately 280 extra deaths from respiratory and cardiovascular diseases per year in Korea, with additional effects on the overall mortality of children aged below five years. Altindag et al. (2017) leveraged birth weight and Asian dust alert to confirm the cost that China's pollution has imposed on nearby nations (Altindag, Baek, and Mocan, 2017).

IV. Estimation Strategy and Data

Although numerous media reports in Korea claim that fine dust from China blows over to Korea, few studies have empirically confirmed this claim. Scientific studies that employ satellite technology support this hypothesis, but their research scope is limited to short time periods and/or geographically narrow areas. To address the limitations of satellite-based studies, we adopt an econometric approach and broadened the research period. This empirical exercise is performed using daily data from 2014 to 2016.

4.1. Estimation Strategy

The underlying model for investigating the effects of PM in China that spreads to Korea is expressed as Equation (1).

$$PM_t^S = Z_t \pi + \varepsilon_t, \text{ where } Z_t = [X_t^{in} \ X_t^{ex}] \text{ and } \pi = [\alpha \ \beta]' \quad (1)$$

In Equation (1), PM_t^S is the variable of interest, the density of PM in Seoul at time t . Z_t is an explanatory variable set consisting of X_t^{in} and X_t^{ex} , which represent the internal factors in Korea and the external factors from China, respectively. α and β are the corresponding coefficient vectors, and ε_t is the standard disturbance term following a normal distribution.

We refer to the literature in choosing the meteorological variables. The internal explanatory vector (X_t^{in}) includes the following: (i) Korea's meteorological conditions (e.g., average daily temperature ($temper_t$), diurnal temperature variation, or the gap between high and low temperature ($temper_gap_t$)), average daily wind speed ($wind_spd_t$), a dummy representing summer effects ($summr_t$), and humidity (hum_t); (ii) the amount of electricity generated by the thermal power

⁵ Asian dust is a meteorological phenomenon in which yellow dust clouds passing over China are carried eastward to Korea by strong, stable westerly winds.

plants in Gyeonggi Province ($power_supply_t$) and the area surrounding the capital of, Seoul; (iii) the use of transport gas and diesel in Seoul ($gas_t, diesel_t$); and (iv) their interaction terms.

Among them, diurnal temperature variation is included in X_t^{in} to account for the impact of temperature reversal on PM10. Temperature reversal is an atmospheric congestion phenomenon in which heavy air is located at a lower elevation, mild air is located at a higher elevation, and no upward and downward air movements from the weight difference occur. This phenomenon occurs when the temperature gap is large. Given such a temperature reversal, air pollutants, such as the PM10 generated on the ground, are not diffused into the atmosphere but continuously accumulate on the ground, thereby increasing the PM density.

A seasonal factor is also included as a proxy for air flow. Although our main interest is to verify the influence of Beijing and Shanghai on Seoul, the data on daily dominant air flow from Beijing and Shanghai to Seoul are unavailable. Nevertheless, a well-documented seasonality in air flow exists. That is, in winter and spring, westerly winds blow from Beijing to Baekryeong Island and then to Seoul, whereas in the summer and fall, southwesterly winds blow from Shanghai to Jeju and then to Seoul. A seasonal dummy, summer, takes the value of 1 from May to October. An interaction term between the season and the PM density in Chinese cities was inserted to account for the effect of air flow.

By including X_t^{in} in the model, we can obtain a controlled relationship between PM_t^S , which is the PM10 density in Seoul, and X_t^{ex} , which are the external factors, such as PM10 density in large cities in China. X_t^{ex} includes the PM10 density in Beijing and Shanghai at time $t-1$, $PM_{t-1}^{Beijing}$ and $PM_{t-1}^{Shanghai}$. Given the distance between Seoul and the cities in China, we assume a time interval for pollutants in China to reach Seoul. After investigating the correlations between the PM densities in Seoul and in the Chinese cities, we observe that the PM density a day before in Chinese cities shows the highest correlation with the PM density in Seoul.⁶ The descriptive statistics for the variables used in the regression models are reported in Table 2.

⁶ We tested various models with $t-1$, $t-2$, and $t-3$ lagged variables and combinations of these. According to AIC and adjusted R^2 , the fitness of a model with $(t-1)$ and $(t-2)$ PM10 densities of Beijing and Shanghai turned out to be the best. However, each $(t-2)$ lagged variable was either statistically insignificant (for $PM_{t-2}^{Beijing}$) or significant only at the 10% significance level (for $PM_{t-2}^{Shanghai}$). Estimation outcomes with various dynamic settings are reported in Appendix 3. After the subsequent model search, we estimated Equation (2) using CO-AR4. The effect of $PM_{t-1}^{Shanghai}$ in determining PM_t^{Seoul} was taken into account in CO-AR4 as it employed $PM_{t-1}^{Shanghai}$, ..., $PM_{t-4}^{Shanghai}$ and $PM_{t-1}^{Beijing}$, ..., $PM_{t-4}^{Beijing}$.

[Table 2] Descriptive Statistics

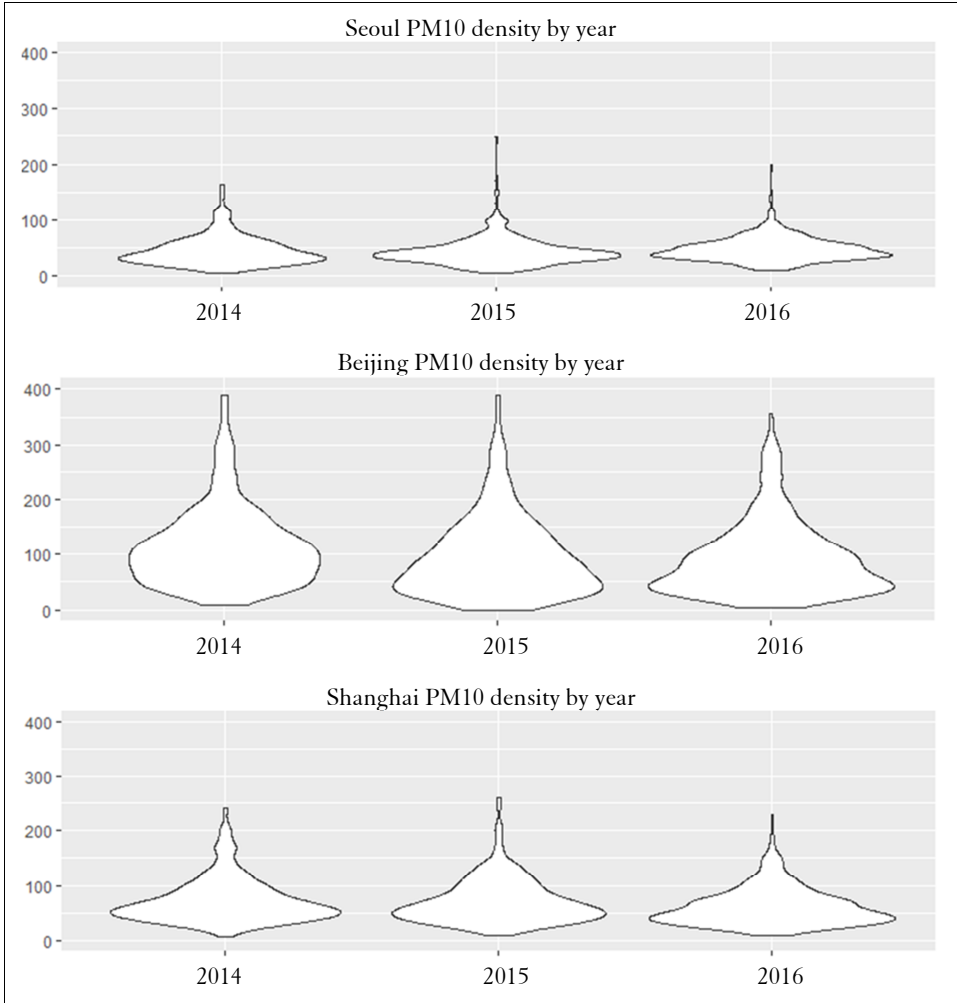
Variables		Description	Average	Standard deviation
X_{t-1}^{ex}	$PM_{t-1}^{Shanghai}$	PM10 in Shanghai at $t-1$	68.6	39.2
	$PM_{t-1}^{Beijing}$	PM10 in Beijing at $t-1$	102.5	74
X_t^{in}	$temper_t$	Daily average temperature	13.5	10.5
	$temper_gap_t$	Daily temperature gap	9.2	10.5
	hum_t	Daily average humidity	60.6	14.7
	$wind_spd_t$	Daily average wind speed	2.5	0.8
	$power_supply_t$	Daily power supply in Gyeonggi	1035.3	383.6
	$diesel_t$	Daily Use of transport diesel	3293.8	314.7
	gas_t	Daily Use of transport gas	4173.1	328.6
Y_t	PM_t^{Seoul}	PM10 in Seoul at t	46.3	29.4

4.2. PM10 Density

Many archived observations are not publicly available in China. In addition, nearly all the data from within China originate from stations operated by provincial environmental agencies that have not been incorporated into China’s national network. Given these restrictions, a third-party source, available at www.tianqihoubao.com, is used. It provides information on the concentration levels of major pollutants (PM, SO₂, NO₂, CO, O₃) in Chinese cities since 2012. Other data are obtained as follows: the PM10 density in Seoul from the NIER of Korea, weather information from the National Weather Data Center of the Korea Meteorological Administration, the use of transport gas and diesel in Seoul from the Korea National Oil Corporation, and the amount of electricity generated by thermal power plants in Gyeonggi Province from the Korea Power Exchange.

Figure 1 depicts the distributions of the PM10 density in three cities, namely, Seoul, Beijing, and Shanghai. The PM10 density in the two Chinese cities decreased over time. This improvement may be attributed to China’s recent efforts to combat air pollution.

[Figure 1] PM10 density in Seoul, Beijing, and Shanghai (2014–2016)



V. Model Search

5.1. Checking Stationarity

Given that the PM10 density data are time series data, stationarity concerns can arise. According to the Dickey–Fuller test, the unit root was not found, and the PM10 data of the three cities to be used in the regression were all stationary. This is confirmed by three Dickey–Fuller statistics, -17.535 , -17.622 , and -18.324 for the PM10 data series observed in Seoul, Beijing, and Shanghai, respectively.⁷

⁷ P-value<.01.

5.2. Checking Granger Causality

According to some Chinese experts, such as the one quoted by Huanqui (“Huanqiu, July 24.2017,” 2017), PM spillover may occur not only from China to Korea but also from Korea to China. If PM10 in Seoul and PM10 in Chinese cities are mutually influential, then the estimation strategies used to capture the interdependencies among multiple time series should be employed. However, if the statistically significant direction of influence is one way, then a univariate approach may be adopted.

To address this issue, we carried out a Granger causality test prior to the estimation to confirm the direction of influence. Considering the flow of air between Seoul and the two Chinese cities, we assumed that the PM density of the previous days in one region could affect the PM density of the next days in the other region. PM_t^{Seoul} , $PM_t^{Beijing}$, and $PM_t^{Shanghai}$ are the PM10 densities in Seoul, Beijing, and Shanghai, respectively. If the past values of variable Y, PM_{t-s}^{Seoul} , contribute to determine the current and future values of X ($PM_{t+f}^{Beijing}$ and $PM_{t+f}^{Shanghai}$, where $f = 0, 1, 2, \dots$), then Y is said to Granger-cause X (C. W. J. Granger, 1969). Conversely, if the past values of X improve the prediction of Y, then X is said to Granger-cause Y.

We tested whether the PM10 density of Chinese cities (Beijing and Shanghai) would affect that of Seoul and then tested the opposite. According to the test outcomes shown in Table 3, Beijing’s PM10 and Shanghai’s PM10 Granger-caused Seoul’s PM10 concentration, but the reverse was not statistically significant. These findings were supported by large F-statistic values from Beijing and Shanghai to Seoul (i.e., 28.05 and 9.80, respectively) and a small F-statistic value for the opposite (i.e., 0.95 and 1.45, respectively) (Table 3). This result implied that the PM10 density in the two Chinese cities Granger-caused the PM10 density in Seoul but that the PM10 density in Seoul was statistically insignificant in determining the PM10 density in the two Chinese cities.

[Table 3] Outcomes of Granger Causality Tests

Cause ↓		Response →		
		PM10 in Seoul	PM10 in Chinese Cities	
			PM10 in Beijing	PM10 in Shanghai
South Korea: PM10 in Seoul		-	0.945	1.445
China:	PM10 in Beijing	28.045***	-	-
	PM10 in Shanghai	9.804***	-	-

Note: *, **, and *** represent statistically significant levels of 0.1, 0.05, and 0.001, respectively.

5.3. Checking Homoscedasticity and Autoregressive Errors

We looked at the residuals to check whether they are spherical or not. The

Breusch–Pagan test confirmed the homoscedasticity assumption for the residuals, and the Breusch–Godfrey test revealed that the residuals are autocorrelated (see Appendix 1 for the test result and residual plots). Accordingly, a Cochrane–Orcutt estimation technique was applied. As a prior procedure for the Cochrane–Orcutt approach, we examined various possible orders of the autoregressive structure for residual ε_t in Equation (1). Then, AR(4) was selected because the Akaike information criterion (AIC) was the lowest with AR(4).⁸

Given the one-way causality and autoregressive errors, we finally selected a Cochrane–Orcutt approach with fourth-order autoregressive errors (henceforth, CO-AR4) and estimated the following transformed models shown in Equation (2):

$$PM_t^{S*} = Z_t^* \pi + v_t = X_t^{in*} \alpha + X_{t-1}^{ex*} \beta + W_t^{\otimes} \delta + v_t \quad (2)$$

, where $PM_t^{S*} = PM_t^S - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot PM_{t-h}^S$, and

$$Z_t^* = Z_t - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot Z_{t-h}^S = \begin{bmatrix} X_t^{in} - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot X_{t-h}^{in} \\ X_{t-1}^{ex} - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot X_{t-1-h}^{ex} \\ W_t^{\otimes} - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot W_{t-h}^{\otimes} \end{bmatrix}^T.$$

In Equation (2), $\hat{\beta}_{h,GLS}$ is the GLS-corrected estimate for the h^{th} order autoregressive error parameter described by Choi et al. (Choi, Hu, and Ogaki, 2005). When $\hat{\beta}_{h,GLS}$ was estimated, the variables included in Equation (2) were identified. For example, X_{t-1}^{ex*} are the predetermined values of $PM_{t-1}^{Beijing}$ and $PM_{t-1}^{Shanghai}$, and $\beta \cdot X_{t-1}^{ex*}$ is expressed as follows:

$$X_{t-1}^{ex*} \beta = \beta_B \cdot (PM_{t-1}^{Beijing} - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot PM_{t-1-h}^{Beijing}) \\ + \beta_S \cdot (PM_{t-1}^{Shanghai} - \sum_{h=1}^4 \hat{\beta}_{h,GLS} \cdot PM_{t-1-h}^{Shanghai}).$$

In addition, the set of interaction terms between the Chinese factors and the weather variables, W_t^{\otimes} , was included in the model to capture the combined effects of the two.

VI. Estimation Outcomes

Owing to autoregressive errors, a conventional OLS approach to Equation (2) did not seem appropriate. Nonetheless, we included its regression results into Table

⁸ Note that the AIC value with AR(4) was 9916.116, the lowest among those from various AR specifications. The AIC value with AR(4) was the lowest for either the full or the reduced model.

4 as a reference. The CO-AR4 estimation outcomes of Equation (2) are presented in Table 4. We also employed a reduced model of the full CO-AR4. The backward elimination method suggested by Draper was used to define the best set of explanatory variables for the reduced model for CO-AR4 without causing specification bias (Draper and Smith, 2014). The F-statistic, 0.427, and its associated p-value, 0.7337, suggested that we could rely on the reduced CO-AR4 model. After applying CO-AR4, we tested whether the assumption for homoscedasticity was violated. The estimation results of the reduced CO-AR4 model are also listed in Table 4.

According to the estimation outcomes of the reduced CO-AR4, the PM10 density in Seoul increased by 0.130 and 0.132 ppm in response to the 1 ppm increase in PM10 density in Beijing and Shanghai on the previous day, respectively. This result is supported by the coefficient estimates for $PM_{t-1}^{Beijing}$ and $PM_{t-1}^{Shanghai}$, 0.130 and 0.132, which are nonzero and statistically significant. However, the PM spillover effect from Beijing decreased by 0.074 ppm in summer when the wind flow hindered the PM10 generated in Beijing from reaching Seoul. Meanwhile, the Shanghai effects were statistically identical regardless of the season.⁹

Among Korea's domestic weather factors, temperature, humidity, and wind speed showed statistical significance. The PM density in Seoul increased by 0.548 ppm when the average daily temperature rose by 1 °C. The daily average humidity and wind speed decreased the PM10 density in Seoul by 0.163 and 2.67 ppm per 1% point humidity increase and 1 m/s wind speed increase, respectively. However, the season factor (summer) itself was insignificant in determining the PM in Seoul.

Regarding internal factors other than the meteorological conditions, the amount of thermal power generation increased the PM10 density in Seoul by 0.010 ppm for every 1 MWh increase in power supply. For transportation, the interaction term between diesel and wind speed was negative and statistically significant, indicating that the diesel effect decreased when the wind speed increased.

As shown in the last column of Table 4, we also employed weighted least squares (WLS) in Equation (4) and defined it as CO-WLS. The estimation model was added because using the estimated errors from the reduced CO-AR4, we tested whether the assumption of homoscedasticity was violated. The Breusch–Pagan test confirmed that the errors of the reduced CO-AR4 satisfied the assumption of constant variance. However, we found some observations with Cook's distance exceeding the corresponding thresholds. Given these observations, a WLS approach to the CO-AR4 model, CO-WLS, was attempted. In other words, CO-WLS was

⁹ We considered PM10 in Shanghai and that in Beijing as a pair. As the summer effect of PM10 in Beijing, represented by the interaction term of $PM_{t-1}^{Beijing}$ and *summer*, is statistically significant in determining PM10 in Seoul, we included both summer effects in the model. The inclusion of an irrelevant variable can lead to a loss in efficiency. However, we included it to prevent a missing variable bias, along with the fact that it is a variable of interest.

[Table 4] Estimation Outcomes for PM10 in Seoul

			FULL		REDUCED		
			OLS	CO-AR4	CO-AR4	CO-WLS	
X_{t-1}^{ex}	$PM_{t-1}^{Shanghai}$	PM10 in Shanghai at $t-1$	0.115 *** (0.026)	0.133 *** (0.024)	0.132 *** (0.024)	0.098 *** (0.017)	
	$PM_{t-1}^{Beijing}$	PM10 in Beijing at $t-1$	0.123 *** (0.013)	0.130 *** (0.014)	0.130 *** (0.014)	0.105 *** (0.010)	
X_t^{in}	$summer_t$	Summer effect	-13.076 ** (5.009)	-2.938 (6.037)	-3.570 (5.982)	-5.918 (4.160)	
	$temper_t$	Daily average temper	0.576 *** (0.162)	0.509 * (0.216)	0.548 * (0.211)	0.611 *** (0.146)	
	hum_t	Daily average humidity	-0.077 (0.076)	-0.126 . (0.072)	-0.163 ** (0.058)	-0.112 ** (0.040)	
	$wind_spd_t$	Daily average wind speed	-3.229 ** (0.985)	-2.595 ** (0.901)	-2.669 ** (0.872)	-3.126 *** (0.605)	
	$power_supply_t$	Power supply in Gyeonggi	0.011 *** (0.003)	0.010 ** (0.003)	0.010 ** (0.003)	0.011 *** (0.002)	
	$diesel_t$	Use of transport diesel	0.036 * (0.015)	0.019 (0.015)	0.011 (0.007)	0.012 * (0.005)	
	$diesel_t \times wind_spd_t$	Domestic interaction term1	-0.015 ** (0.006)	-0.009 (0.005)	-0.006 * (0.002)	-0.006 ** (0.002)	
	$temper_gap_t$	Daily temperature gap	0.849 * (0.362)	0.250 (0.3)			
	gas_t	Use of transport gas	-0.013 (0.014)	-0.009 (0.13))			
	$gas_t \times wind_spd_t$	Domestic interaction term2	0.004 (0.005)	0.004 (0.005)			
	W_t^{\otimes}	$PM_{t-1}^{Shanghai} \times summer_t$	Summer effect of PM10 in Chinese cities at $t-1$	0.084 . (0.044)	-0.028 (0.046)	-0.023 (0.046)	-0.020 (0.033)
		$PM_{t-1}^{Beijing} \times summer_t$		-0.066 ** (0.025)	-0.076 ** (0.026)	-0.074 ** (0.025)	-0.063 *** (0.018)
Intercept			18.692 * (8.377)	12.267 ** (4.421)	14.374 *** (3.637)	14.676 *** (2.494)	
AIC			10240.0	9876.2	9871.5	9186.7	
Adjusted R^2			0.238	0.157	0.158	0.228	

Note: Standard errors are shown in parentheses; and *, **, and *** represent statistically significant levels of 0.1, 0.05, and 0.001, respectively.

used for robustness check to demonstrate that the validity our findings is not critically dependent on outliers. In the WLS model, the weights were equally set to “1/residual square.” The impacts of PM10 density in two Chinese cities on the PM10 density in Seoul estimated by CO-WLS were significant, although they were quantitatively smaller than those of the CO-AR4 estimation results. This result was

supported by the parameter estimates of $PM_{t-1}^{Shanghai}$ and $PM_{t-1}^{Beijing}$, 0.098 and 0.105, which were smaller than those of CO-AR4 (0.132 and 0.130, respectively) but still statistically significant at the 1% significance level.

VII. Conclusion

Cross-border PM spillover has become a sensitive issue in Northeast Asia. As the public concerns about PM pollution increase, so does the interest in identifying its origins. This study empirically tested this cross-border PM spillover hypothesis. By including several domestic factor variables in the model, we can obtain a controlled relationship among the PM densities in Seoul and two major cities in China, namely, Beijing and Shanghai.

For our empirical exercise, we used the daily average PM10 concentration level data from 2014 to 2016 and several domestic explanatory variables, such as meteorological conditions, thermal power generation, and transportation fuel consumption. For the first empirical exercise, we used the time series data on PM10 density in Beijing, Shanghai, and Seoul to estimate the direction and extent of the spillover effects. A series of Granger causality tests showed that the PM10 concentration levels in Beijing and Shanghai were Granger causes of the PM10 density in Seoul, but not vice versa.

Then, we used a CO-AR4 model to explore the relationship between the PM10 density in Seoul and its possible determinants. In addition to external factors, such as the PM10 concentration levels in Beijing and Shanghai, we looked at Korea's internal factors, such as meteorological conditions, domestic economic activities, and energy consumption. According to the CO-AR4 estimation outcomes, the PM10 density in Seoul increased by 0.130 and 0.132 ppm in response to a 1 ppm increase in PM10 density in Beijing and Shanghai on the previous day, respectively. However, the PM spillover effect from Beijing decreased by 0.074 ppm from May to October when the wind flow hindered the PM10 generated in Beijing from reaching Seoul.

Among Korea's domestic weather factors, temperature, humidity, and wind speed exhibited statistical significance. The PM density in Seoul increased by 0.548 ppm when the average daily temperature rose by 1 °C. The daily average humidity and wind speed reduced the PM density in Seoul by 0.163 and 2.67 ppm per 1% point and 1 m/s increases, respectively. As for internal factors other than the meteorological conditions, the amount of electricity generated by the thermal power plants in Gyeonggi showed statistical significance. The amount of thermal power generation increased the PM density in Seoul by 0.010 ppm for every 1 MWh increase in power supply. According to the CO-AR4 model, the impact of diesel

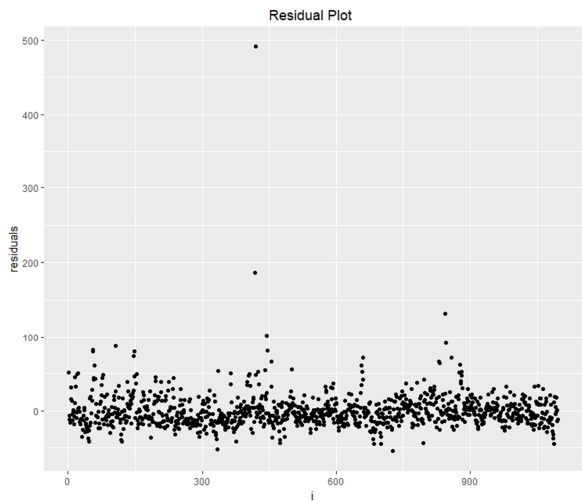
consumption for transportation on the PM density in Seoul was statistically negligible. However, we found some errors in the CO-AR4 estimation outcomes with a large Cook's value. To reduce the influence of these observations, we estimated the model with WLS. The PM spillover effect from the two Chinese cities decreased, and the impact of domestic diesel consumption for transportation was statistically significant. The PM10 density in Seoul increased by 0.012 ppm in response to 1-kl increase in transportation diesel consumption. Regardless of the model, the diesel effect decreased by 0.006 ppm on a windy day.

To our knowledge, this work is the first attempt to adopt an econometric approach to measuring the PM10 spillover effect between Korea and China. Given the cross-border PM spillover effect, China's reinforced air quality policies in recent years are expected to have a significant and positive impact not only on its own air quality but also on the air quality in Northeast Asia. Meanwhile, given the cross-border spillover effect and the remaining gap with the WHO air quality standards, the countries in the region, individually and collectively, must exert extensive effort not only for the health of their citizens but also for those of the citizens of neighboring countries.

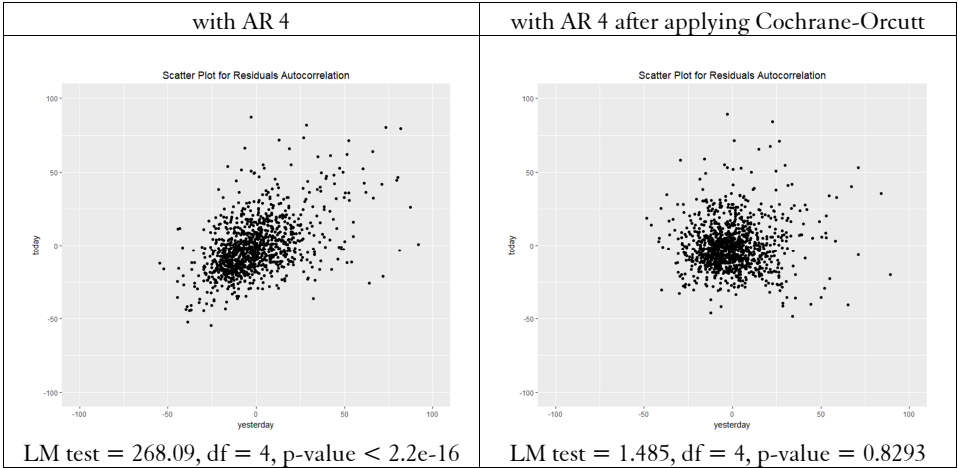
In this study, we only conducted the Granger causality test for the nexus among the PM density levels in Beijing, Shanghai, and Seoul. Although the PM density in Beijing and Shanghai was the highest in China during the study period (2014–2016), air pollution in other areas, such as the east coast cities in China, had an impact on the PM10 density in Seoul. They are located close to Korea, and their pollution-emitting facilities have been relocated since 2016 as a result of the public outcry over PM in major cities. However, the spillover effect of air pollution in these areas on determining the PM density in Seoul was not considered in this study and must be addressed in future research.

Appendix 1. Detect and Solve Homoscedasticity and Autoregressive Errors

1) Outcomes of a Homoscedasticity Test using a Studentized Breusch–Pagan Test \sim BP = 20.29, df = 14, p-value = 0.1212



2) Outcomes of the Breusch–Godfrey Test for Serial Correlation



Appendix 2. Pearson’s Correlation Coefficients Among the PM10 levels in Beijing, Shanghai, and Seoul

	lag	PM_t^{Seoul}	$PM_t^{Beijing}$	$PM_t^{Shanghai}$
PM_{t-h}^{Seoul}	$h = 0$	1	0.232	0.227
	$h = 1$	0.567	0.138	0.188
	$h = 2$	0.257	0.078	0.107
	$h = 3$	0.186	0.032	0.075
$PM_{t-h}^{Beijing}$	$h = 0$	0.232	1	0.060
	$h = 1$	0.353	0.561	0.136
	$h = 2$	0.250	0.203	0.138
	$h = 3$	0.154	0.104	0.067
$PM_{t-h}^{Shanghai}$	$h = 0$	0.227	0.06	1
	$h = 1$	0.239	0.092	0.540
	$h = 2$	0.149	0.125	0.268
	$h = 3$	0.119	0.134	0.162

Appendix 3. Estimation Results of PM10 Density in Seoul with Various Time Lags of Cross-border Determinants

	Lag 1,2,3		Lag 1,2		Lag 1,3		Lag 2,3		Lag 1		Lag 2		Lag 3	
	Estimate	Pr(> t)	Estimate	Pr(> t)	Estimate	Pr(> t)	Estimate	Pr(> t)	Estimate	Pr(> t)	Estimate	Pr(> t)	Estimate	Pr(> t)
(Intercept)	22.402	0.010 *	21.528	0.012 *	21.310	0.014 *	22.708	0.010 *	18.692	0.026 *	21.520	0.013 *	22.057	0.013 *
$PM_{Shanghai_{t-1}}$	0.137	0.000 ****	0.137	0.000 ****	0.117	0.000 ****			0.115	0.000 ****				
$PM_{Beijing_{t-1}}$	0.117	0.000 ****	0.113	0.000 ****	0.122	< 2e-16 ****			0.123	< 2e-16 ****				
$PM_{Shanghai_{t-2}}$	-0.057	0.071	-0.062	0.029 *			0.010	0.727			0.001	0.970		
$PM_{Beijing_{t-2}}$	0.010	0.587	0.019	0.214			0.086	0.000 ****			0.082	0.000 ****		
$PM_{Shanghai_{t-3}}$	-0.017	0.544			-0.039	0.126	-0.017	0.572					-0.008	0.762
$PM_{Beijing_{t-3}}$	0.015	0.343			0.016	0.205	-0.007	0.640					0.036	0.007 **
summer _t	-19.419	0.001 ****	-17.281	0.001 ****	-18.756	0.001 ****	-28.064	0.000 ****	-13.076	0.009 ****	-25.700	0.000 ****	-30.584	0.000 ****
temper_gap _t	0.847	0.020 *	0.893	0.014 *	0.794	0.028 *	1.344	0.000 ****	0.849	0.019 *	1.368	0.000 ****	1.419	0.000 ****
temper _t	0.538	0.001 ****	0.539	0.001 ****	0.559	0.001 ****	0.601	0.000 ****	0.576	0.000 ****	0.595	0.000 ****	0.717	0.000 ****
hum_t	-0.080	0.294	-0.074	0.330	-0.083	0.276	0.047	0.541	-0.077	0.314	0.050	0.514	0.084	0.285
wind_spd _t	-3.682	0.000 ****	-3.609	0.000 ****	-3.479	0.000 ****	-4.161	0.000 ****	-32.29	0.001 ****	-4.183	0.000 ****	-3.874	0.000 ****
power_supply _t	0.010	0.000 ****	0.010	0.000 ****	0.011	0.000 ****	0.012	0.000 ****	0.011	0.000 ****	0.012	0.000 ****	0.013	0.000 ****

gas_t	-0.013	0.337	-0.012	0.374	-0.014	0.307	-0.014	0.343	-0.013	0.343	-0.013	0.351	-0.018	0.230
$diesel_t$	0.037	0.013	*	0.036	0.017	*	0.038	0.011	*	0.036	0.021	*	0.037	0.020
$PM_{t-1}^{Shanghai} \times summer_t$	0.028	0.607		0.018	0.739		0.059	0.205		0.084	0.058			
$PM_{t-1}^{Beijing} \times summer_t$	-0.086	0.004	**	-0.084	0.005	**	-0.078	0.002	**	-0.066	0.008	**		
$PM_{t-2}^{Shanghai} \times summer_t$	0.081	0.204		0.133	0.013	*						0.000	****	
$PM_{t-2}^{Beijing} \times summer_t$	0.017	0.635		0.021	0.470							0.222		
$PM_{t-3}^{Shanghai} \times summer_t$	0.086	0.110					0.121	0.007	**				0.147	0.001
$PM_{t-3}^{Beijing} \times summer_t$	0.003	0.916					0.013	0.592					-0.002	0.939
$gas_t \times wind_spd_t$	0.004	0.412		0.004	0.467		0.005	0.381		0.004	0.443		0.007	0.215
$diesel_t \times wind_spd_t$	-0.016	0.006	**	-0.015	0.007	**	-0.016	0.005	**	-0.015	0.007	**	-0.018	0.003
Adjusted R-Square	0.2429		0.2429	0.2429		0.2423		0.1858		0.2376		0.1875		0.1581
AIC	10240.26		10236.33			10237.1		10316.05		10240.03		10309.75		10348.81

References

- Altindag, D. T., D. Baek, and N. Mocan (2017), “Chinese Yellow Dust and Korean Infant Health,” *Social Science and Medicine*, 186, 78–86.
- Chen, X., and J. Ye (2015), “When the Wind Blows: Spatial Spillover Effects of Urban Air Pollution,” *Environment for Development Discussion Paper Series*, 15(15).
- Choi, C.-Y., L. Hu, and M. Ogaki (2005), “Structural Spurious Regressions and A Hausman-type Cointegration Test,” Rochester Center for Economic Research Working Paper(517).
- Chudnovsky, A. A., H. J. Lee, A. Kostinski, T. Kotlov, and P. Koutrakis (2012), “Prediction of Daily Fine Particulate Matter Concentrations using Aerosol Optical Depth Retrievals from the Geostationary Operational Environmental Satellite (GOES),” *Journal of the Air and Waste Management Association*, 62(9), 1022–1031.
- Cuchiara, G. C., X. Li, J. Carvalho, and B. Rappenglück (2014), “Intercomparison of Planetary Boundary Layer Parameterization and its Impacts on Surface Ozone Concentration in the WRF/Chem Model for a Case Study in Houston/texas,” *Atmospheric Environment*, 96, 175–185.
- Draper, N. R., and H. Smith (2014), *Applied Regression Analysis* (Third Edit): John Wiley & Sons.
- Drifte, R. (2005), “Transboundary Pollution as an Issue in Northeast Asian Regional Politics (Working paper),” Retrieved from <http://eprints.lse.ac.uk/id/eprint/25201>
- Goldberg, M. (2008), “A Systematic Review of the Relation Between Long-term Exposure to Ambient Air Pollution and Chronic Diseases,” *Reviews on Environmental Health*, 23(4), 243–298.
- Granger, C. W. J. (1969), “Investigating Causal Relations by Econometric Models and Cross-spectral Methods,” *Econometrica*, 37(3), 424–438.
- Hsu, N. C., R. Gautam, A. M. Sayer, C. Bettenhausen, C. Li, M. J. Jeong, and B. N. Holben (2012), “Global and Regional Trends of Aerosol Optical Depth over Land and Ocean using SeaWiFS Measurements from 1997 to 2010,” *Atmospheric Chemistry and Physics*, 12(17), 8037–8053.
- Huanqiu News Article, July 24. 2017. (2017), Retrieved from <http://world.huanqiu.com/exclusive/2017-07/11023145.html>
- IPCC. (2014), *Climate Change 2014: Synthesis Report*, Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.
- Jia, R., and H. Ku (2015), “Is China’s Pollution the Culprit for the Choking of South Korea? Evidence from the Asian Dust (Working Paper),” Retrieved from <http://www.sole-jole.org/16345.pdf>
- Kim, H. C., S. Kim, B. U. Kim, C. S. Jin, S. Hong, R. Park, and A. Stein (2017), “Recent Increase of Surface Particulate Matter Concentrations in the Seoul Metropolitan Area, Korea,” *Scientific Reports of Nature*, 7(1), 1–7.
- Kim, I. (2014), “Messages from a Middle Power: Participation by the Republic of Korea in Regional Environmental Cooperation on Transboundary Air Pollution Issues,” *International Environmental Agreements: Politics, Law and Economics*, 14(2), 147–162.

- Korea's National Institute of Environmental Research (NIER) (2017), The KORUS-AQ Rapid Science Synthesis Report, Retrieved from https://espo.nasa.gov/sites/default/files/documents/KORUS-AQ_RSSR.pdf
- Korean Ministry of Environment (2015), Climate Statistics: Korean Ministry of Environment.
- Mosteller, D. (2016), Air Pollution's Hazy Future in South Korea, Retrieved from <https://datadriven.yale.edu/air-quality-2/air-pollutions-hazy-future-in-south-korea-2/>
- Park, R. J., and S. W. Kim (2014), "Air Quality Modeling in East Asia: Present Issues and Future Directions," *Asia-Pacific Journal of Atmospheric Sciences*, 50(1), 105–120. <https://doi.org/10.1007/s13143-014-0030-9>
- Republic of Korea (2006), UN. National Reporting Guidelines for CSD-14/15 Thematic Areas - Atmosphere/Air Pollution.
- Shi, Y., H. C. Ho, Y. Xu, and E. Ng (2018), "Improving Satellite Aerosol Optical Depth-PM_{2.5} correlations using Land use Regression with Microscale Geographic Predictors in a High-density Urban Context," *Atmospheric Environment*, 190(February), 23–34.
- Shim, C. (2017), Policy Measures for Mitigating Fine Particle Pollution in Korea and Suggestions for Expediting International Dialogue in East Asia: JICA Research Institute Working Paper Series, (150).
- Sumitomo Mitsui Advanced Finance for Ecology (2013), An Ill Wind Blows: The Current State and Impact of Cross-border PM 2.5: SAFE Corporate Environmental Magazine Feature Article on PM_{2.5} Part 1.
- Tao, M., L. Chen, L. Su, and J. Tao (2012), "Satellite Observation of Regional Haze Pollution over the North China Plain," *Journal of Geophysical Research Atmospheres*, 117(12), 1–16.
- The International Council on Clean Transportation and DieselNet (2018), PM Standard by Country, Retrieved from <https://www.transportpolicy.net/>
- van Donkelaar, A., R. V. Martin, M. Brauer, R. Kahn, R. Levy, C. Verduzco, and P. J. Villeneuve (2010), "Global Estimates of Ambient Fine Particulate Matter Concentrations from Satellite-based Aerosol Optical Depth: Development and Application," *Environmental Health Perspectives*, 118(6), 847–855.
- WorldBank and Institute for Health Metrics and Evaluation (2016), The Cost of Air Pollution, <https://doi.org/10.1787/9789264210448-en>
- Yale Center for Environmental Law & Policy and The Center for International Earth Science Information Network at Columbia University's Earth Institute (2018), 2018 Environmental Performance Index, Retrieved from <https://epi.envirocenter.yale.edu/epi-topline>
- Zhang, Q., X. Jiang, D. Tong, S. J. Davis, H. Zhao, G. Geng, and D. Guan (2017), "Transboundary Health Impacts of Transported Global Air Pollution and International Trade," *Nature*, 543, 705–709.