Research on the relationship between urban form and urban smog in China

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Abstract: The present study aims at exploring whether aspects of urban form (compactness ratio and elongation ratio) are associated with urban smog (particulate matter) in China. Quantitative indicators relating to urban form and urban smog were selected and quantified for 30 Chinese cities, and the reference years 2000, 2007 and 2010, by using a combination of compiled statistical data, remote sensing and geographical information system data. Panel data analysis was used to evaluate the degree of association between measures of urban form and urban smog, while controlling for urban population, built-up area green coverage rate, power consumption, SO_2 emissions, gross value of industrial output, gross industrial output and buses per capita. The results indicate that urban compactness and urban elongation were positively correlated to urban particulate matter. It is therefore recommend to consider the implication of urban form on smog as part as urban planning, as part of ongoing strategies to mitigate the deleterious consequences of air-pollution.

Keywords: urban smog; urban compactness; urban elongation ratio; passive urbanization

1 Introduction

Sources of air pollution are numerous. Air pollutants can originate from natural and anthropogenic sources (Boubel, 1994), and be classified as primary or secondary pollutants (Kibble and Harrison, 2005). Primary pollutants are those released directly from specific sources of pollution, for example, combustion particles from coal-fired power plants and motor vehicles, or mineral dust from desert wind storms. Once emitted into the atmosphere, some of these primary pollutants could be altered by light energy, heat or the presence of other chemicals to form secondary pollutants.

Despite these many sources, evidence is mounting that air pollution has some kind of relationship with urban form (Rydell et al., 1968; Bereitschaft and Debbage, 2013). For example, urban form is related to a reduction in wind speed and increase in temperature relative to the surrounding rural areas, which can cause a gradient in pollutants from the hot, less windy city-centers to the cooler, windy edges, resulting in "dust domes" or "haze hoods” (William, 1967). Bereitschaft and Debbage (2013) found in a study of 86 U.S. metropolitan areas that increased urban sprawl was associated with increased air pollution, when controlling for climate, land area and population. In China, air pollution is a particular public health concern, and has been linked to an estimated 1.2 million premature deaths in 2010 alone (Scott, 2013). The dense haze surrounding many of China’s northern cities has caused reductions in visibility (Wang et al., 2006). Current particle pollution levels are well above international guidelines. For example, PM2.5 levels (indicating air particles smaller than 2.5 µm in diameter) in Beijing 2009 – 2013 were on average 135 ± 63 µg m^{-3}, with a maximum of 355 µg m^{-3} (Zhang et al 2013), which is over 13 times the World Health Organization's recommended standard for annual averages (10 µg m-3) (WHO 2006)[1] and 4 times over the Chinese annual-average air quality limit (35 µg m^{-3}). Municipal and regional authorities throughout China have paid great attention to urban sustainable development and encouraged innovative policies aimed to reduce urban smog, but until recently there has been a lack of empirical investigation on the

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[1] http://whqlibdoc.who.int/hq/2006/WHO_SDE_PHE_OEH_06.02_eng.pdf

relationship between urban smog and urban form, particularly in China. Thus the aim of the present research is to explore this issue in China by using indicators for urban form and urban smog. The results provide a promising basis for policy-making to promote urban air pollution mitigation through urban planning.

2 Background

Marquez and Smith (1999) described an initial attempt to develop a framework evaluating the effect of urban form on air quality by integrating land use, transport and airshed models. Simulation results of Borrego et al (2006) indicated that more compact cities with mixed land use provide better air quality compared to disperse and network cities. Recently, Martins (2012) presented that urban sprawl showed an aggravation of annual average PM10 values (air particles smaller than 10 μm in diameter), with increases in urban sprawl increasing the frequency that daily limit values in Porto are exceeded. Simulations by De Ridder et al (2008) also indicated that simulated pollutant concentrations of ozone and particulate matter increased with urban sprawl. Similarly, Stone (2008) reported that large US metropolitan regions ranking highly on a quantitative index of sprawl experienced a greater number of ozone exceedances than more spatially compact metropolitan regions. On the other hand, developing corridor cities (linear corridors emanating from the central area with upgraded public transport) was suggested as a way to mitigate air pollution, compared to allowing radial urban sprawl (Manin et al., 1998). Another urban form aspect related to urban smog is population density, with decreased density being correlated with increased ozone production (Stone 2008) in addition to increased transportation distance per capita.

Like other air pollutants, urban aerosols can be of primary or secondary origin. Primary aerosols can originate from combustion engines, industrial emissions, blowing desert or soil particles, biological organic matter; secondary aerosols are formed in the atmosphere from volatile precursors, like SO2, NH3, NOx, and secondary organic aerosols (SOA) formed from volatile organic carbon (VOC) precursors, many of which can originate from combustion and industrial emissions.

With some exceptions, primary aerosols are generally > PM 2.5, and secondary aerosols are < PM 2.5 (Seinfeld and Pandis, 2006). There are many potential reasons for aerosol abundance and aerosol composition that can be related to seasons as well as the local climate and location within China (Chan and Yao, 2008).Generally, there is increased anthropogenic particle emissions in winter due to heating, though in Beijing and surrounding cities, spring is generally associated with the highest levels of air pollution due to dust storm events (Senlin et al, 2007; Zhang et al, 2013). Increased winds and rain from monsoon seasons can also decrease aerosol concentrations in Southern China (Wai and Tanner, 2005[a]; Wai and Tanner, 2005[b]). Recent studies in Beijing, Tianjin and Hebei area have found PM 2.5 levels in all seasons are on average not substantially different, with average levels ranging from 70 - 200 µg/m3 depending on sampling station; however, the most substantially polluted days tend to be in winter and spring, likely due to increased coal burning and dust storm events (Zhao et al 2013, Zhang et al 2013). In earlier studies of PM10 over all of China (Song et al 2009), as well as in studies of PM2.5 and PM 10 in the Pearl River Delta (Cao et al 1994), levels were generally higher in winter, and tended be lower in the Pearl River Delta and Central Southern China than in the Northern China (Song et al. 2009).

3 Methodologies

Parameters related to urban smog and form were compiled for 30 Chinese cities from a combination of data in Chinese statistical yearbooks as well as remote sensing and geographical information system data. As urbanization has changed rampantly in these cities in the past decades, three time points were selected for the compilation of parameters: the years 2000, 2007 and 2010. It would have been beneficial to include more cities to improve the power of the statistics, unfortunately, the Chinese statistical yearbooks generally provided the full data for these 30 cities (consisting of central municipalities and provincial capitals). These 30 cities are all industrial areas with a high population density, and prone to air pollution.

3.1 Urban smog

The parameters selected to represent urban smog is PM10. Unfortunately, PM10 for the 30 cities were only available for 2007 and 2010 in the Chinese statistical yearbooks. Therefore for the year 2000, Aerosol Optical Depth (AOD) was chosen instead. AOD is a satellite-based metric that is compiled on a regional scale. It is defined as the degree to which aerosols prevent the transmission of light by absorption or scattering. The relationship between AOD and average regional PM measurements have been explored extensively in the literature, with no real consensus being formed on how two relate these two parameters globally, though on the local, region specific scale, correlations can be made (Schaap et al 2009, Song et al. 2009). AOD was calculated using the following method. The Moderate Resolution Imaging Spectroradiometer (MODIS), onboard the Earth Observing System (EOS) Terra (EOS AM) and Aqua (EOS PM) polar-orbiting satellites, was launched in 1999 and 2002, respectively. MODIS is designed to have a global coverage every 1 to 2 days and has 36 spectral bands. MODIS aerosol product (MOD04/MCD04) can provide daily ambient AOD over ocean and land. The Level 2 product of MOD04/MCD04 has a spatial resolution of 10×10 km at nadir. The AOD retrieved at 0.550 μm was used in this study. MOD04_L2 daily data in 2000 was used for the computation of the annual average AOD over 30 major cities in China. However, It is noted that due to the influence of weather and intrinsic limitations of the aerosol retrieval algorithm used in MODIS aerosol products in land, MODIS aerosol products often have missing values (Gupta, Patadia et al. 2008). In order to estimate average AOD over a specific city, we first selected the “valid” pixels, defined as those which provide AOD value numbers more than 100 times in one year, and calculated the average AOD values for these pixels.

3.2 Urban form

Unlike air quality, there are no recognized indicators for measuring urban form. Urban form indicators in use are open to widely differing interpretations, and are generally tailored for the aims of a specific study. As some examples, Huang, Lu, and Sellers (2007) employed five urban form indicators for a global comparative study, namely compactness, centrality, complexity, porosity and density. McMillan (2007) used domestic land-use and pedestrian access as urban form indicators to be related to perceived traffic safety and actual traffic safety, which partly echoed the indicators used by Song (2005). The majority of studies that parameterize urban form tend to focus on compactness and sprawl (Wentz, 2000; Tsai, 2005; Colaninno, Roca, Pfeffer, 2011), with geometric measures of elongation and compactness being popular choices (Liu, Song, Arp, 2012; Schindler, Caruso, 2014). Urban sprawl has been correlated with parameters relating to smog, such as vehicle tailpipe emissions and local meteorology (Stone, 2008). Ewing, Pendall and Chen (2003) reported that metropolitan areas with higher levels of urban sprawl were generally associated with more vehicles per household, less public transportation use, and less pedestrian commuting, which was echoed later by the findings of Bereitschaft and Debbage (2013) that higher levels of urban sprawl, or elongation-like urban morphologies, generally exhibit higher concentrations and emissions of air pollution and CO_2. On the basis of this previous literature, two indicators representing urban form were chosen, an urban elongation ratio (ER) and an urban compactness ratio (CR).

The ER measures the extended degree of a region, based on the following equation proposed in 1969 by Webbity (Haggett, 1997):

$$ER = L/L' \tag{1}$$

Where L is the length of the long axis of a region, and L' is the length of the short-axis of a region. The more extended the urban shape is, the higher the ratio is. The urban area is defined as that within the urban land boundary, as identified here using Landsat images and related thematic maps. 30 Landsat TM images (2000, 2007 and

2010) were employed to interpret urban land areas, using ERDAS IMAGING 9.1 and ArcGIS9.3 for data processing. We used both automated photo-interpretation and manual interpretation to digitize the built-up area of case cities from remote sensing images. For this, Landsat images were viewed as near infrared (NIR), red, and green false color composite (represented by red, green, and blue bands in Landsat). Then the thematic land-cover maps, urban street maps, and administrative maps were re-projected and geometrically corrected in accordance with the Landsat imagery in each city. Finally, the built-up areas, such as streets, residential area, and industrial zone, were digitized by identifying object features including shape, texture, size, color, and the association with neighboring objects. During this process, we also used auxiliary information, such as thematic maps and Google earth images. A popular TM band combination of five, four and three in RGB (red, green and blue) color space was used to facilitate the difference of urban land and non-urban land.

There is some discussion in the literature about the best way to parameterize urban compactness. Newman and Kenworthy (1989) related compactness to urban density, based on molecular and molar measures. Schwarz (2010) parameterized urban compactness using landscape metrics and population related indicators, similar to Burton (2002), who argued that urban compactness is a complex phenomena. Finally, Song (2005), Huang, Lu, and Sellers (2007) developed a series of compactness indicators including density, land-use mix, and pedestrian access etc. The metric of urban compactness used here is the model proposed by Thinh et al. (Thinh, Arlt et al. 2002). This model was designed to quantitatively evaluate the urban spatial form. The formula is based on Newton's law of gravitation, though instead of gravitation increasing with increasing mass and decreasing distance of two entities, here for urban compactness (i.e. urban gravity) increases with increasing constructed land and decreasing distance. The formula is as follows:

$$T = \frac{\sum \frac{1}{c} \frac{Z_i Z_j}{d^2(i,j)}}{N(N-1)/2} \tag{2}$$

where T is the average gravity of a specific urban space, i.e., the urban compactness;

Z_i and Z_j the construction land area for grid *i* and *j*, respectively; *d*(*i*,*j*) the Euclidean distance between grid *i* and *j*; c the constant (usually 100 m^2 in application); and *N* the total number of grids in the study area. The value of *T* generally has a positive correlation with the compactness of urban construction space. In practice, it is convenient to rasterize the urban land use data into a certain size or more commonly and as is done here using the remote sensing classification data. Taking the Landsat based land use classification data (Fig. 1), the pixel size is 30×30m, the grid size is set to be 60×60 m, and Z_i and Z_j are the construction land area in grid *i* and *j*, respectively.

Fig. 1 Illustration of the grid dividing principal in the urban compactness computation model

Using *T* can reflect the compactness of urban construction land in space. When comparing different cities, however, due to *d* being in the denominator and varying more than *Z*, *T* is inherently sensitive to the area of urban construction land, i.e., large area cities usually have small *T*, and vice versa. In order to facilitate comparisons across cities, a Normalized Compactness Index (NCI) was proposed herein, to account for these variations in *T*. As a general geometric principle, a circular city is supposed to have the highest compact degree given the same urban area. Thus, here the NCI is obtained by dividing *T* by the maximum compactness for a circular city with the area the same as the given city, and isis calculated as follows:

$$NCI = \frac{T}{T_{max}} = \frac{M(M-1)}{N(N-1)} \times \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} \frac{Z_i Z_j}{d^2(i,j)}}{\sum_{i'=1}^{n}\sum_{j'=1}^{n} \frac{S_{i'} S_{j'}}{d'^2(i',j')}} \qquad (3)$$

Where T_{max} is the compactness of the equivalent circle-shaped city; $S_{i'}$ and $S_{j'}$ the construction land area in grid *i'* and *j'*, respectively; and *M* the total number of the equivalent circle-shaped city. NCI ranges between 0 and 1. For a city with a fixed area, the NCI will approach 1 as the shape of the city is more close to circle. We used Landsat TM and ETM+ images as the data source. The cities in our study were masked out and classified into construction and non-construction lands, respectively. The construction land category was further classified into buildings (e.g., residential, commercial, service and public facilities), traffic and other land use types; the non-construction land was also further classified into subcategories (water body, wetland, woodland, bare land, etc.).

3.3 Panel Data Analysis and Control Variables

The relationship between urban form and smog over time was explored through use of panel data analysis, which separates the cross-sectional and time series dimensions. We estimated panel regression models measured by the log of the urban smog parameter as a function of urban compactness and urban elongation and control variables through use of the following equation:

$$y_{it} = \alpha_i + \beta' x_{it} + \gamma' z_{it} + \mu_{it} \text{ , } i = 1\ldots\ldots N,\ t = 1\ldots\ldots T \qquad (4)$$

where *i* denotes the size of the cross section (30 cities) and *t* (2000, 2007 and 2010) denotes the dimension of the time series, α_i is a scalar, β and γ are $k \times 1$ coefficient and vectors, β' and γ' are the transpose of β andγ, x_{it} and z_{it} are $1 \times k$ vectors of observations of the independent variables (here urban form descriptors and control variables), and y_{it} is the observation of the dependent variable for individual *i* at time *t* (here urban smog descriptors). μ_{it} represents the effects of other factors that are not only unique to individual units, but also to time periods, and that can be characterized by an independently and identically distributed random variable with zero mean and variance (σ^2). Panel data sets are being used increasingly and successfully in applied studies (Mainardi, 2005; Mikhad and Zemcik, 2009).

In order to more accurately explore the relationship between urban form and urban smog, it is necessary to control for confounding variables as part of the panel data analysis. Therefore, control variables were selected based on the strong theoretical or empirically-informed ties to air quality, as well as data availability. These included urban population (Lai and Cheng 2009), built-up area green coverage rate（Li et al., 2012）, power consumption, SO2 gas emissions (Xie, 2014), the gross value of industrial output, built up area, public transport (buses per million people) and heating systems (Zhang, 2014). These selected control variables are presented in Table 1, with data obtained from Chinese Urban Statistical Yearbook (2000, 2007 and 2010). The dummy variable was *heating system*, in which the value of zero was assigned for no urban heating system, and the value of 1 for having an urban heating system.

Table 1. Descriptive statistics for the control variables

Variables		**Mean**	**Maximum**	**Minimum**	**Standard Deviation**
Urban population	2000	2.32×10^2	923.19	45.43	195.95
(*Million people*)	2007	4.21×10^2	1510.99	87.97	348.89
	2010	4.37×10^2	1542.77	91.42	357.18
Built-up area green coverage	2000	30.16	44.8	11.4	8
(*%*)	2007	35.03	60.42	5.55	9.54
	2010	38.78	47.68	26.35	4.27
Power consumption	2000	9.39×10^5	5.32×10^6	1.19×10^5	1.22×10^6
(*Million kwh*)	2007	1.82×10^6	9.90×10^6	2.82×10^5	1.97×10^6
	2010	2.28×10^6	1.15×10^7	3.52×10^5	2.42×10^6
SO_2 emissions	2000[a]	40.16	144.5	0.01	38.97
(*Tons/square kilometer*)	2007[b]	1.24×10^5	6.73×10^5	174	1.26×10^5
(*Tons*)	2010[b]	1.01×10^5	5.86×10^5	103	1.03×10^5
Built up area	2000	1.73×10^2	550	34	118.79
(*Square kilometer*)	2007	3.31×10^2	1226	64	261.37
	2010	3.80×10^2	1350	43	285.69
The gross value of industrial output	2000	7.61×10^6	5.22×10^7	4.73×10^5	9.85×10^6
(*Million yuan*)	2007	2.58×10^7	1.84×10^8	1.46×10^6	3.76×10^7
	2010	3.92×10^7	2.38×10^8	3.07×10^6	5.01×10^7
Buses per million people	2000	9.3	27.6	3.2	5
(-)	2007	11.25	22.02	4.87	3.68
	2010	13	21.12	4.16	3.76
Heating system (-)		0.5	1	0	0.51

[a] SO_2 emissions (*Tons/ Square kilometer*); [b] Industrial SO_2 emissions (*Tons)* Panel data

analysis works best when the sample data population (e.g. number of cities in our case) is as large as possible, as the larger the sample population the greater the degrees of freedom and the lesser the colinearity among explanatory variables (Hsiao, 2003). Steyerberg et al (1999) demonstrated that the selection bias decreases as the events per predicting variable increases. Peduzzi et al (1996) recommends not to have more than one independent predictor variable per 10 data points, though Vittinghoff and McCulloch (2007) responded a minimum of 10 outcome events per predictor variable may be too conservative under certain circumstances. In this study, the sample populations is 90 [i (30 cities)*t (3 years)], and the maximum number of explanatory variables is 10 (ER, CR, 7 control variables and one dummy variable), therefore we start with 9 outcome events per predictor variable; though some of the control variables are removed in the most parsimonious model, due to a lack of their statistical significance.

4 Results and discussion

4.1 Results

Annual averages of AOD in 2000 and PM10 in 2007 and 2010 for the selected 30 Chinese cities are presented in Table 2. Note that the AOD and PM10 data in this Table are normalized to the maximum value, so that all values range from 0.0 to 1.0, and the data is further separated into three categories, representing low, medium and high values. The average value of normalized AOD was 0.13 (i.e. 13% the maximum value), and most of cities had average annual aerosol optical depth ranging from 0.00 to 0.29, with two north China cities (Beijing and Shenyang) ranging from 0.30 to 0.40, and four cities having values above 0.50 (including Jinan, Tianjin, Xian and Zhengzhou). The average value of urban PM10 was 0.10, and most of cities had average annual PM10 ranging from 0.09 to 1.00, with seven (2007) and nine (2010) cities ranging from 0.04 to 0.08, and five (2007) and three (2010) cities having values above 0.13.

Table 2Urban aerosol optical depth and PM10 normalized to the maximum level for Chinese Cities during 2000, 2007 and 2010

Category	Cities
Aerosol optical depth(2000)	
0.00-0.29	Chongqing; Fuzhou; Guangzhou; Guiyang; Herbing; Haikou; Hangzhou; Hefei; Hohhot; Kunming; Lanzhou; Nanchang; Nanjing; Nanning; Shanghai; Shijiazhuang; Taiyuan; Wuhan; Urumqi; Xining; Yinchuan
0.30-0.40	Beijing; Shenyang
0.50-1.00	Jinan; Tianjin; Xian; Zhengzhou
PM10 (2007)	
0.04-0.08	Fuzhou; Guangzhou; Haikou; Hohhot; Kunming; Nanchang; Nanning; Changchun; Changsha; Chengdu; Chongqing; Guiyang; Herbing; Hangzhou; Hefei; Jinan; Nanjing; Shanghai; Shenyang; Taiyuan; Tianjin; Wuhan
0.09-0.12	Xining; Yinchuan; Zhengzhou
0.13-1.00	Beijing; Lanzhou; Shijiazhuang; Urumqi; Xian
PM10 (2010)	
0.04-0.08	Changsha; Fuzhou; Guangzhou; Guiyang; Haikou; Hohhot; Kunming; Nanning; Shanghai
0.09-0.12	Beijing; Changchun; Chengdu; Chongqing; Herbing; Hangzhou; Hefei; Jinan; Nanchang; Nanjing; Shenyang; Shijiazhuang; Taiyuan; Tianjin; Wuhan; Xining; Yinchuan; Zhengzhou
0.13-1.00	Lanzhou; Urumqi; Xian

The results of calculated urban compactness ratio and elongation ratio for 30 cities in 2000, 2007 and 2010 are presented in Table 3. The two ratios were the largest in 2007 (average value of urban compactness was 0.24 and average value of urban elongation was 4.07). There were more cities with a compactness ratio ranging from 0.16 to 0.45 in 2007 (24 cities) and 2010 (21 cities) than in 2000 (15 Cities). There were also more cities with an elongation ratio over 3.00 in 2007 (15 cities), compared with 2000 (2 cities) and 2010 (1 city).

Table 3 Urban form descriptors for 30 Chinese cities during the years 2000, 2007 and 2010

Category	Cities
Compactness Ratio (2000)	
0.05-0.15	Beijing; Chengdu; Chongqing; Guangzhou; Guiyang; Hangzhou; Lanzhou; Nanjing; Tianjin; Urumqi; Xining; Yinchuan

0.16-0.20	Fuzhou; Haikou; Jinan; Nanchang; Shanghai; Shenyang; Wuhan;
0.21-0.45	Changchun; Changsha; Herbing; Hefei; Hohhot; Kunming; Nanning; Shijiazhuang; Taiyuan; Xian; Zhengzhou
Compactness Ratio (2007)	
0.05-0.15	Chongqing; Guangzhou; Guiyang; Hefei; Tianjin; Wuhan
0.16-0.20	Fuzhou; Nanjing
0.21-0.45	Beijing; Changchun; Changsha; Chengdu; Herbing; Haikou; Hangzhou; Hohhot; Jinan; Kunming; Lanzhou; Nanchang; Nanning; Shanghai; Shenyang; Shijiazhuang; Taiyuan; Urumqi; Xian; Xining; Yinchuan; Zhengzhou
Compactness Ratio (2010)	
0.05-0.15	Beijing; Guangzhou; Guiyang; Lanzhou; Nanjing; Taiyuan; Tianjin; Wuhan; Xining
0.16-0.20	Changchun; Changsha; Chengdu; Jinan; Kunming; Nanchang; Shenyang; Urumqi; Yinchuan
0.21-0.45	Chongqing; Fuzhou; Herbing; Haikou; Hangzhou; Hefei; Hohhot; Nanning; Shanghai; Shijiazhuang; Xian; Zhengzhou

Elongation Ratio (2000)	
1.00-2.99	Beijing; Changchun; Changsha; Chengdu; Chongqing; Fuzhou; Guangzhou; Guiyang; Herbing; Haikou; Hefei; Hohhot; Jinan; Kunming; Zhengzhou; Nanchang; Nanjing; Nanning; Shanghai; Shenyang; Shijiazhuang; Taiyuan; Tianjin; Wuhan; Urumqi; Xian; Xining
3.00-4.00	Lanzhou; Yinchuan
4.01-16.99	-
Elongation Ratio (2007)	
1.00-2.99	Beijing; Changchun; Chengdu; Guiyang; Herbing; Haikou; Kunming; Nanchang; Shanghai; Shenyang; Shijiazhuang; Xian; Xining; Zhengzhou
3.00-4.00	Changsha; Guangzhou; Hangzhou; Hefei; Jinan; Nanjing; Nanning; Taiyuan; Wuhan
4.01-16.99	Chongqing; Fuzhou; Hohhot; Lanzhou; Tianjin; Urumqi; Yinchuan
Elongation Ratio (2010)	
1.00-2.99	Beijing; Changchun; Changsha; Chengdu; Chongqing; Fuzhou; Guangzhou; Guiyang; Herbing; Haikou; Hangzhou; Hefei; Hohhot; Jinan; Kunming; Nanchang; Nanjing; Nanning; Shanghai; Shenyang; Shijiazhuang; Taiyuan; Tianjin; Wuhan; Urumqi; Xian; Xining; Yinchuan; Zhengzhou
3.00-4.00	-
4.01-16.99	Lanzhou

Table 2. Parameters for Urban Smog (aerosol optical depth and PM10 normalized to the maximum level), and Urban Form (Compactness Ratio (CR) and Elongation Ratio (ER) for the 30 Chinese Cities included in this study.

City	**Urban Smog**			**Urban Form**					
	AOD$^{a)}$ **2000**	**PM10** $^{a)}$ **2007**	**PM10** $^{a)}$ **2010**	**CR 2000**	**CR 2007**	**CR 2010**	**ER 2000**	**ER 2007**	**ER 2010**
Beijing	0.30-0.40	0.13-1.00	0.09-0.12	0.05-0.15	0.21-0.45	0.05-0.15	1.00-2.99	1.00-2.99	1.00-2.99
Changchun	**?**	0.04-0.08	0.09-0.12	0.21-0.45	0.21-0.45	0.16-0.20	1.00-2.99	1.00-2.99	1.00-2.99
Changsa	**?**	0.04-0.08	0.04-0.08	0.21-0.45	0.21-0.45	0.16-0.20	1.00-2.99	3.00-4.00	1.00-2.99
Chengdu	?	0.04-0.08	0.09-0.12	0.05-0.15	0.21-0.45	0.16-0.20	1.00-2.99	1.00-2.99	1.00-2.99
Chongqing	0.00-0.29	0.04-0.08	0.09-0.12	0.05-0.15	0.05-0.15	0.21-0.45	1.00-2.99	4.01-16.00	1.00-2.99
Fuzhou	0.00-0.29	0.04-0.08	0.04-0.08	0.16-0.20	0.16-0.20	0.21-0.45	1.00-2.99	4.01-16.00	1.00-2.99

Guangzhou	0.00-0.29	0.04-0.08	0.04-0.08	0.05-0.15	0.05-0.15	0.05-0.15	1.00-2.99	3.00-4.00	1.00-2.99
Guiyang	0.00-0.29	0.04-0.08	0.04-0.08	0.05-0.15	0.05-0.15	0.05-0.15	1.00-2.99	1.00-2.99	1.00-2.99
Haikou	0.00-0.29	0.04-0.08	0.04-0.08	0.16-0.20	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
Hangzhou	0.00-0.29	0.04-0.08	0.09-0.12	0.05-0.15	0.21-0.45	0.21-0.45	1.00-2.99	3.00-4.00	1.00-2.99
Hefei	0.00-0.29	0.04-0.08	0.09-0.12	0.21-0.45	0.05-0.15	0.21-0.45	1.00-2.99	3.00-4.00	1.00-2.99
Herbing	0.00-0.29	0.04-0.08	0.09-0.12	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
Hohot	0.00-0.29	0.04-0.08	0.04-0.08	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	4.01-16.00	1.00-2.99
Jinan	0.50-1.00	0.04-0.08	0.09-0.12	0.16-0.20	0.21-0.45	0.16-0.20	1.00-2.99	3.00-4.00	1.00-2.99
Kunming	0.00-0.29	0.04-0.08	0.04-0.08	0.21-0.45	0.21-0.45	0.16-0.20	1.00-2.99	1.00-2.99	1.00-2.99
Lanzhou	0.00-0.29	0.13-1.00	0.13-1.00	0.05-0.15	0.21-0.45	0.05-0.15	3.00-4.00	4.01-16.00	4.01-16.00
Nanchang	0.00-0.29	0.04-0.08	0.09-0.12	0.16-0.20	0.21-0.45	0.16-0.20	1.00-2.99	1.00-2.99	1.00-2.99
Nanjing	0.00-0.29	0.04-0.08	0.09-0.12	0.05-0.15	0.16-0.20	0.05-0.15	1.00-2.99	3.00-4.00	1.00-2.99
Nanning	0.00-0.29	0.04-0.08	0.04-0.08	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	3.00-4.00	1.00-2.99
Shanghai	0.00-0.29	0.04-0.08	0.04-0.08	0.16-0.20	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
Shenyang	0.30-0.40	0.04-0.08	0.09-0.12	0.16-0.20	0.21-0.45	0.16-0.20	1.00-2.99	1.00-2.99	1.00-2.99
Shijiazhuang	0.00-0.29	0.13-1.00	0.09-0.12	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
Taiyuan	0.00-0.29	0.04-0.08	0.09-0.12	0.21-0.45	0.21-0.45	0.05-0.15	1.00-2.99	3.00-4.00	1.00-2.99
Tianjin	0.50-1.00	0.04-0.08	0.09-0.12	0.05-0.15	0.05-0.15	0.05-0.15	1.00-2.99	4.01-16.00	1.00-2.99
Urumqi	0.00-0.29	0.13-1.00	0.13-1.00	0.05-0.15	0.21-0.45	0.16-0.20	1.00-2.99	4.01-16.00	1.00-2.99
Wuhan	0.00-0.29	0.04-0.08	0.09-0.12	0.16-0.20	0.05-0.15	0.05-0.15	1.00-2.99	3.00-4.00	1.00-2.99
Xian	0.50-1.00	0.13-1.00	0.13-1.00	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
Xining	0.00-0.29	0.09-0.12	0.09-0.12	0.05-0.15	0.21-0.45	0.05-0.15	1.00-2.99	1.00-2.99	1.00-2.99
Yinchuan	0.00-0.29	0.09-0.12	0.09-0.12	0.05-0.15	0.21-0.45	0.16-0.20	3.00-4.00	4.01-16.00	1.00-2.99
Zhengzhou	0.50-1.00	0.09-0.12	0.09-0.12	0.21-0.45	0.21-0.45	0.21-0.45	1.00-2.99	1.00-2.99	1.00-2.99
No # Cities									
low category	21	22	9	12	6	9	28	14	29
middle category	2	3	18	7	2	9	2	9	0
high category	4	5	3	11	22	12	0	7	1

a) Normalized to maximum value

To select the appropriate model to use in the panel data analysis, various statistical tests were employed, including the *F*-test, redundant fixed effects test (RFE), the Hausman test and Breusch Pagan and Lagrangian Multiplier (BP-LM) test. The RFE test indicated that the pooled model is better than the fixed effects model (p-value > 0.05) (Hausman, 1978), and the Hausman test indicated that the random effects model is better than the fixed effects model (p-value > 0.05) (Hausman, 1978). The dependent variable in all models was the log of the normalized urban smog index (Table 2). We started with all variables in model 1 and then eliminated the insignificant ones, until a parsimonious model was obtained (model 3) (see table 4). The parsimonious model has 7 predictor variables (CR, ER, Built-up green coverage, Power consumption, SO_2 emissions, gross value of industrial output and buses per million people); as the sample population is 90, we fulfill the criteria of having more than 10 events per predictor variable.

Table 4 Panel Data Analysis results of the relationship between urban smog and urban form along with selected control variables

Dependent variable= urban smog

Independent variable	**Model1**	**Model2**	**Model3**
Urban compactness (CR)	0.56 (3.67)∗	0.57 (3.12) ∗	0.47 (2.50)∗
Urban elongation (ER)	0.42 (3.31) ∗	0.42 (2.69) ∗	0.42 (2.52)∗
Urban population	-0.07 (-0.29)		
Built-up area green coverage	-0.96 (-4.47) ∗	-0.96 (-3.65) ∗	-1.09 (-4.04)∗
Power consumption	0.75 (4.80) ∗	0.74 (4.20) ∗	0.86 (4.91)∗
SO_2 emissions	0.69 (33.63) ∗	0.69 (27.46) ∗	0.69 (25.53)∗
Built up area	0.41 (1.68)	0.37 (1.44) ∗	
The gross value of industrial output	0.52 (3.79) ∗	0.53 (3.25) ∗	0.41 (3.24)∗
Buses per million people	-0.51 (-3.18) ∗	-0.47 (-2.76) ∗	-0.48 (-2.56)∗
Heating system	0.19 (1.59)	0.19 (1.32)	
Diagnostics			
Adjusted R-squared	0.848	0.850	0.852
S.E. of regression	1.206	1.198	1.192

∗ Indicates statistical significance at the 5% level; t-values in parentheses

4.2 Discussion

As presented in Table 4, Urban compactness (CR) was positively correlated with urban smog (normalized AOD / PM10). This is somewhat expected, as compared with other countries, population densities of cities are quite high in China (Kenworthy and

Hu, 2002); furthermore, urban infrastructure investments are relatively limited, which has exhausted urban environmental carrying capacity (Jenks and Burgess, 2000). This correlation is also expected based on the observations by Zhou et al (1983) and Li, Ran and Tao (2008) that emissions of aerosol and PM in high population density districts, such as Beijing, are generally higher than low population density districts, due to increased fossil fuel consumption, due to less heating and transportation.. Transportation in high population densities cities is not necessarily pedestrian-oriented. In 2012, the total amount of vehicles in China was 2.33 hundred million, which was a 3.67% increase compared to 2011.[2] Uncontrolled growth in urbanization and motorization in the city of Karachi, Pakistan, has been blamed in part for a transportation system that is socially, economically, and environmentally unsustainable (Qureshi and Lu, 2007). Vehicle exhaust pollution has also aggravated in China. The air quality in large cities has deteriorated due to photochemical smog, which are typical of vehicle pollution (He, Huo and Zhang, 2002). According to the research of Chinese Academic of Science,[3] in Beijing, 20-30% of the smog-pollution is caused by vehicle emissions. Other elevations in compact urban areas would be anticipated from increased domestic heating, food preparation as well as the formation of haze hoods, as presented in the Introduction. Two possible ways to reduce the influence of urban compactness on urban smog are evidenced through the control variables "built up green area coverage" and "buses per million people", which negatively correlated with urban smog, as increase in these variables would indicate less population density and less vehicles within an urban area.

Urban elongation was also positively correlated with urban smog in Table 4. Martins (2012) reported for the Porto region in Portugal that an index for urban sprawl (similar to urban elongation) had more of an aggravating effect on PM10 than an index for urban compactness. However, in our study, the urban smog seemed equally correlated with the chosen parameters for elongation and compactness. An important aspect in China related to urban elongation is the rapid aggregation of

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[2] http://www.chinairn.com/news/20120718/936214.html

[3] http://env.people.com.cn/n/2014/0113/c1010-24102913.html

industrial parks, named “Kai fa qu”, on the outer vicinities of urban areas (Lian, 2011). According to the findings of Hao, Cao and Wang (2013), the level of industrial aggregation was positively correlated with the level of urban aggregation. He et al (2012) found that the industrial aerosol and soil dust are possibly two dominant influencing factors on northern urban smog.

When considering the rapid changing dynamic of urban form parameters in the years 2000, 2007 and 2010 (Table 4), much of this is attributable to the two pathways of urbanization in China referred to as "passive urbanization" and "active urbanization". Passive urbanization is when the government appropriates rural to urban areas (Yu, Yang, and Xiong, 2013; Zhang and Gu, 2006), which would increase sprawl or elongation. Active urbanization occurs when rural residents / farmers move into the city, increasing either compaction or elongation, depending on if the settling is done mostly in a central or outlying area. Passive urbanization is done both to expand industrial and urban areas (Lin, 2007). Many industrial firms from within cities have been migrated to these outer industrial parks / development zones, causing an increase of industrial pollution in these areas (He, 2007). Between 1984 and 2005, China’s built-up areas dramatically expanded from 8,842 to 32,520 km^2, a growth by 260 percent (China State Statistical Bureau, 2006).

These trends can account for some of the trends seen in the control variables. The gross value of industrial output was positively correlated with urban smog, corresponding with the growth of industrial zones / elongation. Similarly, SO_2 emissions and power consumption also corresponded with increased smog, and which is typically associated with coal burning and other industrial processes, which would be expected to increase with the formation of industrial parks.

5. Conclusions and suggestions

The presented research provides an empirical analysis of the relationship between urban form (as described by compactness and elongation) and urban smog (as described by AOC and PM10) for 30 cities in China. While controlling for built-up area green rate, power consumption, SO_2 emissions, and gross value of industrial output etc, the results indicated that urban compactness and elongation could be contributing factors to urban smog in China. Chinese urban form is characterized by relative high urban population density, motorization-oriented habitation and high rate of industrial aggregation. Meanwhile, the implications of passive and active urbanization, which increase urban compactness and urban elongation simultaneously, increase the prevalence of urban smog in China. Changes in urban planning to minimize increased compaction and elongation may be a strategy to mitigate urban smog pollution in China, such as through including green area and the effeicency of public transportation infrastructure.

In China, currently established measures of reducing urban smog have focused on directly decreasing aerosol emissions from industrial companies (Zhang et al., 2013) and urban transportation,[4] similar to previous successful efforts in other countries including the US Clean Air Act. Some local governments have been able to enforce this successfully. For instance, the government in the Northern Province Liaoning fined eight cities 54.2 million Yuan for their air pollution.[5] Other local governments provided financial support. For example, Beijing established smog reductions plans, and allocated for this financial support of 760 billion Yuan.[6] However, pollution-abatement subsidies have been criticized as being inefficient instruments by theoretical studies (Liu and Cui,2011). Furthermore, subsidy policies are often criticized because according to the polluter pays principle the cost should be borne by the polluter, and not the taxpayer.

The results of this study gives indication that further research is needed on potential urban planning steps that could help reduce smog. Though increased urban compactness was associated with increased smog, Williams et al. (2000) and Burton (2002) saw some advantages of urban compactness, such as resource and economic efficiency, including mass transit efficiency. On the other hand, Tony (1996), Rudlin and Falk (1999), argued against the process of urban compaction because higher

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[4] http://news.xinhuanet.com/2014-01/09/c_125978474.htm

[5] http://zt.21so.com/20131211/wumai.html

[6] http://zqrb.ccstock.cn/html/2014-01/21/content_397503.htm

density led to traffic congestion, air pollution and overcrowding. China is becoming more automobile-dependent (Qureshi and Lu, 2007) and not providing highly efficient public transportation or taxes for fossil fuel-powered automobiles (including tolls, fuel taxes and parking) would encourage this trend, thereby worsening air quality (He, Huo and Zhang, 2002). Considering the negative correlation between urban smog and "buses per million people", it is possible that urban smog and urban compactness may be exhibit less of a relationship if improved infrastructure of public transportation was implemented in the compact cites (as indicated by the negative correlation of urban smog with "buses per million people" in Table 4). Further, increased green areas within the more compact areas would lead to lower smog (as indicated by the negative correlation of urban smog with "built-up green area coverage rate" in Table 3), to both reduce compaction and also introduce vegetation that can act as air filters. Due to the correlation with power consumption and SO_2 in Table 4, more sustainable energy consumption patterns (e.g. reduction in the use of coal, increasing the use of electric cars, solar panels) and industrial practices in these areas would also be expected to mitigate smog in compact areas.

Some limitations are worth mentioning from this research. The indicators of measuring urban form and urban smog are limited. Thus, the research only provides an empirical correlation of certain aspects. But this explorative research provides a starting point for further research on urban form and urban smog in China.

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