



Research  
Air Pollution Control—Review

## Progress of Air Pollution Control in China and Its Challenges and Opportunities in the Ecological Civilization Era



Xi Lu<sup>a,b,#</sup>, Shaojun Zhang<sup>a,b,#</sup>, Jia Xing<sup>a,b,#</sup>, Yunjie Wang<sup>a</sup>, Wenhui Chen<sup>a</sup>, Dian Ding<sup>a</sup>, Ye Wu<sup>a,b</sup>, Shuxiao Wang<sup>a,b,\*</sup>, Lei Duan<sup>a,b</sup>, Jiming Hao<sup>a,b,\*</sup>

<sup>a</sup> State Key Joint Laboratory of Environment Simulation and Pollution Control, School of Environment, Tsinghua University, Beijing 100084, China

<sup>b</sup> State Environmental Protection Key Laboratory of Sources and Control of Air Pollution Complex, Beijing 100084, China

### ARTICLE INFO

#### Article history:

Received 27 December 2019

Revised 18 March 2020

Accepted 26 March 2020

Available online 19 June 2020

#### Keywords:

Air pollution control

Ecological civilization

PM<sub>2.5</sub>

Low-carbon development

Ozone

### ABSTRACT

China's past economic growth has substantially relied on fossil fuels, causing serious air pollution issues. Decoupling economic growth and pollution has become the focus in developing ecological civilization in China. We have analyzed the three-decade progress of air pollution controls in China, highlighting a strategic transformation from emission control toward air quality management. Emission control of sulfur dioxide (SO<sub>2</sub>) resolved the deteriorating acid rain issue in China in 2007. Since 2013, control actions on multiple precursors and sectors have targeted the reduction of the concentration of fine particulate matter (PM<sub>2.5</sub>), marking a transition to an air-quality-oriented strategy. Increasing ozone (O<sub>3</sub>) pollution further requires O<sub>3</sub> and PM<sub>2.5</sub> integrated control strategies with an emphasis on their complex photochemical interactions. Fundamental improvement of air quality in China, as a key indicator for the success of ecological civilization construction, demands the deep de-carbonization of China's energy system as well as more synergistic pathways to address air pollution and global climate change simultaneously.

© 2020 THE AUTHORS. Published by Elsevier LTD on behalf of Chinese Academy of Engineering and Higher Education Press Limited Company. This is an open access article under the CC BY-NC-ND license (<http://creativecommons.org/licenses/by-nc-nd/4.0/>).

### 1. Introduction

The past 30 years have witnessed dramatic economic growth in China. The gross domestic product (GDP) of China expanded by a factor of 43 from 1990 to 2017, with an average annual growth rate of more than 10% [1]. This marvelous achievement in China's economy was primarily fueled by fossil fuels—particularly coal, the main source for the emission of a variety of air pollutants and carbon dioxide (CO<sub>2</sub>). China surpassed the United States in 2007 in terms of CO<sub>2</sub> emissions and in 2009 in terms of energy consumption, and has become the world's largest energy-consuming and carbon-emitting country. In 2017, China was responsible for 23.2% of global energy use and 27.6% of global CO<sub>2</sub> emissions [2]. China experienced its highest annual emissions of sulfur dioxide (SO<sub>2</sub>) in 2007, nitrogen oxides (NO<sub>x</sub>) in 2012, and primary fine particulate matter (PM<sub>2.5</sub>) in 2006 [3,4], at respectively 2.5, 4.6, and 1.5 times the corresponding values for 1990 in China

(Fig. 1 [5,6]). In 2017, approximately 70% of all 338 municipal cities in China still did not meet the National Ambient Air Quality Standard (NAAQS) [7], with non-attainment of PM<sub>2.5</sub> as the most prominent air pollution problem. China faces the dual pressures of the improvement of national air quality and the mitigation of climate change.

Over the past three decades, China has been making continuous efforts to decouple air pollution and carbon intensity from economic growth, with the ultimate goal of achieving a society based on sustainable development and ecological civilization. In the early stages (the 1970s and 1980s), air pollution controls in China were primarily implemented at the local level, with the aim of controlling dust emissions. Later on, acid rain emerged as a serious problem in China, influencing more than 30% of the total territory, particularly in the southern and southwestern regions. Reducing SO<sub>2</sub> emissions from coal combustion sectors has since become more regulatory since the 1990s [8]. These efforts have led to China's total emissions of SO<sub>2</sub> and primary PM<sub>2.5</sub> peaking before 2010 [5].

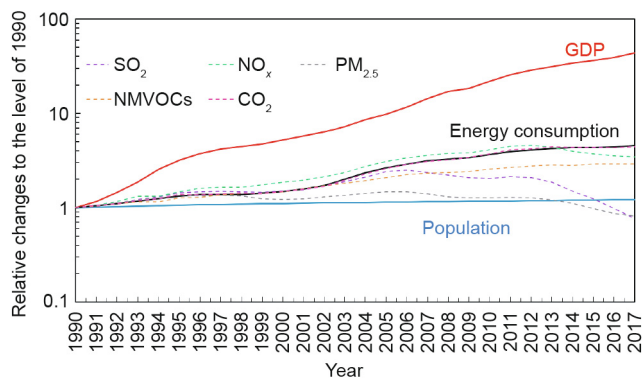
At present, tackling the PM<sub>2.5</sub> issue is the most challenging due to its complex formation, with contributions from multiple precursors and sources. Air quality management in China has thus

\* Corresponding authors.

E-mail addresses: [shxwang@tsinghua.edu.cn](mailto:shxwang@tsinghua.edu.cn) (S. Wang),

[hjm-den@tsinghua.edu.cn](mailto:hjm-den@tsinghua.edu.cn) (J. Hao).

# These authors contributed equally to this work.



**Fig. 1.** Trends in GDP, energy consumption, and population, as well as emissions of  $\text{SO}_2$ ,  $\text{NO}_x$ , primary  $\text{PM}_{2.5}$ , non-methane volatile organic compounds (NMVOCs), and  $\text{CO}_2$  between 1990 and 2017, with values for 1990 taken as 1. GDP, energy consumption, and population data were obtained from the National Bureau of Statistics of China. Emission data of air pollutants ( $\text{SO}_2$ ,  $\text{NO}_x$ ,  $\text{PM}_{2.5}$ , and NMVOCs) for 2010–2017 was obtained from Tsinghua University's multi-resolution emission inventory for China (MEIC) [5].  $\text{CO}_2$  emission data was obtained from the International Energy Agency (IEA) [6].

transformed into a new model: In contrast to its initial focus on control actions for a single pollutant or in a single sector, air quality management has now been redesigned to mitigate multiple precursors from multiple sectors through extensive coordination at the regional or national level. More recently, China has launched a series of movements including the short-term “blue sky defense battle” between 2018 and 2020, and the long-term “beautiful China” targets through 2035. The latter require fundamental improvements in air quality in China; in particular, they require all cities in China to achieve an annual concentration of  $\text{PM}_{2.5}$  below  $35 \mu\text{g}\cdot\text{m}^{-3}$  (the Interim Target-1 level recommended by the World Health Organization (WHO)) by 2035. To realize this goal, in-depth adjustment of the industrial structure and de-carbonization of energy systems will be critical, in addition to more stringent end-of-pipe control measures.

On the other hand, carbon intensity (measured by tonnes of  $\text{CO}_2$  per unit of GDP) in China has declined dramatically since 1980, reflecting initial actions toward the improvement of energy efficiency and, more recently, the strategy for climate change mitigation. At the 2015 Paris Climate Conference (COP21), China pledged to reduce its carbon intensity by 60%–65% relative to the 2005 level, and to peak its carbon emissions by 2030 or earlier. Increasing clean and low-carbon energy sources plays a central role in China's transition from relative to absolute carbon-economy decoupling by 2030. Efforts to reduce emissions of air pollutants and greenhouse gases (GHGs), particularly  $\text{CO}_2$ , have become inevitably interlinked along this path. In 2015, the United Nations General Assembly (UNGA) proposed the Sustainable Development Goals (SDGs) for 2030 [9], which emphasize a synergistic pathway with multiple benefits as an important means of coping with the complex challenges toward sustainability. Since 2012, the Chinese government has carried forward a national strategy of ecological civilization construction in order to decouple economic growth from ecological impacts and build a “beautiful China.” It is clear that both air pollution control and climate change mitigation play pivotal roles in this progress. This paper offers a comprehensive review of China's continuous efforts to address the changing targets of air pollution control over the past 30 years. In order for China to address the emerging challenges of air pollution in the ecological civilization era, which include ozone ( $\text{O}_3$ )-related issues and continuous reduction in  $\text{PM}_{2.5}$  concentrations, an integrative pathway is necessary that fully considers the

synergistic effects between air pollution control and climate change mitigation.

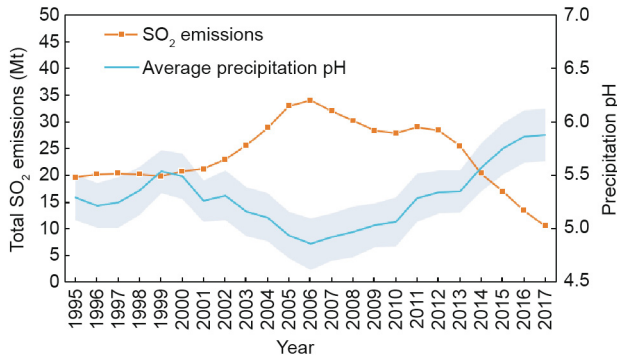
## 2. From acid rain to haze pollution: Central issues in ecological civilization

In the 1980s, acid rain became China's first serious cross-regional air pollution problem, as the extensive use of coal and other fossil fuels led to substantial emissions of  $\text{SO}_2$  and  $\text{NO}_x$ . These gaseous pollutants were oxidized and transformed in the atmosphere to form sulfuric and nitric acids, which could be transported over hundreds of kilometers, creating acid deposition on a regional scale [8]. In particular, in southern and southwestern China, the sulfur contents in coal were as high as 4% in some areas, and the acid-neutralizing effect of alkaline dust was much weaker than in northern China, resulting in the most severe acid deposition in the country [8]. Acid rain posed a severe threat to the environment, agricultural production, humans, and complex ecological systems (e.g., ground vegetation and biodiversity). The societal costs of acid rain in China were estimated to be as high as 32 billion USD [8].

A significant negative correlation (Pearson's  $r = -0.90$ ) can be seen between the national average precipitation acidity (i.e., average pH values) and total  $\text{SO}_2$  emissions over the past 20 years (Fig. 2 [10]). The trough of average precipitation pH values and the peak of total  $\text{SO}_2$  emissions both took place in 2006, when about one third of China's total territory area was at risk of acid rain. In terms of bulk sulfur and nitrogen deposition, the severity of the acid rain in China in 2006, when acid rain was at its peak, was estimated in the same range (if not higher) as that in central Europe in the 1980s [8].

Chinese policymakers took a longer path to recognize issues with  $\text{PM}_{2.5}$  pollution in comparison with their response to the acid rain issue. The landmark Harvard Six Cities Study for the eastern United States suggested that ambient fine particle exposures—and not exposures of coarse particle fractions—were specifically associated with daily mortality [11], leading to the first  $\text{PM}_{2.5}$  NAAQS being established by the US Environmental Protection Agency (EPA) in 1997 [12]. Shortly after, researchers in China began to measure ambient  $\text{PM}_{2.5}$  concentrations and analyze their chemical composition. A research group at Tsinghua University pioneered the continuous measurement of ambient  $\text{PM}_{2.5}$  concentrations in Beijing in 1999, and reported annual average values of between 115 and  $127 \mu\text{g}\cdot\text{m}^{-3}$  [13]. As opposed to coarser particles, secondary aerosols (e.g., ammonium nitrate and sulfate, and organic aerosols) rather than primary ones were found to be responsible for the majority of  $\text{PM}_{2.5}$ , suggesting that controlling dust emissions alone (which had been ongoing since 1970s) could not sufficiently reduce overall  $\text{PM}_{2.5}$  concentrations.

High  $\text{PM}_{2.5}$  concentrations, which are known as haze episodes, frequently hit many Chinese cities in winter, significantly deteriorating visibility and harming people's health. However,  $\text{PM}_{2.5}$  was not officially listed in China's NAAQS until the amendment in 2012, when China adopted the WHO Interim Target-1 level ( $35 \mu\text{g}\cdot\text{m}^{-3}$ ) as the annual mean limit for  $\text{PM}_{2.5}$  concentrations. Since then, an official  $\text{PM}_{2.5}$  monitoring network has been gradually established to diagnose more ground-level  $\text{PM}_{2.5}$  concentrations, with hourly results available to the public. In 2013, 96% of the 74 key cities in China did not meet the NAAQS annual  $\text{PM}_{2.5}$  concentrations. The most polluted cities were located in the Beijing–Tianjin–Hebei (BTH) region, where the annual averages were 2–3 times the limit. During a serious haze pollution episode in January 2013, hourly  $\text{PM}_{2.5}$  concentrations were recorded as high as  $800 \mu\text{g}\cdot\text{m}^{-3}$  [14]. Image spectroradiometers onboard satellites provided the measurement of aerosol optical depth (AOD), which is used to characterize continuous trends in ground-level



**Fig. 2.** Trends in national average precipitation pH and total SO<sub>2</sub> emissions between 1995 and 2017. Precipitation pH data were obtained from 74 sites with continuous observations in the Acid Rain Monitoring Networks [10]; the blue belt indicates the interval within one standard deviation of the mean. Emission data for SO<sub>2</sub> were derived from the MEIC model.

PM<sub>2.5</sub>. These satellite-based measurements confirmed that the BTH region faced the most serious PM<sub>2.5</sub> pollution within China, with significant deterioration during 2004–2007 [15].

The Global Burden of Disease Study (GBD) 2015 identified ambient PM<sub>2.5</sub> exposure as a significant risk factor in the disease burden in China and as responsible for 1.1 million deaths in that year [16]. Based on the GBD project, the World Bank Group estimated that the premature mortality costs due to air pollution (including indoor exposure) were close to 10% of the GDP equivalent in 2013—significantly higher than those of developed countries in Europe and North America [17]. Undoubtedly, given the scale of the impacts of air pollution on public health, improvement of air quality has become a central domain in developing ecological civilization in China.

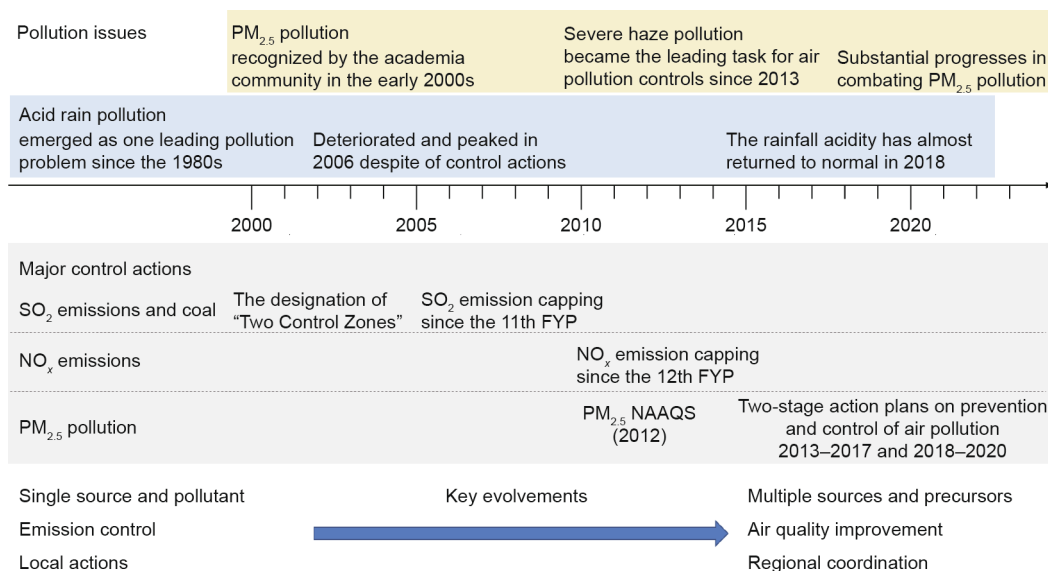
### 3. From emission control to air quality management: Key tasks in ecological civilization

With rapid societal development in China over the past 30 years, air quality management in China has been chasing shifting targets, starting with acid rain and NO<sub>x</sub> emissions, and now focusing on

PM<sub>2.5</sub> pollution (Fig. 3). Control actions on acid rain were primarily focused on SO<sub>2</sub> emissions from coal combustion in the 1990s. However, the increasing trend of SO<sub>2</sub> emissions was not reversed until 2007, when diversified technological and policy instruments were widely utilized. Such progress could not have been achieved without innovative improvement in political accountability, verification of SO<sub>2</sub> emissions, and financial incentives for power plants to install and operate flue gas desulfurization (FGD). NO<sub>x</sub> has more sources than SO<sub>2</sub>, including coal burning, mobile sources, and other fossil fuel combustion sectors. Therefore, control actions on NO<sub>x</sub> emissions must be more extensive and complex, which requires cooperation among multiple sectors. Controlling PM<sub>2.5</sub> concentrations not only requires consistently mitigating multiple precursors from numerous sectors, but also demands consideration of complex influences from both meteorological and atmospheric chemistry conditions. Indeed, comprehensive control actions with multi-party coordination on provincial and even national levels have been implemented in China to minimize the adverse ecological and social impacts of PM<sub>2.5</sub> pollution. Since 2013, combating PM<sub>2.5</sub> pollution has marked a strategic transfer from emission control toward air quality management in China, which is oriented toward benefiting the public health and social welfare. The fundamental improvement of PM<sub>2.5</sub> concentrations has become a key indicator of the success of ecological civilization construction in China.

#### 3.1. Decoupling SO<sub>2</sub> emissions

The Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution Air (short for Air Pollution Law, hereinafter) was formulated in 1987 [18]. In its first version, industrial hazardous gases, dust, and odor pollution were major focuses. Sulfur-containing gases were only regulated for a few industrial sectors, including refineries, ammonia production, coking, and non-ferrous metal metallurgy. In 1995, the Air Pollution Law was amended by adding a separate chapter on controlling air pollution, sourced from coal combustion, to tackle SO<sub>2</sub> emissions and the acid rain issue. The amendment in 1995 requested the designation of priority control zones for acid rain and SO<sub>2</sub> pollution (the Two Control Zones (TCZ) policy) [19]. Control zones for acid rain were designated for areas with high acidity of precipitation (pH ≤ 4.5),



**Fig. 3.** Key milestones and policy evolution of air pollution control in China, including control actions on the emissions of SO<sub>2</sub> and NO<sub>x</sub> and on ambient concentrations of PM<sub>2.5</sub>. FYP: Five-Year Plan.

high sulfur deposition exceeding the critical load, and high SO<sub>2</sub> emissions. The designation of control zones for SO<sub>2</sub> pollution was based on the surpassing of ambient SO<sub>2</sub> concentrations according to NAAQS limits. The TCZ policy proposed by the State Environmental Protection Administration (SEPA), now the Ministry of Ecology and Environment (MEE), were approved by the State Council of the People's Republic of China (State Council, hereinafter) as a nationwide plan. The plan designated 11.4% of the national territory (a total area of  $1.09 \times 10^6$  km<sup>2</sup>) as control zones, set a target to cap the total emissions of SO<sub>2</sub> by 2010, limited the production and use of high-sulfur coal, and required the adoption of desulfurization units for newly built and renovated coal-fired power plants within the TCZ.

Although installation of FGD systems was required by the 2000 amendment of the Air Pollution Law, the implementation was not well enforced [20]. Consequently, the total SO<sub>2</sub> emissions increased by nearly 50% during 2002–2006, driven by a surge in coal consumption (Fig. 2), which worsened acid rain pollution nationwide [10]. This situation was improved during the 11th Five-Year Plan (FYP) (2006–2010), which set a concrete target to mitigate total SO<sub>2</sub> emissions by 10%. The 11th FYP was considered a milestone during which the air pollution control target became mandatory for provincial and municipal governments for the first time, enforced by the fact that local leaders could receive a sanction in cadre evaluation if they failed [21].

In addition to political accountability, verification of SO<sub>2</sub> emissions and financial incentives were utilized as measures to achieve reduction targets [20]; for example, continuous emission monitoring systems (CEMSs) have been required at coal-fired power plants since 2007, in order to report the operational condition of FGD and supervise real-time SO<sub>2</sub> concentrations in the flue gas. Power plants with FGD operating properly (i.e., used in at least 90% of the electricity generated) have enjoyed a price premium of 0.015 CNY per kilowatt-hour. Otherwise, they are fined by an amount no less than the price premium. As a result, the adoption rate of FGD among thermal power plants increased from 14% in 2005 to 86% in 2010. SO<sub>2</sub> emissions from power generation, the largest anthropogenic source in China, decreased by 23% from 2005 to 2010, even though electricity generation increased by approximately 80% over the same period. Total SO<sub>2</sub> emissions in China decreased by 14% from 2005 to 2010 according to official statistics [22], representing the huge success of the 11th FYP in controlling SO<sub>2</sub> emissions. The results from satellite observations confirmed the same trend [23]. Acid rain issues were greatly mitigated [10].

### 3.2. Decoupling NO<sub>x</sub> emissions

On the other hand, regulations on NO<sub>x</sub> emissions in China have lagged in comparison with actions on SO<sub>2</sub>. The sources for NO<sub>x</sub> are distributed across multiple sectors involving high-temperature combustions, resulting in increased difficulty and higher costs for NO<sub>x</sub> emission mitigation. In addition to coal combustion processes, including thermal power and industrial plants, on-road vehicles and other fossil fuel combustion processes are important sources of NO<sub>x</sub> emissions. Total NO<sub>x</sub> emissions increased threefold during 1990–2010 (Fig. 1), as indicated by both bottom-up estimation and satellite observations [24]. Many large cities have experienced rapid motorization since 2000, resulting in their urban nitrogen dioxide (NO<sub>2</sub>) concentrations exceeding NAAQS. Increasing trends in the ratios of nitrate (NO<sub>3</sub><sup>-</sup>) to sulfate (SO<sub>4</sub><sup>2-</sup>) in both ambient PM<sub>2.5</sub> mass [25] and precipitation [10] have been observed in many regions of China. However, national mandates on controlling NO<sub>x</sub> emissions were not established until the 12th FYP (2011–2015).

Learning from the success of SO<sub>2</sub> emission control during the 11th FYP, the Chinese government set a similar mandatory target of reducing NO<sub>x</sub> emissions by 10% during the period of the 12th

FYP. Thermal power plants, which were responsible for at least 30% of the national total emissions of NO<sub>x</sub>, were the main sector targeted by policymakers [26]. Many of the policy instruments and management practices adopted to control SO<sub>2</sub> emissions (e.g., CEMS) were tailored for the mitigation of NO<sub>x</sub> emissions. In 2010, coal-fired power plants relied heavily on low-NO<sub>x</sub> burners (LNB), a moderate technology designed to control NO<sub>x</sub> emissions, while the adoption rate of more advanced technology, such as selective catalytic reduction (SCR), accounted for only 12%. During the 12th FYP, China tightened the emission standards for coal-fired power plants and set a NO<sub>x</sub> limit of 100 mg·m<sup>-3</sup>—the most stringent standard in the world at that time. The NO<sub>x</sub> limit was further revised to 50 mg·m<sup>-3</sup> in 2015, which is known as the ultra-low emission standard [27]. As a result, more than 80% of thermal power plants installed SCR to control NO<sub>x</sub> emissions by 2015. The total NO<sub>x</sub> emissions decreased by 10.9% during the 12th FYP, which reversed the rapid increasing trend of emissions from the previous decade [28,29].

### 3.3. Decoupling PM<sub>2.5</sub> concentration

The inclusion of PM<sub>2.5</sub> as a criteria pollutant in the 2012 amendment of NAAQS marked the strategic and philosophical evolution of air pollution control in China. Ambient PM<sub>2.5</sub> concentration had become a more transparent indicator for city residents in China than other air pollutants, as the public began to gradually understand the relationship between PM<sub>2.5</sub> pollution, atmospheric visibility, and health impact. The major objectives of the Air Pollution Prevention and Control Action Plan 2013–2017 (Action Plan, hereinafter) launched by the State Council were to improve air quality and reduce heavy pollution days nationwide. Unlike the previous TCZ policy, which was proposed by the SEPA and then approved by the State Council, the formulation of the Action Plan was directly led and promoted by the State Council for the first time in the area of air pollution control. The higher level of authority for the Action Plan reflected the fact that combating PM<sub>2.5</sub> pollution would require more coordinated efforts at the national and regional levels than ever before. Three developed regions—the BTH, Yangtze River Delta (YRD), and Pearl River Delta (PRD)—had more stringent targets: to reduce average PM<sub>2.5</sub> concentrations by 25%, 20%, and 15%, respectively, during 2013–2017. Although variation in the meteorological conditions from 2013 to 2017 slightly favored reducing PM<sub>2.5</sub> concentrations in the BTH region, emission reductions were the dominating factor for the decline of ambient PM<sub>2.5</sub> concentration and the associated health benefits [30]. Beijing set a goal that the annual average concentration in 2017 should be below 60 µg·m<sup>-3</sup>, which required a reduction within five years that was greater than 30% of the 2013 level (89 µg·m<sup>-3</sup>).

Fig. 4 summarizes the major action plans for PM<sub>2.5</sub> control implemented by the Chinese government since 2013. The policy framework shows a clear trend of gradually involving more categories of PM<sub>2.5</sub> precursors and emission sectors, since traditional control actions for a single pollutant or a single sector were insufficient to deliver substantial reductions in PM<sub>2.5</sub> concentrations as required by the Action Plan. One notable case is that the estimated SO<sub>2</sub> emissions and sulfate concentrations inversed from satellite observations both rebounded slightly during 2010–2011 due to a rapid increase in industrial activities and less stringent controls on the iron and steel industries (Fig. 2), which surpassed the reduction benefits of SO<sub>2</sub> controls on power plants [31]. As a result, a series of stringent control measures have been implemented in the industrial polluting sector, including strengthening industrial emissions standards, phasing out small-scale high-polluting factories, phasing out outdated industrial capacities, and upgrading industrial boilers [32]. Compared with the control measures for



other precursors (e.g., SO<sub>2</sub> and NO<sub>x</sub>), the controls on volatile organic compounds (VOCs) and NH<sub>3</sub> emissions before 2013 were considered to be inadequate in China [33]. A response surface modeling (RSM) study of the nonlinear response of PM<sub>2.5</sub> to precursor emissions in the most polluted BTH region suggested that PM<sub>2.5</sub> concentrations are primarily sensitive to the emissions of NH<sub>3</sub> and organic compounds (e.g., non-methane volatile organic compounds (NMVOCs), intermediate volatility organic compounds (IVOCs), and primary organic aerosols (POAs)) [34]. National emissions of NMVOCs had to be reduced by at least 36% from the 2012 level by 2030 in order to meet the PM<sub>2.5</sub> NAAQS [35]. Therefore, a comprehensive VOC control program was launched in 2015, targeting the petrochemical industry, organic chemical industry, surface coating industry, and packaging and printing industry. The RSM technique also quantified that the rapid increase in NH<sub>3</sub> emissions during 1990–2005 resulted in increases of 50%–60% in sulfate and nitrate aerosol concentrations [36,37]. Nitrate aerosol has become the leading component in PM<sub>2.5</sub> concentrations in the BTH region, and dominates severe haze events [38,39]. One important cause is that, now that ambient SO<sub>2</sub> concentrations have been successfully reduced, ammonia-rich and high-humidity conditions more significantly favor the partitioning of nitrate toward the particle phase [39,40]. Consequently, the central government has recognized the importance of ammonia emission controls and has focused on the agriculture sector (e.g., crop farming and aquaculture) in the following Three-Year Action Plan (2018–2020).

As Fig. 4 indicates, control actions have been implemented nationwide, with additional efforts in several key regions. Improving air quality and mitigating PM<sub>2.5</sub> pollution in core megacities (e.g., Beijing and Shanghai) have been attained due to increased attention from the central government and the public. Beijing provides a success story in tackling PM<sub>2.5</sub> pollution after two-decade-long efforts. In fact, the implementation of comprehensive air pollution controls in Beijing was initiated as early as the 1990s. During the 2008 Olympic Games, extensive control measures were implemented in Beijing and surrounding provinces to guarantee good air quality [41]. Ambient concentrations of CO, SO<sub>2</sub>, PM<sub>10</sub>, and NO<sub>2</sub> continuously decreased, even as the municipal GDP, resident population, vehicle stock, and energy consumption increased significantly from 1998 to 2013. Owing to the success of air pollution controls for the 2008 Olympic Games, the local leaders recognized that ambient PM<sub>2.5</sub> concentrations resulted from a combination of emissions, meteorological conditions, and

atmospheric chemistry. Furthermore, PM<sub>2.5</sub> pollution would not have been successfully controlled without coordinated efforts from surrounding provinces. Thus, the government experienced valuable learning regarding the source apportionment of PM<sub>2.5</sub> pollutions. The results from an analysis of one-year continuous monitoring data indicated that local sources were actually responsible for about two thirds of PM<sub>2.5</sub> mass concentrations in Beijing for 2013. Among local contributors, mobile sources (primarily on-road vehicles) were estimated to play a leading role (31%), followed by coal combustion (22%), industrial production (18%), and dust (15%) [42].

Prior to 2013, control measures focused on large-capacity coal-fired power plants and urban coal combustion. Since 2013, urban and rural households in Beijing have been provided with more subsidies to expedite the replacement of coal with electricity or natural gas. Households that choose coal-to-electricity heating renovation can receive subsidies equivalent to two thirds of the equipment acquisition costs and are eligible for a discount of up to 78% off the heating electricity bill [43]. As of 2017, the majority of Beijing households have become coal-free except for some in remote rural areas. As a direct result of coal elimination, the current annual SO<sub>2</sub> concentration is now below 10 μg·m<sup>-3</sup> in Beijing, indicating a reduction of over 90% in the past two decades. Further efforts are now being made to transform Beijing into a coal-free city.

Beijing also pioneered in controlling vehicle emissions in China. The formulation and enforcement of local standards for vehicle emissions and fuel quality in Beijing are essentially one step ahead of national requirements [44]. Advanced on-road monitoring techniques—particularly remote sensing and portable emission measurement systems—have been adopted to improve in-use compliance, and sustainable transportation systems have been promoted through substantial incentives and traffic management [43]. Urban residents are embracing more sustainable travel modes than ever, as bicycles, subway, and ground bus are now responsible for more than 60% of total trips in Beijing. New energy vehicles—mostly battery electric vehicles—accounted for nearly 3% of the total vehicle population by 2017.

Total emissions of major precursors in Beijing and the surrounding region are estimated to have decreased more rapidly during 2013–2017 than ever before. Emissions in Beijing have been reduced by 83% for SO<sub>2</sub>, 43% for NO<sub>x</sub>, 42% for VOCs, and 59% for primary PM<sub>2.5</sub>. Comprehensive air quality modeling results have

Implementation period								Targeted precursor					Sector					
2013	2014	2015	2016	2017	2018	2019	2020	PM	SO <sub>2</sub>	NO <sub>x</sub>	VOC	NH <sub>3</sub>	PP	IN	TR	DO	AR	
→ 12th FYP								√	√	√			√	√	√			
→ Air Pollution Prevention and Control Action Plan								√	√	√	√		√	√	√			
→ Enhancing air pollution control in the energy industry								√	√	√	√		√	√	√	√		
→ Action Plan for Retrofitting and Upgrading of Coal-Fired Power Plants								√	√	√	√		√					
→ Industrial Green Development Plan									√	√	√			√				
→ The Plans for the Protection of the Ecological Environment in Beijing–Tianjin–Hebei Coordinated Development								√	√	√	√		√	√	√	√		
→ Action Plan on VOC Reduction in Key Industries											√			√				
→ Comprehensive Work Plan for Energy Conservation and Emission Reduction									√	√	√		√	√	√	√		
→ Three-Year Action Plan								√	√	√	√	√	√	√	√	√	√	

Nationwide
  Key regions (e.g., BTH, YRD)
  Nationwide with more efforts in key regions

Fig. 4. Summary of the major control regulations in China since 2013. PP: power plants; IN: industry; TR: transport; DO: domestic sector; AR: agriculture.

confirmed that emission mitigation was the primary cause of the significant progress in alleviating  $PM_{2.5}$  pollution from  $89 \mu\text{g}\cdot\text{m}^{-3}$  in 2013 to  $58 \mu\text{g}\cdot\text{m}^{-3}$  in 2017 [45] (and  $42 \mu\text{g}\cdot\text{m}^{-3}$  in 2019). Among control measures, the renovation of coal boilers together with the promotion of cleaner residential fuels are estimated to have reduced  $PM_{2.5}$  concentrations by more than  $10 \mu\text{g}\cdot\text{m}^{-3}$ . Emission controls on local vehicles have also effectively mitigated  $NO_x$  emissions, and account for a reduction of  $2 \mu\text{g}\cdot\text{m}^{-3}$  in  $PM_{2.5}$  concentration [43].

Guided by the Action Plan, many other cities in China are following Beijing to reduce ambient  $PM_{2.5}$  concentrations (Fig. 5). The strategic promotion of ecological civilization has spurred stronger willingness, clearer targets, more supportive legislation and incentives, and more stringent implementation of control actions from central to local governments. All these factors work together with public engagement to achieve progress in controlling  $PM_{2.5}$  pollution and improving public health and welfare. During the Action Plan 2013–2017,  $PM_{2.5}$  concentrations decreased by 23% on average for all the municipal cities in China [46] (Fig. 5). Reductions for the three key regions were even more significant: 40% for BTH, 34% for YRD, and 28% for PRD. The average  $PM_{2.5}$  concentration in the PRD region was  $34 \mu\text{g}\cdot\text{m}^{-3}$ , which adhered to NAAQS for  $PM_{2.5}$ . This progress has motivated some local governments to actively target more stringent limits on  $PM_{2.5}$

(e.g., the WHO Interim Target-2 level of  $25 \mu\text{g}\cdot\text{m}^{-3}$ ) and to resolve other pressing issues such as  $O_3$  pollution.

#### 4. Emerging $O_3$ challenges in ecological civilization construction

Despite the substantial progress that has been made in addressing acid rain and  $PM_{2.5}$  issues (Fig. 6(a)), an unwanted increase in ground-level  $O_3$  pollution is still found in many regions across China, posing the more complex challenge of simultaneous continuous mitigation of  $PM_{2.5}$  pollution with an emerging  $O_3$  issue. As  $O_3$  pollution has been steadily increasing in most parts of eastern China [47–50],  $O_3$  issues have recently gained more attention among both the public and academia in China. As illustrated in Fig. 6(b), the average 90th percentiles in daily maximum 8 h  $O_3$  concentrations increased by 20.1% and 24.5% in 74 key cities and BTH, respectively, from 2013 to 2017. Furthermore, densely populated regions of eastern China were exposed to high  $O_3$  pollution, increasing the impacts of  $O_3$  on human health. The results of the GBD project [51] estimated that respiratory mortality attributable to long-term  $O_3$  exposure in adults is between 274 000 and 316 000 in China, equal to approximately 25%–29% of the deaths attributable to  $PM_{2.5}$  exposure [16]. It is necessary to integrate  $O_3$  and  $PM_{2.5}$  controls to ensure a pollution-free and healthy living environment in the development of ecological civilization in China.

The biggest challenge of  $O_3$  pollution control is rooted in the complexity of atmospheric photochemistry formation, which depends on  $O_3$ – $NO_x$ –VOC sensitivity diagnosis and meteorological conditions [52,53]. A control strategy designed to reduce emissions of  $NO_x$  or VOCs for  $PM_{2.5}$  control may result in an unexpected disadvantage for  $O_3$  reduction. Strong VOC-limiting systems are found in most urban areas of China [54,55], and  $O_3$  in urban areas can be enhanced if few or no VOCs are controlled simultaneously with  $NO_x$  [56]. For example, during the Action Plan from 2013 to 2017, anthropogenic emissions of  $NO_x$  were reduced by 21% but VOCs increased by 2% nationwide [57]. Such combinational changes in  $NO_x$ –VOC emissions might be beneficial for reducing  $PM_{2.5}$  but not for  $O_3$ , as shown by the decreased  $PM_{2.5}$  concentrations and increased  $O_3$  concentrations during the same period (Fig. 6). It is notable that the increase in  $O_3$  concentration during the Action Plan period is also attributable to variation in meteorological conditions—specifically, changing temperature and short-wave radiation. Therefore, full consideration of meteorology and optimal mitigation of precursor emissions are critical in designing an effective  $O_3$  control strategy.

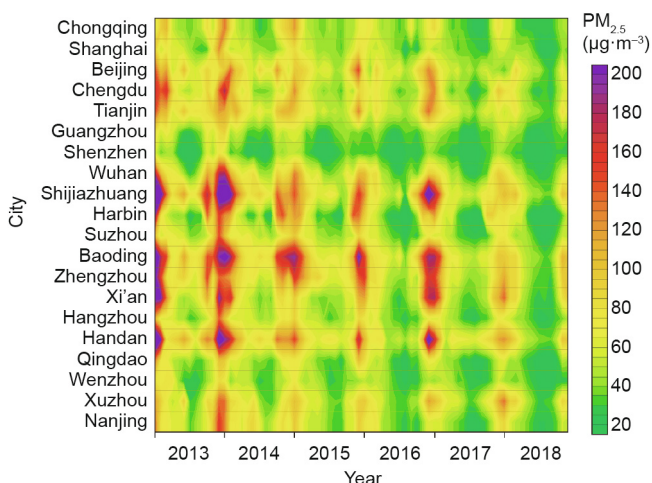


Fig. 5. Trend of annual average  $PM_{2.5}$  concentrations for the top 20 most populated cities in China from 2013 to 2018. Observed  $PM_{2.5}$  concentrations were obtained from the China National Environmental Monitoring Center.

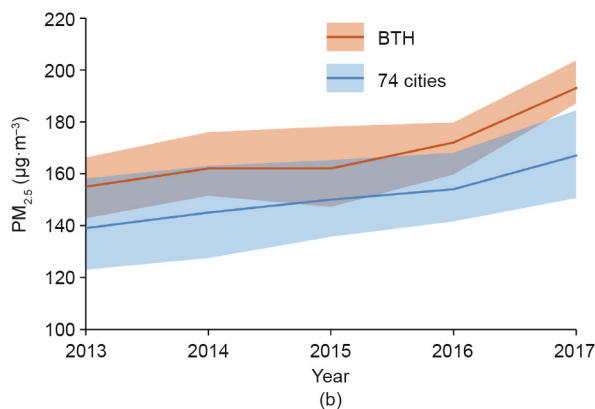
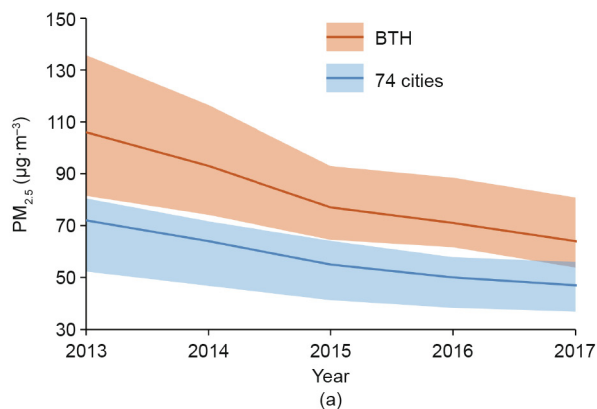


Fig. 6. (a) Trends in observed annual  $PM_{2.5}$  concentrations for BTH and 74 key cities between 2013 and 2017; (b) trends expressed as 90th percentiles in maximum daily average 8 h  $O_3$  concentration. Shaded areas represent the 25th–75th percentile range. Data are available in Table S1 in Appendix A. Details on monthly  $PM_{2.5}$  concentration for 74 key cities are summarized in Tables S2–S6.  $PM_{2.5}$  and  $O_3$  concentrations in the BTH region and 74 cities were obtained from the China National Environmental Monitoring Center.

In addition, O<sub>3</sub> pollution can be intensified by PM<sub>2.5</sub> mitigation through the reaction pathway affecting the termination of O<sub>3</sub> formation chemistry. Reduced particle surfaces, where the heterogeneous reactions involving atmospheric oxidants occur, may lead to termination reactions of some O<sub>3</sub>-consuming radicals (e.g., N<sub>2</sub>O<sub>5</sub> and HO<sub>2</sub>) [58,59]. The reduced PM<sub>2.5</sub> suppresses the termination reactions of those radicals, thus enhancing the photochemical reactions and O<sub>3</sub> concentrations. Li et al. [60] found that the suppressed aerosol sink of hydroperoxyl (HO<sub>2</sub>) radicals, due to the decrease of PM<sub>2.5</sub> over the 2013–2017 period, was an important factor in the increase of O<sub>3</sub> in the North China Plain.

Another complexity comes from the influence of the aerosol radiative effect (ADE) on O<sub>3</sub> formation. It is well known that the ADE induces a reduction in solar radiation that leads to lower photolysis rates and less O<sub>3</sub> production [61]. Thus, it is expected that substantial PM<sub>2.5</sub> reductions might lead to a potential risk of rising O<sub>3</sub> levels [54,62,63]. Furthermore, the ADE can change vertical temperature profiles and corresponding atmospheric stability, atmospheric ventilation, cloud cover, and rainfall, which may also influence O<sub>3</sub> concentrations. The mechanism of meteorological influences on O<sub>3</sub> exhibits strong seasonality, which is more complicated than the direct impacts from ADE on radiation [64–68]. A previous study suggested that O<sub>3</sub> would increase by up to 2%–3% in eastern China due to an ADE-induced rise in precursor concentrations, stemming from the stabilization of the atmosphere, which is associated with reduced planetary boundary layer height and ventilation [68].

In conjunction with strengthened PM<sub>2.5</sub> controls, O<sub>3</sub> pollution could be more challenging in future. It is important to consider the O<sub>3</sub> response to the PM<sub>2.5</sub> control policy, and to achieve optimum benefit for human health and ecological civilization. The control strategy should focus on common precursors (i.e., NO<sub>x</sub> and NMVOCs) for both O<sub>3</sub> and PM<sub>2.5</sub>; in particular, it should implement simultaneous NMVOC controls with NO<sub>x</sub> controls under VOC-limited conditions, such as in urban areas and during winter time. The challenge is that the control of NO<sub>x</sub> and VOCs is usually more difficult and expensive than that of other pollutants. Development of cost-effective control technologies can largely facilitate the control process. Many studies have found that regional emission sources have much more influence on O<sub>3</sub> concentration than local sources [53,69], suggesting that more intensive regional joint controls are necessary.

## 5. Synergy between climate mitigation and air pollution control: A win–win opportunity in the ecological civilization era

Significant co-benefits exist from air pollution control and mitigation of GHG emissions in China, as both air pollutants and GHGs are generally emitted from the same sources, such as the combustion of fossil fuels—particularly coal [70]. In 2013, coal combustion was responsible for 8 Gt or 81% of total CO<sub>2</sub> emissions in China, and contributed 35%–46% (i.e., 18–28 μg·m<sup>-3</sup>) of total PM<sub>2.5</sub> concentrations nationwide [71]. Over the past decades, the energy efficiency promoted from a series of FYPs in China has resulted in evident savings in coal consumption and thus reductions in both air pollution and carbon emissions. In northern China, household heating has undergone coal-to-clean fuel transitions since 2013, which are regarded as a necessary means to tackle the heavy PM<sub>2.5</sub> pollution in winter [43]. Measures to control coal consumption with a priority on reducing air pollutants also reduce CO<sub>2</sub> emissions simultaneously. As promoted in the Action Plan 2013–2017, options such as restricting the use of fossil fuels with high emission factors, promoting zero-emission vehicles, and renewable energy have all led to an important reduction in both air pollutants and GHGs [72–74].

In the short and medium term, air pollution controls are expected to act as a leading driver in the synergistic path for realization of these co-benefits. The Three-Year Action Plan for winning the “blue sky defense battle” (2018–2020) explicitly emphasizes the co-benefits between air pollution control and climate change mitigation. In particular, it introduces more stringent measures to limit coal use and encourage clean transportation in China, which will be clearly beneficial for reducing CO<sub>2</sub> emissions in the related sectors. In addition, the 2035 “beautiful China” targets for air quality (e.g., to decrease annual PM<sub>2.5</sub> to below 35 μg·m<sup>-3</sup>) are expected to impose stricter requirements for de-carbonization of the energy system in China in comparison with the climate targets that were committed to in China’s Nationally Determined Contributions (NDCs) [75]. In the long term, the WHO air quality guidelines (AQGs) are considered to be the ultimate objective for air quality improvements in China during the process of developing ecological civilization through 2050.

At the same time, realizing the global 1.5 °C/2 °C climate target will require China to significantly reduce its GHG emissions by 2050 and beyond, as indicated in a large body of scenario analysis [76–78]. Thus, both climate mitigation and air pollution control will become increasingly challenging in the long term. It is clear that a dynamic understanding of the synergistic effects between air pollution control and climate mitigation in China at different stages is critical for the country to follow a cost-effective pathway toward sustainable development and ecological civilization.

Air quality and climate systems can enhance each other through a number of positive feedback loops. Global climate change may weaken the strength of Asian monsoons, resulting in frequent heat waves and episodes of stagnant air, and leading to increasing O<sub>3</sub> and PM<sub>2.5</sub> concentrations and high pollution episodes [79,80]. In addition, the increase of tropospheric O<sub>3</sub> together with the reduction of stratospheric O<sub>3</sub> will likely increase surface and low-level atmospheric temperatures, which will in turn exacerbate O<sub>3</sub> pollution, creating a vicious circle [81–84]. Mitigation of climate change benefits air quality improvement in this sense, and vice versa. It is true that the operation of control equipment for air pollutants will consume additional energy, which—if the energy is fossil based—will lead to a marginal increase in GHG emissions [85–87]. Furthermore, some mitigation efforts on aerosols such as sulfates for the purpose of reducing acid rain and PM<sub>2.5</sub> pollution might escalate warming effects [88–93], as they act as cooling radiative forcing in the climate system. Given the existing significant reduction in SO<sub>2</sub> emissions in China, the warming effects due to further reduction in sulfate aerosols are expected to be limited in the future. It is clear that the co-benefits largely exceed the tradeoffs between air pollution control and GHG mitigation, in the case of China.

## 6. Conclusions and implications

This paper reviews the three-decade progress that has been made in air pollution control in China, highlighting a strategic transformation from emission control toward air quality management. In the early stage (starting in the 1980s), acid rain was the prominent cross-regional air pollution problem in China. Substantial control measures on SO<sub>2</sub> emissions from coal combustion sectors—particularly the SO<sub>2</sub> emission capping policy since the 11th FYP—resolved the deterioration of the acid rain issue around 2007. The 2012 NAAQS amendment introduced PM<sub>2.5</sub> as a newly added criteria pollutant, marking a transition to an air-quality-oriented strategy. Emission controls for a single pollutant or a single sector are inadequate to meet the stringent target of PM<sub>2.5</sub> concentration reduction. China has implemented a series of control action plans, which have been redesigned to mitigate multiple precursors from multiple sectors through extensive coordination at



the regional or national level. During the Action Plan 2013–2017, many cities effectively decoupled PM<sub>2.5</sub> concentrations and socioeconomic development. PM<sub>2.5</sub> concentrations decreased by an average of 23% during this period for all municipal cities in China. Beijing has set a successful example as a pioneer in energy transition and transportation emission controls, and has achieved significant progress in mitigating PM<sub>2.5</sub> pollution from 89 µg·m<sup>-3</sup> in 2013 to 58 µg·m<sup>-3</sup> in 2017.

Despite these achievements in air quality management, China is facing both challenges and opportunities to realize “beautiful China” targets and to absolutely decouple emissions of CO<sub>2</sub> and air pollutants from economic growth in the ecological civilization era. On the one hand, O<sub>3</sub> pollution is emerging and could be more challenging in future. On the other hand, the coupling of climate mitigation and air pollution control exhibits significant co-benefits that can lead to a win–win opportunity. An integrated management framework linking energy, environment, health, and climate units should be established for the synergistic mitigation of PM<sub>2.5</sub>, O<sub>3</sub>, and GHGs. Such a framework is expected to consist of three core components: a precise and dynamic system to reproduce the condition of the regional and global atmospheric environment; a coordinated and highly efficient treatment system to simultaneously mitigate the emissions of air pollutants and GHGs; and a comprehensive decision-supporting platform to form a synergistic mitigation pathway and optimal technical combination. Ultimately, a multi-objective and multi-benefit roadmap is required in order for China to attain air quality standards domestically and to fulfill international agreements concerning GHGs, and so forth.

## Acknowledgements

This work is supported by the National Key Research Development Program of China (2016YFC0208901 and 2017YFC0212100), and the National Natural Science Foundation of China (71722003 and 71690244). Dr. Shuxiao Wang acknowledges support from the Samsung Advanced Institute of Technology and the Tencent Foundation through the XPLOER PRIZE.

## Compliance with ethics guidelines

Xi Lu, Shaojun Zhang, Jia Xing, Yunjie Wang, Wenhui Chen, Dian Ding, Ye Wu, Shuxiao Wang, Lei Duan, and Jiming Hao declare that they have no conflict of interest or financial conflicts to disclose.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.eng.2020.03.014>.

## References

- [1] World Bank Group. China overview 2017 [Internet]. Beijing: The World Bank in China; 2019 [cited 2020 Apr 23]. Available from: <https://www.worldbank.org/en/country/china/overview#1>.
- [2] British Petroleum. BP statistical review of world energy. London: British Petroleum Co.; 2018.
- [3] Zheng C, Zhao C, Li Y, Wu X, Zhang K, Gao J, et al. Spatial and temporal distribution of NO<sub>2</sub> and SO<sub>2</sub> in Inner Mongolia urban agglomeration obtained from satellite remote sensing and ground observations. *Atmos Environ* 2018;188:50–9.
- [4] Zhang K, Zhao C, Fan H, Yang Y, Sun Y. Toward understanding the differences of PM<sub>2.5</sub> characteristics among five China urban cities. *Asia-Pac J Atmos Sci* 2019;5:1–10.
- [5] Zheng B, Tong D, Li M, Liu F, Hong C, Geng G, et al. Trends in China's anthropogenic emissions since 2010 as the consequence of clean air actions. *Atmos Chem Phys* 2018;18:14095–111.
- [6] International Energy Agency. CO<sub>2</sub> emissions from fuel combustion. Paris: International Energy Agency; 2019.
- [7] Report on the state of the ecology and environment in China [Internet]. Beijing: Ministry of Ecology and Environment of the People's Republic of China; 2018 Aug 1 [cited 2020 Jun 8]. Available from: <http://english.mee.gov.cn/Resources/Reports/soe/SOEE2017/201808/P020180801597738742758.pdf>.
- [8] Larssen T, Lydersen E, Tang D, He Y, Gao J, Liu H, et al. Acid rain in China. *Environ Sci Technol* 2006;40(2):418–25.
- [9] Griggs D, Stafford-Smith M, Gaffney O, Rockström J, Öhman MC, Shyamsundar P, et al. Sustainable development goals for people and planet. *Nature* 2013;495(7441):305–7.
- [10] China Meteorological Administration. Annual report of acid rain monitoring in China. Beijing: China Meteorological Administration; 2014. Chinese.
- [11] Schwartz J, Dockery DW, Neas LM. Is daily mortality associated specifically with fine particles? *J Air Waste Manage* 1996;46(10):927–39.
- [12] Identification of nonattainment classification and deadlines for submission of state implementation plan (SIP) provisions for the 1997 & 2006 fine particle National Ambient Air Quality Standards (NAAQS)—fact sheet [Internet]. Washington, DC: United States Environmental Protection Agency; [updated 2020 May 22; cited 2020 Jun 8]. Available from: [https://www.epa.gov/sites/production/files/2016-04/documents/20140428\\_factsheet\\_nonattainment.pdf](https://www.epa.gov/sites/production/files/2016-04/documents/20140428_factsheet_nonattainment.pdf).
- [13] He K, Yang F, Ma Y, Zhang Q, Yao X, Chan CK, et al. The characteristics of PM<sub>2.5</sub> in Beijing, China. *Atmos Environ* 2001;35(29):4959–70.
- [14] Andersson A, Deng J, Du K, Zhang M, Yan C, Sköld M, et al. Regionally-varying combustion sources of the January 2013 severe haze events over eastern China. *Environ Sci Technol* 2015;49(4):2038–43.
- [15] Ma Z, Hu X, Sayer AM, Levy R, Zhang Q, Xue Y, et al. Satellite-based spatiotemporal trends in PM<sub>2.5</sub> concentrations: China, 2004–2013. *Environ Health Persp* 2015;124(2):184–92.
- [16] Cohen AJ, Brauer M, Burnett R, Anderson HR, Frostad J, Estep K, et al. Estimates and 25-year trends of the global burden of disease attributable to ambient air pollution: an analysis of data from the Global Burden of Diseases Study 2015. *Lancet* 2017;389(10082):1907–18.
- [17] World Bank Group. The cost of air pollution. Washington, DC: World Bank Group; 2016.
- [18] Standing Committee of the National People's Congress. Law of the People's Republic of China on the Prevention and Control of Atmospheric Pollution. (Sep 5, 1987). Chinese.
- [19] Hao J, Wang S, Liu B, He K. Designation of acid rain and SO<sub>2</sub> control zones and control policies in China. *J Environ Sci Health A* 2000;35(10):1901–14.
- [20] Schreifels JJ, Fu Y, Wilson EJ. Sulfur dioxide control in China: policy evolution during the 10th and 11th Five-Year Plans and lessons for the future. *Energy Policy* 2012;48:779–89.
- [21] Jin Y, Andersson H, Zhang S. Air pollution control policies in China: a retrospective and prospects. *Int J Environ Res Pub Health* 2016;13(12):1219.
- [22] Ministry of Ecology and Environment of the People's Republic of China. China environmental statistical bulletin 2010. Beijing: Ministry of Ecology and Environment of the People's Republic of China; 2012. Chinese.
- [23] Wang S, Zhang Q, Martin RV, Philip S, Liu F, Li M, et al. Satellite measurements oversee China's sulfur dioxide emission reductions from coal-fired power plants. *Environ Res Lett* 2015;10(11):114015.
- [24] De Foy B, Lu Z, Streets DG. Satellite NO<sub>2</sub> retrievals suggest China has exceeded its NO<sub>x</sub> reduction goals from the twelfth Five-Year Plan. *Sci Rep* 2016;6:35912.
- [25] Yang F, Tan J, Zhao Q, Du Z, He K, Ma Y, et al. Characteristics of PM<sub>2.5</sub> speciation in representative megacities and across China. *Atmos Chem Phys* 2011;11(11):5207–19.
- [26] Zhao B, Wang SX, Liu H, Xu JY, Fu K, Klimont Z, et al. NO<sub>x</sub> emissions in China: historical trends and future perspectives. *Atmos Chem Phys* 2013;13(19):9869–97.
- [27] Liu X, Gao X, Wu X, Yu W, Chen L, Ni R, et al. Updated hourly emissions factors for Chinese power plants showing the impact of widespread ultralow emissions technology deployment. *Environ Sci Technol* 2019;53(5):2570–8.
- [28] Ministry of Ecology and Environment of the People's Republic of China. China environmental statistic bulletin 2015. Beijing: Ministry of Ecology and Environment of the People's Republic of China; 2016. Chinese.
- [29] Liu F, Zhang Q, Zheng B, Tong D, Yan L, Zheng Y, et al. Recent reduction in NO<sub>x</sub> emissions over China: synthesis of satellite observations and emission inventories. *Environ Res Lett* 2016;11(11):114002.
- [30] Ding D, Xing J, Wang S, Liu K, Hao J. Estimated contributions of emissions controls, meteorological factors, population growth, and changes in baseline mortality to reductions in ambient PM<sub>2.5</sub> and PM<sub>2.5</sub>-related mortality in China, 2013–2017. *Environ Health Perspect* 2019;127(6):067009.
- [31] Geng G, Zhang Q, Tong D, Li M, Zheng Y, Wang S, et al. Chemical composition of ambient PM<sub>2.5</sub> over China and relationship to precursor emissions during 2005–2012. *Atmos Chem Phys* 2017;17(14):9187–203.
- [32] Zhang Q, Zheng Y, Tong D, Shao M, Wang S, Zhang Y, et al. Drivers of improved PM<sub>2.5</sub> air quality in China from 2013 to 2017. *Proc Natl Acad Sci USA* 2019;116(49):24463–9.
- [33] Wang J, Zhao B, Wang S, Yang F, Xing J, Morawska L, et al. Particulate matter pollution over China and the effects of control policies. *Sci Total Environ* 2017;584–585:426–47.
- [34] Zhao B, Wu W, Wang S, Xing J, Chang X, Liou KN, et al. A modeling study of the nonlinear response of fine particles to air pollutant emissions in the Beijing–Tianjin–Hebei region. *Atmos Chem Phys* 2017;17(19):12031–50.
- [35] Wang S, Zhao B, Wu Y, Hao J. Target and measures to prevent and control ambient fine particle pollution in China. *China Environ Manage* 2015;2:37–43. Chinese.
- [36] Wang S, Xing J, Jang C, Zhu Y, Fu JS, Hao J. Impact assessment of ammonia emissions on inorganic aerosols in East China using response surface modeling technique. *Environ Sci Technol* 2011;45(21):9293–300.



- [37] Wang L, Wei Z, Yang J, Zhang Y, Zhang FF, Su J, et al. The 2013 severe haze over the southern Hebei, China: model evaluation, source apportionment, and policy implications. *Atmos Chem Phys* 2013;13(11):28395–451.
- [38] Xu Q, Wang S, Jiang J, Bhattacharai N, Li X, Chang X, et al. Nitrate dominates the chemical composition of PM<sub>2.5</sub> during haze event in Beijing, China. *Sci Total Environ* 2019;689:1293–303.
- [39] Wang Y, Chen Y, Wu Z, Shang D, Bian Y, Du Z, et al. Mutual promotion between aerosol particle liquid water and particulate nitrate enhancement leads to severe nitrate-dominated particulate matter pollution and low visibility. *Atmos Chem Phys* 2020;20(4):2161–75.
- [40] Li H, Zhang Q, Zheng B, Chen C, Wu N, Guo H, et al. Nitrate-driven urban haze pollution during summertime over the North China Plain. *Atmos Chem Phys* 2018;18:5293–306.
- [41] Wang S, Zhao M, Xing J, Wu Y, Zhou Y, Lei Y, et al. Quantifying the air pollutants emission reduction during the 2008 Olympic Games in Beijing. *Environ Sci Technol* 2010;44(7):2490–6.
- [42] Beijing Municipal Ecology and Environment Bureau. Source apportionment results of PM<sub>2.5</sub> in Beijing. Beijing: Beijing Municipal Ecology and Environment Bureau; 2014. Chinese.
- [43] United Nations Environment Programme. A review of 20 years' air pollution control in Beijing. Nairobi: United Nations Environment Programme; 2019.
- [44] Wu Y, Zhang S, Hao J, Liu H, Wu X, Hu J, et al. On-road vehicle emissions and their control in China: a review and outlook. *Environ Sci Technol* 2017;57(4):332–49.
- [45] Cheng J, Su J, Cui T, Li X, Dong X, Sun F, et al. Dominant role of emission reduction in PM<sub>2.5</sub> air quality improvement in Beijing during 2013–2017: a model-based decomposition analysis. *Atmos Chem Phys* 2019;19(9):6125–46.
- [46] Ministry of Ecology and Environment of the People's Republic of China. China environmental statistic bulletin 2017. Beijing: Ministry of Ecology and Environment of the People's Republic of China; 2018. Chinese.
- [47] Tang G, Li X, Wang Y, Xin J. Surface ozone trend details and interpretations in Beijing, 2001–2006. *Atmos Chem Phys* 2009;9(22):8813–23.
- [48] Gao W, Tie X, Xu J, Huang R, Mao X, Zhou G, et al. Long-term trend of O<sub>3</sub> in a mega city (Shanghai), China: characteristics, causes, and interactions with precursors. *Sci Total Environ* 2017;603–4:425–33.
- [49] Li J, Lu K, Lv W, Li J, Zhong L, Ou Y, et al. Fast increasing of surface ozone concentrations in Pearl River Delta characterized by a regional air quality monitoring network during 2006–2011. *J Environ Sci* 2014;26(1):23–36.
- [50] Verstraeten WW, Neu JL, Williams JE, Bowman KW, Worden JR, Boersma KF. Rapid increases in tropospheric ozone production and export from China. *Nature Geosci* 2015;8:690–5.
- [51] Malley CS, Henze DK, Kuylenstierna JCI, Vallack HW, Davila Y, Anenberg SC, et al. Updated global estimates of respiratory mortality in adults  $\geq$  30 years of age attributable to long-term ozone exposure. *Environ Health Persp* 2017;125(8):087021.
- [52] Tang G, Wang Y, Li X. Spatial-temporal variations in surface ozone in northern China as observed during 2009–2010 and possible implications for future air quality control strategies. *Atmos Chem Phys* 2012;12(5):2757–76.
- [53] Xing J, Wang SX, Jang C, Zhu Y, Hao JM. Nonlinear response of ozone to precursor emission changes in China: a modeling study using response surface methodology. *Atmos Chem Phys* 2011;11(10):5027–44.
- [54] Anger A, Dessens O, Xi F, Barker T, Wu R. China's air pollution reduction efforts may result in an increase in surface ozone levels in highly polluted areas. *Ambio* 2016;45(2):254–65.
- [55] Wang N, Lyu X, Deng X, Huang X, Jiang F, Ding A. Aggravating O<sub>3</sub> pollution due to NO<sub>x</sub> emission control in eastern China. *Environ Sci Technol* 2019;677:732–44.
- [56] Xing J, Ding D, Wang S, Zhao B, Jiang C, Wu W, et al. Quantification of the enhanced effectiveness of NO<sub>x</sub> control from simultaneous reductions of VOC and NH<sub>3</sub> for reducing air pollution in the Beijing-Tianjin-Hebei region, China. *Atmos Chem Phys* 2018;18(11):7799–814.
- [57] Zheng B, Tong D, Li M, Liu F, Hong C, Geng G, et al. Trends in China's anthropogenic emissions since 2010 as the consequence of clean air actions. *Atmos Chem Phys* 2018;18(19):14095–111.
- [58] Lou S, Liao H, Zhu B. Impacts of aerosols on surface-layer ozone concentrations in China through heterogeneous reactions and changes in photolysis rates. *Atmos Environ* 2014;85(2):123–38.
- [59] Tie X, Madronich S, Walters S, Edwards DP, Ginoux P, Mahowald N, et al. Assessment of the global impact of aerosols on tropospheric oxidants. *J Geophys Res* 2005;110(D3):204.
- [60] Li K, Jacob DJ, Liao H, Shen L, Zhang Q, Bates KH. Anthropogenic drivers of 2013–2017 trends in summer surface ozone in China. *Proc Natl Acad Sci USA* 2019;116(2):422–7.
- [61] Benas N, Mourtanou E, Kouvarakis G, Baics A, Mihalopoulos N, Vardavas I. Surface ozone photolysis rate trends in the Eastern Mediterranean: modeling the effects of aerosols and total column ozone based on Terra MODIS data. *Atmos Environ* 2013;74:1–9.
- [62] Wang J, Allen DJ, Pickering KE, Li Z, He H. Impact of aerosol direct effect on East Asian air quality during the EAST-AIRE campaign. *J Geophys Res* 2016;121(11):6534–54.
- [63] Bian H, Han S, Tie X, Sun M, Liu A. Evidence of impact of aerosols on surface ozone concentration in Tianjin, China. *Atmos Environ* 2007;41(22):4672–81.
- [64] Xing J, Mathur R, Pleim J, Hogrefe C, Gan CM, Wong DC, et al. Air pollution and climate response to aerosol direct radiative effects: a modeling study of decadal trends across the northern hemisphere. *J Geophys Res* 2016;120(23):12221–36.
- [65] Wang JD, Wang SX, Jiang JK, Ding A, Zheng M, Zhao B, et al. Impact of aerosol-meteorology interactions on fine particle pollution during China's severe haze episode in January 2013. *Environ Res Lett* 2014;9(9):094002.
- [66] Jacobson MZ. Control of fossil-fuel particulate black carbon and organic matter, possibly the most effective method of slowing global warming. *J Geophys Res* 2002;107(D19):1–22.
- [67] Jacobson MZ. Short-term effects of controlling fossil-fuel soot, biofuel soot and gases, and methane on climate, Arctic ice, and air pollution health. *J Geophys Res* 2010;115(D14):209.
- [68] Xing J, Mathur R, Pleim J, Hogrefe C, Gan CM, Wong DC, et al. Air pollution and climate response to aerosol direct radiative effects: a modeling study of decadal trends across the northern hemisphere. *J Geophys Res* 2015;120(23):12221–36.
- [69] Xing J, Wang S, Zhao B, Wu W, Ding D, Jiang C, et al. Quantifying nonlinear multi-regional contributions to ozone and fine particles using an updated response surface modeling technique. *Environ Sci Technol* 2017;51(20):11788–98.
- [70] Oh I, Yoo WJ, Yoo Y. Impact and interactions of policies for mitigation of air pollutants and greenhouse gas emissions in Korea. *Int J Environ Res Public Health* 2019;16(7):1161.
- [71] Ma Q, Cai S, Wang S, Zhao B, Martin RV, Brauer M, et al. Impact of coal burning on ambient PM<sub>2.5</sub> pollution in China. *Atmos Chem Phys* 2017;17:4477–91.
- [72] Ke W, Zhang S, Wu Y, Zhao B, Wang S, Hao J. Assessing the future vehicle fleet electrification: the impacts on regional and urban air quality. *Environ Sci Technol* 2016;51(2):1007–16.
- [73] He X, Zhang S, Wu Y, Wallington TJ, Lu X, Tamor MA, et al. Economic and climate benefits of electric vehicles in China, the United States, and Germany. *Environ Sci Technol* 2019;53(18):11013–22.
- [74] Liang X, Zhang S, Wu Y, Xing J, He X, Zhang KM, et al. Air quality and health benefits from fleet electrification in China. *Nat Sustain* 2019;2(10):962–71.
- [75] Li H, Tan X, Guo J, Zhu K, Huang C. Study on an implementation scheme of synergistic emission reduction of CO<sub>2</sub> and air pollutants in China's steel industry. *Sustainability* 2019;11(2):352.
- [76] Pan X, Chen W, Clarke LE, Wang L, Liu G. China's energy system transformation towards the 2 °C goal: implications of different effort-sharing principles. *Energy Policy* 2017;103:116–26.
- [77] Millar RJ, Fuglestedt JS, Friedlingstein P, Rogelj J, Grubb MJ, Matthews HD, et al. Emission budgets and pathways consistent with limiting warming to 1.5 °C. *Nat Geosci* 2017;10:741–50.
- [78] Meinshausen M, Meinshausen N, Hare W, Raper SCB, Frieler K, Knutti R, et al. Greenhouse-gas emission targets for limiting global warming to 2 °C. *Nature* 2009;458:1158–62.
- [79] Fiore AM, Naik V, Leibensperger EM. Air quality and climate connections. *J Air Waste Manage* 2015;65(6):645–85.
- [80] Silva RA, West JJ, Zhang Y, Anenberg SC, Lamarque JF, Shindell DT, et al. Global premature mortality due to anthropogenic outdoor air pollution and the contribution of past climate change. *Environ Res Lett* 2013;8(3):034005.
- [81] Bloomer BJ, Stehr JW, Piety CA, Salawitch RJ, Dickerson RR. Observed relationships of ozone air pollution with temperature and emissions. *Geophys Res Lett* 2009;36(9):L09803.
- [82] Rasmussen D, Fiore A, Naik V, Horowitz LW, McGinnis SJ, Schultz MG. Surface ozone-temperature relationships in the eastern US: a monthly climatology for evaluating chemistry-climate models. *Atmos Environ* 2012;47:142–53.
- [83] Miller AJ, Nagatani RM, Tiao GC, Niu XF, Reinsel GC, Wuebbles DJ, et al. Comparisons of observed ozone and temperature trends in the lower stratosphere. *Geophys Res Lett* 1992;19(9):929–32.
- [84] Rood RB, Douglass AR. Interpretation of ozone temperature correlations: 1. theory. *J Geophys Res-Atmos* 1985;90(D3):5733–43.
- [85] Feng X, Lugovoy O, Qin H. Co-controlling CO<sub>2</sub> and NO<sub>x</sub> emission in China's cement industry: an optimal development pathway study. *Adv Clim Change Res* 2018;9(1):34–42.
- [86] Mao X, Zeng A, Hu T, Zhou J, Xing Y, Liu S. Co-control of local air pollutants and CO<sub>2</sub> in the Chinese iron and steel industry. *Environ Sci Technol* 2013;47(21):12002–10.
- [87] Zhou J, Mao XQ, Hu T, Zeng A, Xing YK, Corsetti G. Implications of the 11th and 12th Five-Year Plans for energy conservation and CO<sub>2</sub> and air pollutants reduction: a case study from the city of Urumqi, China. *J Clean Prod* 2015;112:1767–77.
- [88] Liu S, Xing J, Zhao B, Wang J, Wang S, Zhang X, et al. Understanding of aerosol-climate interactions in China: aerosol impacts on solar radiation, temperature, cloud, and precipitation and its changes under future climate and emission scenarios. *Curr Pollut Rep* 2019;5:36–51.
- [89] Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, et al. Climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge: Cambridge University Press; 2013.
- [90] Liao H, Chang W, Yang Y. Climatic effects of air pollutants over China: a review. *Adv Atmos Sci* 2015;32(1):115–39.
- [91] Shindell D, Faluvegi G. Climate response to regional radiative forcing during the twentieth century. *Nat Geosci* 2009;2(4):294.
- [92] Kaiser DP, Qian Y. Decreasing trends in sunshine duration over China for 1954–1998: indication of increased haze pollution? *Geophys Res Lett* 2002;29(21):2042.
- [93] Chen LX, Zhang B, Zhu WQ, Zhou XJ, Luo YF, Zhou ZJ, et al. Variation of atmospheric aerosol optical depth and its relationship with climate change in China east of 100 E over the last 50 years. *Theor Appl Climatol* 2009;96(1–2):191–9.