Characterizing multi-pollutant air pollution in China: Comparison of three air quality indices

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A B S T R A C T

Multi-pollutant air pollution (i.e., several pollutants reaching very high concentrations simultaneously) frequently occurs in many regions across China. Air quality index (AQI) is used worldwide to inform the public about levels of air pollution and associated health risks. The current AQI approach used in China is based on the maximum value of individual pollutants, and does not consider the combined health effects of exposure to multiple pollutants. In this study, two novel alternative indices – aggregate air quality index (AAQI) and health-risk based air quality index (HAQI) – were calculated based on data collected in six megacities of China (Beijing, Shanghai, Guangzhou, Shijiazhuang, Xi'an, and Wuhan) during 2013 to 2014. Both AAQI and HAQI take into account the combined health effects of various pollutants, and the HAQI considers the exposure (or concentration)-response relationships of pollutants. AAQI and HAQI were compared to AQI to examine the effectiveness of the current AQI in characterizing multi-pollutant air pollution in China. The AAQI and HAQI values are higher than the AQI on days when two or more pollutants simultaneously exceed the Chinese Ambient Air Quality Standards (CAAQS) 24-hour Grade II standards. The results of the comparison of the classification of risk categories based on the three indices indicate that the current AQI approach underestimates the severity of health risk associated with exposure to multi-pollutant air pollution. For the AQI-based risk category of ‘unhealthy’, 96% and 80% of the days would be ‘very unhealthy’ or ‘hazardous’ if based on AAQI and HAQI, respectively; and for the AQI-based risk category of ‘very unhealthy’, 67% and 75% of the days would be ‘hazardous’ if based on AAQI and HAQI, respectively. The results suggest that the general public, especially sensitive population groups such as children and the elderly, should take more stringent actions than those currently suggested based on the AQI approach during high air pollution events. Sensitivity studies were conducted to examine the assumptions used in the AAQI and HAQI approaches. Results show that AAQI is sensitive to the choice of pollutant irrelevant constant. HAQI is sensitive to the choice of both threshold values and pollutants included in total risk calculation.

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1. Introduction

China has been experiencing serious air pollution problems in recent decades due to the rapid industrialization and urbanization, and increasing energy consumption. Numerous studies have demonstrated associations between air pollution and various health effects (e.g., Brunkreuff and Forsberg, 2005; Burnett et al., 2000; Dockery, 2001; Dominici et al., 2005; Le Tertre et al., 2002). Recognizing the severity of the air pollution situation and the dense population in China, Chinese scientists have recently started air pollution-health effects studies (e.g., Cao et al., 2012; Chen, 2007; Chen et al., 2013; Guo et al., 2013; Kan and Chen, 2002; Kan and Gu, 2011). The results of these studies confirmed that air pollution threatens public health in China. Therefore, it is important to inform the public about the levels of air pollution and associated health risks so that people can take measures to protect their health.

Air pollution levels are determined by the concentrations of a complex mixture of air pollutants. Currently SO$_2$, NO$_2$, CO, O$_3$, PM$_{12.5}$, and PM$_{2.5}$ are defined as the six criteria pollutants around the world in quantifying air pollution levels. The concentrations among the pollutants can be different by orders of magnitude, and their unit-concentration health effects are significantly different as well. Therefore, it is difficult for the general public to use the concentrations directly to characterize the levels of air pollution. Alternatively, the use of an index ranging from good to unhealthy is more understandable.
for the general public, and becomes the most common way to interpret air pollution levels in many countries and regions (Shooter and Brimblecombe, 2005). Such indices were firstly developed in early 1970’s (Babcock, 1970), and have been evolving since then. Currently, Air Quality Index (AQI) is the most used index worldwide.

The most commonly used index is the United States Environmental Protection Agency (US EPA) AQI. The AQI ranges from 0 to 500 and is calculated based on the concentrations of the six criteria pollutants. For a given location on a given day, a sub-AQI for every pollutant is calculated, and the maximum of all the sub-AQIs is defined as the overall AQI. It indicates “unhealthy” air quality when AQI is greater than 100. This method has been criticized because it does not appropriately represent the combined health effects of exposure to multiple pollutants. A few studies have been conducted to develop alternative approaches that take into account the combined health effects of various pollutants. Swamee and Tyagi (1999) proposed to combine the sub-AQIs to form an Aggregate AQI (AAQI). Kyrkilis et al. (2007) followed this idea and developed an AAQI for the area of Athens, Greece. The results indicate that the AAQI estimates the air pollution exposure more effectively than the US EPA AQI. Another novel alternative AQI approach is based on the health risk associated with exposure to multiple air pollutants (Cairncross et al., 2007; Sicard et al., 2012; Stieb et al., 2008; Wong et al., 2013). The health-risk based AQI (HAQI) considers the established exposure (or concentration)-response relationships. HAQI has shown improvement over the existing AQIs in various countries and regions. Currently, Canada is using the Air Quality Health Index approach that is based on the approach proposed by Stieb et al. (2008).

The Chinese Ministry of Environmental Protection (MEP) adopted the US EPA AQI approach and developed the Chinese AQI system in 2012 (MEP, 2012a). Wong et al. (2013) developed a HAQI for Hong Kong, but no studies have been conducted to investigate the differences between the AQI, AAQI, or HAQI approaches in designating air pollution severities in mainland China. In a recent study, the spatial and temporal distributions of six criteria pollutants in China were revealed (Wang et al., 2014), making it possible for the first time to investigate how effective different AQI approaches in characterizing the severity of multiple-pollutant air pollution in China. In this study, AAQI was defined based on the sub-AQI of individual pollutant and HAQI was developed using the exposure-response relationships from the health studies in China. Both HAQI and AAQI were compared with the MEP’s AQI in different seasons and locations. Health implications of the three indices under different pollution situation were also investigated. Hypotheses in each approach were discussed and future research needs to establish an optimal index that can provide the general public with the accurate information for the purpose of health protection were highlighted.

2. Methods

2.1. Study areas and data source

Six representative cities (Beijing, Shanghai, Guangzhou, Shijiazhuang, Xi’an, and Wuhan) were selected for comparative analyses of AQI, AAQI, and HAQI in China. The locations of the six cities are shown in Fig. 1. Beijing, Shanghai, and Guangzhou are located in the North China Plain, the Yangtze River Delta, and the Pearl River Delta, respectively. These regions have become hot spots of air pollution studies in China due to dense population, more developed economy, and frequent pollution events. A recent study investigating the air pollution status in China during 2013–2014 revealed that Shijiazhuang, Wuhan, and Xi’an are the most heavily polluted provincial capital cities in the North, South-East, and West China, respectively (Wang et al., 2014). The six cities located...
in different regions exhibit significant differences in pollution types and levels (Table 1). Therefore, analyses of different AQI approaches using data from the six cities could provide an overall understanding on the effectiveness of different AQSs to characterize the air pollution exposures in China.

Hourly concentrations of SO₂, NO₂, PM_{2.5}, PM_{10}, CO, and O₃ in the six cities of interest were downloaded from the publishing website of China National Environmental Monitoring Center (http://113.108.142.147:20035/emcpublish/). One-year data from March 1st, 2013 to February 28th, 2014 are included in this study. This dataset has been used in previous studies (Hu et al., 2014; Wang et al., 2014; Zhang et al., 2015) to investigate the temporal and spatial variations in the gaseous and particulate pollutants in China. Detailed monitoring techniques have been provided in these studies, therefore, only a brief description of the dataset is provided here. All the measurements were conducted at the national air quality monitoring sites located in each city. At each site, automated monitoring systems have been installed and used to measure the ambient concentration of SO₂, NO₂, O₃, CO, PM_{2.5}, and PM_{10} according to China Environmental Protection Standards HJ 193–2013 (http://www.es.org.cn/download/2013/7-12/2627-1.pdf), and HJ 655–2013 (http://www.es.org.cn/download/2013/7-12/2626-1.pdf). The values from the monitoring sites at each city were automatically reported to the China National Environmental Monitoring Center and published after being validated based on Technical Guideline on Environmental Monitoring Quality Management HJ 630 and published after being validated based on Technical Guideline on Environmental Monitoring Quality Management HJ 630–2011 (http://kjs.mep.gov.cn/hjbhbz/bzwb/other/qq/201109/W020120130585014685198.pdf). The citywide average concentrations were calculated by averaging the concentrations at all sites in each city. This is the same method that the Chinese MEP uses to report daily concentrations of air pollutants to the public.

2.2. Air quality indices

2.2.1. MEP’s AQI (or AQI)

The MEP’s AQI approach is based on the Chinese Ambient Air Quality Standards (CAAQS). The sub-AQI of each pollutant is calculated using Eq. (1) with the monitored pollutant concentrations:

\[
\text{AQI}_i = \frac{(C_i - C_{i,m})}{C_{i,1} - C_{i,1}} \times (C_{i,m} - C_{i,j-1}) + \text{AQI}_{i,j-1} \text{, } j > 1
\]

where AQI is the index of pollutant i; C_{i,m} is the monitored ambient concentration of pollutant i; j is the health category index so that C_{i,j-1} and C_{i,j} are the upper limit concentrations for the jth and j-1th health categories, encompass C_{i,m}; AQI_{j-1} is the upper limit concentration for C_{i,j-1}. Table 2 gives the health categories and the corresponding ranges of AQI values and pollutant concentrations. The overall AQI is the maximum of the sub-AQI of all pollutants, as shown in Eq. (2):

\[
\text{AQI} = \max\{\text{AQI}_1, \text{AQI}_2, \ldots, \text{AQI}_n\}, n = 1, 2, \ldots, 6.
\]

The AQI approach converts the pollutant concentrations to a number on a scale of 0 to 500, with a sub-AQI of 100 corresponding to the upper limit of the CAAQS 24-hour Grade II standards (MEP, 2012b). The overall AQI represents the overall health risk of the air pollution.

### Table 1

<table>
<thead>
<tr>
<th>City</th>
<th>Population (M)</th>
<th>Death rate</th>
<th>PM_{2.5} (μg/m³)</th>
<th>PM_{10} (μg/m³)</th>
<th>SO₂ (μg/m³)</th>
<th>NO₂ (μg/m³)</th>
<th>O₃ (μg/m³)</th>
<th>CO (μg/m³)</th>
<th>Pollution days</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beijing (BJ)</td>
<td>12.97</td>
<td>4.31</td>
<td>87 ± 67</td>
<td>109 ± 62</td>
<td>26 ± 23</td>
<td>51 ± 23</td>
<td>96 ± 58</td>
<td>1.3 ± 0.8</td>
<td>105/182/345</td>
</tr>
<tr>
<td>Shanghai (SH)</td>
<td>14.27</td>
<td>5.24</td>
<td>56 ± 41</td>
<td>80 ± 47</td>
<td>20 ± 14</td>
<td>41 ± 18</td>
<td>103 ± 45</td>
<td>0.9 ± 0.3</td>
<td>49/100/342</td>
</tr>
<tr>
<td>Guangzhou (GZ)</td>
<td>8.22</td>
<td>6.17</td>
<td>52 ± 28</td>
<td>72 ± 35</td>
<td>20 ± 9</td>
<td>49 ± 21</td>
<td>96 ± 51</td>
<td>1 ± 0.2</td>
<td>43/89/329</td>
</tr>
<tr>
<td>Shijiazhuang (SZ)</td>
<td>10.05</td>
<td>6.73</td>
<td>144 ± 109</td>
<td>287 ± 139</td>
<td>89 ± 60</td>
<td>64 ± 27</td>
<td>99 ± 62</td>
<td>1.8 ± 1.3</td>
<td>224/261/303</td>
</tr>
<tr>
<td>Xi’an (XA)</td>
<td>7.96</td>
<td>4.69</td>
<td>97 ± 83</td>
<td>181 ± 127</td>
<td>40 ± 29</td>
<td>51 ± 18</td>
<td>79 ± 43</td>
<td>2 ± 0.8</td>
<td>127/185/315</td>
</tr>
<tr>
<td>Wuhan (WH)</td>
<td>8.22</td>
<td>5.54</td>
<td>92 ± 70</td>
<td>135 ± 75</td>
<td>43 ± 20</td>
<td>53 ± 25</td>
<td>114 ± 54</td>
<td>1.3 ± 0.5</td>
<td>117/193/306</td>
</tr>
</tbody>
</table>

* Population is the permanent resident population; the temporary resident population is not included.

* O₃ standards are daily maximum 8-hour standards.

* Standards are 24-hour Grade III standards. The Grade II standards correspond to an AQI value of 100.

* The first number represents pollution days with at least two pollutant exceeding Grade II standards, the second number represents pollution days with at least one pollutant exceeding the Grade II standards, and the third number represents total days with valid air quality monitoring data from 03/01/2013 to 02/28/2014.

### Table 2

<table>
<thead>
<tr>
<th>Pollutant concentrations (μg/m³)</th>
<th>CAAQS standards</th>
<th>MEP’s AQI (or AQI)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO</td>
<td>500</td>
<td>750</td>
</tr>
<tr>
<td>NO₂</td>
<td>50</td>
<td>40</td>
</tr>
<tr>
<td>PM_{2.5}</td>
<td>150</td>
<td>75</td>
</tr>
<tr>
<td>PM_{10}</td>
<td>150</td>
<td>75</td>
</tr>
<tr>
<td>SO₂</td>
<td>100</td>
<td>80</td>
</tr>
<tr>
<td>O₃</td>
<td>200</td>
<td>150</td>
</tr>
<tr>
<td>CO</td>
<td>300</td>
<td>250</td>
</tr>
<tr>
<td>O₃</td>
<td>400</td>
<td>350</td>
</tr>
<tr>
<td>PM_{2.5}</td>
<td>500</td>
<td>600</td>
</tr>
</tbody>
</table>

The AQI below SO₂, NO₂, CO, O₃, and PM_{2.5} are calculated using the MEP’s AQI approach. The sub-AQI of each pollutant is calculated using Eq. (1) with the monitored pollutant concentrations.
In a general form, relative risk (RR) for each pollutant is estimated based on the Cairncross’s concept (Cairncross et al., 2007) can be used. Based on AQI, the total excess risk (ER) of exposure to multiple pollutants exposure-response characteristics (Cairncross et al., 2007; Sicard et al., 2009). Therefore, AAQI (and HAQI) values are capped at 500 when they exceed 500.

2.2.3. Health-risk based AQI (HAQI)

While the AAQI attempts to account for the combined health effects of multiple air pollutants, it does not explicitly account for the established exposure-response relationships of the pollutants. Health-risk based indices were proposed in a few studies to include such exposure-response characteristics (Cairncross et al., 2007; Sicard et al., 2009). To define a health-risk based AQI, the total excess risk (ER) of exposure to multiple pollutants based on the Cairncross’s concept (Cairncross et al., 2007) can be used. In a general form, relative risk (RR) for each pollutant is estimated based on health effect studies, using Eq. (4):

\[ RR = \exp(\beta(C - C_0)), \quad C > C_0 \]

where \( \beta \) is the exposure-response relationship coefficient, representing the excess risk of health effect (such as mortality) per unit increase of pollutant (such as 1 mg/m\(^3\) of PM\(_{2.5}\)); \( C \) is the concentration of pollutant, and \( C_0 \) is the threshold concentration, below which it is believed the pollutant demonstrates no obvious adverse health effects (i.e. \( RR = 1 \)). For this study, \( \beta \) values were chosen from a systematic review of Chinese studies on short-term exposure to air pollution and daily mortality of all ages (Shang et al., 2013). The \( \beta \) values are 0.38\%, 0.32\%, 0.81\%, 1.30\%, and 0.48\% per 10 mg/m\(^3\) increase of PM\(_{2.5}\), PM\(_{10}\), SO\(_2\), NO\(_2\), and O\(_3\), respectively, and the \( \beta \) value is 3.7\% per 1 mg/m\(^3\) increase of CO.

It is presumed that the air pollution poses little or no health risk when concentrations of pollutants are below the Grade II standards. Therefore, the upper limits of CAAQS 24-hour Grade II standard for the six pollutants were used as the \( C_0 \) values as a first estimation. However, some studies indicated these values cannot be regarded as threshold values below which a zero adverse response may be expected (WHO, 2005), thus two sensitivity analyses were conducted by changing \( C_0 \) values to 1) the upper limits of CAAQS 24-hour Grade I standards and 2) no threshold values (i.e., \( C_0 = 0 \)) except CO. The results of the sensitivity analyses are included in Results and discussion.

ER of each pollutant, which is defined as RR-1, is summed up to calculate the total excess risk (ER\(_{\text{total}}\)) for simultaneous exposure to several air pollutants:

\[ ER_{\text{total}} = \sum_{i=1}^{n} ER_i = \sum_{i=1}^{n} (RR_i - 1). \tag{5} \]

Therefore, higher values of ER\(_{\text{total}}\) reflect higher health risks. Fig. 2 shows the average ERs of the six pollutants in the six cities using the chosen \( \beta \) values, the measured daily concentrations, and the \( C_0 \) values of the CAQS 24 h Grade II standards. ER\(_{\text{total}}\) varies significantly among the six cities, with the minimum of 0.6% in Guangzhou and the maximum of 9.9% in Shijiazhuang. PM\(_{2.5}\) contributes to ER\(_{\text{total}}\) the most in Beijing, Shanghai, and Wuhan; PM\(_{10}\) contributes to ER\(_{\text{total}}\) the most in Shijiazhuang and Xi’an, followed by PM\(_{2.5}\); O\(_3\) and NO\(_2\) also have certain contributions, especially in Guangzhou. O\(_3\), NO\(_2\) and PM\(_{2.5}\) have similar ER values. Note linearly adding up the pollutants’ risks could over-estimate ER\(_{\text{total}}\) if certain pollutants are highly correlated, such as PM\(_{2.5}\) and PM\(_{10}\). Therefore, adding the six pollutants’ ER is an upper bound estimation of ER\(_{\text{total}}\). Analysis was conducted to estimate the ER\(_{\text{total}}\) by adding ERs of PM\(_{2.5}\), O\(_3\), and NO\(_2\) in Results and Discussion.

Previous studies converted ER\(_{\text{total}}\) to an arbitrary index, ranging from 0–10, for warning the public about the air pollution risks (Cairncross et al., 2007). Such an index cannot be directly compared to AAQI and HAQI as they are on different scales. In this study, the HAQI with the same scale as AQI and AAQI (0–500), is introduced based on ER\(_{\text{total}}\) and the equivalent pollutant concentrations. The equivalent concentration (\( C^* \)) of a criteria pollutant is defined as the concentration level at which the ER of the pollutant equals to ER\(_{\text{total}}\), as shown in Eq. (6):

\[ RR^* = ER_{\text{total}} + 1 = \exp(\beta(C^* - C_0))] \tag{6} \]

where RR\(^*\) represents the relative risk calculated based on the equivalent pollutant concentration. Using the RR\(^*\) value determined from ER\(_{\text{total}}\) using Eq. (5), the equivalent pollutant concentration of the ith criteria pollutant (\( C_i^* \)) can be determined using Eq. (7):

\[ C_i^* = \ln(RR^*)/\beta_i + C_{0,i} \tag{7} \]

where \( \beta_i \) and \( C_{0,i} \) are the \( \beta \) and \( C_0 \) values of the ith pollutant, respectively. Because RR\(^*\) is the total risk of exposing to multi-pollutants, the...
equivalent concentration represents the combined health risk of all criteria pollutants and is naturally higher than the actual concentration of the pollutant. The HAQI based on the equivalent concentration of the ith criteria pollutant (sub-HAQI, or HAQIi) can then be determined using $C_i^*$ as shown in Eq. (8), instead of the actual concentration $C_i$ in Eq. (1).

$$\text{HAQI}_i = \frac{(\text{AQI}_{i,j} - \text{AQI}_{i,j-1})}{(C_{i,j} - C_{i,j-1})} + \frac{\text{AQI}_{i,j-1}}{j \geq 1} C_0/C_1$$

The overall HAQI is determined as the maximum of all sub-HAQIs, as shown in Eq. (9)

$$\text{HAQI} = \max(\text{HAQI}_1, \text{HAQI}_2, ..., \text{HAQI}_n), \ n = 1, 2, ..., 6. \quad (9)$$

HAQI includes the information of exposure to multi-pollutants and considers the exposure-response relationships of different pollutants. HAQI is a function of $C_0$ and ERtotal. Its value changes if a different set of $C_0$ values is chosen or different pollutants are used to calculate the ERtotal. Impact of different choices of these parameters on HAQI is further discussed in Results and discussion.

Note that although Eq. (9) appears to be similar to Eq. (2), they are very different from the health risk perspective. Eq. (2) implies that the overall health risk of exposure to criteria pollutants is determined only by the pollutant that has the highest AQI value. However, each sub-HAQI in Eq. (9) already considers the relative risk due to multiple pollutants. Choosing the maximum sub-HAQI as the overall HAQI is a more cautious way of estimating the multi-pollutant health risk, considering the fact that the indices are going to be used to warn the general public, especially the sensitive population, of potential air pollution exposure.

It should also be noted that the AAQI and HAQI methods assume that effects of individual pollutants are additive. However, non-additive

Fig. 3. Comparison of AAQI and HAQI to AQI in the six cities on unhealthy days (i.e., AQI > 100). The AAQI and HAQI are designed to have the same scale as AQI, i.e. 0–500. Therefore, AAQI and HAQI are capped at 500 when their calculated values exceed 500.
interactions (such as synergism or antagonism) among pollutants might occur in reality (Mauderly et al., 2010). In the past years, various methods, such as the statistical interaction approach, the indicator approach, the source apportionment techniques and risk-based metrics, have been developed to estimate total health risk of multi-pollutant exposure (Dominici et al., 2010; Oakes et al., 2014). Despite the progress in characterizing multi-pollutant exposure metrics, the actual interaction among pollutants remains unclear and additional research is needed to quantify the combined effects of multiple pollutants.

3. Results and discussion

Fig. 3 shows the comparison of AAQI and HAQI to AQI on unhealthy days (i.e., when AQI > 100). In all cities, AAQI is always higher than AQI. HAQI equals to AQI on healthy days (i.e., when AQI ≤ 100) and on single-pollutant unhealthy days (i.e., when AQI > 100 and only one pollutant exceeds its CAAQS 24-hour Grade II standard). HAQI is higher than AQI on multi-pollutant unhealthy days (i.e., when AQI > 100 and more than one pollutants exceed the CAAQS 24-hour Grade II standards). Table 1 shows that of all the unhealthy days during 2013–2014, 58%, 49%, 48%, 86%, 69%, and 61% of the days are with at least two pollutants simultaneously exceeding the CAAQS 24-hour Grade II standards in Beijing, Shanghavi, Guangzhou, Shijiazhuang, Xi’an and Wuhan, respectively. The fact that AQI is lower than both AAQI and HAQI on these days indicates that the current AQI approach likely under-reports the severity of the health risk associated with exposure to multi-pollutant air pollution.

On unhealthy days, AAQI is highly correlated with AQI in all cities, with correlation coefficients ($R^2$) in the range of 0.73 (Guangzhou) to 0.93 (Shijiazhuang). HAQI has weaker correlation with AQI, compared to AAQI. $R^2$ of HAQI to AQI is from 0.69 (Guangzhou) to 0.85 (Beijing). Fig. 3 shows that AAQI is generally greater than HAQI on the “unhealthy for sensitive groups” days (i.e., 100 < AQI ≤ 150), but HAQI generally becomes greater on the “very unhealthy” days (i.e., when AQI ≥ 200). The results suggest that the AAQI approach, without considering the actual pollutants’ exposure-response relationships, could still underestimate the health risks on these high pollution days, even though it takes into account the combined effects of all pollutants.

Air quality is divided into six health risk categories based on the AQI value and certain actions are advised at the different health risk levels to protect health (Table 1). Misclassification of air pollution levels will result in inappropriate protection recommendations to the public and consequently cause potential health problems. Fig. 4 shows the total number of days (sum of the six cities) of different risk categories based on AAQI and HAQI ratios. The standard deviations of AAQI/AQI ratios range from 1.46 (Beijing) to 1.63 (Guangzhou). The difference in the mean AAQI/AQI ratios among the six cities is much less than in HAQI/AQI ratios. The standard deviations of AAQI/AQI ratios range from 0.16 (Shijiazhuang) to 0.24 (Wuhan), showing small variations among the cities. However, the standard deviations of HAQI/AQI ratios show much greater variations, ranging from 0.13 (Shanghai) to 0.38 (Shijiazhuang). The results reveal large discrepancy resulted from different approaches to characterize the multi-pollutant air pollution, and merit further investigation to identify a more accurate index.

The pollutant irrelevant constant $\rho$ is the key parameter in AAQI calculation. Previous studies suggested values of $\rho$ should be between 2 and 3 (Khanna, 2000; Swamee and Tyagi, 1999), but the most proper value for $\rho$ is still under debate. A $\rho$ value of 2.0 was firstly chosen in the AAQI calculation and sensitivity study was further conducted to examine the effect of $\rho$ on the AAQI results by choosing $\rho$ values of 1.5, 2.5, and 3.0. Fig. 5(b) shows the comparison of mean and standard deviation of AAQI/AQI ratios with the four different $\rho$ values. The results indicate that the choice of the $\rho$ values greatly influences the AAQI values. AAQI decreases with the increase of the $\rho$ value. The AAQI/AQI ratios show small variations among the six cities, ranging from 1.85 to 2.1 with $\rho = 1.5$, 1.46 to 1.63 with $\rho = 2.0$, 1.30 to 1.42 with $\rho = 2.5$, and 1.21 to 1.30 with $\rho = 3.0$, despite the quite different pollution status in the six cities. This result indicates that AAQI is not sufficient to distinguish the risks due to different air pollution conditions.

HAQI calculation is dependent on $C_0$ and $ER_{total}$. The CAAQS 24-hour Grade II standards were used firstly as $C_0$. However, some studies indicated that guideline values cannot be regarded as threshold values below which a zero adverse response may be expected (Bell et al., 2006; Gent et al., 2003). Some studies indicated that $PM_{10}$, $PM_{2.5}$, $O_3$, $SO_2$ and $NO_2$ may not have apparent threshold values below which the risk of adverse health effect is zero (WHO, 2005). Therefore, two
The HAQI approach developed in this study is sensitive to the parameter of C0 and approach to calculating ER\text{total}, with the main differences occurring in Shijiazhuang and Xi'an. Wang et al. (2014) revealed significant fraction of coarse PM in China, especially in the western and northern China. Therefore, when estimating the total risk, PM10 or at least the coarse PM particles should be considered in the health risk estimation, in addition to the PM2.5 risk.

The comparison indicates that during multi-pollutant air pollution events, when concentrations of several pollutants are simultaneously high and cause risk to public health, the current AQI approach underestimates the health risk. A combined index considering multi-pollutant exposure provides a more accurate description of the air pollution health risk in China than the current AQI. The findings in this study have important health implications. Results of AAQI and HAQI both indicate that the health risk is mostly ‘at least one category’ higher than what the current AQI suggests on days of AQI greater than 150. Based on this finding, the public, especially the sensitive groups (children, older adults, people with lung and heart diseases), should take more stringent actions to avoid the adverse effects of air pollution. This is especially true for heavily polluted cities and in the winter season when the pollution level is higher.

Even though the HAQI approach developed in this study is sensitive to the parameter of C0 and approach to calculating ER\text{total}, the main finding is that health risk of exposure to multi-pollutant pollution is likely higher than what the current AQI suggests still holds true. The HAQI developed in this study is not aimed to replace the current AQI, but can serve as an alternative approach. HAQI considers the specific exposure-response relationships of multi-pollutants and therefore delivers more accurate pollution risk information. To identify an optimal HAQI approach that best captures the exposure-health associations in China, further investigation is needed on several aspects. As previously discussed, studies on the pollutants’ threshold values and the total risk of multi-pollutant exposure are needed. Studies on associations between HAQI and observed health effects are also needed. Some
studies found that HAQI (in different forms from the HAQI in this study) is significantly associated with observed health effects (Hong et al., 1999; To et al., 2013). Such studies should be conducted in all of China to examine the effectiveness of any proposed AQI approach in characterizing air pollution risks.

4. Conclusions

The AAQI and HAQI are higher than AQI on days when two or more pollutants simultaneously exceed the CAAQS 24-hour Grade II standards. The health risk categories are commonly used to inform the general public about the air pollution levels. The comparison result suggests that for the AQI-based risk category of ‘unhealthy’, 96% and 80% of the days would be ‘very unhealthy’ or ‘hazardous’ if based on the AAQI and HAQI, respectively; and for the AQI-based risk category of ‘very unhealthy’, 67% and 75% of the days would be ‘hazardous’ if based on the AAQI and HAQI, respectively. Therefore, the current AQI approach underestimates the severity of the health risk associated with the exposure to multi-pollutant air pollution. During high air pollution events, the health risk of air pollution is more serious relative with the exposure to multi-pollutant air pollution. During high air pollution events, the health risk of air pollution is more serious relative with the exposure to multi-pollutant air pollution.

sensitivity studies show that the AAQI is affected by the choice of pollutant irrelevant constant \(p\) and the HAQI is sensitive to the choice of the threshold values and the pollutants included in the total risk calculation. Future studies should focus on the investigation of the pollutants’ threshold values, the total risk of multi-pollutant exposure, and associations between the indices and observed health effects.

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References


