Science of the Total Environment 663 (2019) 329–337

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Air pollution reduction in China: Recent success but great challenge for the future

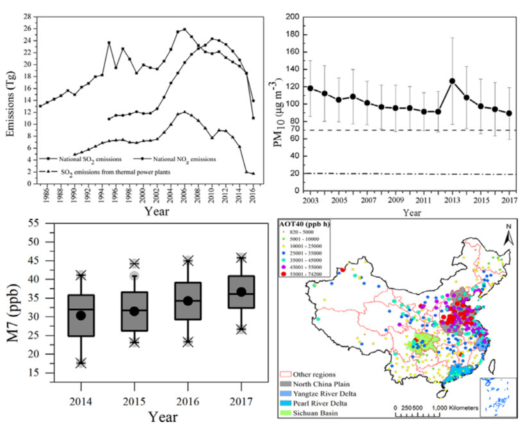
Yingying Zeng [a], Yuanfei Cao [b], Xue Qiao [b,c,d], Barnabas C. Seyler [a], Ya Tang [a,c,d,*]

[a] Department of Environment, College of Architecture and Environment, Sichuan University, No. 24, South Section One, First Ring Road, Chengdu, Sichuan 610065, China
[b] Institute of New Energy and Low-Carbon Technology, Sichuan University, No. 24, South Section One, First Ring Road, Chengdu, Sichuan 610065, China
[c] Healthy Food Evaluation Research Center, Sichuan University, No. 24, South Section One, First Ring Road, Chengdu, Sichuan 610065, China
[d] State Key Laboratory of Hydraulics and Mountain River Engineering, Sichuan University, No. 24, South Section One, First Ring Road, Chengdu, Sichuan 610065, China

HIGHLIGHTS

- SO_2, NO_x, and particulate matter have been reduced but O_3 pollution is increasing.
- Significant emission reduction has occurred largely by robust administrative power.
- Economic costs should be considered for sustainable pollution control measures.
- More research is needed to quantify O_3 effects on vegetation and human health.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 30 September 2018
Received in revised form 16 January 2019
Accepted 18 January 2019
Available online 23 January 2019

Editor: Alessandra De Marco

Keywords:
Air pollution
Pollution reduction
Policies
Challenges
Administrative power

ABSTRACT

China's rapid economic growth has caused severe air pollution, raising serious concerns about the growing evidence of its negative health, environmental, and economic impacts. Consequently, the Chinese government has implemented a number of policies and measures to reduce air pollution. Relying on published information over the last three decades in China, we analyzed trends in air pollutant emissions (SO_2 and NO_x) and concentrations of particulate matter (PM) and ozone (O_3). During the past decade, SO_2 and NO_x emissions had declined throughout China and concentrations of $PM_{2.5}$ and PM_{10} had considerably decreased in most cities, but average reported 90th MDA8 O_3, M7, and AOT40 O_3 for 31 capital cities showed an increasing trend between 2013 and 2017. Despite progress in air pollution reduction and an increasing number of "clear sky" days, PM concentrations throughout China remain higher than the World Health Organization guidelines, and urban smog and haze remain a major threat to human health and the environment. Thus far, significant emission reductions have occurred largely through robust administrative power, especially when emission reductions were tied to the performance evaluations and promotion of government officials. Similar to most already-industrialized nations, China is now shifting away from SO_2-dominated to NO_x- and O_3-dominated air pollution. Existing technologies and improved operations of existing control equipment appear unlikely to achieve sufficient reductions in NO_x and O_3 pollution. Considering the complex relationship between O_3, NO_x, VOCs, weather, and

* Corresponding author at: Department of Environment, College of Architecture and Environment and Healthy Food Evaluation Research Center, Sichuan University, No. 24, South Section One, First Ring Road, Chengdu, Sichuan 610065, China.
E-mail address: tangya@scu.edu.cn (Y. Tang).

https://doi.org/10.1016/j.scitotenv.2019.01.262

socio-economic changes in China, it is necessary to increase research on impacts of increasing ozone on plants and to adopt novel technologies and implemented to further reduce air pollution to levels that will protect human health and the environment.

1. Introduction

Environmental pollution is a common consequence of industrialization and urbanization, leading to reduced life expectancy and individual well-being (Straf et al., 2013). Since countries are undergoing different stages of industrialization and urbanization, public awareness of environmental pollution and challenges with environmental controls vary widely across the world. During the early stages of industrialization, people are often unaware of or less concerned with the detrimental effects of environmental pollution on human health and ecosystems. Instead, they usually welcome industries that boost the local economy by providing jobs and reducing overall poverty. But as economic prosperity rises, so does the public's concern for environmental protection.

China has undergone the world's fastest growth rate in per capita income, with annual growth of 7% over the past two decades. The country's rapid economic expansion has also resulted in world history's largest rural-to-urban migration (Zhang and Song, 2003), and, as of 2017, about 58.5% of China's population now lives in urban areas. Paralleling the experiences of most developed countries during their early stages of industrialization and urbanization, China's rapid economic growth has yielded significant volumes of pollution emissions and environmental damage (Economy, 2004; Lin et al., 2010; Guan et al., 2012). Increased environmental pollution can be partly attributed to the relocation of factories from around the world to China. On the one hand, production for the world has contributed greatly to China's economic development; however, on the other hand, it has severely polluted the local and regional environment. Consequently, the nation now experiences some of the most polluted skies and waterways in the world (Vennemo et al., 2009).

The history of air pollution control in England and the United States indicates that only when pollution causes substantiated damage to human health and human well-being does pollution become a major concern for local people (Davis, 2002). Only with increased public awareness about the health damages caused by polluted environments are strict pollution control measures taken by governments. This has been observed throughout the history of global industrialization (Brimblecombe, 2006; Davis, 2002; Samet, 2011), including in China today. At this stage of China's development, there is growing domestic consensus that severe pollution and ecological degradation are inhibiting sustainable development. Consequences of industrialization and urbanization, including degraded ecosystems, depleted natural resources, and impaired human health, as well as the significant effects of ozone on plants, all have major economic impacts (Selin et al., 2009). The economic implications of the wide-ranging damage on human health and the environment have been suggested to exceed the benefits of GDP growth (Daly, 1997).

Responding to severe pollution and the growing public concern, the Chinese government has adopted a series of measures to reduce environmental pollution and improve regulations on pollution emissions (Geng et al., 2012). In particular, China's State Council has adopted three nationwide action plans to mitigate air, water, and soil pollution. Compared with soil and water pollution, air pollution has gained more attention, as it has wider spatial impacts and is more visible. National, provincial, and city-level air quality monitoring stations have been established across the country and air quality data are now publicly accessible. Based on understanding the characteristics and sources of air pollution in China, and the lessons of air pollution control from developed countries, the Chinese government has enforced policies and measures specifically targeting air pollution. As a result, some types of pollution, such as acid deposition, smog, and haze, have reduced over the past ten years (Cheng et al., 2013; Gu et al., 2013; Lin et al., 2014; Bao et al., 2015; He et al., 2017), with steep declines evident over the past five years. However, surface ozone (O_3) has become an increasing problem in China, with its resultant urban haze extending over a growing number of fast-growing regions and detrimental effects on plant growth (Feng et al., 2014; Li et al., 2018a, 2018b).

This paper seeks to clarify the nature of current environmental pollution problems in China and establish baselines against which to evaluate progress, which will also assist policymakers set priorities for future attention. As many developing countries are also experiencing or will experience industrialization and urbanization, China's experiences with air pollution control can provide valuable insight. Therefore, the main objectives of this study are to: (1) review China's air pollution control history; (2) evaluate whether air pollution control policies and measures that were previously successful will continue to be in the future, and (3) summarize research progress and discuss future implications of ozone effects on plants in China.

2. Materials and methods

Evaluating publicly accessible information, we analyzed the long-term trends of air pollution emissions and pollutant concentrations in China, with a special focus on parameters that reflect China's economic development. Energy consumption and annual emission data of sulfur dioxide (SO_2) from 1985 to 2016 were derived from the China National Statistical Yearbooks (http://www.stats.gov.cn). Annual emission data of nitrogen oxides (NO_x) were collected from various sources for different periods, including 1995–1999 (Zhang et al., 2007), 2000–2010 (Shi et al., 2014), and 2011–2016 (China National Statistical Yearbooks). To quantify air quality changes from 2003 to 2016, the annual average concentrations of particulate matter (PM), including both $PM_{2.5}$ and PM_{10} (i.e., PM with aerodynamic diameters ≤2.5 μm and ≤10 μm, respectively), and the 90th percentile of maximum daily 8-h average ozone (90th MDA8 O_3) measured at national air quality stations (NAQSs) were derived from the China National Statistical Yearbooks (http://www.stats.gov.cn). Hourly data of O_3 between 2014 and 2017 were derived from the China National Environmental Monitoring Centre (http://www.cnemc.cn). Hourly concentrations of O_3 and $PM_{2.5}$ began to be measured at NAQSs in 2013, and by the end of 2017 there were approximately 1500 NAQSs across China. Using the hourly data of O_3, we further calculated the annual mean of the weekly averages from 9:00 a.m. to 16:00 p.m. (i.e., M7), used to assess the effects of O_3 on plants in the daytime. For deciduous broad-leaved forest species and evergreen broad-leaved forest species, we calculated both AOT40 and AOT40f, which are the O_3 indexes used to estimate the risk level posed to forests. AOT40 is the annual sum of the excess of hourly concentrations over the cut-off of 40 ppb during light hours (6 a.m.–8 p.m.), and AOT40f are those only for the six months of the primary growing season (April to September) for forest trees (Paoletti et al., 2007).

3. Results

3.1. Changes in air pollution

Acid deposition has been a serious problem in China since the 1980s. As the major precursor of acid deposition in China, SO_2 has long been one of the most important targets of air pollution control efforts. While the number of targeted pollutants has varied between two and twelve since the 1980s, SO_2 has always been one of the targets (Xue et al., 2014). SO_2 is emitted mostly from coal combustion, the primary energy source in China (Fig. 1). With increasing coal consumption, total SO_2 emissions peaked first in 1995 and then in 2006 (25.89 million tons, i.e., teragram or Tg), but started to decline after 2007, especially since 2011, until reaching 11.03 Tg in 2016 (Fig. 2). As major consumers of coal, thermal power plants contribute about 30–40% of China's total SO_2 emissions. Annual SO_2 emissions from coal-fired power plants during the period from 1990 to 2016 reached a peak of 12.04 Tg in 2006, declining thereafter, following a similar trend with the SO_2 total emissions output, despite a continuous increase in national energy consumption (Figs. 1 and 2). Apart from power plants, industry and other sectors contributed another 50–60% of the total SO_2 emissions (He et al., 2012; China FAQs Project, 2012).

Declining SO_2 emissions over the same timeframe as rising power generation is partially due to declining consumption of sulfur-containing coal. Coal consumption rate (coal consumption per unit electricity supplied) is usually used to measure the energy efficiency of power plants. The higher the number, the less efficient the plant is at yielding energy. Despite fluctuations, China's coal consumption rate has exhibited a declining trend over time, from 385.01 gce/kWh (g coal equivalent per Kilowatt hour) in 1991 to 290.29 gce/kWh in 2016, representing an improvement in energy efficiency of 25% over the last 25 years (Fig. 3). SO_2 emission intensity (SO_2 emissions per unit coal consumption) showed a similar decline of nearly 20-fold, from a high of 1.77×10^{-2} Gg/Tg in 1991 to a low of 0.09×10^{-2} Gg/Tg in 2016. This significant change is also attributed to the central government policy that mandated the installation of flue gas desulfurization (FGD) systems in coal-fired power plants and reform generator units.

The overall temporal NO_x pattern is similar to the SO_2 pattern, though lagging in time. Total NO_x emissions increased slightly from 10.9 Tg in 1995 to 11.8 Tg in 2001. But it increased quickly from 12.6 Tg in 2002 to the peak of 24.3 Tg in 2010. After 2010, China's NO_x emissions reduced dramatically to 14.0 Tg in 2016 (Fig. 2).

National PM standards have changed greatly over the past four decades. Before 2000, only total suspended particulate matter was used to evaluate air quality and this was neither widely monitored nor enforced. In 2001, PM_{10} was introduced as a more refined measure to assess air quality. Until 2013, this was one of the most important air pollutants under evaluation, at which point both $PM_{2.5}$ and O_3 became standard for assessing air quality (Wang and Hao, 2012). The mean PM_{10} concentration for the 31 provincial capital cities in 2003 was 118 μg m^{-3}, about 1.5 times that of the National Ambient Air Quality Standards (NAAQS-2012, 70 μg m^{-3}) and about 6 times that of the ambient

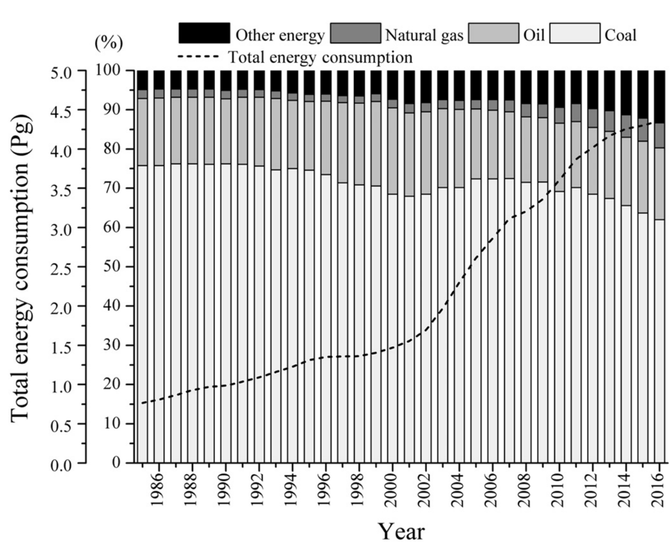


Fig. 1. Energy consumption in China (1985–2016). Note: Other energy includes hydro-, wind, solar energy, bioenergy and tidal energy, etc.

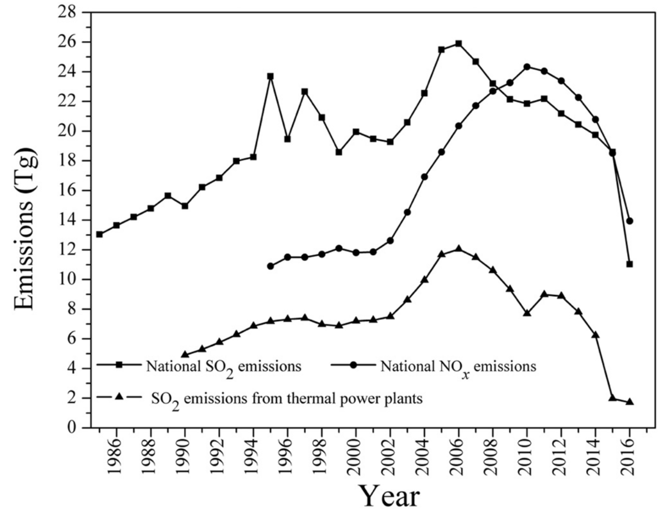


Fig. 2. National SO_2, NO_x, and SO_2 emissions from thermal power plants (1985–2016).

air quality guideline set by the World Health Organization (WHO, 20 μg m^{-3}). It declined to 91 μg m^{-3} in 2012, but jumped again to 126 μg m^{-3} in 2013. Beginning in 2014, it started decreasing again and this trend has continued, reaching 89 μg m^{-3} in 2017 (Fig. 4), which was still higher than the NAAQS-2012 standard of 70 μg m^{-3} and about 4.5 times that of the WHO guideline. Measured at NAQSs since 2013, $PM_{2.5}$ has been experiencing a downward trend. The annual mean $PM_{2.5}$ concentration for the 31 capital cities was 74 μg m^{-3} in 2013, decreasing to 43 μg m^{-3} in 2017 (Fig. 4), which remained about 4 times the WHO guideline. Hence, even though PM_{10} and $PM_{2.5}$ have both reduced considerably during the years between 2013 and 2017, by 29% and 42%, respectively, they remain significantly above the WHO guidelines.

In contrast to the recent declining trends of SO_2, NO_x, PM_{10}, $PM_{2.5}$, O_3 now constitutes the most important increasing air pollutant. The average value of 90th MDA8 O_3 for the 31 capital cities increased from 61.6 ppb to 76.0 ppb between 2013 and 2017, an increase of 23% in five years (Fig. 5a). Over the same timeframe, the percentage of capital cities with 90th MDA8 O_3 values exceeding 70 ppb grew from 32% (2013) to 68% (2017). In addition, average reported M7 in the 31 cities increased from 30.1 ppb in 2014 to 36.6 ppb in 2017 (Fig. 5b). Similarly, average reported AOT40 in the 31 cities increased from 15,746 ppb h in 2014 to 35,016 ppb h in 2017 (Fig. 5c). Li et al. (2018a) assessed the risk that ambient O_3 exposure posed to forests in China between 2015 and 2016, showing that in different geographical regions the annual average

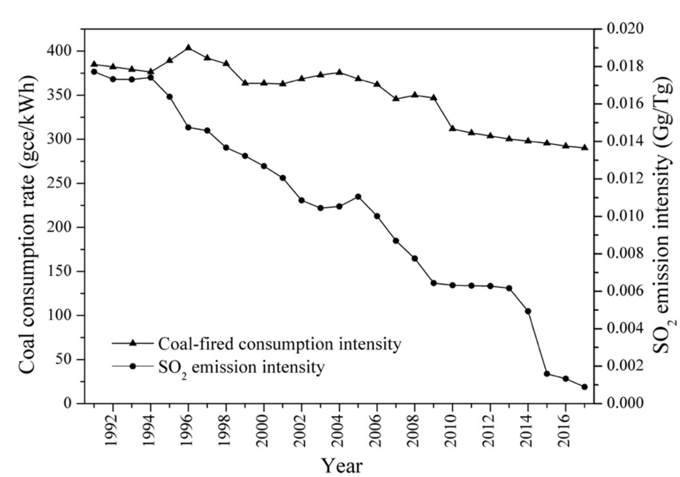


Fig. 3. Coal consumption rate and SO_2 emission intensity in coal-fired power plants (1990–2016).

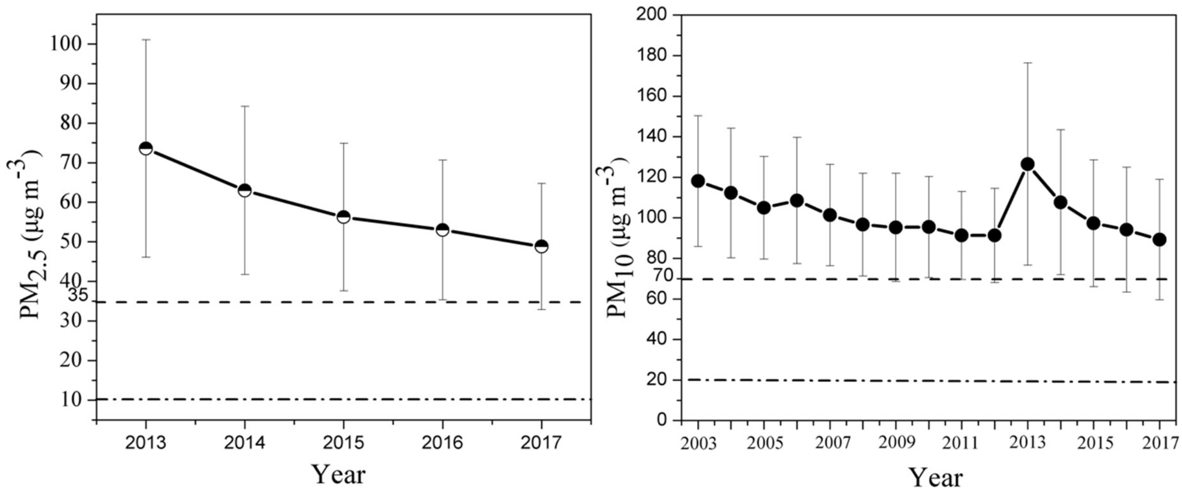


Fig. 4. Annual average concentrations of $PM_{2.5}$ and PM_{10} in 31 provincial capital cities and municipalities in China (2013–2017). Note: Horizontal lines represent the annual average value target of the Chinese National Ambient Air Quality Standards (NAAQS-2012) and the ambient air quality guideline set by the World Health Organization (WHO), respectively.

AOT40 exceeded by as much as three to five times the critical level recommended for forest protection (CLRTAP, 2015). Moreover, AOT40 values in 2016 were significantly higher than in 2015 across most of China. In general, AOT40 values in 2017 were higher than those in 2016, indicating that the effect of O_3 on forests in China is becoming more serious.

The reported increase in O_3 pollution was more significant in some economic centers, including the Beijing-Tianjin-Hebei region (North China Plain), Yangtze River Delta, Pearl River Delta, and Sichuan Basin (Wang et al., 2016, 2017; Gong et al., 2018). In 2017, the average value of AOT40 in 363 cities nationwide was 35,016 ppb h (Fig. 6). By region, the North China Plain faces the most serious O_3 pollution problem in China, with an average AOT40f of 44,012 ppb h, and the average AOT40 of the North China Plain is 50,350 ppb h, which was higher than AOT40f. The AOT40f of O_3 for cities across this region was 22,420–56,800 ppb h, about two to six times the critical load for trees (10,000 ppb h) proposed by Bytnerowicz et al. (2008). AOT40 of O_3 in most Yangtze Delta cities ranged from 25,840–74,200 ppb h, about two to seven times the critical loads for plants, mentioned above. In the Sichuan Basin, AOT40 of O_3 was approximately 11,800–68,910 ppb h,

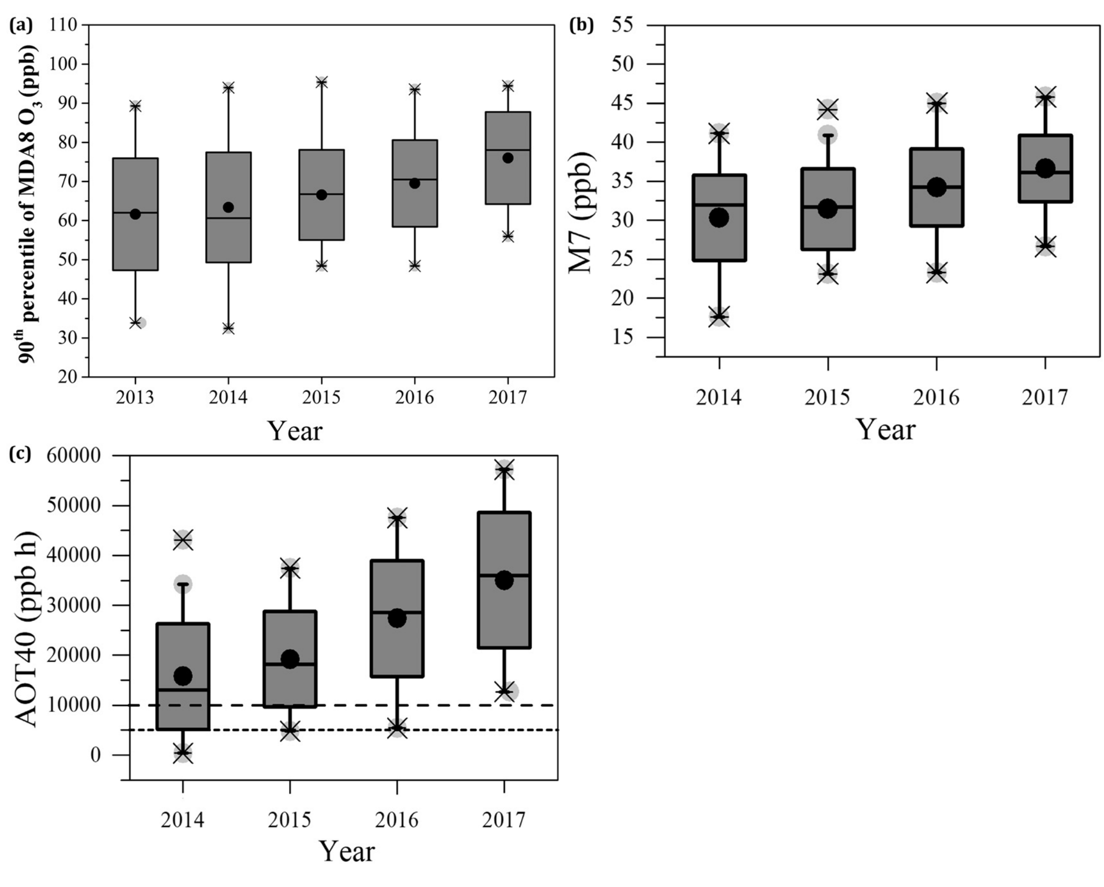


Fig. 5. 90th MDA8 O_3, M7 and AOT40 of O_3 in 31 provincial capital cities and municipalities in China (2013–2017). Note: Horizontal lines represent the critical loads for forests set by Bull (1996), Bytnerowicz et al. (2008) and CLRTAP (2015), respectively.

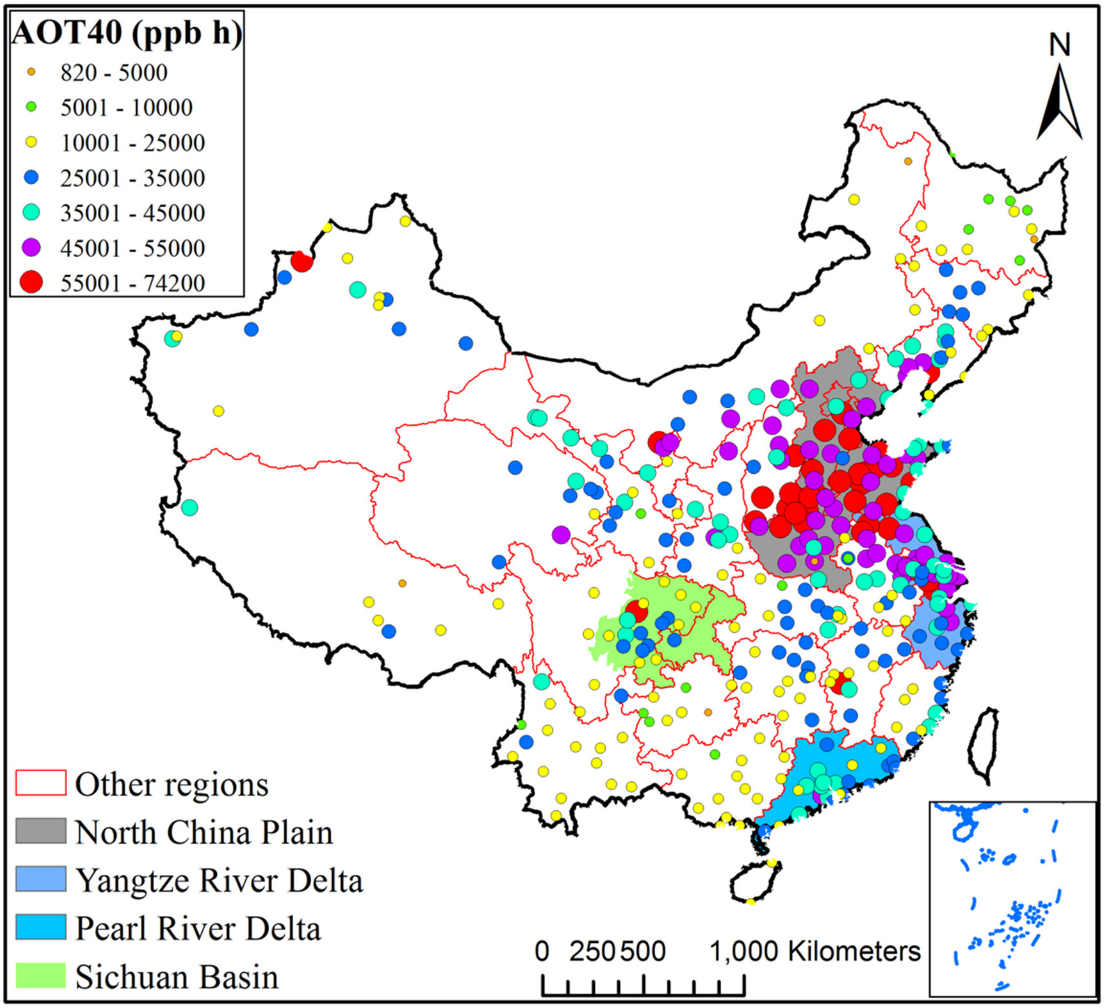


Fig. 6. Spatial distributions of AOT40 of O_3 (ppb h) in 363 cities in China (2017).

with an average AOT40 of 27,798 ppb h. In the Pearl Delta, strongly affected by the summer monsoon, an average AOT40 of 31,363 ppb h was reported, substantially lower than the preceding three regions. In most cities in other regions, such as those on the Qinghai-Tibetan and Yunnan-Guizhou plateaus, the AOT40 did not exceed 25,000 ppb h. These relatively lower levels indicate that trees do not experience high concentrations of O_3 there during the growing season.

3.2. Air pollution control policies and measures

A compiled summary of all major Chinese national-level air pollution control plans, policies, and regulations is presented in Table 1. As acid deposition (particularly that caused by SO_2 emissions) was identified as a major problem in China in the 1980s–2000s, the central government prioritized reducing SO_2 and NO_x emissions by 10% during the periods of the 9th through the 12th Five-Year Plans (China FAQs Project, 2012). The most important regulations to control SO_2 emissions was to reduce SO_2 from coal-fired power plants through nationwide installation of desulfurization equipment and shutting down small coal-fired power plants in 2005 and 2006 (Xu, 2011). Other government-mandated measures included increased use of low-sulfur coal and petrol, application of state-of-the-art "clean" production technologies in the coal industry, phasing out small coal mines, replacing coal with natural gas, and facilitating development of clean energy, such as wind and solar power. Special regulations concerning NO_x emission reductions were announced in 2013 in the Air Pollution Prevention and Control Action Plan (CAAC, 2013). Installation of selective catalytic reduction (SCR) equipment was required at all power plants beginning in 2011 onwards. The tightened emission standards of air pollutants from power plants and other industries have caused substantial reductions in SO_2 emissions. However, NO_2 levels continue to rise, as they also originate from automotive combustion and agricultural sources.

Table 1
Plans, policies, and regulations of the Chinese government to control air pollution.

Name	Year (revised)
The 10th Five-year Plan	2001–2005
The 11th Five-year Plan	2006–2010
The 12th Five-year Plan	2011–2015
Air Pollution Prevention and Control Law	1987 (1995, 2000)
Regulation on Management of Ozone Depleting Substances	2010
Air Pollution Prevention and Control Action Plan	2013
Ambient Air Quality Standards	1982 (1996, 2012)
Rules on Supervision and Management of Automobile Exhaust Pollution	1990
Thermal power plant emission standards for atmospheric pollutants	1991 (1996, 2003, 2011)
Management Rules for Prohibiting Burning and Comprehensive Utilization of Straw	1999
Rules on the Standard for Compulsory Retirement of Motor Vehicles	2013
China III emission standard for cars, nationwide	2007
China IV emission standard for gasoline cars, nationwide	2011
China IV emission standard for diesel cars, nationwide	2015

Power plants are not just important sources of acid deposition (e.g., SO_2 and NO_x) but also of primary particulate matter (PPM). Thus, since 2004, newly built power plants are required to meet the Standard for Emission of Air Pollutants from Power Plants (GB13223-2003), mandating PPM concentrations in flue gas lower than 50 mg m^{-3}. In 2003, China announced the Clean Production Promotion Law and the Environmental Impact Assessment Law, both of which require improving the efficiency of resource utilization, reducing pollutant emissions, and developing a circular economy. To reduce vehicle emissions, the relevant standards have been modified three times between 2007 and 2015, including the China III standards for all types of cars, China IV standards for gasoline cars, and China IV standards for diesel cars 2015 (Wu et al., 2017). These actions have proven effective at reducing vehicular emissions, despite surging vehicular populations over the past decade in most cities. While vehicular emissions have even decreased in some places (S. Wang et al., 2010), this is not uniformly true. To further control emissions, the National Ambient Air Quality Standards (NAAQS) were modified in 2012 to be more stringent (MEE, 2012a; MEE, 2012b), and for the first time $PM_{2.5}$ and O_3 standards were included in NAAQS-2013 (MEE, 2012c).

4. Discussion

4.1. National policy implementation: critical for air pollution reduction

During most of the past four decades, air pollution in China has been measured principally by focusing on SO_2 emissions and the resultant extensive acid deposition effects of specific air pollution control measures, for which considerable efforts have been made to date (Wang et al., 2014). Specifically, SO_2 emission reductions from such key enterprises as coal-fired thermal power, iron/steel plants, have been prioritized for air pollution control since 1996. It is now mandatory to install online monitoring devices in the key enterprises and manage them strictly. A number of policies and measures, including various incentive regimes, have been implemented during different Five-Year Plans (Xue et al., 2014). The emission standards of the Air Pollutants for Thermal Power Plants (GB13223-1991) were first implemented in 1991, but then revised in 1996, 2003, and 2011. The emission standards for SO_2, NO_x, PM, and smoke have become increasingly stringent. Currently, the emission concentration limit for SO_2 is 200 mg m^{-3} for existing boilers and 50 mg m^{-3} for boilers in key regions such as Beijing. All these efforts have contributed to a dramatic reduction of SO_2 emissions since 2006.

The central government started to establish other major air pollutant emission targets for key industrial sectors and regions during the 9th Five-Year Plan (1996–2000) along with economic restructuring in order to reduce SO_2 and other emissions. As a result, the SO_2 emission standard was stricter from 1996 to 2000 than that in 1995. However, most local governments prioritized economic development over environmental protection and there was a clear SO_2 emission rise during the 10th Five-Year Plan (2001–2005), resulting in widespread noncompliance of emission control standards (OECD, 2007). As a consequence, the central government placed greater emphasis on SO_2 emission reduction goals in the 11th Five-Year Plan (2006–2010), establishing stringent targets against the 2005 levels to reduce the national energy consumption per unit GDP output (by 20%) and SO_2 emission (10%) by 2010.

To achieve these targets, a novel set of political instruments was devised that included compulsory reduction targets set for provincial governors and managers of major state-owned power companies, a modified evaluation system for government administrators and corporate officials, and stronger enforcement of existing laws by the central government. Most significant was the enforcement of several regulations in the power sector—all new thermal power units and most already in operation had to have flue gas desulfurization (FGD) systems installed. Moreover, smaller units with less energy efficiency had to be gradually shut down. These measures yielded dramatic effects. Although the trend of increasing SO_2 emissions continued into the first year (2006) of the 11th Five-Year Plan (Fig. 2), it began to decline again afterwards by an average annual rate of 2.8%, reaching 22 Tg by 2010, a 14.3% reduction against the 2005 level (MEE, 2011a). With gradual replacement of coal with oil, gas, and renewable energy sources from 2011 onward, reported SO_2 emissions further declined to 11.03 Tg by 2016, a significant reduction of 50% in just fifteen years (Fig. 2).

Particulate matter reduction has also resulted from policy changes. To meet increasing demand for electricity, thermal power generation increased by 4.6 times between 1998 and 2016, presenting a huge challenge for air pollution reduction goals. In 2003, China issued much stricter PM emission standards for power plants (GB13223-2003), which stipulated that all newly built and rebuilt power plants must not emit PM concentrations above 50 mg m^{-3}. Since then, >92% of coal-fired power plants have installed electrostatic precipitators (ESP), and fabric filters have also been used in units with installed capacity exceeding 600 MW. As a result, between 1990 and 2005, PM emission factors in various industries across China reportedly fell by as much as 7% to 69%. However, the overall effectiveness of PM control technologies was offset by greatly increasing emissions from other industries such as steel, cement, and aluminum. As a result, reported overall PM concentrations remained high (Lei et al., 2011), increasing significantly in 2013 (Fig. 4).

As appreciation of PM's considerable health impacts has increased and the visible problems with air pollution have grown, $PM_{2.5}$ pollution now constitutes a widespread concern throughout the country, with $PM_{2.5}$–dominated smog and haze becoming a major public health concern. With the prevalence of smog and haze across China, increasing "blue sky" days in major cities became a key criterion for local government performance evaluations, especially in the Beijing-Tianjin-Hebei, Yangtze Delta, and Pearl Delta regions during the implementation of the Air Pollution Prevention and Control Action Plan. With continued efforts, country-wide average reported concentrations of PM_{10} and $PM_{2.5}$ decreased significantly, by 29% and 42%, respectively, from 2013 to 2016.

NO_x emissions from power plants have also been greatly reduced with the installation of selective catalytic reduction (SCR) facilities (D. Liu et al., 2013). In 2013 alone, SCR facilities were installed in about 50% of power plants, greatly impacting NO_x emissions over the following years. According to the National Bureau of Statistics (NBS), NO_x emissions dropped to 13.93 Mt. in 2016, which is 42% lower than the peak in 2011.

In contrast to the significant NO_x emissions reduction from power plants, emissions from vehicles have been increasing. Vehicles, both on-road and off-road, contribute greatly to air pollution and many studies have confirmed that air pollution in Chinese megacities was directly associated with increasing vehicular traffic (Hao et al., 2000; Chan and Yao, 2008; Yang et al., 2011; Cui et al., 2015). China's vehicle population increased 8.3–fold between 1998 and 2016. By the end of 2016, the vehicle population exceeded 300 million, of which 217 million were private cars (China National Statistical Yearbook, 2017), even though some megacities, such as Beijing, Guangzhou, Hangzhou, Shanghai, Shenzhen, and Tianjin, adopted policies restricting car purchases in order to limit the number of private cars within their jurisdictions.

To reduce vehicle emissions, the central government has established integrated emission control policies since the 1990s, including implementation of emission standards for new vehicles, emission controls for in-use vehicles, fuel quality improvements, promotion of sustainable transportation, alternative-fuels, and advanced vehicles, and traffic management programs (Wu et al., 2011, 2016). The State Council set a target to phase out all yellow-label cars (i.e., gasoline cars emitting higher than China I standard and diesel cars higher than China III standard) and aging high-fuel-consumption vehicles. Many provinces/municipalities, such as Beijing, Tianjin, Hebei, Shanxi, and Liaoning, have surpassed their targets in recent years. Phasing out heavily-polluting vehicles has also had comprehensive effects for improving urban air quality, reducing CO_2, and black carbon emissions (Gao et al., 2014). In 2016, 4.04 million of these vehicles were taken out of use, a 106.5% completion of the annual target set in 2015 (MEE, 2017a). Recently China has issued standards for new vehicles and engines based on the European Union Standards. At the national level, Phase III standards (similar to Euro III) began to be put into effect in July 2007 (MEE, 2011b). Phase IV emission standards (similar to Euro IV) for gasoline cars (2011) and diesel cars (2015) were also implemented. Studies suggest that increasingly stringent standards for vehicle emissions could mitigate the total vehicle emissions in 2030 of HC (hydrocarbon), CO (carbon monoxide), NO_x, and $PM_{2.5}$ by approximately 39%, 57%, 59%, and 79%, respectively, compared with the 2013 level (Wu et al., 2016).

4.2. Use of command and control policies

Reducing emissions is probably the most effective way to control air pollution, but the effectiveness of its implementation varies depending on local economic and political circumstances. State-mandated pollution-control policies have contributed to a significant reduction of air pollution in China, despite the fact that overall energy consumption and economic output have continued to grow. While some prior studies suggest that air pollution reduction is difficult and takes much time (e.g., Ge et al., 2009), China has achieved considerable success in reducing air pollution in a relatively short time. Like developed nations where air pollution emissions were reduced with the implementation of national-level laws, such as the Clean Air Act in the United States (Popp, 2003), air pollutant emissions in China have been substantially reduced through the implementation of two recent action plans under the Law on the Prevention and Control of Air Pollution.

Considering the history of air pollution control measures since the 1990s, the significant emission reduction achievements in China have largely been due to robust administrative power, especially when emission reductions were directly linked to the performance evaluations and promotion of government officials. In the context of the Chinese political system, it is widely understood that when the implementation of a policy relates to the evaluation and promotion of officials, the policy is most effectively implemented. Reducing air pollution was incorporated into the evaluation criteria for local government annual performance reviews, with air quality controls and the number of “blue sky” days both being evaluation targets. A local official failing to meet the targets for three consecutive years would be ineligible for promotion during the next five years (Xue et al., 2014). Consequently, these policies have made air pollution control one of the top priorities for China's local governments.

While the public is generally pleased with the increasing number of “blue sky” days and the reduction of smog and haze in many regions in recent years, it is important to note that these achievements so far were probably the easiest aspects of air pollution to control (the “low hanging fruit” so to speak). That is, these emission reductions were largely achieved through exercised administrative power, which, in many cases, were achieved without considering economic costs. For instance, substantial improvements in air quality were achieved through control strategies (such as temporary closure of many factories) for certain significant events, such as the 2008 Olympic Games, 2014 APEC Summit, etc. However, creating conditions to sustain these significant reductions is now providing local and national policymakers with major challenges in setting realistic goals and reasonable timetables, while also implementing appropriate incentives for promotion review for government officials.

4.3. Increasing ozone: new challenge and urgent need for focused research effort on air pollution control and in ozone's effects on plants

Recent changes in air pollution control measures in China recall the history of developed countries, where crippling episodes of pollution triggered widespread awareness of the problem's severity. In those nations, policies on strict air pollution control were implemented after acute incidents of severe human health problems occurred, for example, in London, UK, and Donora, Pennsylvania, US (Bell and Davis, 2001; Davis, 2002). Similarly, China's progress reducing air pollution thus far reflects the critical importance of sound government policy to reduce and control major emissions of air pollution across numerous sectors. For example, the reduction of some conventional air pollutants in China is a consequence of strong administrative power (i.e., command and control), with forced closure of factories utilizing obsolete technology having considerable impact. Our preliminary analyses of monitoring data indicate that air pollution in most areas of China has been undergoing a gradual process of reduction, and, similar to most already-industrialized nations, China has largely on the process shifting away from SO_2-dominated to NO_x- and O_3-dominated air pollution. At a national scale, the average SO_4^{2-} deposition declined from 40.54 kg S ha^{-1} yr^{-1} in the 1990s to 34.87 kg S ha^{-1} yr^{-1} in the 2010s (Yu et al., 2017). In contrast, nitrogen deposition increased from the 1990s to the 2010s and, overall, average annual bulk deposition of N increased by approximately 8 kg N ha^{-1} from the 1980s to the 2000s (X. Liu et al., 2013). As a result, the national average equivalent ratio of $SO_4^{2-}/(NO_3^- + NH_4^+)$ in precipitation decreased from 1.44 in 2011 to 0.97 in 2016 (MEE, 2012d, 2017b).

Building on scientific research and technological advancements, economic growth and air pollution reduction in China and other industrializing countries can now be achieved with much lower costs to human health. Technology has already contributed to the effective reduction in SO_2 through the flue gas desulfurization system, NO_x through the selective catalytic reduction technology, and particulate matter (Carmichael et al., 2002; Zhao et al., 2013). However, concentrations of both PM_{10} and $PM_{2.5}$ remain much higher than the WHO guidelines. Further reduction poses a big challenge and will likely depend on the development and implementation of new control technologies.

Similarly, despite the documented progress made in reducing overall air-pollution, increasing numbers of motor vehicles across China are contributing to a rising emission of ozone's precursors, VOCs and NO_x, with associated increase of surface ozone concentrations are rising throughout the country, and now presenting a major challenge for air pollution control in China. Considering the complex relationship between ozone, NO_x, VOCs, weather, and socio-economic changes in China, further reduction of O_3 to the internationally accepted levels will not be easy since the number of vehicles will likely continue to increase further (T. Wang et al., 2010). With the transition away from the predominance of SO_2 air pollution to NO_x- and O_3-dominated air pollution, existing technologies and improved operations of existing control equipment appear unlikely to achieve sufficient reductions in O_3 and NO_x pollution. Selective catalytic reduction technology can reduce NO_x, which in turn will reduce the formation of ozone. But, nevertheless, focused research on technology development and control mechanisms are needed in order to reduce the contributors to these major pollutants.

It is well established that exposure to elevated O_3 can cause many negative effects to human health (Brauer et al., 2015; Li et al., 2015), agricultural crops (Laila et al., 2018), and forest plants (Matyssek et al., 2010). AOT40 has a good correlation with the biomass of many forest species and has been widely used (Marco, 2009; Paoletti, 2009; Beltman et al., 2013). Although the industrial emissions of ozone's precursors (NO_x, VOCs) have been considerably controlled, AOT40 values in China increased by 31% annually between 2014 and 2017 (Fig. 5c), indicating that ozone pollution is becoming severer. Monitoring data showed that O_3 concentration in most parts of China exceeded the critical load of AOT40. Deciduous broad-leaved species were generally considered to be less resistant to O_3 than evergreen broadleaf and conifer species (Anav et al., 2011; Li et al., 2017). Among the four regions with significant O_3 pollution (Fig. 6), AOT40 in the North China Plain was the highest, showing that the temperate deciduous forests in

North China may be facing the greatest risk from ozone. Although the evergreen broad-leaved tree species in the Pearl River Delta, Sichuan Basin, and Yangtze River Delta may be more resistant to the harmful effects of O_3, more research is necessary to understand how the high concentrations would affect these forests.

In general, despite increasing ozone pollution impacts, unfortunately, studies on the impacts of ozone on plants, especially on forest plants, in China, are inadequate. A number of studies use open-top chamber chambers (OTC) for ozone fumigation to study the mechanism of ozone-to-tree damage (Zhang et al., 2014; Feng et al., 2011) and some studies have investigated ozone's effect on crop yield (Wang et al., 2012; Zhao et al., 2018), folia injury (Feng et al., 2014; Yang et al., 2014; Cheng et al., 2016), and physiological changes to urban tree species (Li, 2015; Long et al., 2017; Ping et al., 2017; Yang et al., 2017; Li et al., 2018b). Most of the over 1500 ozone monitoring stations in China are located in urban areas. In rural and remote areas where there is more vegetation, there is no corresponding monitoring networks, but one study revealed annual average ozone concentrations were higher in rural areas than urban (Yang et al., 2014). In addition, regional transmission of polluted gases and ozone-rich air masses in the troposphere and stratosphere sink may contribute to increased ozone concentrations in rural areas. In consideration of the large geographical size, rich flora, and highly variable forest species composition, and highly variation of ozone in different regions of China, in contrast with the relatively well-studied forest and urban plants in the US and Europe (Fujioka et al., 1999; Davis and Skelly, 1992), more focused studies on the effects of ozone on both urban and wild plants are needed in China. Thus, reducing ground-level O_3 concentrations poses a particularly difficult challenge for the future.

5. Conclusion

To control air pollution in China, the Chinese government has enforced a number of nation-wide pollution standards and control measures over the last three decades. In the context of China's administrative model, air pollution control through government-mandates has yielded significant reductions in the emission of several major pollutants during a rather short time, highlighting the utility of this approach. However, these pollution-control policies and resultant significant reduction of air pollution in China, have been largely implemented without consideration of their economic costs, which may not be sustainable in the future. China has been undergoing a similar process of air pollution and control that was previously experienced in many developed countries. Although remarkable results have been achieved in air pollution control, concentrations of PM_{10}, $PM_{2.5}$, and O_3 remain much higher than the WHO guidelines. With continued economic growth, the numbers of motor vehicles across China continues to increase rapidly, further contributing to the rising emission of ozone's precursors. Considering the complex relationship between ozone, NO_x, VOCs, the weather, and socio-economic changes in China, further reduction of ground-level O_3 to the internationally accepted levels will require more focus on research and development of new technologies to address these continued air pollution emissions challenges. Consequently, more research is needed to understand the impacts of ozone on forest plants. While further reducing concentrations of both PM_{10} and $PM_{2.5}$ to safe levels requires continued focused effort, reducing ground-level O_3 concentrations poses an increasingly difficult challenge for the future.

Acknowledgements

This study is supported by the National Science Foundation of China (41628102, 71742004), the Program of Introducing Talents of Discipline to Universities (B08037), and $PM_{2.5}$ Monitoring in the Sichuan University Campuses (SCU2015CC0001).

References

Anav, A., Menut, L., Khvorostyanov, D., 2011. Impact of tropospheric ozone on the Euro-Mediterranean vegetation. Glob. Chang. Biol. 17, 2342–2359.
Bao, J., Yang, X., Zhao, Z., Wang, Z., Yu, C., Li, X., 2015. The spatial-temporal characteristics of air pollution in China from 2001–2014. Int. J. Environ. Res. Public Health 12, 15875–15887.
Bell, M., Davis, D., 2001. Reassessment of the lethal London fog of 1952: novel indicators of acute and chronic consequences of acute exposure to air pollution. Environ. Health Perspect. 109, 389–394.
Beltman, B., Hendriks, C., Tum, M., Schaap, M., 2013. The impact of large scale biomass production on ozone air pollution in Europe. Atmos. Environ. 71, 352–363.
Brauer, M., Freedman, G., Frostad, J., Donkelaar, A., Martin, R., Dentener, F., Dingenen, R., et al., 2015. Ambient air pollution exposure estimation for the global burden of disease 2013. Environ. Sci. Technol. 50, 79.
Brimblecombe, P., 2006. The Clean Air Act after 50 years. Weather 61, 311–314.
Bull, K., 1996. Workshop summary. In: Karenlampi, L., Skarby, L. (Eds.), Critical Level for Ozone in Europe: Testing and Finalizing the Concepts (UN-ECE Workshop Report). University of Kuopio, Finland, pp. 7–8.
Bytnerowicz, A., Arbaugh, M., Schilling, S., Frączek, W., Alexander, D., 2008. Ozone distribution and phytotoxic potential in mixed conifer forests of the San Bernardino Mountains, southern California. Environ. Pollut. 155 (3), 398–408.
CAAC: Clean Air Alliance of China, 2013. State council air pollution prevention and control action plan, issue II. http://en.cleanairchina.org/product/6346.html.
Carmichael, G., Streets, D., Calori, G., Amann, M., Jacobson, M., Hansen, J., 2002. Changing trends in sulfur emissions in Asia: implications for acid deposition, air pollution, and climate. Environ. Sci. Technol. 36, 4707–4713.
Chan, C., Yao, X., 2008. Air pollution in mega cities in China. Atmos. Environ. 42, 1–42.
Cheng, Z., Jiang, J., Fajardo, O., Wang, S., Hao, J., 2013. Characteristics and health impacts of particulate matter pollution in China (2001−2011). Atmos. Environ. 65, 186–194.
Cheng, X.Y., Luo, C.W., Liu, Q.J., Meng, S.W., Zhou, Y.S., Zhou, H., 2016. Tropospheric ozone distribution and injury on leaves of sensitive woody plants in Nanchang city, China. Plant Sci. J. 34 (2), 211–219 (In Chinese).
China FAQs Project, 2012. China adopts world-class pollutant emissions standards for coal power plants. http://www.chinafaqs.org/library/chinafaqs-china-adopts.
China National Statistical Yearbook, 2017. http://www.stats.gov.cn/tjsj/ndsj/2017/indexch.htm.
CLRTAP, 2015. Mapping critical levels for vegetation, chapter III of manual on methodologies and criteria for modelling and mapping critical loads and levels and air pollution effects, risks and trends. UNECE Convention on Long-range Transboundary Air Pollution http://www.icpmapping.org.
Cui, H., Chen, W., Dai, W., Liu, H., Wang, X., He, K., 2015. Source apportionment of PM2.5 in Guangzhou combining observation data analysis and chemical transport model simulation. Atmos. Environ. 116, 262–271.
Daly, E., 1997. Beyond growth: the economic of sustainable development. Moving to a Steady-state. Beacon Press, pp. 40–41.
Davis, D., 2002. When Smoke Ran Like Water: Tales of Environmental Deception and the Battle Against Pollution. Basic Books.
Davis, D., Skelly, M., 1992. Foliar sensitivity of eight eastern hardwood tree species to ozone. Water Air Soil Pollut. 62, 269–277.
Economy, C., 2004. The river runs black: the environmental challenge to China's future. Lessons From Abroad. Cornell University Press.
Feng, Z., Niu, J., Zhang, W., Wang, X., Yao, F., Tian, Y., 2011. Effects of ozone exposure on sub-tropical evergreen *Cinnamomum camphora* seedlings grown in different nitrogen loads. Trees (Berlin) 25, 617–625.
Feng, Z., Sun, J., Wan, W., Hu, E., Calatayud, V., 2014. Evidence of widespread ozone-induced visible injury on plants in Beijing, China. Environ. Pollut. 193, 296–301.
Fujioka, M., Roads, O., Chen, S., Miller, R., Mcbride, R., 1999. Oxidant air pollution impacts in the montane forests of southern California: a case study of the San Bernardino mountains. Mt. Res. Dev. 20, 289–290.
Gao, Y., Mao, X., Corsetti, G., Yi, W., 2014. Assessment of co-control effects for air pollutants and greenhouse gases in urban transport: a case study in Urumqi. China Environ. Sci. 34, 2985–2992.
Ge, C., Chen, J., Wang, J., Feng, L., 2009. China's total emission control policy: a critical review. Chin. J. Popul. Resour. Environ. 7, 50–58.
Geng, Y., Fu, J., Sarkis, J., Xue, B., 2012. Towards a national circular economy indicator system in China: an evaluation and critical analysis. J. Clean. Prod. 23, 216–224.
Gong, X., Hong, S., Jaffe, D.A., 2018. Ozone in China: spatial distribution and leading meteorological factors controlling O_3 in 16 Chinese cities. Aerosol Air Qual. Res. 18, 2287–2300.
Gu, D., Wang, Y., Smeltzer, C., Liu, Z., 2013. Reduction in NO_x emission trends over China: regional and seasonal variations. Environ. Sci. Technol. 22, 12912–12919.
Guan, D., Liu, Z., Geng, Y., Lindner, S., Hubacek, K., 2012. The gigatonne gap in China's carbon dioxide inventories. Nat. Clim. Chang. 454–463.
Hao, J., He, D., Wu, Y., Fu, L., He, K., 2000. A study of the emission and concentration distribution of vehicular pollutants in the urban area of Beijing. Atmos. Environ. 34, 453–465.
He, H., Li, C., Loughner, P., Li, Z., Krotkov, A., Yang, K., 2012. SO_2 over central China: measurements, numerical simulations and the tropospheric sulfur budget. J. Geophys. Res. Atmos. 117, 812–819.
He, J., Gong, S., Yu, Y., Yu, L., Wu, L., Mao, H., Song, C., Zhao, S., Liu, H., Li, X., Li, R., 2017. Air pollution characteristics and their relation to meteorological conditions during 2014–2015 in major Chinese cities. Environ. Pollut. 223, 484–496.
Laila, B., Huda, Q., Nihal, N., Reem, B., Wafaa, K.T., Rehab, D., Ibrahim, H., 2018. Effects of low concentrations of ozone (o_3) on metabolic and physiological attributes in wheat (*Triticum aestivum* L.). Pants. Biomed. Pharmacol. J. 11, 1–6.

Lei, Y., Zhang, Q., He, K., Streets, D., 2011. Primary anthropogenic aerosol emission trends for China, 1990–2005. Atmos. Chem. Phys. 11, 17153–17212.
Li, M., 2015. Effects of ozone on photosynthesis of serval plants. Environ. Sci. 36, 1888–1992.
Li, T., Yan, M., Ma, W., Ban, J., Liu, T., Lin, H., Liu, Z., Liu, Z., 2015. Short-term effects of multiple ozone metrics on daily mortality in a megacity of China. Environ. Sci. Pollut. Res. 22, 8738–8746.
Li, P., Feng, Z., Catalayud, V., Yuan, X., Xu, Y., Paoletti, E., 2017. A meta-analysis on growth, physiological and biochemical responses of woody species to ground-level ozone highlights the role of plant functional types. Plant Cell Environ. 40, 2369–2380.
Li, P., Marco, D., Feng, Z., Anav, A., Zhou, D., Paoletti, E., 2018a. Nationwide ground-level ozone measurements in China suggest serious risks to forests. Environ. Pollut. 237, 803–813.
Li, P., Feng, Z., Shang, B., Yuan, X., Dai, L., Xu, Y., 2018b. Stomatal characteristics and ozone dose-response relationships for six greening tree species. Acta Ecol. Sin. 38, 2710–2721.
Lin, J., Nielsen, C.P., Zhao, Y., 2010. Recent changes in particulate air pollution over China observed from space and the ground: effectiveness of emission control. Environ. Sci. Technol. 44, 7771–7776.
Lin, H., Zhang, Y., Liu, T., Xiao, J., Xu, Y., Xu, X., Qian, Z., Tong, S., Luo, Y., Zeng, W., Ma, W., 2014. Mortality reduction following the air pollution control measures during the 2010 Asian Games. Atmos. Environ. 91, 24–31.
Liu, X., Zhang, Y., Han, W., Tang, A., Shen, J., Cui, Z., Fangmeier, A., 2013. Enhanced nitrogen deposition over China. Nature 494, 459–462.
Liu, D., Li, J., Zhang, Y., Xu, Y., Liu, X., Ding, P., Shen, C., Chen, Y., Tian, C., Zhang, G., 2013. The use of levoglucosan and radiocarbon for source apportionment of PM2.5 carbonaceous aerosols at a background site in East China. Environ. Sci. Technol. 47, 10454–10461.
Long, J., Jiang, Y., Liu, J., Cheng, H., 2017. Physiological effects of acute ozone stress on 6 types of landscape shrubs. J. Xiamen Univ. Technol. 25, 77–82 (In Chinese).
Marco, D., 2009. Assessment of present and future risk to Italian forests and human health: modelling and mapping. Environ. Pollut. 157, 1407–1412.
Matyssek, R., Karnosky, F., Wieser, G., Percy, K., Oksanen, E., Grams, E., 2010. Advances in understanding ozone impact on forest trees: messages from novel phytotron and free-air fumigation studies. Environ. Pollut. 158, 1990–2006.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2011a. Report on environmental quality in China, 2010. http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2011b. Annual report on China vehicle emission control, 2010. http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2012a. Ambient air quality standard (GB3095-2012). http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2012b. Notification of GB3095-2012 implementation. http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2012c. Technical regulation on ambient air quality index (on trial). http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2012d. China environment status bulletin, 2011. http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2017a. China environment status bulletin, 2016. http://www.mee.gov.cn.
MEE (Ministry of Ecology and Environment of the People's Republic of China), 2017b. Report on environmental quality in China, 2016. http://www.mee.gov.cn.
OECD (the Organisation for Economic Co-operation and Development), 2007. OECD Environmental Performance Reviews: China 2007. OECD Publishing, Paris https://doi.org/10.1787/9789264031166-en.
Paoletti, E., 2009. Ozone and urban forests in Italy. Environ. Pollut. 157, 1506–1512.
Paoletti, E., Marco, D., Racalbuto, S., 2007. Why should we calculate complex indices of ozone exposure? Results from Mediterranean background sites. Environ. Monit. Assess. 128, 19–30.
Ping, Q., Xu, S., Li, J., He, X.Y., Chen, W., Huang, Y.Q., 2017. Ecophysiological responses of turf-type white clover (*Trifolium repens*) to elevated O_3 concentration. Chin. J. Ecol. 36, 1234–1242 (In Chinese).
Popp, D., 2003. Pollution control innovations and the clean air act of 1990. J. Policy Anal. Manage. 22, 641–660.
Samet, J.M., 2011. The Clean Air Act and health — a clearer view from 2011. N. Engl. J. Med. 365, 198–201.
Selin, E., Wu, S., Nam, M., Reilly, M., Paltsev, S., Prinn, G., 2009. Global health and economic impacts of future ozone pollution. Report - MIT Joint Program on the Science and Policy of Global Change. 4, pp. 940–941.
Shi, Y., Xia, Y., Lu, B., Liu, N., Zhang, L., Li, S.J., Li, W., 2014. Emission inventory and trends of NO_x for China, 2000–2020. J. Zhejiang Univ. Sci. A 15, 454–464.
Straf, K., Cohen, A., Sammet, J., 2013. Air Pollution and Cancer, IARC Scientific Publication 161, International Agency for Research on Cancer. World Health Organization, Lyo, FR.
Vennemo, H., Aunan, K., Lindhjem, H., Seip, M., 2009. Environmental pollution in China: status and trends. Rev. Environ. Econ. Policy 3, 209–230.
Wang, S., Hao, J., 2012. Air quality management in China: issues, challenges, and options. J. Environ. Sci. 24, 2–13.
Wang, S., Zhao, M., Xing, J., Wu, Y., Zhou, Y., Lei, Y., Hao, J., 2010. Quantifying the air pollutants emission reduction during the 2008 Olympic games in Beijing. Environ. Sci. Technol. 44, 2490–2496.
Wang, T., Nie, W., Gao, J., Xue, L., Gao, X., Wang, X., Qiu, J., Poon, C., et al., 2010. Air quality during the 2008 Beijing Olympics: secondary pollutants and regional impact. Atmos. Chem. Phys. 10, 7603–7615.
Wang, X., Zhang, Q., Zheng, F., Zheng, Q., Yao, F., Chen, Z., 2012. Effects of elevated O_3 concentration on winter wheat and rice yields in the Yangtze River Delta, China. Environ. Pollut. 171, 118–125.
Wang, S., Zhao, B., Cai, Y., Klimont, Z., Nielsen, C.P., Morikawa, T., Hao, J., 2014. Emission trends and mitigation options for air pollutants in East Asia. Atmos. Chem. Phys. 14, 6571–6603.
Wang, T., Xue, L., Brimblecombe, P., Lam, Y.F., Li, L., Zhang, L., 2016. Ozone pollution in China: a review of concentrations, meteorological influences, chemical precursors, and effects. Sci. Total Environ. 575, 1582–1596.
Wang, W., Cheng, T., Gu, X., Chen, H., Guo, H., Wang, Y., Meng, C., 2017. Assessing spatial and temporal patterns of observed ground-level ozone in China. Sci. Rep. 7, 3651.
Wu, Y., Wang, R., Zhou, Y., Lin, B., Fu, L., He, K., Hao, J., 2011. On-road vehicle emission control in Beijing: past, present, and future. Environ. Sci. Technol. 45, 147.
Wu, X., Wu, Y., Zhang, S., Liu, H., Fu, L., Hao, J., 2016. Assessment of vehicle emission programs in China during 1998–2013: achievement, challenges and implications. Environ. Pollut. 214, 556–567.
Wu, Y., Zhang, S., Hao, J., Liu, H., Wu, X., Hu, J., Walsh, M., Wallington, T., Zhang, K., Stevanovic, S., 2017. On-road vehicle emissions and their control in China: a review and outlook. Sci. Total Environ. 574, 332–349.
Xu, Y., 2011. Improvements in the operation of SO_2 scrubbers in China's coal power plants. Environ. Sci. Technol. 45, 380–385.
Xue, B., Mitchell, B., Geng, Y., Ren, W., Müller, K., Ma, Z., Tobias, M., 2014. A review on China's pollutant emissions reduction assessment. Ecol. Indic. 38, 272–278.
Yang, F., Tan, J., Zhao, Q., Du, Z., He, K., Ma, Y., Duan, F., Chen, G., 2011. Characteristics of $PM_{2.5}$ speciation in representative megacities and across China. Atmos. Chem. Phys. 11, 5207–5219.
Yang, W., Lv, T., An, S., Zhao, D., 2014. Comparison of near-ground ozone concentrations between urban and rural forests. Acta Ecol. Sin. 34, 5670–5678 (In Chinese).
Yang, N., Wang, X., Zhang, Y., Zheng, F., Chen, Y., 2017. Ozone injury to marigold (*Tagetes erecta* Linn.), petunia (*Petunia hybrida* Vilm.) and morning glory (*Pharbitis purpurea* (L.) Voigt) and their different physiological responses. Asian J. Ecotoxicol. 12 (6), 141–149 (In Chinese).
Yu, H., He, N., Wang, Q., Zhu, J., Gao, Y., Zhang, Y., Yu, G., 2017. Development of atmospheric acid deposition in China from the 1990s to the 2010s. Environ. Pollut. 231, 182–190.
Zhang, K., Song, S., 2003. Rural–urban migration and urbanization in China: evidence from time-series and cross-section analyses. China Econ. Rev. 14, 386–400.
Zhang, Q., Streets, G., He, K., Wang, Y., Richter, A., Burrows, P., Lei, Y., 2007. NO_x emission trends for China, 1995–2004: the view from the ground and the view from space. J. Geophys. Res. Atmos. 112, D22306.
Zhang, W., Feng, Z., Wang, X., Niu, J., 2014. Impacts of elevated ozone on growth and photosynthesis of *Metasequoia glyptostroboides* Hu et Cheng. Plant Sci. 226, 182–188 (In Chinese).
Zhao, B., Wang, S., Liu, H., Xu, J., Fu, K., Klimont, Z., Hao, J., He, K., Cofala, J., Amann, M., 2013. NO_x emissions in China: historical trends and future perspectives. Atmos. Chem. Phys. 13, 9869–9897.
Zhao, H., Zheng, Y., Wei, L., Guan, Q., 2018. Seasonal variation in surface ozone and its effect on the winter heat and rice in Nanjing, China. Environ. Sci. 39, 3418–3425 (In Chinese).