The Climate and Air Quality Co-benefits of Controlling Black Carbon and Organic Carbon: A Review of Emissions, Impacts and Policies

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Preface

This book is a series of review reports produced under the Chinese-Norwegian Project on Emission, Impact, and Control Policy for Black Carbon and its Co-benefits in Northern China (ChiNorBC). The project is jointly implemented by the Chinese Research Academy of Environmental Sciences (CRAES) and the Norwegian Environment Agency (NEA), in partnership with the Chinese Academy of Environmental Planning (CAEP), the Norwegian Institute of Public Health (NIPH) and Center for International Climate Research (CICERO), with financial support from the Norwegian Ministry of Foreign Affairs.

There is no internationally agreed definition of black carbon (BC) and organic carbon (OC). BC is the light-absorbing component of fine particles and is produced by incomplete combustion of fossil fuel, biofuel and biomass. BC is always co-emitted with OC. Emissions of BC and OC affects the climate and have adverse health effects. Reductions of BC and OC will have co-benefits for climate, air quality and health.

ChiNorBC will develop improved emission inventories for BC/OC emissions in China using the most recent, best available national statistics and measurements obtained in the project. Based on this, new estimates of effects of BC/OC on climate, air quality, and health will be provided. The project will further raise scientific, governmental, and public awareness and enhance the understanding of the positive impacts of BC/OC emissions reductions. Ultimately the ChiNorBC will provide Chinese policy makers with policy solutions for reducing BC/OC emissions in China which maximizes the co-benefits.

For a more comprehensive description of the project, and to get access to all the project reports, please visit the project web site http://chinorbc.net/.

The authors have contributed to this book in their individual capacity and their organizations have been mentioned for identification purposes. The development and coordination of this book was led by Miaomiao Cheng and Xiaoyan Zhu; Chapter 1 was written by Xiaoyan Zhu, Yongjie Wei, Miaomiao Cheng and Vigdis Vestreng; Chapter 2 was written by Guorui Zhi, Yanjun Wang, Yuzhe Zhang, Wenjing Jin; Chapter 3 was written by Xiaohui Du, Jun Xu, Marianne Tronstad Lund, Ole-Kristian Kvissel; Chapter 4 was written by Yongjie Wei, Xiaojing Zhu, Zhigang Li, Bingqian Liu, Per Schwarze, Shilpa Rao, Marit Låg, Vegard S. Grytting and Alfonso Diz-Lois Palomares; Chapter 5 was written by Miaomiao Cheng, Yu Chen and Wenyong Guo; Chapter 6 was written by Ingeborg Rønning, Scott Randall and Vigdis Vestreng; and Chapter 7 was written by Xuying Wang, Yixuan Zheng; Individual chapters of the book was reviewed and edited by the whole project team, in particular Miaomiao Cheng, Vigdis Vestreng, Tor Skudal and Fan Meng.

Content

1 Introduction

Black carbon (BC) and organic carbon (OC) constitute important parts of atmospheric fine particulate matter ($PM_{2.5}$). By reducing the emissions we can obtain both climate and health benefits. Emission control measures will in many cases also mitigate other harmful pollutants, thus many co-benefits can be achived by optimizing reduction strategies.

Studies have shown that the warming effect of BC is second only to that of greenhouse gases. The warming effect is because BC heats the atmosphere, but also because it melts snow and ice, when deposited on the ground. The climate effect depends on where on the Earth the BC emissions occur. BC only stays in the atmosphere a short time (days), and is therefor not globally well mixed. Thus there are large regional differences. OC is always co-emitted with BC (we therefore often use the term BC/OC in this book), but has a cooling effect on the atmosphere. Thus it is important to look at the net climate effect of control measures.

Emissions of BC and OC have adverse health effects. Even though there are uncertainties in terms of emission data and climate and health effects, we have enough knowledge to act.

In this book, we present the status of BC/OC emission inventories in China (Chapter 2), impact on air quality and climate change (Chapter 3), health effects (Chapter 4) and status of BC/OC control in both China (Chapter 5) and Norway (Chapter 6). Finally we give an overview on how to reduce BC/OC emission in China in the future (Chapter 7).

1.1 What is BC and OC?

Particulate matter (PM), often expressed as 10 or 2.5 μm mass median aerodynamic diameter (PM_{10} or $PM_{2.5}$), is recognized as one of the most dangerous pollutant to health (Cassee et al., 2013). BC is an important part of PM and appears always combined with OC and other compounds to form PM, which is present in ambient air (Pace, 2002). BC is formed through the incomplete combustion of fossil fuels, biofuel and biomass, and is emitted from both anthropogenic and natural sources It consists of pure carbon in several linked forms. BC warms the earth by absorbing sunlight and re-emitting heat to the atmosphere and by reducing albedo (the ability to reflect sunlight) when deposited on snow and ice.

BC absorbs a wide range of solar radiation, from ultraviolet (UV) to infrared (IR). The diameter of BC is generally between 0.1-1 μm (Qin et al., 2001). Due to the small particle size, it is easily spread over a long distance in the atmosphere.

EC is an operational definition that encompasses carbonaceous particles which can only be oxidized at temperatures > 340℃ and are thermally stable in inert gases below approximately 4000 K (Bond et al., 2013; J.A.Ogren et al., 1983; Petzold et al., 2013). Even though EC and BC are measure by different methods and no fixed correlation between the two terms have been established, BC and EC are often used interchangeably in emission inventories and other studies.

OC refers to the fraction of carbonaceous aerosols which contains organic compounds (chemical compounds that contain carbon-hydrogen bonds as well as other elements).

There are also other terms used to characterize different carbon containing componds, for example brown carbon (BrC) (Zhi et al., 2015), black smoke (BS) (Quincey et al., 2011), Equivalent black carbon (eBC) and refractory black carbon (rBC), (Lack et al., 2014), carbon black (CB) (Long et al., 2013), and light-absorbing carbon (LAC).

Even though there are many different terms, in epidemiological studies, BC, EC and BS are commonly used to estimate the association with concentrations of BC exposure and human health.

1.2 Why Concern: What are the effects on health, air quality and climate of BC/OC?

Due to the rapid economic development in China in recent decades, the combustion of coal, other fossil fuels and biofuels have increased considerably. Large amounts of BC/OC have been emitted into the atmosphere.This situation has not only caused heavy air pollution in megacities of China. BC and OC are also complex climate forcers of regional climate change due to different lifetimes and properties. Air pollution and climate change are closely linked, and reductions in BC/OC can result in substancial co-benefits.

Northern China which includes about half of the Chinese population and the vast area north of the Huai River-Qin Mountain line in China, is experiencing severe air pollution, particularly during the winter season due to increased coal and firewood combustion for heating and unfavorable weather conditions, such as calm winds and a strong inversion layer. BC/OC produces adverse effects on air quality, health and climate change and has become one of the hotspot issues in environmental and climatic research.

BC/OC are two of the short-lived climate forcers (SLCFs). BC is an important particulate pollutant and the cause of regional air pollution and decreased visibility. BC contributes about 5%-15% to the total aerosol mass concentration in urban air. Furthermore, BC together with BrC is one of the most radiatively important aerosol components in the atmosphere. BC plays an important role in regional and global climates as well as extreme weather. BC aerosols enhance the occurrence of extreme haze pollution episodes in megacities in China. In addition, BC constitutes the main fraction of fine particles (0.5-1 μm) and ultrafine particles (0.05-0.12 μm), which have affected human health. Due to the porous structure of organic compounds, other chemicals, including toxic, are easily absorbed.

Accumulating evidence has identified the health risks associated with short-lived climate forcers, particularly BC, secondary organic and inorganic aerosols. Exposure to BC/OC may lead to decreased lung function, obesity and effects on neurocognition in human being.Toxicological and health risk assessment studies conducted in northern China will help to better understand the relationship between diverse exposure type and the incidence of disease. Additional air quality metrics (e.g., BC/OC and secondary organic and inorganic aerosols) may be valuable when evaluating air quality, climate change and health risks.

BC, OC and other aerosols affect the climate through their interactions with solar radiation and clouds. Most aerosol species have a cooling climate impact, while BC stands out by absorbing solar radiation and warming the atmosphere. There are significant uncertainties, but the best estimate of the global aerosol effect on climate is a net cooling that has masked a notable fraction of the anthropogenic warming induced by greenhouse gas emissions to date. Additionally, aerosols influence local and regional climate and weather, such as the precipitaiton, haze events, and extreme weather, although the exact nature and extent of their role is still a topic of ongoing research. The significant range in estimates of aerosol's climate effects stem from uncertainties in their distribution, atmospheric processing, and optical and microphysical properties, as well as uncertainties in their sources. Despite progress over the past decades, representing aerosols remain a challenge for global and regional models of the atmosphere, with validation in many cases hampered by lack of observations.

1.3 The Overview of the ChiNorBC-Project

The Ministry of Commerce of the People's Republic of China (MOFCOM) and the Norwegian Ministry of Foreign Affair (MFA) entered into an agreement about the the Chinese- Norwegian Project on Emission, Impact and Control Policy for BC and it's Co-benefits in Northern China (ChiNorBC) dated at 29 November 2019. The agreement is in accordance with the memorandum of understanding (MoU) regarding Economic and Technical Cooperation signed by the Government of the people's Republic of China and the Government of the kingdom of Norway, as well as the memorandum of understanding signed by the Ministry of Ecology and Environment of the People's Republic of China (MEE) and the Norwegian Ministry of Climate and Environment (MCE).

The implementation period of the project is 3 years (from October 2019 to December 2022) and the project fund is 24.16 million Norwegian kroner. This project will first develop a baseline by analyzing the current emission and concentration levels in northern China, as well as existing strategies to reduce the impacts of BC/OC. The study area contains Beijing-Tianjin-Hebei (BTH) and the surrounding areas (Beijing, Tianjin, Hebei, Inner Mongolia, Shanxi, Henan and Shandong Provinces), Northeast China (Heilongjiang, Jilin, and Liaoning Provinces) and Northwest China (Shanxi, Ningxia, Gansu and Xinjiang Provinces). Current practices in Norway and other countries will be assessed to gain experience. With this background, the project will develop or update relevant emission inventories and model air quality and climate effects. Effects on health will also be studied and quantified. Finally, the project will develop and propose policy scenarios towards 2035 for reducing the emissions and impacts of BC/OC. This book documents the baseline for the studies undertaken and documented in the ChiNorBC project. For further reading please consult our website, http://chinorbc.net/.

2 Review on BC/OC Emission Inventories

The project document for emission inventory stipulates preparation of "Review literature on the BC/OC emission inventories for northern China and estimation methodology in China and other countries." We will actually consider that the inventories to be built by the project are not only for northern China, but for all of China. Prior to systematic drafting of this review, the two parties of this project, from China and Norway, initially collaborated to formulate the review framework and then interactively commented on and revised the report.

The first challenge, however, is understanding exactly what BC is. Researchers around the world find it difficult to agree on terminology that considers all aspects of specific properties, definitions, measurement methods and related uncertainties. This leads to much ambiguity in the scientific literature on measurements and numerical models, where BC may be referred to with different names or based on different properties of the particles, with no clear definition of the terms. For example, global emission inventories and modelling studies (Bond et al., 2007; Granier et al., 2011; Junker and Liousse, 2008; Lee et al., 2012; Vignati et al., 2010), as well as scientific assessments depend on BC data sets that do not include information on measurement methods (Solomon et al., 2007; Bond et al., 2013). For this reason, Petzold et al. (2013) proposed definitions of terms and recommendations for reporting measurements of BC, EC, light absorption, refractory carbon, and other properties related to this distinct fraction of the carbonaceous aerosol. Soot is a useful qualitative description when referring to carbonaceous particles formed from incomplete combustion, though non-carbonaceous matter is also included in soot. Soot and BC have been used interchangeably in much existing literature to refer to light absorbing carbon, especially in early publications when BC was not so deeply concerning the scientists. In this review, we prefer to use BC in most cases, without excepting the use of other terms, such as EC, when necessary.

Available literature explains various aspects of BC (Bond & Bergstrom, 2006; UNEP, 2015; UNEP/WMO, 2011). BC is the light-absorbing component of carbonaceous particles; the light absorbing property comes from its graphitic bonds. BC is produced by incomplete combustion of fossil fuel, biofuel and biomass. BC is emitted by various devices and processes, such as diesel cars and trucks, residential stoves, forest fires, agricultural open burning and some industrial facilities. The warming effects of BC on the atmosphere are caused by soot particles absorbing direct or reflected sunlight and then reemitting heat to the atmosphere and by reducing albedo when deposited on snow and ice. The warming impact of BC on climate is about 700 times stronger than CO_2 when BC's global warming potential (GWP) is estimated over a 100-year time scale. In a short-term perspective the impact is even larger due to the short impact of BC in the atmosphere (GWP (20) = 2700) (Aamaas and Berntsen, 2021). BC also influences cloud formation and impacts regional circulation and rainfall patterns. In addition, BC impacts human health as a primary component of PM in air pollution.

In recent decades, on the one hand, the rapid growth of China's economy has been accompanied by a considerable increase in the combustion of coal, other fossil fuels and biofuels, resulting in an increase in BC/OC emissions (Cheng et al., 2017; Zhi et al., 2017). On the other hand, China's continually intensified clean air pursuit has prompted China to accelerate the transfer from solid fossil energy to clean energy (gas, solar energy, electricity), especially in the residential sector and mobile sources (Pelegov and Eremenko., 2021; Zhang et al., 2020). Therefore there are multiple benefits for sustainable development from establishing an efficient policy system for the emission reduction of BC/OC on both regional and national levels in China.

BC/OC emission data related to northern or all of China are published by different institutions and are often inconsistent in total or in specific sectors due to the substantial differences in emission factors chosen, activity levels recognized and methodologies applied (Li et al., 2017a). The inconsistent emission results often lead to misunderstanding and confusion and add difficulty to identifying major emission sources and formulating control policy. For this reason, a more comprehensive and convincing emission inventory of BC/OC for northern China is necessary to provide scientific data supporting the formulation and implementation of policies by incorporating new emission factors derived from measurements, investigated activity levels, and improved methodology.

Among the several targets of the review, the analysis of BC/OC emission factors is of top importance. This will contribute to the development of emission factor tables for mobile and residential sources, which are generally considered major sectors in a BC/OC emission inventory.

2.1 BC/OC Emission Inventories

2.1.1 Approaches to Establishing Air Pollutant Emission Inventories

There are usually two approaches to establishing air pollutant emission inventories: bottom-up and top-down. The "bottom-up" approach works out total emissions of individual pollutants based on the detailed fuel consumption (activity level) data of sectors multiplied by corresponding emission factors (Bond et al., 2004; Cao et al., 2006). The "top-down" approach infers emissions from one or more indicators relevant to emissions. Such indicators may be ambient concentrations observed by ground equipment or column concentrations observed by satellite. In top-down rationale, inverse modelling is an important approach, which translates observed concentrations into primary emissions by using measurements of atmospheric concentrations in combination with modeled fields. However, modelling processes such as transport and removal also result in biases. In most cases, a bottom-up approach is preferred in establishing basic emission inventories; such inventories can be used for model inputs so as to simulate ambient concentrations.

The top-down approach is more widely used as an important constraining indicator to test, validate and evaluate bottom-up emissions. For example, Kaiser et al. (2012) derived a global average enhancement factor of 3.4 for emissions of the organic aerosols (OA) and BC resulting from biomass burning and summarized several other top-down studies that estimated emissions two to four times greater than the bottom-up estimates. This shows that top-down estimates can expose underestimations of BC emissions from bottom-up estimates. A thesis by Zhao (2019) describes a top-down estimate of regional BC emissions using ground and satellite observations for the Yangtze River Delta Region. Bond et al. (2013) presented more examples to compare the bottom-up emissions and the top-down estimates.

2.1.2 Global emissions

2.1.2.1 General description

Because of the impacts of BC emissions on climate change, local air pollution and human health, there have been many studies on the sources of BC emissions since the 1990s. The emission figures and sources of BC in different parts of the world have been explored and are summarized below.

Cooke et al. (1999), at the Mixte CNRS-CEA laboratory of France, calculated the global-scale emissions of carbonaceous aerosols from fossil fuel usage. Using a top-down method, they estimated the global BC emissions from the burning of fossil fuels in 1984 to be approximately 5.1-6.4 million tonnes, of which China contributed about 1.15-1.46 million tonnes, accounting for about 22.8% of global BC emissions. The contribution of BC emissions from mobile/transport sources was 17%-23%.

Penner et al. (1993), in the U.S., estimated that global BC emissions (biomass burning excluded) were about 12.61 million tonnes in 1980, of which China contributed about 2.68 million tonnes.

Bond et al. (2004) calculated global BC emissions, using the bottom-up approach, by assigning emission factors on the basis of fuel type and economic sectors (regions and countries). They estimated that global BC emissions in 1996 were about 8 million tonnes, of which China contributed about 1.49 million tonnes or about 18.6% of the total. Bond et al. also estimated in detail the contributions of different sources to BC emissions. The “open burning” source contributed the largest part of BC emissions, about 42%. About 10% of global BC emissions were from industrial sources, 24% from residential sources and 24% from mobile sources (Figure 2-1).

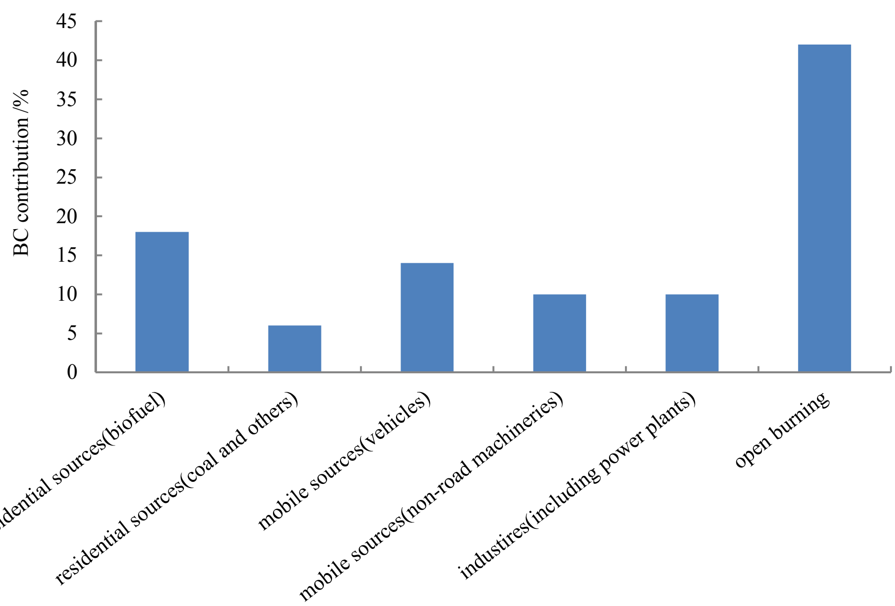


Figure 2-1 Global BC contribution of different sources in 1996(Bond et al., 2004)

Bond et al. (2007) estimated that global BC emissions in 2000 were about 8.4 million tonnes.

Asia, parts of Africa and Latin America (Central and South America) contributed most of the BC emissions.

Lamarque et al. (2010) showed that 75% of the world's BC emissions in the year 2000 (biomass burning included) came from three main regions: Asia (China, India), Africa, and Latin America. Asia accounted for 40% of the emissions, while Africa and Latin America accounted for about 23% and 12%, respectively (Figure 2-2).

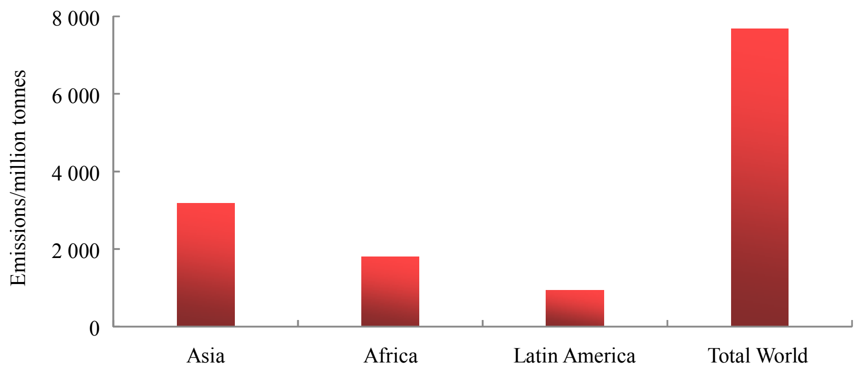


Figure 2-2 Global and regional BC emissions

In addition, Lamarque et al. (2010) divided global BC emissions into six major sources. They estimated that global BC emissions were still dominated by open biomass combustion (including wildfires), which accounted for about 35%, while BC emissions from domestic stoves and heating accounted for about 25% of the global total (Figure 2-3). In developing countries, BC from combustion was mainly due to the burning of coal, biomass or animal waste, by which China, India and Africa accounted for nearly two thirds of global anthropogenic BC emissions.

Several inventories covering more recent years are now available. For the sixth cycle of the Coupled Model Intercomparison Project (CMIP6) and IPCC Assessment Report, the Community Emission Data System (CEDS) provided new historical emission estimates for anthropogenic aerosol and precursor emissions, up to 2014 (Hoesly et al., 2018). Emission estimates in this first CEDS release are generally slightly higher than in other global inventories, with global anthropogenic BC and OC emissions in 2014 of 8 and 19.7 million tonnes while in 2010 of 7.7 million tonnes and 18.7 million tonnes, respectively. Two other global inventories with 2015 as the most recent year, the Emissions Database for Global Atmospheric Research version 5 (EDGARv5) (Crippa et al., 2019; https://edgar.jrc.ec.europa.eu/dataset_ap50) and the ECLIPSEv5 created with the Greenhouse Gas - Air Pollution Interactions and Synergies

(GAINS) model (Amann et al., 2011; Klimont et al., 2017), both have lower global BC and OC estimates. Global anthropogenic BC emissions are approximately 5 million tonnes in both these inventories, while OC emissions are 13 million tonnes and 11.7 million tonnes in ECLIPSEv5 and EDGARv5, respectively.

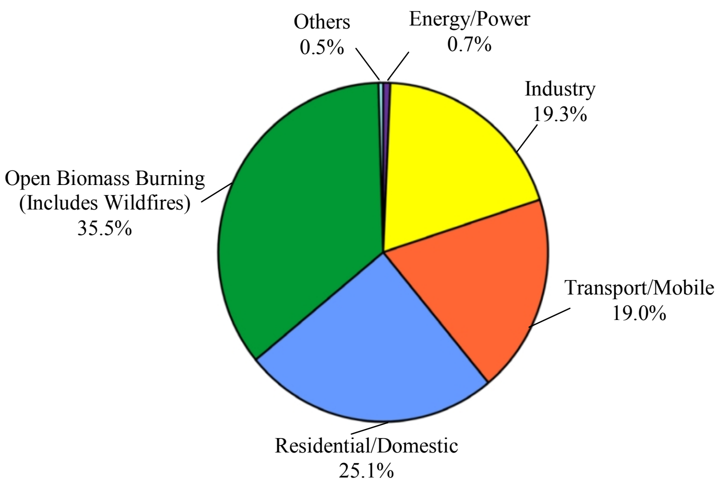


Figure 2-3 Main sources of global BC

These inventories also differ somewhat in their regional and global temporal trends. For instance, while CEDS shows a rapid increase in Asian emissions over the 2000-2014 period, the evolution is flatter in ECLIPSEv5 and levels off in 2014 and 2015 in EDGAR5. An updated release of the CEDS emissions, extending to year 2019 (O'Rourke et al., 2021), shows a levelling off in global emissions since 2014, driven by a combination of increasing emissions in some regions, such as South Asia and Africa south of the Sahara and a decline in emissions in China, the latter in line with recent observational evidence (Kanaya et al., 2020). A similarity across inventories is in the sectoral and regional distributions, where the largest individual contributions to global emissions are from the residential and commercial sectors and from China and the rest of the Asia-Pacific region, for both pollutants. While significant work has been undertaken to improve the estimates of global and regional emission inventories of aerosols and precursors, uncertainties in magnitude and spatiotemporal patterns remain.

2.1.2.2 Global BC emissions and emission source trends from 1960 to 2017

The global BC emissions show an inverse U-shaped temporal trend, which is mainly driven by the interaction between the positive effects of population growth, per capita energy consumption and vehicle fleet, and the negative effects of residential energy switching, stove upgrading, phasing out of beehive coke ovens and reduced emission factors (EFs) for vehicles and industrial processes. Urbanization caused an important increase in urban emissions as well as a significant decline in overall rural emissions.

As Figure2-4 shows, the global BC emissions increased steadily from 1960 to 2015. That study also used EDGAR and CEDS to estimate the global BC emissions (Xu et al., 2021).

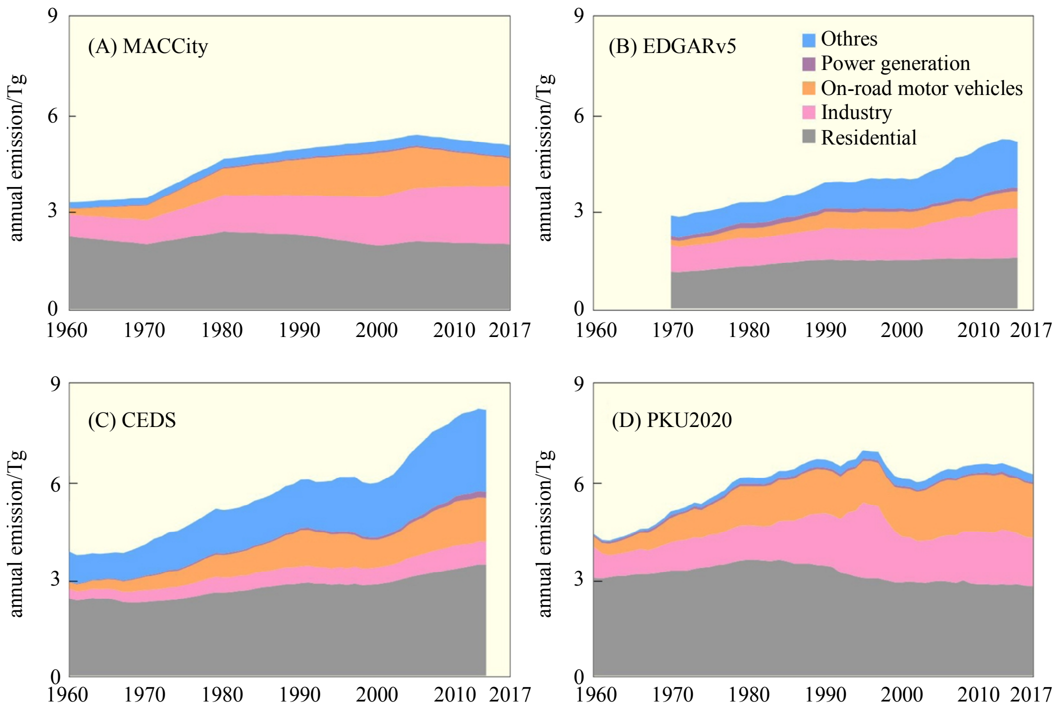


Figure 2-4 Temporal trends of annual BC emissions from major anthropogenic sectors (residential, on-road motor vehicles, industry, and others) from 1960 to 2017 reported by three other emission inventories for comparison. The compared inventories are (A) MACCity, (B) 19 Emissions Database for Global Atmospheric Research (EDGARv5), (C) Community Emission Data System (CEDS), (D) Peking University emission inventories, 2020 version (PKU 2020)

2.1.3 Regional Emissions

2.1.3.1 The Asia-Pacific Regional Emission Inventories Led by the U.S.

Compared with developed countries (such as European countries and the U.S.), China's research on the emission inventory of air pollutants was launched much later due to the late start of industrial development. For research on air pollution in China, one of the most widely used emission inventories is the one called "TRACE-P emission inventory" (Streets et al., 2003). This inventory is an outcome of the project "Transport and Chemical Evolution over the Pacific, TRACE-P" (Jacob et al., 2003; Streets et al., 2006) from the National Aeronautics and Space Administration (NASA) and the project "Asian Pacific Regional Aerosol Characterization Experiment, ACE-Asia" (Huebert et al., 2003). These projects were funded by the National Oceanic and Atmospheric Administration (NOAA) and National Science Foundation (NSF). The target of these studies was to understand and quantify the effects of chemical outflows from Asia on the U.S. Emissions were estimated at 2.54 Tg BC and 10.4 Tg OC for all major anthropogenic sources, including biomass burning, in 64 regions of Asia. China dominated Asian emissions, accounting for 41.3% of total BC and 32.7% of total OC. China's energy data in "TRACE-P" came from the research results of Sinton and Fridley (2000) rather than official energy statistics. The other data in "TRACE-P" came from the results of the project "RAINS-ASIA" (Amann et al., 2004). Information on pollutant emission factors were taken from several different sources including Fu et al. (2001) for motor vehicles, Andreae and Merlet (2001) for biomass burning and Bouwman et al. (1997) for Ammonia (NH_3). Aerial experiments have also been carried out over the western Pacific in combination with ground observations, satellite data and model simulations for emission inventories preparation (Ohara et al., 2007).

Then, in 2006, the project "Intercontinental Chemical Transport Experiment-Phase B, INTEX-B, " launched by NASA, prepared the INTEX-B emission inventory (Zhang et al., 2009). This inventory is a continuation of the TRACE-P emission inventory. It uses official energy statistics and gives a relatively accurate picture of emissions across Asia, particularly in China. Emissions in 2006 were estimated at 2.97 Tg BC and 6.57 Tg OC for all major anthropogenic sources, excluding biomass burning. China also dominated Asian emissions, representing 60.6% of the total BC and 48.7% of Asian total OC. The authors estimated 2001 emissions for China using the same methodology and found that all components showed an increasing trend during 2001-2006, including 14% for both BC and OC, though for most sectors net emission factors were fundamentally changed due to dramatic economic growth and dynamic technology penetration. For example, in power plants the net emission factors of $PM_{2.5}$ declined from 2.0g/kg coal to 1.2g/kg coal.

2.1.3.2 The Asian Emission Inventory Led by Japan

Besides the Asia-Pacific emission inventory led by the U.S., International Institute for Applied Systems Analysis (IIASA) and the World Bank launched the project "RAINS-Asia" which provided data support for acid rain mitigation in Asia in the 1990s. On this basis, Japan's National Institute of Environmental Research developed the Regional Emissions Asian Inventory (REAS) (Ohara et al., 2007), which is a relatively complete and comprehensive emission inventory of Asia (biomass burning excluded). REAS-1.1 version integrated historical emission data (up to 2007) and forecasted future emissions (up to 2020). It was subsequently updated to REAS-2.1 (Kurokawa et al., 2013), in which the spatial resolution was further improved. This version of REAS revised the data of different regions in China by referring to the INTEX-B emission inventory. It took not only the effect of pollution control technologies on emission factors into account in detail, but also China's basic energy consumption in the statistical yearbook of each province. Asian emissions for BC and OC in 2008 were estimated at 3.03 Tg and 7.72 Tg in this inventory, up 35% and 21% compared with 2000. By country, China and India were respectively the largest and second largest contributors because of their continuous increases in energy consumption, industrial activities and infrastructure development.

2.1.4 China Emissions

The BC emissions of China[①] calculated by different scholars differ considerably, but all show a large contribution to global BC emissions. Streets et al. (2001) studied the 1995 emissions of China by sector and province. Using the bottom-up method and emission factors by sectors and fuel types (based on consumption of various fuel types) derived from different studies, they estimated that the BC emissions of China were about 1.342 million tonnes in 1995. The residential sector contributed the largest part of BC, about 83.3%. The industry sector contributed about 7.2% of the total BC emissions, ranking second. The power generation sector contributed about 8,300 tonnes of BC emissions, accounting for about 0.6% of the total. The total BC emissions from mobile sources was 43,400 tonnes, accounting for 3.2% of the total. Field combustion (open burning) released about 74,700 tonnes BC, representing about 5.6% of the total. They also forecasted that China's BC emissions would fall to 1.224 million tonnes by 2020 after adoption of particle/BC emission control technology. But Streets et al. expected a

① In this chapter the emissions of China may be inclusive or exclusive of Hongkong, Macao, and/or Taiwan, just depending whether the original literature cited have the data of Hongkong, Macao, and/or Taiwan.

faster increase in the number of vehicles in China. Therefore, they used the BC emission factor of 1995 and predicted that BC emissions from mobile sources in China would rise to 139,300 tonnes by 2020, accounting for 11.4% of the expected BC total.

The first BC-related emission inventory made by Chinese scholars was most likely that by Cao et al. (2006), with the Chinese Academy of Meteorological Sciences. This study was later expanded with more pollutant types (Cao et al., 2011) (biomass burning included). Based on the method developed for the INTEX-B emission inventory, Tsinghua University updated the statistical data and the calculation method, and recalculated China's emissions in the INTEX-B emission inventory (Lei et al., 2011) (biomass burning excluded). The uncertainties of the emissions in the inventory were also quantified (Zhao et al., 2011). Subsequently, Tsinghua University established the first multi-scale air pollutant emission inventory, Multi-resolution Emission Inventory for China (MEIC) based on the method for emission inventory construction for technology and dynamic processes (biomass burning excluded). MEIC (V1.0) was released in 2012 (Lu et al., 2011). It includes 10 pollutants [SO_2, NO_x, CO, NH_3, non-metherne Volatile Orgomic Compounels (nmVOC), PM_{10}, $PM_{2.5}$, BC, OC, CO_2] and more than 700 emission sources (including power plant, industrial, agricultural, residential, and traffic). MEIC is adapted to user needs by setting different temporal and spatial resolutions. The latest version of MEIC, V1.2, was released in 2015 (Tong et al., 2020).

At present, there are many universities and scientific research institutions in China carrying out emission inventory studies, but these focus on selected types of pollutants or selected areas (Ni et al., 2014; Qi et al., 2017; Wang et al., 2012; Zhang et al., 2013; Zhao et al., 2012).

Cao et al. (2007) also calculated a detailed high-resolution emission inventory of BC for China in the year 2000 by sectors, provinces and months. The latest activity data include fossil and biomass fuels and socio-economic statistics from government agencies, mostly at county levels. Total BC emissions were about 1.50 million tonnes in 2000, and the burning of coal and biomass contributed about 87% of the total. BC emissions from industrial coal combustion were 497,100 tonnes, accounting for 33.15% of the total. BC emissions from domestic coal combustion were 520,800 tonnes, accounting for 34.73%. BC emissions from industrial straw combustion were 100,100 tonnes, accounting for 6.68%. BC emissions from traffic sources were only 26,800 tonnes, accounting for 1.79% of total emissions. The authors attributed the low contribution of traffic to the fact that the BC emission factor for heavy-duty vehicles was estimated by using the BC emission factor of light-duty vehicles in tunnel tests, which might substantially deviate from the actual situation.

Later, Cao et al. (2011) updated the emission factors and estimated BC emissions of China at 1.4 million tonnes in 2007, which was lower than that found in the previous study. Mobile sources contributed 7.4% of the total, which was higher than previously reported. Cao et al. believed the emission factors used in their new paper had smaller relative errors due to newer emission measurement data, which led to more accurate emission estimation.

BC emissions of China in 2007 were also calculated by Ni et al. (2014), using emission factors obtained from domestic and foreign literature and the statistical data bulletin of China. They estimated emissions at about 963,000 tonnes (biomass burning included), less than that reported by the studies mentioned above. They estimated the contribution of traffic BC emissions at about 8.9%.

Zhang et al. (2013) from Peking University calculated the BC emission inventory (biomass burning included) for 2008 in China (except for Hong Kong, Macao, and Taiwan) based on the national statistical data, the latest measured emission factor data in China, and a vehicle emission factor model developed in line with actual vehicle emissions in China. They estimated that total BC emissions of China in 2008 were 1,604,940 tonnes, of which industrial and residential sources were 695,000 tonnes and 636,000 tonnes, respectively, together accounting for 82.9% of total BC emissions. BC emissions from transport sources were 194,630 tonnes, accounting for 12.1%, which was higher than reported by Cao et al. (2007, 2011). According to Zhang et al. (2013), this was mainly due to the small number of emission factor samples chosen by Cao et al., the deviation in obtaining activity data and BC emission factors being used without considering the changes in emission standards for motor vehicles. By collecting and collating data on BC emissions and establishing a database of emission factors more in accordance with China's actual vehicle emission situation, Zhang et al. (2013) expected improved accuracy in calculating vehicle emission.

Qin et al. (2011) updated China's BC emission inventory for 2009 (biomass excluded). Using a top-down approach to update annual changes in fuel consumption activity levels, they found that China's BC emissions in 2009 amounted to 1.88 million tonnes, of which mobile sources emitted about 241,000 tonnes, accounting for about 12.8%.

Wang et al. (2012) calculated a more detailed BC emission inventory for 2007 for China (biomass burning included) by making a more detailed classification of industry types and using updated BC emission factors. Based on the sub-industry types of fuel consumption from surveys, such as of statistical literature, they estimated that BC emissions in China were 1.957 million tonnes in 2007, which was higher than reported by other studies. Of the total, residential coal contributed about 27.5% and was the largest source of emissions, followed by coke production and civil biomass combustion sources, accounting for 17.3% and 12.7%, respectively. Motor vehicle emissions accounted for about 9.4%.

Fu et al. (2018) calculated China's BC inventory for 2012 (biomass burning excluded) using statistical data. According to their research, total BC emissions were about 1.887 million tonnes, of which BC from household sources were 818,000 tonnes, about 43.3%, making households the largest source of emissions. Transportation sources contributed 178,000 tonnes, about 9.4% of the total.

Zhou et al. (2018) used the multi-regional input-output (MRIO) analysis (top-down BC estimation method) to compile national and provincial BC emissions (unclear on biomass burning contribution). They estimated that total BC emissions in China in 2010 were 1.259 million tonnes, of which emissions from transportation, storage and postal service accounted for about 14%.

Some scholars in China studied BC emissions at the provincial or regional level. Li et al. (2017) from South China University of Technology adopted "bottom-up" and "top-down" methods, using collected anthropogenic activity level data and the latest emission factors measured, to establish the BC and OC emission inventory of anthropogenic sources in Guangdong Province in 2012. Their research showed anthropogenic emissions of BC and OC were 53,500 tonnes and 78,800 tonnes, respectively. BC emissions were mainly from road mobile sources and biomass combustion sources, 30.1% and 29.4%, respectively. The uncertainty ranges of BC and OC emission source inventories established were 66%-154% and 63%-126%, respectively.

Xu et al. (2018) from Dongguan Research Institute of Sun Yat-sen University calculated BC emission characteristics of motor vehicles in Guangdong province, using a bottom-up method and the BC emission factors recommended in the COPERT model. Their study showed that in 2014 the emissions were about 645.69 million tonnes. Since 2012, BC emissions of motor vehicles decreased by 17.4%. Xu et al. believed the decreasing trend might be related to the implementation of upgraded National Emission Standards for heavy-duty diesel vehicles and fuel products following National IV requirement in 2013.

Huang et al. (2017) from Nanjing University used the COPERT model and BC emission factors based on real-time measurement data of Chinese vehicles to establish the motor vehicle air pollutants emission inventory of Jiangsu Province in 2012. Based on the measured emissions data, the BC and OC emissions of the vehicles were 15,700 tonnes and 5,100 tonnes, respectively. Based on the COPERT model calculation, the BC and OC emissions were 4,700 tonnes and 3,000 tonnes, respectively. In Huang et al. (2017), the motor vehicles' BC and OC emissions in Jiangsu were 11,900 tonnes and 2,800 tonnes, respectively. The research showed that the COPERT model likely underestimated the BC emissions of motor vehicles in China, mainly due to low emission factors for heavy vehicles.

In parts 2.2 to 2.4 in this review report, we have tried to piece together the BC emission inventories on global, regional and national scales. We acknowledge that there is not enough information to fully and clearly account for whether differences among inventories arise from rapidly changing emissions, improvements in statistics, different methods or all of these factors.

In the following we will focus on two large sources with substantial uncertainty, namely the mobile and residential sectors.

2.1.5 Mobile Sources

Estimates of BC emissions from mobile sources and their contribution to anthropogenic BC emissions are summarized in Table 2-1. The total BC emissions in China, as reported in the literature, range from 1.05 million tonnes (in 2000) to 1.8 million tonnes (in 2006), the contribution of mobile sources/transportation to total BC emission varies from 1.8% to 32.3%, displaying substantial variation between different reports.

Table 2-1 Mobile source BC emission from different studies

Author	Base Year	Method	Results (million tonnes BC)	Mobile/Transportation's contribution
Cooke et al., 1999	1984	Top-down	Global: 6.40, China: 1.46(about 22.8%)	none
Streets et al., 2001	1995	Bottom-up	China: 1.342	3.23%, including road transport 1.16%, non-road transport 1.74%, and ships 0.33%
Bond et al., 2004	1996	Bottom-up	Global: 7.95, China: 1.4(18.7%)	18.8% (Global), including on-road (diesel fuel) 9.96%, non-road (diesel fuel) 7.28%, and others (gasoline) 1.57%
Streets et al., 2003	2000	Bottom-up	Asia: 2.54, China: 1.05 (41.3% of Asia)	19% (Asia), no further classification
Zhang, et al., 2009	2006	Top-down/local bottom-up (China)	Asia: 2.97, China: 1.8 (60.6% of Asia)	About 14% (Asia, vehicle), no further classification

Author	Base Year	Method	Results (million tonnes BC)	Mobile/Transportation's contribution
Cao, et al., 2007	2000	Bottom-up	China: 1.5	1.8% (China), including diesel fuel 1.7% and gasoline fuel 0.1%
Cao, et al., 2011	2007	Bottom-up	China: 1.4	7.4% (China), no further classification
Zhang, et al., 2013	2008	Bottom-up	China: 1.6	Transportation Sources 12.1%, including road transport (vehicle) 12.06% and aviation 0.04%
Wang, et al., 2012	2007	Top-down (classify sub-industry fuel consumption)	China: 1.957	9.4% (vehicles, China), including vehicle (diesel) 8.0%, vehicle (gasoline) 1.4%, and aviation (kerosene) 0.085%
Qin, et al., 2011	2009	Bottom-up	China: 1.88	Transportation Sources 12.8%, no further classification
Fu, et al., 2018	2012	Bottom-up	China: 1.887	Transportation Sources (9.4%), including trucks (diesel) 3.8%, diesel passenger vehicles (car, coach, bus, etc.) 5.5%, and others (gasoline) 0.1%

At present there is little literature about off-road mobile BC emissions in China. Two reviews focused on BC emission quantities of marine vessels and their effect on sea ice (Bai et al., 2016; Yun, 2020). The former study showed that in 2013, BC emissions from international marine vessels accounted for about 2% of the global total of BC emissions and about 9% of all diesel combustion BC emissions, about one third of which were emitted in areas north of 40°. The latter study reviewed the progress and legislative trends of IMO regarding BC emissions from ships, and introduced definitions, methods for BC surveys, measures for reducing BC emissions and advice on the BC emission reduction issue for ships in the Arctic region, in consideration of China's rights and interests.

2.1.6 Residential Sector

Table 2-2 summarizes residential sector BC and OC emissions found in some available emission inventories. According to an estimate by Peking University, in 2007 a total of 1,957 Gg (Giga gram, 10^9 tonnes) BC was emitted in China, of which 989 Gg was emitted by the residential sector, accounting for 50.5% of total emissions. Industrial emissions of 646 Gg accounted for 33.0% of the total, transport emissions of 188 Gg for 9.6%, power plant emissions of 50.7 Gg for 2.6%, 77.7 Gg from biomass opening burning for 4.0% and other sources for 3.4% (Wang et al., 2012). Results of a Tsinghua University MEIC emission inventory for China also showed that residential emissions were the largest BC emission source in China. Total residential emissions in 2010 accounted for about 50% of the total emissions that year, which is consistent with the research conclusions of several scholars (Streets et al., 2001; Ohara et al., 2007; Lei et al., 2011; Lu et al., 2011).

Table 2-2 BC/OC emissions in China's residential sector

References	Inventory year	BC/Gg	OC/Gg
Bond et al., 2004	1996	546	1,689
Streets et al., 2003	2000	781	2,572
Ohara et al., 2007	2000	938	2,497
Cao et al., 2006	2000	818	2,651
Lei et al, 2011	2006	700	2,610
Zhao et al., 2011	2006	841	2,528
Zhang et al., 2009	2006	1,002	2,606
Cao et al., 2011	2007	651	1,546
Wang et al., 2012	2007	989	/
Zhang et al., 2013	2008	636	/
Kurokawa et al., 2013	2008	715	2,496
Winijkul and Bond, 2016	2010	860	2,280
Lu et al., 2011	2010	936	2,790
Zhao et al., 2013	2010	809	2,228
Li et al., 2017a	2010	848	2,481
Li et al., 2017b	2010	908	2,752
Mean ± SD	/	829±116	2,464±318

An important reason why residential emissions are so large is that many Chinese rural residents use a variety of cooking stoves for heating/cooking, with coal and biomass as fuels. The characteristics of low combustion efficiency, no emission control measures but high emission intensity and low emission height are collectively responsible for the high share (Zhang et al., 2020). Especially during winter heating, the high concentration of air pollutants discharged by residential coal burning over a long time is one of the main causes of autumn/winter haze in northern China (Liu et al., 2016). According to a survey by the Chinese Academy of Environmental Sciences, 97% of rural households in northern China use coal. Such accounts for nearly 80% of domestic energy consumption (Zhi et al., 2015), though the rate has gone down due to intense promotion of clean energy in northern China. Almost all such coal is used for heating in winter. This means that large quantities of coal are consumed in winter in inefficient stoves without any control measures, releasing large amounts of pollutants. According to the latest estimates by Zhu et al. (2019), 51% of national BC emissions in 2014 came from the residential source, 80% of which came from rural areas.

2.2 BC/OC Emission Factors in Key Sectors

Data described in this section are from peer reviewed literature. The following sources were not evaluated because emission factors from Europe (EEA/EMEP Guidebook, https://www.eea.europa.eu/publications/emep-eea-guidebook-2019#additional-files) and North America (U.S. AP-42 Compilation of Air Emissions Factors, https://www.epa.gov/air-emissions-factors-and-quantification/ap-42-compilation-air-emissions-factors) were found not to be representative of Chinese conditions: The IPPC emission factor database (https://www.ipcc-nggip.iges.or.jp/EFDB/main.php) does not currently contain emission factors for BC and OC. Work has started under the IPCC to develop globally applicable methodology for SLCFs including BC and OC, in line with the IPCC 2006 Guidelines and 2019 Refinement for GHG.

2.2.1 Household Coal

Accurate understanding of pollutant emissions is the basis for evaluating the effectiveness of emission reduction (Seinfeld, 2008), while the emission factor is the important basis for the total emission estimate. In estimating BC emission factors, the emission factor of total particulate matter is often multiplied by the proportion of BC in the PM, leading to a high uncertainty (Streets et al., 2001; Zhi, 2008).

In the last two decades, many scholars have started to directly measure the emission strength of BC/OC after dilution of high concentration flue gas by using dilution systems. Great progress has been made in in-depth research on the mechanisms and emission factors of residential coal combustion. It has been found that the relationship between residential coal emission factors and coal maturity is a bell-shaped distribution (coal maturity denotes the degree of coalification; coalification is a process in which volatile matter is converted into coal of increasingly higher rank with anthracite as the final product). In other words, BC emission factors are relatively low when coal maturities are at the high or low end, but relatively high in the middle range of coal maturity (Zhi et al., 2008). Emission factors can be even hundreds of times greater in the middle range than that at the high or low end of maturities (Chen et al., 2006; Chen et al., 2009a). However, coal with low maturity (e.g., lignite) is fragile and does not burn well, while coal with high maturity (anthracite) is expensive and hard to burn. As a result, residents prefer to use moderately mature coal that has higher emission factors and leads to higher soot pollution in China. In addition, the combustion form of coal and the type of stove have significant impacts on emission factors. Making coal into briquette (honeycomb briquette) increases the specific area of a piece of coal several times, which is conducive to oxygen-supported combustion and can reduce the emission factor of BC of bituminous coal by 35 times (Chen et al., 2009b). If briquettes are used with efficient stoves, the BC emissions will be further reduced (Zhi et al., 2009).

Table 2-3 and Table 2-4 give the emission factors of BC and OC for residential coal combustion from different sources. These data were processed with a statistical tool and the distributions are shown in Figure 2-15 and Figure 2-16.

Table 2-3 BC emission factors of coal (g/kg)

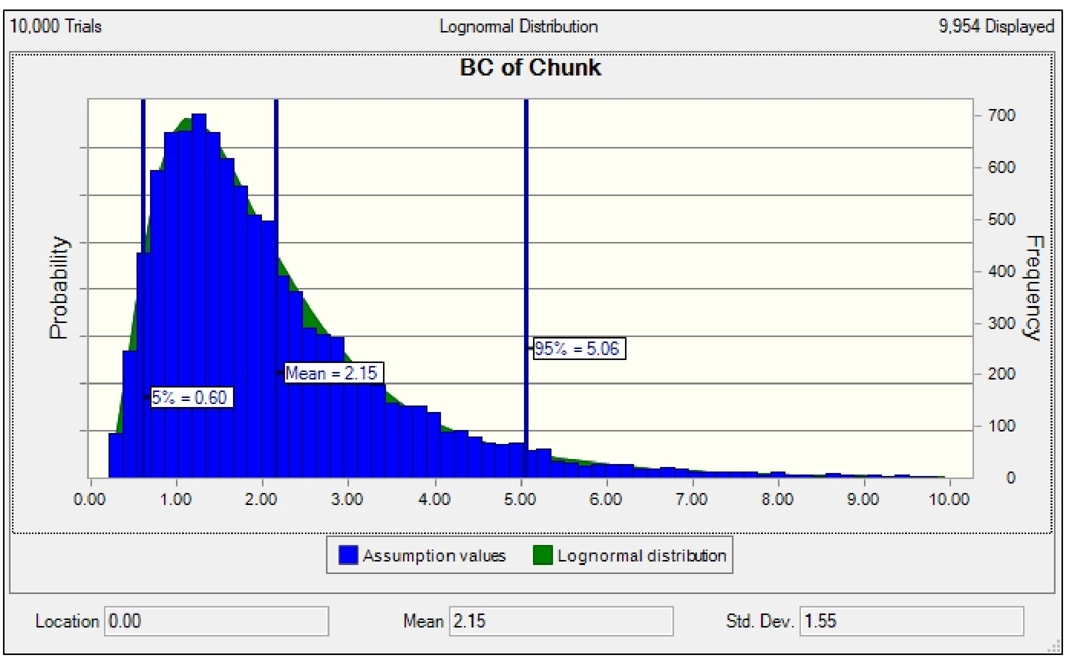
BC	Chunk		Briquette	
	EFs	References	EFs	References
	0.68	Li et al., 2016b	0.65	Li et al., 2016b
	2.75	Zhang et al., 2008	0.09	Zhang et al., 2008
	0.28	Chen et al., 2005	0.32	Liu et al., 2007
	3.51	Liu et al., 2007	0.03	Liu et al., 2007
	3.81	Zhi et al., 2008	0.08	Zhi et al., 2008
	3.05	Chen et al., 2009a	0.09	Chen et al., 2009a
	0.51	Zhi et al., 2009	0.05	Zhi et al., 2009
	1.23	Zhi et al., 2009	0.09	Zhi et al., 2009
	2.89	Zhi et al., 2009	0.16	Zhi et al., 2009
	0.18	Zhi et al., 2009	0.03	Zhi et al., 2009
	3.05	Chen et al., 2009b	0.09	Chen et al., 2009b

BC	Chunk		Briquette	
	EFs	References	EFs	References
	0.83	Shen et al., 2014	0.006	Shen et al., 2014
	0.31	Shen et al., 2014	0.004	Shen et al., 2014
	4.80	Shen et al., 2014	0.18	Shen et al., 2014
	2.04	Chen et al., 2015a	0.67	Chen et al., 2015a
	0.006	Shen et al., 2014		
	2.80	Tian et al., 2017		
	4.92	Sun et al., 2018		
	2.00	Thompson et al., 2019		
	3.7	Streets et al., 2003		
	2.63	Ministry of Ecology and Environment of People's Republic of China, 2014		
	0.23	Shen et al., 2010		
	3.32	Chen et al., 2006		
Average ± SD	2.15±1.58		0.17±0.21	

Table 2-4 OC emission factors of coal (g/kg)

OC	Chunk		Briquette	
	EFs	References	EFs	References
	2.49	Li et al., 2016b	4.02	Li et al., 2016b
	5.93	Zhi et al., 2008	4.16	Zhi et al., 2008
	5.39	Liu et al., 2007	1.24	Liu et al., 2007
	8.29	Chen et al., 2006	4.15	Liu et al., 2007
	2.98	Zhang et al., 2008	2.27	Zhang et al., 2008
	5.5	Chen et al., 2009a	3.74	Chen et al., 2009a
	6.22	Zhi et al., 2009	5.48	Zhi et al., 2009
	9.76	Zhi et al., 2009	4.94	Zhi et al., 2009
	5.76	Zhi et al., 2009	6.17	Zhi et al., 2009
	4.31	Zhi et al., 2009	2.50	Zhi et al., 2009

OC	Chunk		Briquette	
	EFs	References	EFs	References
	1.00	Shen et al., 2014	0.007	Shen et al., 2014
	0.66	Shen et al., 2014	0.02	Shen et al., 2014
	5.90	Shen et al., 2014	4.80	Shen et al., 2014
	0.80	Chen et al., 2015a	1.99	Chen et al., 2015a
	0.65	Chen et al., 2015b	1.15	Chen et al., 2015b
	0.10	Shen et al., 2014		
	7.82	Chen et al., 2005		
	1.90	Tian et al., 2017		
	3.65	Sun et al., 2018		
	9.70	Thompson et al., 2019		
	3.12	Ministry of Environmental protection of People's Republic of China, 2014		
	3.00	Streets et al., 2003		
Average ± SD	4.32±2.95		3.11±1.95	



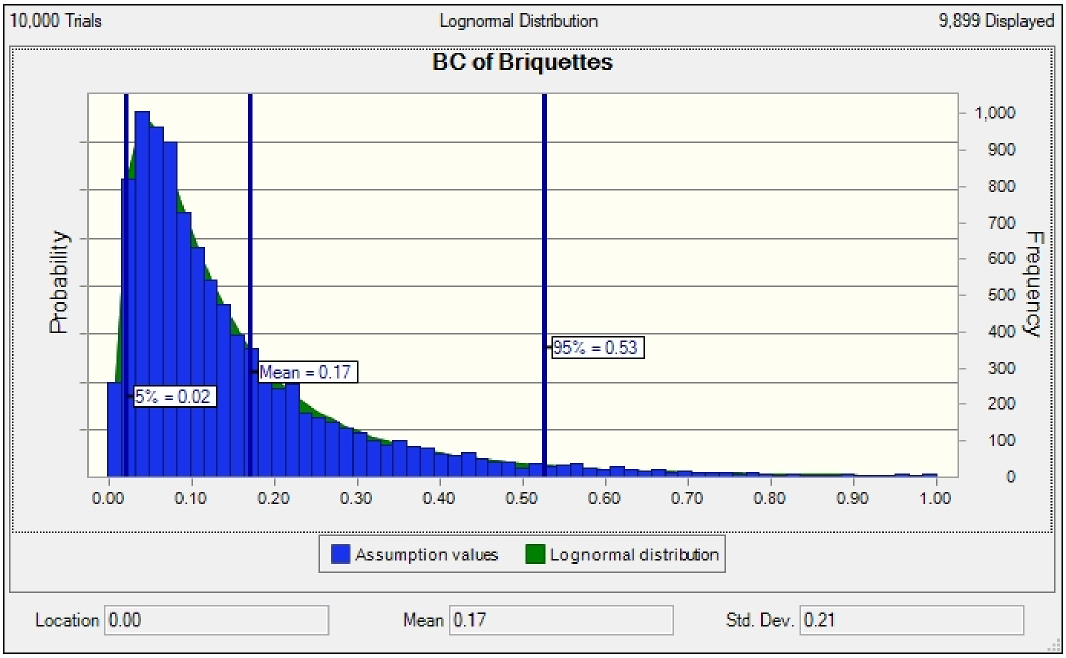
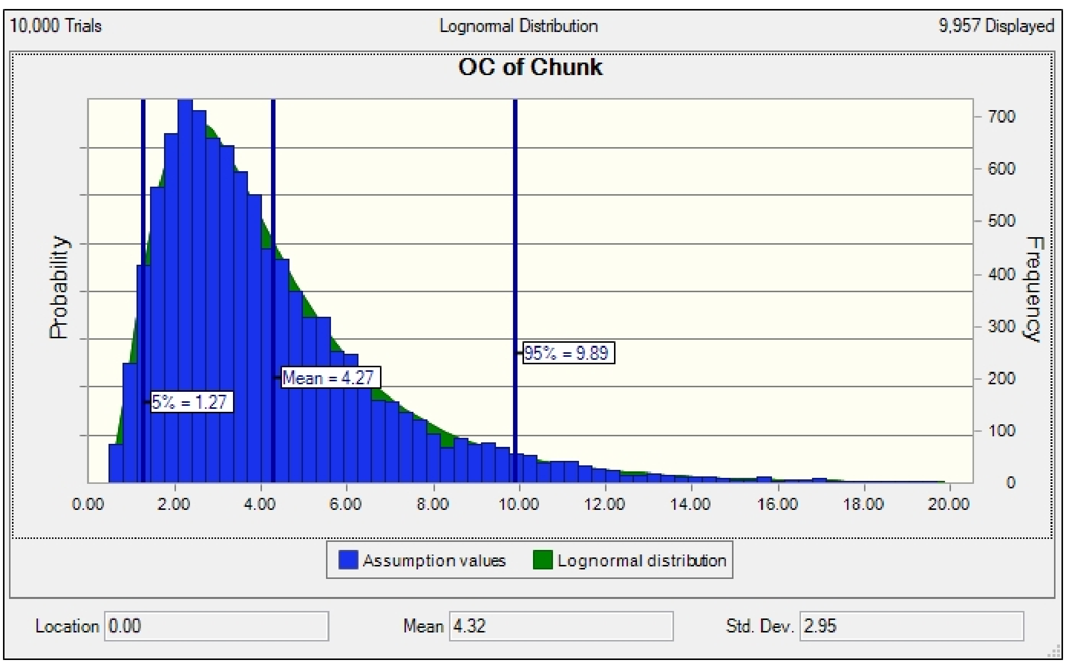


Figure 2-15 BC emission factor uncertainty distribution of coal and briquette



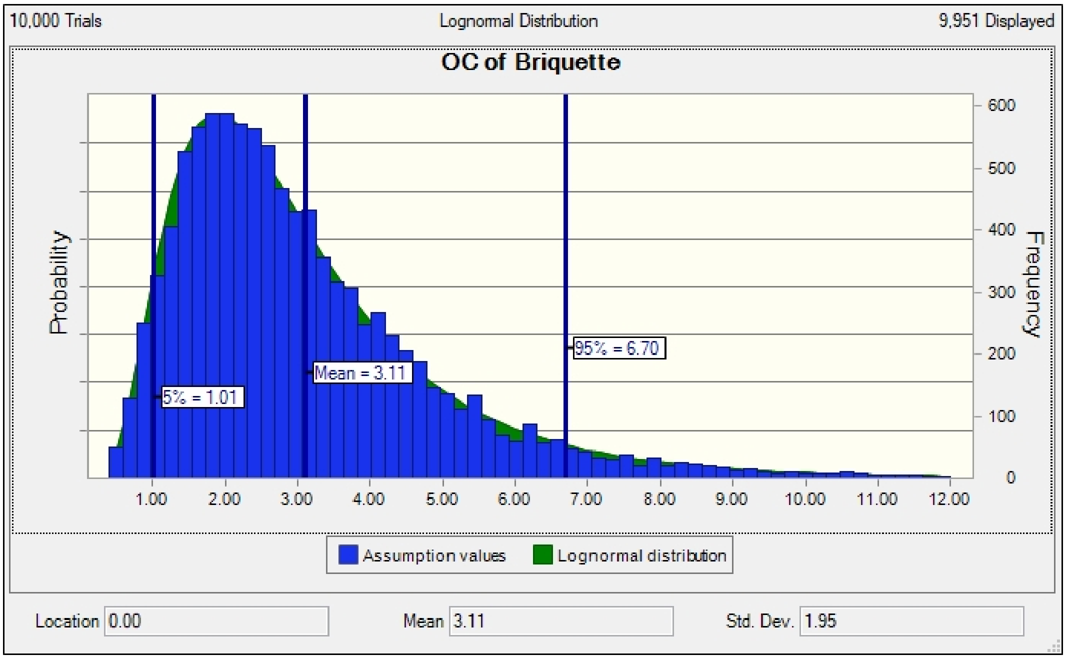


Figure 2-16 OC emission factor uncertainty distribution of coal and briquette

2.2.2 Mobile Sources

At present, there is no official test method for measuring BC emissions for mobile sources in China. The main measurement method for mobile source BC is to measure PM and then analyze the EC/soot part from the PM. This method can sometimes provide PM_{10} and $PM_{2.5}$ measurements. Measurements of PM are complex, as some of the emitted PM is filterable and some is coagulated. Table 2-5 summarizes the official U.S. PM sampling method (EPA, 2006). For sampling diesel PM, the EPA mainly uses 40CFR Part1065. The measurement of BC mainly uses the method of thermo-optical analysis, but it is not the official test standard.

Table 2-5 EPA of Particle Sampling and Measurement Method

Method	PM Type	Filtration Temperature/℉	Purpose	CFR Reference
EPA Method 5	Filterable	248±25	General	40CFR 60 Appendix A-3
EPA Method 5A	Filterable	108±18	Asphalt Roofing	40CFR 60 Appendix A-3

Method	PM Type	Filtration Temperature/℉	Purpose	CFR Reference
EPA Method 5B	Filterable	320±25	Utility Plants	40CFR 60 Appendix A-3
EPA Method 5D	Filterable	248±25	Positive Pressure Baghouses	40CFR 60 Appendix A-3
EPA Method 5E	Filterable and Total Organic Material	248±25	Wool Fiberglass	40CFR 60 Appendix A-3
EPA Method 5F	Filterable	320±25	Non sulfate Filterable PM	40CFR 60 Appendix A-3
EPA Method 5G	Filterable and Condensable	＜90	Wood Heaters-Dilution	40CFR 60 Appendix A-3
EPA Method 5H	Filterable and Condensable	＜248 and >68	Wood Heaters	40CFR 60 Appendix A-3
EPA Method 5I	Filterable	248±25	Low level general	40CFR 60 Appendix A-3
EPA Method 17	Filterable	stack temperature	General	40CFR 60 Appendix A-6
EPA Method 201	Filterable 10μm	stack temperature	General-Particle Sizing	40CFR 51 Appendix M
EPA Method 201A	Filterable 10 μm/2.5 μm	stack temperature	General-Particle Sizing	40CFR 51 Appendix M
EPA Method 202	Condensable	85	General-Condensable PM	40CFR 51 Appendix M
EPA Conditional Test Method-039	Total 10 μm/2.5 μm (Filterable and Condensable)	85	General-Dilution based PM	
Example State, VCS, and International Methods				
CARB 5	Filterable	248±25		
CARB 501	Filterable, multiple aerodynamic sizes	stack temperature	General-Particle Size	
ASTM D6831-05a	Filterable	stack temperature	Continuous PM	
ISO 9096 and EN 13284	Filterable			
VDI 2066 Part.10 method and in the Norm EN 13284-1	Filterable 10 μm/2.5 μm			

As for measurements of BC of mobile sources in real-time driving conditions, the emissions vary greatly with respect to differences in fuel type, vehicle type and running conditions. The sampling temperature has great influence on the sampling of PM and the measurement of BC. The general sampling method always requires pre-dilution with ambient air.

BC emission factors of mobile sources derived from different reports can be separated into two categories. One is BC emissions directly per fuel consumed (g/kg fuel) or per mileage traveled (g/km). The other is to derive the BC emission factor from the emission factors of particles ($EFPM_{10}$ or $EFPM_{2.5}$) by assigning a fraction of particles to BC or OC. Such BC factors are affected by different types/fuels used and standards as well as conditions, etc. Different BC factors reported in the literature are given in Table 2-6 and Table 2-7. BC and/or OC fractions in particles are given in Table 2-8.

Table 2-6 BC/OC emission factors from the literature

	Low-income countries/(g/kg)		Middle-income countries/(g/kg)		High-income countries/(g/kg)		BC/OC Origin	Reference
	BC	OC	BC	OC	BC	OC		
Diesel	/	/	/	/	2.0	/	Literature review of EC measurement with thermal-optic method	Cooke and Wilson, 1996
Gasoline	/	/	/	/	0.1	/		
Diesel	10	5.0	10	5.0	2.0	1.0	Literature review of EC measurement with thermal-optic method	Cooke et al., 1999
Gasoline	0.15	0.73	0.15	0.73	0.03	0.07		
Gasoline	/	/	/	/	0.049	/	Literature review of PM factors multiplied BC fractions in PM	Cao et al., 2007
Diesel	/	/	/	/	1.1	/		
Gasoline, Light-duty vehicle	/	/	0.26	/	0.13	/	Literature review of PM factors and BC fractions in PM	Klimont et al., 2009
Diesel, Light-duty vehicle	/	/	3.1	/	0.55	/		
Diesel, heavy-duty truck	/	/	1.45	/	0.042	/		

	Low-income countries/(g/kg)		Middle-income countries/(g/kg)		High-income countries/(g/kg)		BC/OC Origin	Reference
	BC	OC	BC	OC	BC	OC		
Diesel, Light-duty truck	2.2(China 0),0.85 (China Ⅰ)	/	0.76(China Ⅱ),0.46 (China Ⅲ)	/	0.27 (China Ⅳ)	/	Literature review of BC factors by optical measurement with a micro Aethalometer	Song, et al., 2012
Diesel, Light-duty cars	1.5(China 0),0.6 (China Ⅰ)	/	0.4(China Ⅱ),0.2 (China Ⅲ)	/	0.15 (China Ⅳ)	/		
Diesel, Heavy-duty truck	2.3(China 0), 1.24 (China Ⅰ)	/	0.55(China Ⅱ),0.33 (China Ⅲ)	/	0.2 (China Ⅳ)	/		
Gasoline, Light-duty vehicle	0.09(China 0), 0.054 (China Ⅰ)	/	0.029 (China Ⅱ), 0.018 (China Ⅲ)	/	0.009 (China Ⅳ)	/		
faGasoline, large/heavy-duty vehicle	0.14(China 0), 0.08 (China Ⅰ)	/	0.04(China Ⅱ),0.025 (China Ⅲ)	/	0.015 (China Ⅳ)	/		
Gasoline	/	/	/	/	0.07	/	Calculated by EF_{PM} (CPOERT model) multifield by F_{BC} (Bond, et al., 2004)	Qin, et al., 2011
Diesel	/	/	/	/	0.25	/		
Gasoline	/	/	/	/	0.056	/	Calculated by EF_{PM} (Huang, et al., 2014) multifield by F_{BC} (Bond, et al., 2004)	Zhou, et al., 2018
Diesel	/	/	/	/	1.22	/		

Table 2-7 BC emission factors (g/km)

	2006	2013	Reference
Gasoline	0.011	0.0025	Krecl, et al., 2017
Diesel	0.0948	0.0234	
Gasoline	0.0005	0.0003	TRANSPHORM database; Krecl, et al., 2017
Diesel	0.0929	0.0238	

Table 2-8 BC/OC fractions by PM (g/kg or g/km)

	Emission Factor	$F_{2.5}/F_{1.0}$	F_{BC}	F_{OC}	Reference
Gasoline	$EF_{PM_{10}}$:0.124-0.392(g/kg)	$F_{2.5}$:0.74-0.95	0.32	/	Streets, et al., 2003
Diesel	$EF_{PM_{10}}$:0.34-4.33(g/kg)	$F_{2.5}$:0.95	0.52	/	
Gasoline, large passenger bus	$EF_{PM_{2.5}}$:0.08(g/km)	/	0.27	0.58	Li, et al., 2017
Diesel, large passenger coach	$EF_{PM_{2.5}}$:0.71(g/km)	/	0.51	0.32	
Gasoline, light-duty vehicles	$EF_{PM_{2.5}}$:0.02(g/km)	/	0.27	0.58	
Diesel, light-duty vehicles	$EF_{PM_{2.5}}$:0.12(g/km)	/	0.51	0.32	
Gasoline, heavy-duty truck	$EF_{PM_{2.5}}$:0.07(g/km)	/	0.27	0.58	
Diesel, heavy-duty truck	$EF_{PM_{2.5}}$:0.78(g/km)	/	0.51	0.32	
Gasoline, light-duty truck	$EF_{PM_{2.5}}$:0.064(g/km)	/	0.27	0.58	
Diesel, light-duty truck	$EF_{PM_{2.5}}$:0.253(g/km)	/	0.51	0.32	
Bus	$EF_{PM_{2.5}}$:0.775(g/km)	/	0.51	0.32	
Taxi	$EF_{PM_{2.5}}$:0.018(g/km)	/	0.27	0.58	
Motorcycle	$EF_{PM_{2.5}}$:0.03(g/km)	/	0.12	0.53	
Fishing vessels	$EF_{PM_{2.5}}$:2.16(g/kg)	/	0.44	0.33	
Agricultural machinery	$EF_{PM_{2.5}}$:4.0(g/kg fuel)	/	0.31	0.44	
Agricultural farm vehicle	$EF_{PM_{2.5}}$:2.78(g/kg)	/	0.31	0.44	
Construction machinery	$EF_{PM_{2.5}}$:6(g/kg)	/	0.31	0.44	
Diesel, on-road/ standards in place	EF_{PM}:1.5	$F_{1.0}$:0.86	0.66	0.21	Bond et al., 2004
Diesel, on-road/ standards beginning	EF_{PM}:3.5	$F_{1.0}$:0.86	0.66	0.21	
Diesel super emitters	EF_{PM}:12	$F_{1.0}$:0.86	0.66	0.21	
Diesel farm vehicles	EF_{PM}:4.0	$F_{1.0}$:0.86	0.66	0.21	
Diesel nonfarm off-road vehicles	EF_{PM}:5.5	$F_{1.0}$:0.86	0.66	0.21	
Diesel and heavy oil, ships	EF_{PM}:1.8	$F_{1.0}$:0.86	0.66	0.21	
Gasoline, all vehicles/standards in place	EF_{PM}:0.15	$F_{1.0}$:0.85	0.34	0.36	
Gasoline, all vehicles/standards beginning	EF_{PM}:0.5	$F_{1.0}$:0.85	0.34	0.36	
Gasoline super emitters	EF_{PM}:2.0	$F_{1.0}$:0.85	0.34	0.36	
Gasoline, two-stroke/standard	EF_{PM}:15	$F_{1.0}$:0.95	0.05	0.79	
Gasoline, two-stroke/high-emission	EF_{PM}:30	$F_{1.0}$:0.95	0.05	0.79	

	Emission Factor	$F_{2.5}/F_{1.0}$	F_{BC}	F_{OC}	Reference
practice					
Diesel, Heavy-duty trucks - EURO I	/	/	0.65	/	COPERT model
Diesel, Heavy-duty trucks - EURO II	/	/	0.65	/	
Diesel, Heavy-duty trucks - EURO III and above	/	/	0.70	/	
Diesel, Buses - EURO I	/	/	0.65	/	
Diesel, Buses - EURO II	/	/	0.65	/	
Diesel, Buses - EURO III and above	/	/	0.70	/	
Diesel, Electronically controlled injection vehicle			0.56	/	Zheng X, 2016
Diesel, Mechanical injection vehicle			0.43	/	

In addition, Yan et al. (2014) investigated the traffic flow impacts on BC emissions, including hourly profiles for total traffic volume, fleet composition by vehicle category and average speed on a typical freeway (the North Fourth Ring Road in Beijing) during 2009. By applying the Emission Factor Model for the Beijing Vehicle Fleet (EMBEV) in combination with previous studies on vehicle emissions of BC, the BC emission factors and emission intensity from on-road vehicles were derived. In combination with simultaneously measured meteorological data in Beijing, dispersion of road traffic BC emissions was simulated with the AERMOD model in a roadside environment and was further validated with concurrently observed BC concentration data. Results showed that the hourly average BC emission factor was very strongly correlated with the proportion of the traffic volume of heavy-duty diesel vehicles (for example, diesel-powered passenger buses and freight trucks). Due to the traffic restrictions on truck use in the urban area of Beijing during daytime (6 a.m. to 11 p.m.), the average BC emission factor was (9.3±1.2) mg/(km·veh) during daytime but increased to (29.5±11.1) mg/km during nighttime. Two peaks of BC emission intensity were observed synchronized with traffic volume peaks (106.1±13.0) g/(km·h) during the morning rush period (7 a.m. to 9 a.m.) and (102.6±6.2) g/(km·h) during the evening rush period (5 a.m. to 7 a.m.). During the day, light-duty passenger cars were the largest contributor (1.07±1.57 μg/m^3, about 40% contribution) among all vehicle categories, followed by the public bus fleet (0.58±0.85 μg/m^3). During nighttime, trucks became the dominant contributor (2.44±2.31 μg/m^3) to the BC concentration on the road, which contributed to about 70% of the concentration.

2.3 Activity Data for Key Sectors and Other Sectors

2.3.1 Household Coal

The preferred method of estimating the coal use in the residential sector is to use official statistics. Many emission inventories make use of such data as inputs. In China, however, researchers found risks in directly utilizing official statistics on rural coal (Zhi et al., 2017; Zhang et al., 2020). With economic development and improvement of living conditions in China, the demand for heating in rural areas has greatly increased (Liu et al., 2013; Ma et al., 2019; Zhu et al., 2019). Especially in 2009, under the State Council's "Project to Renovate the Dilapidated Houses in Rural Areas" many farmers took the opportunity to renovate their houses and upgrade their heating approach, from a stove-direct-heating mode to a mini-boiler-water-circulation-heating mode (Ministry of Housing and Urban-Rural Development of the People Republic of China, 2009). This dramatically improves heating of the house and indoor air quality (Shan et al., 2015). At the same time, rural winter heating coal consumption greatly increased, thus rural air quality deteriorated (Liao et al., 2017). Compared with coal use in industry and power plants, the use and circulation of rural coal are almost in a vacuum in terms of official statistics and supervision (Zhang et al., 2020), Therefore, the management of rural coal use is still in a largely chaotic and non-standard condition (Peng et al., 2019; Zhi et al., 2017; Cheng et al., 2017).

Under the circumstances, the sufficient quality activity levels of residential coal consumption can only be obtained by means of investigation. Fortunately, quite a few scholars have investigated the true situation of coal consumption for rural heating in winter in recent years (Chen et al., 2018; Gao et al., 2016; Li et al., 2016a; Li et al., 2015; Liu et al., 2013; Peng et al., 2019; Ru et al., 2015; Wang and Jiang, 2017; Wang et al., 2017; Xiao et al., 2017; Zhang et al., 2015; Zhang et al., 2014; Zhao et al., 2015; Zhi et al., 2017). Zhi et al. (2017) found that coal consumption in Baoding was nearly five times that shown by the data in the government's statistical yearbook. Even in the same region the results of different studies vary considerably. For example, Liao et al. (2017) found coal consumption in Beijing to be four times greater than reported by Li et al. (2015). All of the above evidence indicates that the activity level of residential coal consumption is still very uncertain. It is of particular importance that northern China is endeavoring to switch its rural household coal use to clean energy (gas or electricity), the rate of switching to clean energy must be considered when calculating the BC/OC emissions from rural household coal.

Meanwhile, data from investigations in a village or region usually indicate total consumption of a whole year or winter instead of daily or monthly consumption. Moreover, once updating is needed, a new investigation has to be reorganized, which entails substantial manpower and material resources. Fortunately, there have been recent advances in building algorithms for allocation of year/winter totals to daily shares. Based on meteorological elements observed (temperature, relative humidity, wind speed, and sunshine hours), daily fractions of coal use can be derived (Zhang et al., 2020). With the algorithm, rural coal heating consumption data (Figure 2-17) and daily specific fractions of coal consumption (Figure 2-18) in China's "2+26" cities for the 2018-2019 heating season were calculated (Zhang et al., 2020). These developments have helped determine and allocate BC/OC emissions from rural household coal.

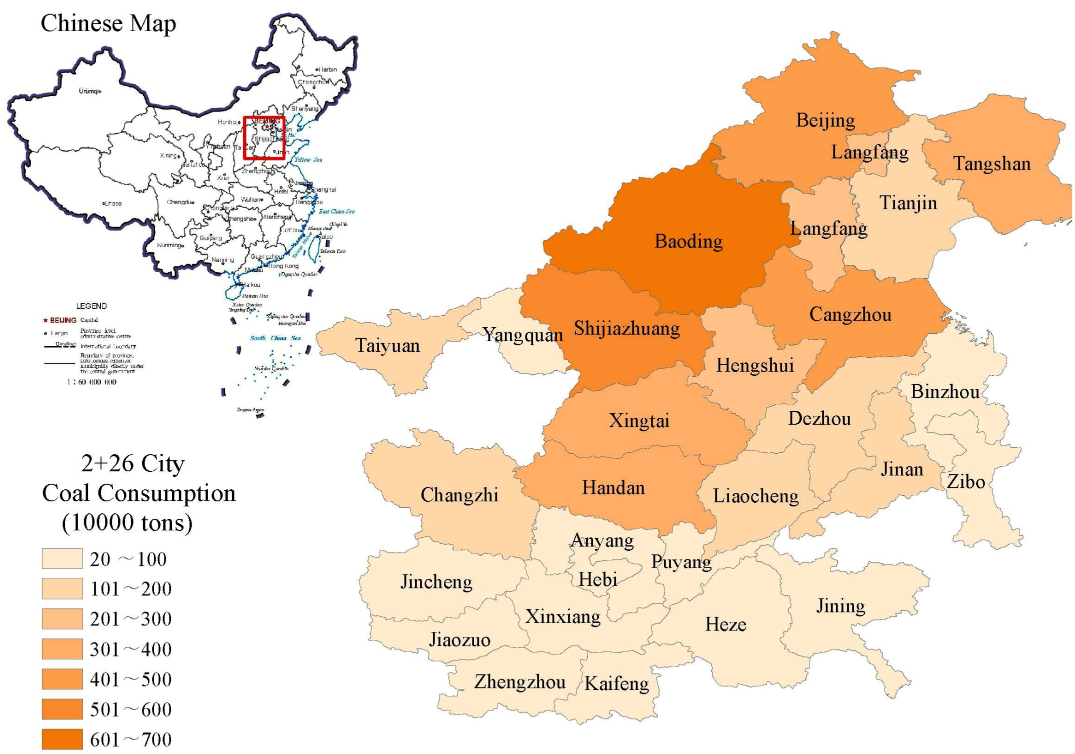


Figure 2-17 Algorithm-deduced rural coal heating consumption of China's "2+26" cities for the 2018-2019 heating season

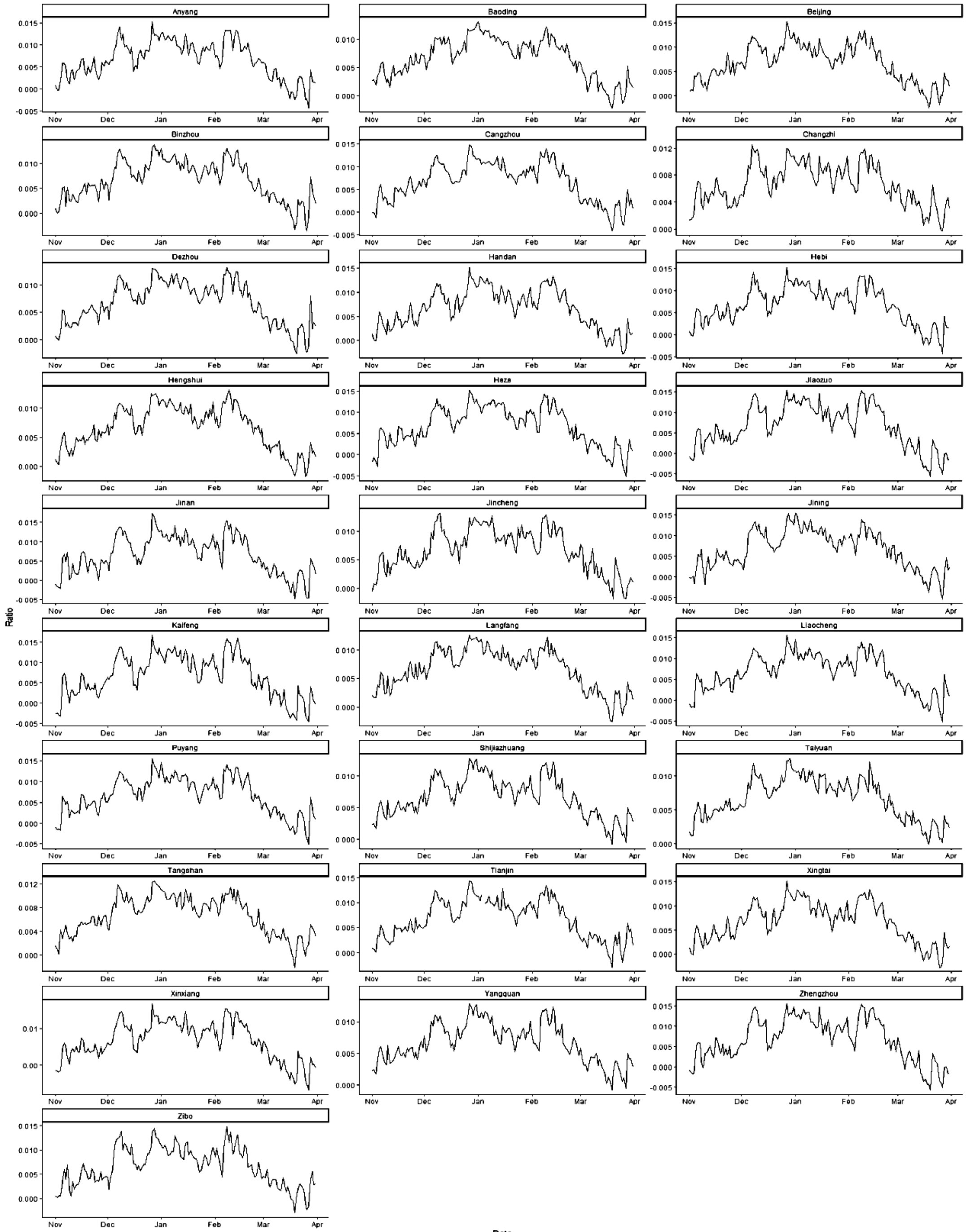


Figure 2-18 Temporal profile of the daily share of coal consumption for each “2+26” city during winter

2.3.2 Mobile Sources

At present, the authoritative data are those found in the "Road Motor Vehicle Air Pollutant Emission Inventory Development Technical Guidelines" issued by the Ministry of Ecology and Environment in 2015 (MEE of China, 2015). The Guidelines divide motor vehicles into 11 categories according to their application, weight and size, among which passenger vehicles are classified into four types: large, medium, small and micro, while trucks are divided into five categories: heavy-duty, medium-duty, light-duty, micro and low-speed. Motorcycles are divided into two types: ordinary and light. Buses and taxis are listed separately from passenger cars. The annual average driving distances (VKT, km) for all types of road vehicles are given in Table 2-9.

Table 2-9 VKT recommended by the guide

Vehicle Type	VKT/km
Small, micro passenger cars	18,000
Taxi passenger cars	120,000
Medium passenger cars	31,300
large passenger cars/Coach	58,000
Bus	60,000
light-duty, micro size truck	30,000
Medium-duty truck	35,000
Heavy-duty truck	75,000
Motorcycles	6,000
Low speed truck	30,000
Three-wheel truck	23,000

Qin Y et al. (2011) estimated the distance traveled by different vehicles by year (Table 2-10). We note that there are no published activity data later than 2009. Since 2009, Chinese authors have used the data recommended by the Guidelines mentioned above.

Table 2-10 Vehicle distance travelled (10,000 km) of each vehicle category, 2000-2009

Type	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Buses and Coaches	4.23	3.95	3.86	3.74	4.06	4.04	4.08	4.25	4.25	4.22
Passenger cars	3.75	3.80	3.85	3.90	3.95	4.00	4.00	4.00	4.00	4.00
Heavy-duty vans	5.50	5.38	6.80	6.87	7.07	7.21	7.75	8.34	8.92	9.54
Other-duty vans	3.00	2.92	3.68	3.70	3.80	3.86	4.13	4.43	4.74	5.07
Motorcycles	1.30	1.31	1.32	1.33	1.34	1.35	1.35	1.35	1.35	1.35

For the activity level of various types of non-road mobile machinery in China, recommended values are also given in the "Technical Guidelines for the Development of Inventories of Atmospheric Pollutants from non-road Mobile Sources" issued by the Ministry of Ecology and Environment (MEE of China, 2015). The guide classifies non-road sources into engineering machinery, agricultural machinery, small general machinery, diesel generators, ships, diesel locomotives and so on. Construction machinery includes excavators, bulldozers, loaders, forklifts, road rollers, pavers, graders and other machinery. The agricultural machinery includes mainly tractors, agricultural transport vehicles (Farm Machinery Licensing), combine harvesters, drainage and irrigation machinery and other machinery. Small general-purpose machines refer to small spark-ignition engines for non-road mobile machines, such as lawn mowers, powered primarily by gasoline. A diesel generator is a mobile electric generator that runs on diesel fuel at a constant speed. Vessels include inland and coastal vessels, capable of carrying out transportation and operations in navigable waters or while moored. A diesel locomotive is a railway locomotive powered by an internal combustion engine, primarily on diesel fuel and driven by a transmission to move its wheels. For construction machinery, agricultural machinery, small general machinery and diesel generators, the recommended average number of hours of use per year is shown in Table 2-11.

Table 2-11 Recommend working hours of non-road machinery per year

Types		Working hours per year/(h/a)
Construction machinery	Excavators	770
	Bulldozers	770
	Loaders	770
	Forklifts	770
	Road rollers	770
	Pavers	770
	Graders	770
	Others	770
Agricultural machinery	Large and medium tractors	500
	Small tractors	500
	Combine Harvesters	150
	Drainage and irrigation machinery	380
	Others	380
Small general-purpose machinery	Hand-held	50
	Non-hand-held	125
Diesel generators		770

It is presumed that the estimated railway locomotive activity level data are derived from diesel locomotive fuel consumption. The fuel consumption of diesel locomotives can be calculated by Railway Department Statistics or according to passenger and cargo turnover and daily freight production. The diesel locomotive fuel consumption coefficient of freight railways, passenger and freight turnover, daily freight production and freight railway diesel locomotive fuel consumption coefficient can be obtained from relevant China Statistical Yearbooks or China Traffic Yearbooks. The ship fuel consumption is used to represent the activity level of inland river and coastal ships. The fuel consumption of inland water and coastal vessels is obtained by using data in the “Ministry of Transport of the People’s Republic of China Bulletin” or data on passenger and cargo turnover from the “Highway and Waterway Transport Industry Development Statistics Bulletin” or the “China Traffic Yearbook.”

This review gives us a great understanding of pollutant emission factors and emission quantities in China and abroad, especially for residential sector and mobile sources. It also makes great preparations for us to generate our own emission inventories later, such as providing literature references (Annex Ⅰ) for our emission factors, which is one of the essential sources of emission factors.

If you are interested in emission inventories, please pay attention to the other relevant content of the website for more information on emission inventories.

3 Impact Assessment of BC/OC on Air Quality and Climate Change

This part is a literature survey and summary of the impact of BC/OC and other pollutants on climate change and air quality from the scientific literature in recent years. We note that it is not a systematic literature review BC/OC play an important role in air quality and climate change in China and high concentrations of carbon aerosols may exacerbate the severity of smog and increase the complexity of air pollution.

3.1 BC/OC Research Status and Knowledge in Climate Impact

Aerosols play a key role in shaping regional climate and air quality. Depending on type, they scatter and absorb solar radiation, causing a direct (i.e. aerosol-radiation) radiative forcing (RF_{ari}) and affect the properties of clouds (indirect, aerosol-cloud, radiative forcing, RF_{aci}). Globally, the combined effect of aerosols is an effective radiative forcing (ERF[①]) estimated to be −1.3 W/m^2 over the industrial era (1750-2014) (Forster et al., 2021), but with considerable uncertainty [−2.09 to −0.6 W/m^2]. Considering the full 1750-2019, the total aerosol ERF is assessed to be smaller in magnitude −1.1 [−1.7 to −0.4] W/m^2 due to recent emission changes, but aerosols have nevertheless masked a significant fraction of the greenhouse gas induced warming to date (Samset et al., 2018a).

BC and OC are two of the key aerosol species that receive attention in the science and policy arena due to their dual impacts on climate and air quality. Originating from incomplete

① The change in the net energy flux at the top of the atmosphere of the Earth system due to an imposed perturbation, such as changes in greenhouse gas or aerosol concentrations, after allowing for rapid adjustments in e.g. cloud and vertical temperature (Forster et al., 2021).

combustion of fossil fuels, biofuels and biomass, OC is the fraction of carbonaceous aerosols which contain compounds of carbon and BC is soot made almost purely of carbon. Here we outline some key points and remaining knowledge gaps related to their atmospheric distribution and climate effects. Note, however, that this is not a systematic review. For a more comprehensive overview, we refer to existing assessments (Bond et al., 2013; Lee et al., 2013).

While most aerosols have a cooling impact on the climate, BC acts to warm the atmosphere by absorbing solar radiation, causing a positive RF_{ari}. There is a significant spread in estimates of the RF_{ari} of BC aerosols: The IPCC Fifth Assessment report (AR5) reported an RF_{ari} (1850-2011) of 0.4 (0.05 to 0.8) W/m^2, with a slightly lower value of 0.31 W/m^2 estimated by Lund et al. (2018a). The impact of BC aerosols on climate is a complex interplay of many mechanisms, including changes in cloud properites and cover, darkening of snow covered surfaces and rapid adjusments in the atmosphere (Bond et al., 2013). Recent work suggest that the rapid adjusment to BC forcing, due to changes in clouds, lapse rate and atmospheric water vapor, act to offset a substatinal portion of the positive RF_{ari} of these aerosols, resulting in a weaker net ERF and relatively weak global-mean surface temperature response (Stjern et al., 2017; Takemura et al., 2019). The best estimate of BC ERF is assessed to be 0.063 (−0.28 to 0.42) W/m^2 in AR6 (Naik et al., 2021), a smaller positive values than in AR5. BC aerosols contribute further to radiative forcing through depostion on snow and ice, darkening the surface and increasing absorption and snow melt. Considerable uncertainties exist in estimates of BC concentrations of in the snowpack (Dou and Xiao, 2016) and in the processes and snow characteristics that determines the subsequent albedo change, although scientific progress has been in the years (Forster et al., 2021). In AR6, the best estimate of the global instantaneous RF (IRF) due to BC on snow deposition is assessed to be 0.04 [0.00 to 0.09] W/m^2 (Forster et al., 2021), similar to AR5. To account for a higher efficacy (i.e. stronger temperature response per unit RF than for a correponsing RF by CO_2) of the BC-on-snow effect, the IRF estimate is doubled, giving an overall assessed ERF of +0.08 [0.00 to 0.18] W/m^2.

While the impact of BC on global mean surface temperature has been assessed to be lower than previously thought, BC still plays in an important role for regional climate, including through modulating the hydrological cycle. The sensitivity of the regional climate to reductions in aerosol emissions has been found to be especially high in the Asian region (Samset et al., 2018b).

The wide spread and high uncertainty in BC forcing estimates stem from high variability in atmospheric distribution and sign of rapid adjustments in global models, which in turn is related to differences in treatment of processes affecting BC in the atmosphere, as well as uncertainties in emission inventories. Upon emission, the aerosols get transported away from the source regions while undergoing chemical aging and scavenging processes along the way. Insufficient process understanding and model representation, and limited observations for validation, leads to uncertianties in the long-range aerosol transport and atmospheric lifetime. For instance, BC emitted in Asia has been shown to reach the Arctic where the aerosols can be deposited on snow and contribute to further warming (Ikeda et al., 2017; Qi and Wang, 2019). However, the exact role of Asian emissions for Arctic BC levels is, not sufficiently constrained due to uncertainties in and different model treatment of the atmospheric lifetime, aging and wet and dry removal of the aerosols (Bourgeois and Bey, 2011; Browse et al., 2012; Liu et al., 2011; Lund et al., 2012) .

The vertical distribution of BC is also a critical factor for quantifying the subsequent climate impact, as the BC efficacy (i.e., radiative forcing per kg BC) increases strongly with altitude. Global models have had a tendency to have overestimate high-altitude BC concentration, in particular in remote oceanic regions (Schwarz et al., 2010; Katich et al., 2018), which in turn may mean that they overestimate the BC impact (Ban-Weiss et al., 2011; Samset et al., 2015). Several studies have pointed to a too long lifetime of many current models as the main cause of this discrepancy and suggested that a global mean BC of less than approximately 5 days is required for reasonable agreement between modeled and measured vertical BC profiles (Samset et al., 2014; Lund et al., 2018b; Hodnebrog et al., 2014; Wang et al., 2014). Over recent years, considerable work has been undertaken to understand the processes underlying the model-measurement discrepancies and improve the model performance (Fan et al., 2012; Kipling et al., 2013; Zhang et al., 2012; He et al., 2016; Lund et al., 2017; Xu et al., 2019; Liu and Matsui, 2021). Nevertheless, as summarized by the IPCC AR6, *"the lack of global scale observations of carbonaceous aerosol, its complex atmospheric chemistry, and the large spread in its simulated global budget and burdens means that there is only low confidence in the quantification of the present-day atmospheric distribution of individual components of carbonaceous aerosols.*"

Further uncertainty in the climate impact of BC arises from uncertianties in the optical properties, how efficiently the absorb radiation and how that absorption is affected when BC becomes mixed with other aerosol in the atmosphere. Studies also indicate differences between urban and rural environment. A recent review by Samset et al. (2018c) provides the current status of knowledge about aerosol absorption, the main uncertainties and needs for future improvements.

OC is co-emitted with BC but is predominantly scattering (i.e., a negative RF_{ari}). There is also a non-negligible absorbing component, at least for certain sources. These absorbing organic aerosols, often referred to as BrC, is a source of significant uncertainty in estimates of total aerosol absorption. OC also contributes to aerosol-cloud interactions and hence to a negative RF_{aci}. The most recent best estimate ERF of OC is −0.20 [−0.03 to −0.41] W/m^2 (Naik et al., 2021). Global modeling and understanding of OC is confounded by many of the same uncertainties as BC.

3.2 BC/OC Pollution Status in Concerned Regions in China

BC/OC play an important role in air quality and climate change in China (Bahner et al, 2014; Ji et al, 2019; UNEP/WMO, 2011), BC and OC are always emitted together, but the proportion varies depending on the source. Zhang et al. (2014) has highlighted the increasing role of OC in air pollution as well as in climate change in China generally. Their results show that anthropogenic aerosols present in $PM_{2.5}$ (including BC and OC) were in high concentration in some Chinese cities. Hence, a high concentration of carbonaceous aerosols could strengthen the seriousness of haze and increase the complexity of air pollution (Han et al., 2011; Yang et al, 2017).

Although its per-capita emission of BC is generally not high compared to that of developed countries, China is presently the highest emitter of BC globally (Wang et al., 2012; Zhang et al., 2013; Hoesly et al., 2018; Kurokawa & Ohara, 2019; Choi et al., 2020). A recent study by Wang et al. (2014) suggests that the BC emissions in China could even be 2 or 3 times higher than current estimates**.** Due to the heavy use of solid fuels and high population density (Huang et al., 2015), the OC emissions in China are also high compared to BC emissions, it was reported that anthropogenic emissions of OC in Asia were 8.88Tg, among which 2.56Tg were from China in 2000 (Ohara et al., 2007). In 2015, emissions of BC and OC in China were 1.66Tg and 2.69Tg respectively, accounting for 50.0% and 39.8% emissions in Asia.

Although no routine ambient BC and OC measurement are available in China for the last decade, many measurements of the recent status are reported in the literature (Zhang et al., 2019). The routine measurement of the $PM_{2.5}$ component started only after 2016 at “2+26” key cities over Beijing-Tianjin-Hebei and surrounding areas, accompanied with the campaign of “the Battle for Prevention and Control of Pollution”. However, the data were not fully open yet. Therefore, we try to obtain the general status of BC and OC pollution over Northern China by integrating data from all available sources. By compiling the data available from literature and monitoring raw data during 2013-2017, it was found that the spatial coverage of the measurements could represent the situation over Northern China and the measurement was the most abundant over Beijing-Tianjin-Hebei and surrounding areas. However, since the measurement took over at city sites, the data could represent the situation for major city areas only.

Statistics and calculation of long-term averages (2013-2017) of the results of studies on the characteristics of changes in BC/OC concentrations, the concentration of BC/OC in the northern and southwest China (Hu et al., 2015; Yang et al., 2016; Wang et al., 2011; Huang et al., 2014) is higher than that in the southeast and northwest China (Chen et al., 2016; Feng et al., 2014; Zhuang et al., 2014; Tao et al., 2017). Pronounced seasonality of BC and OC concentration existed over Northern China. For cities over the Beijing-Tianjin-Heibei area, the seasonal mean concentration of BC was generally no more than 2μg/m^3 in spring and summer, while it grew up to 3-10μg/m^3 in fall and winter season. For individual city variations, some cities have very distinct seasonality, the seasonality was even more pronounced. For example, the monthly mean BC concentration reached above 18μg/m^3 in January in Xi'an, a city over Northwest of China, and it was only 2μg/m^3 in June or July. However, the seasonal variation of BC concentration was not significant in cities such as Beijing, Nanjing (Zhang et al., 2019), Shanghai (Ming et al., 2017; Chang et al., 2017).

In terms of seasonality in OC, spring and summer concentration were significantly lower than in fall and winter. Summer OC was the lowest among the seasons, where OC concentration never reached levels higher than 20μg/m^3. In fall and winter, the seasonal mean OC was generally above 20μg/m^3, with some cities in BTH being up to 40μg/m^3. The seasonality was more pronounced in Xi'an as well, where OC reached close to 60μg/m^3 in January and down to below 10μg/m^3 in June or July.

Since most of the measurements from the literature were sporadic, which were not designed systematically and operated routinely, the full picture of the status of BC and OC pollution could not be obtained at this stage. Along with the operation of a monitoring network for $PM_{2.5}$ components, the pollution status of BC and OC in a specific part of Northern China, such as over BTH, could be obtained in near future. To obtain the full picture of Northern China, more time and effort should be taken.

3.3 Impact Evaluation Methodology

BC/OC has a profound impact on both air quality and climate change, and as a major component of PM, it has a greater impact on human health than other pollutants. Therefore, in order to assess and study the changes and distribution of BC/OC in the atmosphere and the impact on climate better, we conducted a simulation and scenario impact assessment study of air quality using different Chemical Transport Models (CTMs) models. Although CTMs have been applied more frequently in the past decades to solve scientific problems related to air quality in China, the simulation results obtained vary due to the different input data of CTMs, model configurations and the mechanisms of the models themselves. To determine the CTM model performance for BC/OC simulations in the Chinese context and in studying climate change, ethods to objectively and accurately assess model performance are very important in air quality management applications.

3.3.1 BC/OC Impact Evaluation in the Project

The impact of BC/OC emissions in the study region on air quality, radiative forcing and global temperature will be quantified with a stepwise approach using a suite of modeling tools of varying degrees of complexity. A critical component will be the use of updated national/regional emission estimates.

To determine the impact of the emissions on the atmospheric distribution of aerosols, we will perform simulations with two state-of-the-art chemistry-transport models, the regional Community Multiscale Air Quality (CMAQ) Modeling system (https://www.cmascenter.org/ cmaq/) and the global OsloCTM3 (https://www.cicero.oslo.no/en/osloctm[3]). CTMs are numerical models that simulate emissions, transport, formation, deposition, and the fate of multiple air pollutants in the atmosphere (Brasseur, et al., 2017) and may give air quality forecasts. CTMs simulate these processes to describe the spatiotemporal distribution of aerosols and trace gases, using meteorological information as input (Schaap et al., 2012). CTMs can be classified according to their methodology as Eulerian and Lagrangian models (Jacob et al., 1999), where Eulerian CTMs include weather research and forecasting model with chemistry (WRF-Chem), CMAQ, comprehensive air quality model with extensions (CAMx), global three-dimensional chemical transport model (GEOS-Chem) and model for ozone and related chemical tracers (MOZART).

CTMs have been applied in a variety of capacities, such as various atmospheric chemistry and air-pollution issues, all over the world, including sensitivity analysis for ozone (Simon et al., 2013; Sandu and Zhang. 2008), air quality forecasting for atmospheric pollutants (Odman et al., 2007), atmospheric chemistry and model evaluation research (Brune et al., 2016; Travis et al., 2016), aerosol-radiation-cloud feedbacks on meteorology and air quality (B. Zhang et al., 2015; Qiu et al., 2017); acid deposition, visibility and haze pollution issues (Han et al., 2014; Fan et al., 2015); the source sector contribution, long-range transport for ozone and aerosol concentrations (K. Li et al., 2016; Zhu et al., 2017) and exposure studies (Bravo et al., 2012; Di et al., 2016).

3.3.2 Evaluation Methods

3.3.2.1 Global CTM Simulation

The OsloCTM3 is an offline global 3-dimensional chemistry-transport model driven by 3-hourly meteorological forecast data from the European Center for Medium-Range Weather Forecast (ECMWF) OpenIFS model (Lund et al., 2018a; Søvde et al., 2012). The model can be run in $1.125° \times 1.125°$ and $2.25° \times 2.25°$ (default) horizontal resolutions, it has 60 levels in the vertical, from the surface to 0.1 hPa. The model treats full tropospheric chemistry, as well as all the main climate-relevant aerosol species (BC/OC, sulfate, secondary organic aerosol, nitrate, dust and sea salt). In addition to BC/OC, emissions of SO_2, NO_x, CO, VOCs and NH_3 are needed to run the model and will be updated with results from this project where available.

The OsloCTM3 (and its predecessor version OsloCTM2) is well documented and has been used in a large number of studies of global and regional anthropogenic emissions, as well as in detailed process studies. It is also one of the participating models in the multi-model experiments AeroCom and HTAP2.

Recently, Lund et al. (2018a) performed a comprehensive evaluation of the model performance in terms of present-day aerosols distribution using the most recent published global emission inventory, the Community Emission Data System (CEDS) (Hosely et al., 2018). Global models typically have trouble representing the vertical distribution of BC aerosol compared to observations (Samset et al., 2014). Work to constrain the treatment of BC in the model (Lund et al., 2017; 2018b) has resulted in significant improvements in many regions. Overall, the OsloCTM3 also performs well for surface concentrations and total aerosol optical depth. However, aerosol concentrations are underestimated in broader East Asian region. While using the CEDS emissions give better agreement with BC surface measurements than using older emissions data, the results indicate significant remaining uncertainties in also the recent global inventories. It is also a known issue that the global inventories do not fully capture the recent declining trends in SO_2 emissions in China. The evaluation of the model over China is also complicated by lack of available data. Through this project, we will perform an updated model evaluation with additional observational data provided by the CRAES collaborators.

To extract the contribution (i.e., source apportionment) from emissions in the study region and

China from the total global emissions, we will perform two simulations for each year (baseline and future pathways); one with all global emissions and one with a 20% reduction in the emissions in the study region. The difference between the two, scaled by a factor 5, is then taken as the contribution from the regional emissions to atmospheric concentrations of aerosols.

3.3.2.2 Regional CTM Simulations

The Reginal CTM used in this project is the CMAQ model system (version 5.0.2, https://www.cmascenter.org/index.cfm), it is a core part of the third generation of the Air Quality Forecasting and Assessment System (Models-3) developed by the U.S. National Environmental Protection Agency, which comprehensively considers pollution issues such as PM, photochemical oxidation and acid deposition (Zhang, 2004; Zhang et al., 2005). CMAQ model adopts the concept of "one atmosphere", which includes complex gas-liquid-aerosol chemistry and simple non-homogeneous chemical processes, and has good simulation capability for atmospheric secondary pollutants (O_3, $PM_{2.5}$, and PM_{10}). Such model systems can be used for a variety of purposes such as simulation, assessment and decision studies of air quality at multiple scales and with multiple pollutants. In this study, CMAQ is configured with the chemical mechanism of the Regional Acid Deposition Model version 2 (RADM2). The aerosol chemical species considered within CMAQ (Binkowski et al., 2003) include fine sulfate, nitrate, ammonium, biogenic OC, EC and other unspecified material of anthropogenic origin, etc.

Anthropogenic emission inventories used for modeling in this research are generally large-scale national (MEIC) and even continental (TRAC-P, INTEX-B, REAS, MIX) inventories (Zhang et al., 2009; Streets et al., 2003; Ohara et al., 2007; Li et al., 2017), etc. Kanaya et al. (2019) simulated surface BC concentrations for China from 2009 to 2019 with different bottom-up emission inventories (e.g., MEIC1.3, ECLIPSE v5a and v6b, REAS updated and CEDS) and compared decadal trend with observations, it was found that the decadal trend in the modeled BC concentrations for most emission inventory simulations, except CEDS, are generally in good agreement with the observed ones. Simulation results differ slightly between different emission inventories.

Although significant advantages can be found in CTMs, how to accurately simulate the concentrations of BC/OC is still a challenge, with the problems of inaccurate emission inventories and other imperfect physical and chemical parameterizations (Carmichael et al., 2008, Chen et al., 2019). To obtain a reliable BC/OC concentration along with model simulations, we need more accurate bottom-up emission inventories.

3.3.2.3 Radiative Transfer Calculation

We will calculate the instantaneous top-of-the atmosphere radiative forcing due to aerosol-radiation interactions using the Oslo RTM, which is an offline radiative transfer model. The same model has been used in earlier studies of RF_{ari} (Myhre et al., 2017; Myhre et al., 2013) with recent updates to aerosol optical properties (Lund et al., 2018). The radiative forcing of aerosol-cloud interactions (RF_{aci}) (earlier denoted the cloud albedo effect or Twomey effect) will be calculated using the same radiative transfer model. To account for the change in cloud droplet concentration resulting from anthropogenic aerosols, which alter the cloud effective radius and thus the optical properties of the clouds, the approach from Quaas et al. (2006) is used. This method has also been applied in the above-mentioned earlier studies. Opportunities to estimate the RF using the high-resolution CMAQ output will be explored.

3.3.2.4 GWP and GTP Assessment

Quantifying the impact of individual regions, sectors or other emission sources on surface temperature is challenging using coupled climate model due to the small signal of climate response relative to the natural variability of the system. Instead, more simplified emulators are commonly used. One approach is to use so-called emission metrics. Emission metrics were developed to allow a simple comparison of the impacts of emissions of different species on a common scale and provide a basis for multi-component climate policies. Here we will produce updated values of the two most commonly used emission metrics, the GWP (IPCC, 1990) and the Global Temperature change Potential (GTP) (Shine et al., 2005), based on input from 3.2, and use them to assess the impacts of present and future emissions in China on global climate over different time horizons. The method from Lund et al. (2020) will be used.

The GWP and GTP are both measures of the global mean impact and does not allow for an assessment of the local climate impacts. However, by calculation region-specific metric values, we account for the fact that emissions in China can have a different effect on global climate than the same emission magnitude in another region. By combining the GTP with the emission scenarios developed in the project, we will provide a first-order estimated of the global temperature response to emissions in the study region under different pathways for future development. The emission metrics can also be used to assess the climate implication of different policy packages for emission reduction in the near- and long-term, as well as potential trade-offs. There are important limitations and advantages to using different emission metrics, in particular when comparing short-lived species like aerosols with the long-lived greenhouse gases. While it is ultimately a user-choice which metric to use, we will demonstrate these issues by using both GWP and GTP and evaluating impacts at different time horizons.

3.4 Regional Air Quality Simulation Over Northern China

3.4.1 Summary of Regional Air Quality Simulation Research Activities

CTMs such as CAMx, CMAQ and WRF-Chem have been applied to modeling studies. Those classic CTMs have been widely employed for estimating the formation and transport of air pollutions (Chen et al., 2020) and simulate the impact of pollutant emissions on air quality (Gu et al., 2018).

The results of the simulation study and air quality monitoring data indicate that $PM_{2.5}$ air quality in northern China has gradually improved, but surface ozone pollution has increased in recent years (Li et al., 2019a; Cheng et al., 2019). About 40% decrease of $PM_{2.5}$ over the 2013-2017 period in the North China Plain (Li et al., 2019b) mainly attributed to reductions in anthropogenic emissions (Silver et al., 2020) rather than meteorological variations (Zhang et al., 2019), moreover, natural emissions also have a significant impact on the increase in air quality (Yang et al., 2017). Therefore, accurate and up-to-date emission inventories (Zheng et al., 2019; Zhou et al., 2017; Zhou et al., 2018) are essential for better performance on air quality modeling.

In addition to air quality effects, long-range transport of pollutants from Asia has been shown to exert large impacts on the formation and dissipation of haze (An et al., 2019; Zhang et al., 2019). Long-range transport of local air pollutant emissions from cities, such as $PM_{2.5}$, can also influence air quality in rural areas surrounding or downwind the city (Ding et al., 2016), long-range transport is also another source of secondary PM (Wang et al., 2016) or fine aerosols (Chuang et al., 2003). Source apportionment tools in CAMx and precursor tagging methodology in CMAQ has been used to calculate the local and distant source contributions (Lin et al., 2016) to ozone, precursor, the primary and secondary PM species concentrations among the selected source groupings (Mailroom, 2010). The sources of air pollutants vary widely across China, the major contributors to $PM_{2.5}$ concentrations are industrial or residential over northern China (Zhang et al., 2017b; Qiao et al., 2018; Shi et al., 2017; Liu et al., 2020), the high emission rates of BC and OC emissions from open burning of biomass (Qin and Xie, 2011) and $PM_{2.5}$ emissions from straw combustion (Zhang et al., 2016) are a considerable primary contribution to $PM_{2.5}$ concentration in northern China (Zhang et al., 2017a). Nevertheless, motor vehicles and regional sources are the most important sources of ozone, followed by a point and biogenic sources (Li et al., 2012; Streets et al., 2007). A large proportion of O_3 concentration can be attributed to external contributions (Wang et al., 2019; Gao et al., 2016; Qu et al., 2014).

The O_3 source apportionment attributes in-situ O_3 formation to a NO_x or VOC source based on the index values that determine the sensitivity of O_3 formation to the NO_x or VOC-limit (Wang et al., 2019a), CTMs can also determine the source sensitivities of modeled chemical species to emission (Wang et al., 2019b). The urban centers and the developed industrial areas of northern China are predominantly VOC-sensitive conditions, while both VOCs and NO_x-limited sensitive and NO_x-sensitive conditions predominate in suburban and remote areas, respectively (Xie et al., 2014; Han et al. 2018; Xu et al., 2019). Furthermore, CTMs has been used in different recent studies to study regional photochemical pollution (Tang et al., 2017), it is also possible to investigate what physicochemical processes govern pollutant concentrations at different times and locations through the process analysis tools in CTM, which can reveal, for example, that atmospheric processes can counteract the scavenging effect of chemical reactions on O_3. (Wang et al., 2019a), the relationship between visibility and aerosol optical depth (Lin et al., 2016; Li et al., 2019), and chemical formation and transformation (An et al., 2019) in China.

A comparison of the performances on simulating the $PM_{2.5}$ over East Asia in 2010 among the twelve CTMS (WRF-CMAQ (v4.7.1 and v5.0.2), WRF-Chem (v3.6.1 and v3.7.1), GEOS-Chem, NHM-Chem, NAQPMS, and NU-WRF) can be found in Tan et al. (2019). Ma et al. (2019) applied and evaluated four regional air quality models (WRF-Chem (version 3.9.1), CHIMERE (version 2017r4), CMAQ (version 5.2) and CAMx (version 6.50)) to simulate dust storms in East Asia, they found different model result in large differences about the simulation value of PM_{10}. It is also evident from numerous model performance evaluation studies of CTMs that differences in the physical and chemical mechanisms used in the models affect the simulation results, but the model performance evaluation results all achieve the “goal” range of the benchmark, which is the model performance that most studies are able to achieve (Emery et al., 2017; Huang et al., 2021).

In the National Key Research and Development Project—“Research on Regulatory Air Quality Modeling Technology System”(2018-2021), a data base regional CTM verification has been established, including nationwide routine monitoring data of O_3, NO/NO_2, CO, SO_2, PM_{10}, $PM_{2.5}$ of 2017-2019 of 338 cities from China’s national air quality monitoring network, $PM_{2.5}$ components, VOCs and radicals of 17 severe ozone, $PM_{2.5}$, and sand-storm pollution cases from super research stations, and $PM_{2.5}$ composition data of 2017/18-2018/19 winters of Beijing-Tianjin-Hebei and Surrounding Cities (2+26 cities). The verification of regional CTM models of CMAQ model (version 5.02 and 5.3.2), CAMx model (version 6.20 and 7.1), NAQPMS model and RegAEMS model have been conducted and the targeted air pollution species include SO_2, NO_2, CO, O_3, $PM_{2.5}$, PM_{10} and $PM_{2.5}$ component of sulfate, nitrate, ammonium, BC and OC.

3.4.2 Model Evaluation and the Common Problems

Different CTMs need different validation parameters to evaluate their performance in different ways. Therefore, there is no general measurement standard suitable for all conditions (Chang and Hanna, 2004). Some of the most common metrics were discussed, such as Root mean square error (RMSE), the coefficient of determination(R^2), the correlation coefficient(R), the mean bias (MB), mean error (ME), normalized mean bias (NMB), normalized mean error (NME), etc.(Yu et al., 2006; USEPA, 2007; Wang et al., 2010) and some other uncommon statistical metrics for different CTMs in recent studies (Appel et al., 2007; Appel et al., 2008;).The evaluation metrics are so varied that it is difficult to judge the overall performance of the model (Karroum et al., 2020). Before we decide on the evaluation metrics to be used, we make clear that the objective of our research is primarily to assess the performance of the model, compare the simulated atmospheric levels of air pollutants from CTMs using current and update emission inventories against observations under the base-year within specific relatively clean areas and severely polluted areas. Our final selection of metrics includes NMB, NME and Pearson's r. Relative measurement is particularly useful when comparing model performance between different CTMs (USEPA, 2007), the common metrics of relative measurement are normalization or fractional form, i.e., NMB and NME, meanwhile, we chose MFB and MFE, which are frequently used in model performance evaluation (Zhang et al., 2014; Zhang and Ying, 2010), as the metrics for analysis. Depending on the variance of the frequency distribution of error magnitudes, we also can combine both RMSE and MAE to determine the model performance (Chai et al., 2014). Pearson's correlation coefficient, which passed a significance test (p-value), was used to examine the strength and direction of the linear relationship between the two continuous variables, the model and the monitoring data (Shimadera et al., 2016; Syrakov et al., 2016). Our performance goals and criteria for NMB, NME and R refer to Yamaji et al. (2020) and Emery et al. (2017), while the performance goals and criteria for MFB and MFE refer to Boylan and Russel. (2006) and USEPA. (2007). RMSE and MAE are negatively-oriented scores, which means lower values are better.

In summary, there is some degrees of uncertainty in model simulations. In terms of $PM_{2.5}$ simulation, regional CTMs generally captured $PM_{2.5}$ magnitude in most cities and the trend of variation in daily average over Northern China (Gao et al., 2016; Chen et al., 2017; Liu et al., 2020). However, due to the complexity of each meteorology (Wang et al., 2010; Chen et al., 2020) and chemistry mechanism (Liu et al., 2019; An et al., 2019) involved and the uncertainty in emission inventory (Hoesly et al., 2017), accurately predicting $PM_{2.5}$ remained a challenge. For instance, the diurnal variation of $PM_{2.5}$ was not reproduced well (Petersen et al., 2019). Moreover, predicting the $PM_{2.5}$ level in severe haze events remains a challenge (Li et al.2019; Zhang et al., 2012; Zhang et al., 2017), as meteorology simulation in calm weather was still a problem that needs to be solved (Hu et al., 2016; Bartholdy, 2000). Choi et al. (2019) compared surface $PM_{2.5}$ chemical components simulated by the WRF-Chem and CMAQ, the results show that the simulation results of different models using similar emission inventory had significant inconsistencies.

In terms of $PM_{2.5}$ components, although $PM_{2.5}$ concentration was well reproduced, each $PM_{2.5}$ species was not so good when compared with measurement (Qiao et al., 2019; Zhao et al., 2016). For instance, some secondary inorganic components were significantly underestimated, especially in severe pollution cases in winter (Hayami et al., 2008; Chen et al., 2016; Wang et al., 2014), when sulfate and nitrate experienced an abrupt increase in several hours (Li et al., 2017; Xu et al., 2019). Some new theories or reaction pathways (Cheng et al., 2017; Wang et al., 2018; He et al., 2014) were proposed to fill the gap and related research still went on. Moreover, secondary organic aerosols simulation was another problem (Hu et al., 2016; Huang et al., 2015).

Another issue was air quality simulations require high-performance computational resources and modeling expertise (El-Harbawi, 2013). Improvements in computational efficiency have received increasing attention from researchers in the field of atmospheric pollution modeling in recent years.

3.4.3 Status of BC/OC Simulation

Currently, regional air quality can be simulated using the CMAQ model, CAMx, WRF-Chem, as well as other air quality models. The anthropogenic emission inventory currently used in studies are MEIC, the Regional Emission inventory in Asia version 2 (REAS2), the Regional Emission inventory in Asia version 3.1 (REASv3.1), Emissions Database for Global Atmospheric Research (EDGAR), the Community Emissions Data System (CEDS), transport and Chemical Evolution over the Pacific (TRACE-P), the Intercontinental Chemical Transport Experiment-Phase B (INTEX-B), etc. Most anthropogenic emissions inventories are slow to update (REAS2 (Kurokawa et al., 2013), EDGAR (Crippa et al., 2018), CEDS (Hoesly et al., 2016)) or have been discontinued (TRACE-P (Streets et al., 2003), INTEX-B (Zhang et al., 2009)). Therefore, continued efforts to update and improve the anthropogenic emissions inventory in China based on the latest emissions statistics are essential.

According to the statistics of different anthropogenic emission inventories, the annual BC emissions increased gradually from 1949 (341 Gg/yr) to 1996 (2,189 Gg/yr), with a fluctuating upward trend from 1996 to 2007(about 1,850 Gg/yr) (Wang et al., 2012), while another study shows the annual emissions of BC in China have shown a decreasing trend in recent years from 2009(1,650 Gg/yr) to 2019 (1,100 Gg/yr) (Kanaya et al., 2019). Similar to BC, the annual OC emissions increased gradually from 1990 (341 Gg/yr) to 2000 (2,189 Gg/yr) (Huang et al., 2015), with a fluctuating upward trend from 2000(2,738 Gg/yr) to 2014(4,297 Gg/yr) (Hoesly et al., 2017), while another study shows the annual emissions of BC in China have shown a decreasing trend in recent years from 2015(2,500 Gg/yr) to 2017 (2,100 Gg/yr) (Silver et al., 2020). Statistical results for inventories of anthropogenic emissions from sources vary considerably between the different literature, but the underlying trends are consistent (Zhang et al., 2009; Kurokawa et al., 2019; Crippa et al., 2018; An et al., 2019; Chang et al., 2018). Spatially, China's BC and OC emissions are more consistent with regional economic development and rural population density distribution, showing an east-west trend (Zhang et al., 2013; Cao et al., 2006; Hoesly et al., 2016; Huang et al., 2015). In northern China, there are strong seasonal variations in BC and OC emissions due to residential heating and agricultural open burning, with peaks in winter and lower emissions in spring and summer (Cao et al., 2006; Liu et al., 2016); in southern China, BC is mainly emitted from mobile emission, so seasonal variations are not significant, while OC emissions are relatively high in winter, due to the important contribution of crop residue combustion (Zheng et al., 2012).

An essential component of all modeling studies is the evaluation of model performance for the simulation, there have been several recent regional studies of BC/OC simulation and evaluation using CTMs in China. Some of these studies simulated ground-level BC and OC concentrations in China using CMAQ model system with different emission inventories (Zhang et al., 2004; Zhang, 2005; Zhang et al., 2007; Wang et al., 2010; Chen et al., 2019; Hu et al., 2017), they reported that the simulated values of BC and OC are generally in good agreement with their observed ones, while the simulated values are slightly lower than the observed value in some regions, which the normalized mean bias between model simulation and observation concentrations within -50% (Wang et al., 2010). Underestimation of pollutant concentrations is largely due to underestimation of primary emissions (Hu et al., 2017).

There were relatively few modeling studies on BC and OC compared with $PM_{2.5}$ and O_3 over China so far. One reason should be attributed to the unavailable long-term data of BC and OC via routine measurement at sites representing regional characteristics. Long-term observations of BC mass concentrations in East Asia (2006-2015) and assessment of model performance and uncertainties in Asia by the MICS-Asia project (Han et al., 2010; Chen et al., 2019). The accuracy of the model simulations was improved when Wang et al. (2016) simulated the emissions using the monthly average BC measurements after inversion. It should be noted that the above evaluation work on BC and OC simulation was based on limited observation at some specific sites. However, along with the implementation of the $PM_{2.5}$ component network over Northern China and the detailed compiling of BC and OC emission inventory, further modeling works on BC and OC will be carried out in the project. We have described the research progress related to BC/OC simulation and evaluation of simulation results in subsection 3.1.

BC, OC and other aerosols affect climate through their interactions with solar radiation and clouds. Most aerosol species have a cooling climate impact, while BC stands out by absorbing solar radiation. While surrounded by significant uncertainty, the best estimate of the global aerosol effect on climate is a net cooling that has masked a notable fraction of the greenhouse gas induced warming to date. Additionally, aerosols influence local and regional climate and weather, such as the precipitaiton, haze events and extreme weather, although the exact nature and extent of their role is still a topic of ongoing research. The significant range in estimates of aerosol's climate effects stem from uncertainties in their distribution, atmospheric processing and optical and microphysical properties, as well as uncertainties in their sources. Despite progress over the past decades, representing aerosols remain a challenge for global and regional atmospheric models, with validation in many cases hampered by lack of observations.

China is currently the world's highest emitter of BC due to its heavy use of solid fuels and high population density. OC emissions are also higher in China compared to BC emissions, which were 1.66Tg and 2.69Tg respectively in 2015, accounting for 50.0% and 39.8% of Asia's emissions. Although there have been no conventional measurements of environmental BC and OC in China in the past decade, many recent measurements have been reported in the literature. The spatial distribution of BC/OC concentration showed a trend of high in the north and low in the south. The seasonal average concentration of BC in cities in The Beijing-Tianjin-Hebei region (BTH) was generally less than 2μg/m^3 in spring and summer, and up to 10μg/m^3 in autumn and winter. The average concentration of OC in autumn and winter is generally above 20μg/m^3, some cities of BTH can reach 40μg/m^3.

In the literature, emission trends in the last decade have been compared with observed results, and the simulated BC concentrations in most emission inventories are basically consistent with the observed results. Despite the significant advantages of continuous time series models, it is still a challenge to accurately simulate BC/OC concentrations and the accuracy of emission inventory is an important basis for obtaining reliable BC/OC concentrations.

Different air quality transport models require different validation parameters to evaluate their performance in different ways. Therefore, there is no universal measurement standard that applies to all conditions. The main objective of our study is to evaluate the performance of the model using current and updated emission inventories and to compare the atmospheric pollutant concentrations simulated by the model with the observed values in the base year. The indicators we finally selected included correlation coefficient (R), normalized mean deviation (NMB), normalized mean error (NME), etc.

4 Human Health Effect of BC in Air Pollution

Decades ago, Europe began to pay attention to air pollution from combustion. And they used "black smoke (BS)" as a measurement index to assess air quality. In the initial epidemiological studies, BS was often used as one of the indexes of air pollution to study whether there was an association between mortality and air pollution. However, as technology and measurement methods have improved, new scientific evidence has led to a recognition of the significant role of black particles (black carbon-BC) as one of the short-lived climate forces. BC has also been found to have significant impacts on health (Figure 4-1). These include impacts on cardiovascular system, respiratory and nervous systems. Evidence on the health impacts from BC is still developing.

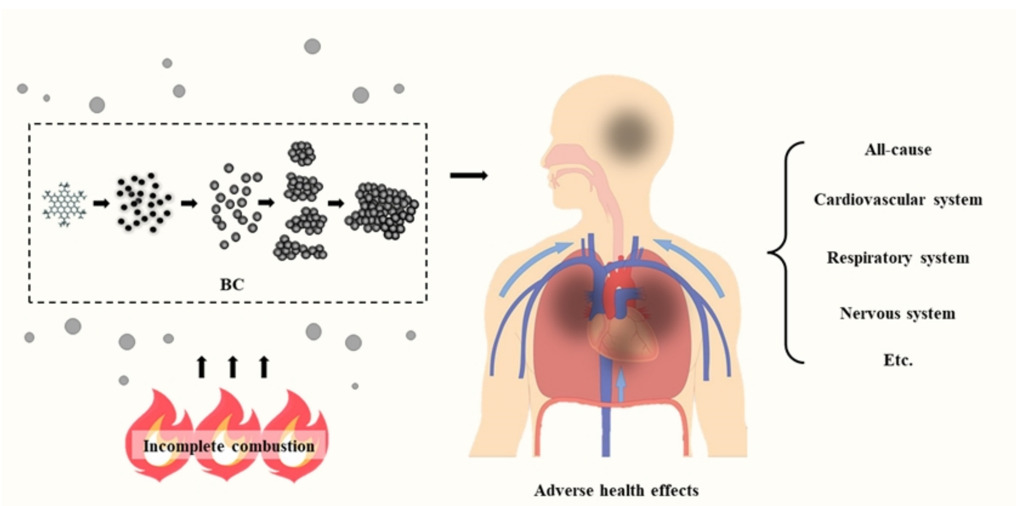


Figure 4-1 Impacts of BC on human health

The development of air pollution standards ideally involves the integration of data from the disciplines of epidemiology, controlled clinical studies, and animal toxicology. Epidemiological studies show statistical associations between health outcomes and exposure; they cannot establish a definite cause-effect relationship. The utility of toxicological studies is to establish this relationship.

In this report, we review all evidence so far on the health impacts from BC in the literature using a systematic review process. We review the global literature on long-term / short-term epidemiological studies, experimental studies, and practice statements from the WHO. This gives us a good overview of the literature that we can then use to develop further experimental studies in the project.

4.1 Health Effects in Epidemiological Studies

We present the results from a systematic review that we conducted. Exposure to BC in a short or long term, people will have varying degrees of adverse effects, especially on the respiratory system, cardiovascular system, and even the nervous system. We identify the need for a systematic quantitative review of the evidence.

The major exposure of BC is inhaled by human body through the respiratory tract. BC in the indoor environment mainly comes from outdoor (Hong et al., 2005; Jia, 2014), and the concentrations of BC are usually lower than the outdoor (Tran et al., 2018). In this study, we mainly research all global articles related to BC in the outdoor environment.

4.1.1 Systematic Analysis

Meta-analysis is an important systematic analysis method. It allows us to express quantitatively by summarizing the research results of scholars and using statistical methods.

According to the Preferred Reporting Items for Systematic Review and Meta-Analyses (PRISMA) guide, four authoritative databases were searched for articles on epidemiology published from the establishment of the database to July 1, 2021. The databases were PubMed, Embase, The Cochrane Library and Web of Science. Searching words were focused on BC, air pollution, mortality and morbidity (searching process see Figure 4-2). Study design was based on an observational epidemiological study, including cohort, case-control, time-series, and case-crossover. We have included all studies on mortality or morbidity of heart and lung diseases, neurological diseases, and other diseases.

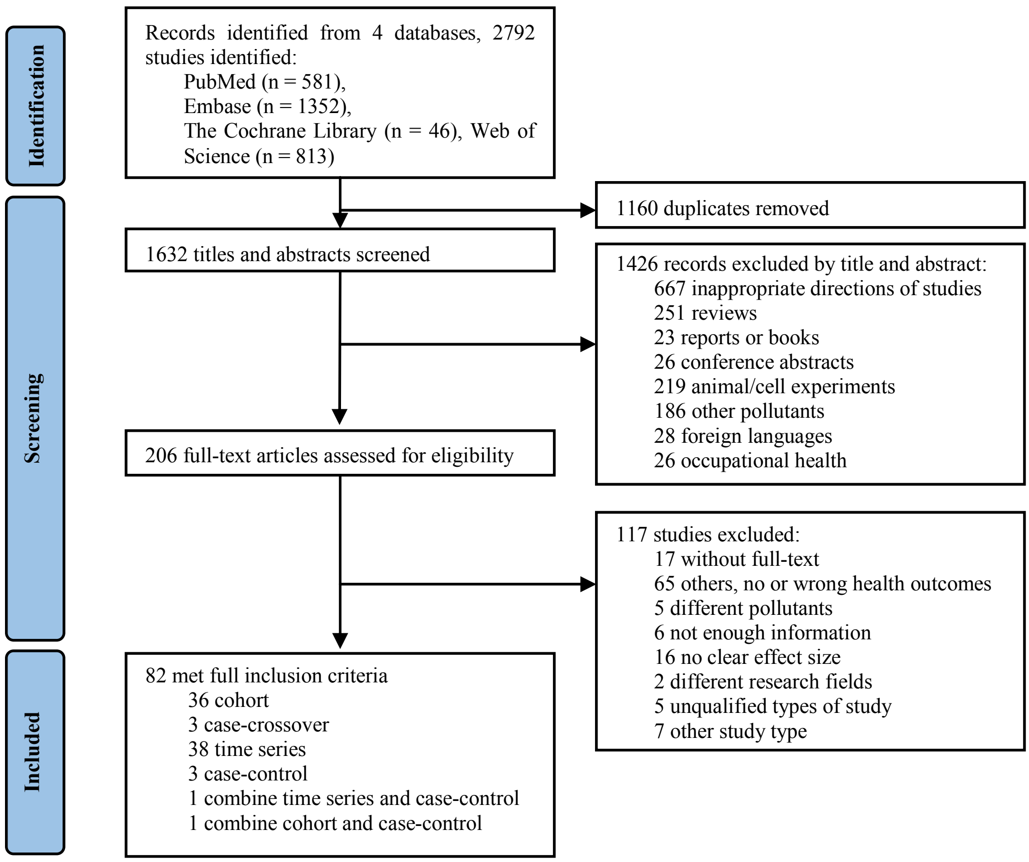


Figure 4-2 PRISMA 2020 flow diagram for this systematic analysis

The search results were 82 eligibility articles. Among them, there were 43 articles about short-term exposure and 40 articles about long-term exposure.

4.1.2 Short-term Health Effects

Through literature review, it is found that short-term exposure to BC affects the lung and cardiorespiratory function of adults with complete immunity or relatively vulnerable children and the elderly, especially susceptible people with asthma, coronary heart disease or other diseases.

The search results of short-term exposure got 43 articles. Table 4-1 was the important information extracted, including study design, study region, study period, subject and sample size, pollutant, health outcome, effect size, international Classification of diseases (ICD), confounding factor, etc.

Table 4-1 Extracted data of short-term exposure articles

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	GAM (Generalized additive Poisson regression models)	Europe		all		lag0-1, lag0-5, lag3-30	CM, RM	percent change	ICD-9 460-519, 390-459	seasonality, long-term trends, temperature, humidity, influenza epidemics, other unusual events, day of the week, and holidays	Analitis et al., 2006
time series	regression model	UK	1994.10-1996.12	all	2300000	lag0-1, lag0-2, lag0-3	ACM, CM, RM	percent change	ICD-9 460-519, 390-459, 390-429, 410-414, 430-438, 493, 490-492, 494-496	long term time trends, seasonal patterns, influenza epidemics, day of the week, and temperature and humidity	Anderson et al., 2001
time series	GAM	UK	2011-2012	all		lag1, lag2, lag0-6, lag1-30	CM, RM	percent change	ICD-10 I00-I99, J00-J99	PM mass	Atkinson et al., 2016
time series	regression model	Spain	1991-1993	all	>750000	lag0, lag1, lag2, lag3, lag4	CM, RM, TM (>70years)	RR	ICD-9 390-459, 460-519	seasonality, temperature, humidity, long term trends, day of the week and holiday, influenza incidence	Ballester et al., 1996
time series	GAM	Spain	1990-1996	all	8249790	lag0, lag1	CM, RM, TM	RR	ICD-9 001-799, 390-459, 460-519	temperature, humidity, long term trends, seasonality, incidence of flu, days of the week, holidays, unusual events	Ballester et al., 2002

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	GLM [generalized linear (possion) model]	UK	1974-1998	⩾50	≈1500000	lag0, lag1-6, lag7-12, lag13-18, lag19-24, lag25-30, lag0-30, lag0-40	ACM, CM, RM, LCM, other mortality	percent change	ICD-9 410-414, 426-429, 434-440, 480-487, 490-496, 162	temperature	Beverl et al., 2014
time series	regression model	Scotland	1974-1988	35-64	15331	lag0-3, lag0-6, lag0-30	ACM, CM, RM	percent change	ICD-9 410-414, 426-429, 434-440, 480-487, 490-496	potential confounding variables measured at baseline: smoking history, social class, BMI, marital status, systolic blood pressure and total cholesterol	Beverl et al., 2012
time series	logistic regression	Netherl-and	1993-1995	50-70	327	lag0, lag1, lag2 and 5-days mean	RSP	OR	/	daily minimum temperature, linear, quadratic and cubic time trend, and weekend/holidays	Boezen et al., 2005
time series	Poisson regression models	UK	1992-1994	all	≈7000000	lag0, lag1, lag2, lag3, lag0-1, lag0-2, lag0-3	ACM, CM, RM, all cancer mortality, all other causes mortality	percent change	ICD-9 <800, 460-519, 490-96, 466, 480-486, 390-459, 410-414, 140-239	trend, seasonality, calendar, deaths from influenza, meteorology, and serial correlation	Bremner et al., 1999
time series	daily time-series analyses	Chile	1998-2006	all	≈498000	lag0, lag1, lag2	CM, RM, TM	RR	ICD-9 <800, ICD-9 390-459, ICD-9 460-519	long term trends, day-of-the-week, and average humidex on the day of death and the day prior to death Santiago de Chile	Cakmak et al., 2009a

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	daily time-series analyses	Chile	2001-2006	all	2198000	lag0, lag1, lag2	total non-accidental morbidity, RM	RR	ICD-9 <800, ICD-9 460-519	long term trends, day of the week, and average humidex on the day of ED visit and the day prior to the visit	Cakmak et al., 2009b
time series	GLM	China	2004-2008	all	2700000	lag0, lag1, lag2, lag3	CM, RM, TM	percent change	ICD-10 A00-R99, I00-I99, J00-J98, S00-T98	temporal trend, day of the week, temperature, relative humidity, SO_2, NO_2	Cao et al., 2012
time series	GLM	Scotland	1981-2001	all	≈1600000	lag0-40	ACM, CM, RM, non-cardiorespiratory causes mortality	percent change	ICD-9 410-414, 426-429, 434-440, 480-487, 490-496	season and other long-term trend, day of week, BS (lag 0, 1 6, 7 12, 13 18, 19 24 and 25 30 days) and temperature (modelled as a double linear relationship, lagged 0, 1 6, 7 12, 13 18, 19 24 and 25 30 days)	Carder et al., 2008
time series	linear regression and autoregressive Poisson model	France	1987-1992	all	≈6140000	lag0, lag1, lag2 and cumulative lags over 4 days	RM, RA, AA, COPD admission	RR	ICD-9 460-519	long term trends, seasonal, weekly and daily patterns, meteorological factors, influenza epidemics, and holidays, a strike of medical residents and nurses in public hospitals	Dab et al., 1996
time series	GAM	France	1988-1997	≥65, all	≈92000	lag0-1, lag0-5	NAM, CM, RM	percent change	ICD-9<800, 460-519, 390-459	/	Filleul et al., 2004

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	GAM	Netherl-and	1986-1994	all	≈14800000	lag0-6	TM, CM, COPD mortality, PM	RR	ICD-9 480-486, 490-496, 390-448	long-term trends, seasonal trends, influenza epidemics, ambient temperature, ambient relative humidity, day of the week and holidays	Fischer et al., 2003
time series	DLM (distributed lag models)	China	2007.4.19-2008.12.31	all	≈7000000	maximum lag of 3 day	CM, RM, TM	percent change	ICD-10 A00-R99, I00-I99, J00-J98	other pollutants	Geng et al., 2013
time series	logistic regression	USA	2014-2016	5-12	2488	——	AP	percent change		SES, BMI, second hand smoking	Gentile et al., 2020
case-crossover	logistic regression	UK	1981-1996	all	14346	lag1-6, lag7-12, lag13-18, lag19-24, lag25-30, lag1-30	PM, community PM (community death from pneumonia), non-community PM	%RR change	ICD-9 480-487	subgroups defined by sex and age	Gittins et al., 2013
time series	GAM	China	2006-2011	all		lag0, lag1, lag2, lag3, lag4, lag0-1, lag0-2, lag0-3, lag0-4	CM	%ER	ICD-10 I00-I99	subgroups defined by age, sex, educational attainment	Gong et al., 2019

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	logistic regression	Netherl-and	1995.7.3-1995.10.6	18-55	60	lag0, lag1, lag2	respiratory symptoms (bronchodilator use, shortness of breath, woken up with breathing problems, pain on deep inspiration, cough and/or phlegm, nasal symptoms, inhaled steroids) and peak expiratory flow	RR	Global Initiative for Asthma	symptom prevalence and medication use, trends in mean morning and evening PEF, exposure to outdoor aeroallergens (data kindly provided by F.Th.M. Spieksma, Leiden, the Netherlands), exposure to environmental tobacco smoke, day of the week and daily maximum temperature (1 h average)	Hilter-man n et al., 1998
time series	poisson time-series model	USA	2000-2006	⩾40		lag0, lag1, lag3	CM, CVD hospitalization	%ER	ICD-9 402, 410, 414, 427, 428, 430-439, ICD-10 I11, I21-I22, I25, I48, I50, I60-I69	temporal and seasonal trends, temperature effects and day of the week	Ito et al., 2011
time series	GAM	USA	2003-2007	all		lag0, lag1, lag2, lag3, lag0-3	TM, CM, RM, IHD mortality, cancer mortality	RR	ICD-10 I00-I99, J00-J99, C00-C48	longer-term temporal trend, as time since the study began, day of week, and daily temperature and humidity	Kim et al., 2015
time series	Poisson time-series regression model	France	1990-1995	all	≈11000000	lag0-1	TM, CM, RM	RR	ICD-9 390-459, 460-519	long-term trends, seasons, days of the week, holidays, influenza epidemics, temperature and humidity	Le Tertre et al., 2002

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	Poisson regression models	USA	1995-1997	⩾65		lag0, lag1, lag2, lag3, lag4	TM, CM	RR	ICD-9 390-448.9	day of the week with indicator variables, and time trends, temperature, and relative humidity with smoothing functions	Mar et al., 2000
time series	Poisson regression models	USA	2000-2003	>65		lag0, lag1, lag2, lag3	ACM, CM, RM	%ER	ICD-10 I00-I99	day of week, smoothing splines of one-day lags of average temperature and humidity [each with 3 degrees of freedom (df)], and a smoothing spline of time with 4 df per year of data	Ostro et al., 2007
time series	GEEs (Generalised estimating equations)	UK	1995.10-1997.10		94	lag1	FEV1, FVC, PEF, worsening symptoms (dyspnoea, sputum purulence or sputum amount, nasal discharge/congestion, wheeze or tight chest and upper respiratory symptoms)	OR		indoor temperature and time spent outdoors	Peacock et al., 2011
time series	Poisson log-linear model	UK	1981-1995	all	≈450000	lag0, lag1	CA, RA	percent change	ICD-9 410-414, 426-429, 434-440, 480-487, 490-496	seasonal and weekday variation, daily temperature and wind speed	Prescott et al., 1998

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	generalized mixed model followed a poisson distribution linear mixed model	USA	2002-2005 spring	10-12	40	lag0, lag1	cough, wheeze, shortness of breath, total symptoms	RR	international study of asthma and allergies in childhood	/	Spira-Cohen et al., 2011
time series	GLM	Serbia	2001-2005	⩾65	≈171000	lag0, lag0-3	daily CM	OR	ICD-10 I00-I99	/	Stanković et al., 2007
time series	Poisson time-series regression model	China	1998-2007	⩾65	40150	lag0, lag1, lag2, lag3, lag0-1	emergency hospital admission for type 2 diabetes mellitus	%ER	ICD-9 250.X0, 250.X2, (X=0-9)	meteorological factors, time trends, public holiday, day of the week, and influenza epidemic	Sun et al., 2016
time series	Poisson regression models	Spain	1985-1991	⩾70		lag0, lag1, lag3, lag5	TM, CM, RM	RR	ICD-9 390-459, 460-519	year, season, day of week, temperature, humidity, influenza, autocorrelation	Sunyer et al., 1996
case-crossover	logistic regression	Japan	2003.4-2007.12	⩾65		lag0, lag1, lag2, lag3, lag0-1, lag0-3	ACM, CM, RM	%ER	ICD-10 I00-I99, J00-J99	ambient temperature, and relative humidity	Ueda et al., 2016
time series	GAM	China	2014-2016		4186	lag0, lag1, lag2, lag0-1, lag0-2	daily hospitalizations for ischemic stroke	percent change	ICD-10 I63	time, day of week, holidays, and weather conditions	Wang et al., 2019

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
case-crossover	Bidirectional time-stratified case-crossover analysis	USA	2006-2011	⩾18	577	lag1, lag2, lag3, lag5, lag7	ICH among patients with a lobar but not deep ICH	OR	acute ICH (symptom onset within 1 day of presentation) confirmed by computed tomography scan.4 Lobar ICH was defined as selective involvement of cerebral cortex, underlying white matter, or both	barometric pressure (continuous), ambient temperature, and dew point temperature (natural cubic splines with 3 df).	Wilker et al., 2018
time series	generalized additive quasi-Poisson regression with polynomial distributed lag model (PDLM)	China	2011-2013	all		lag0, lag1, lag2, lag3, lag4, lag0-1, lag0-2, lag0-3, lag0-4	NAM, CM, RM, IHD mortality, stroke mortality, MI mortality, COPD mortality	percent change	ICD-10 A00-R99, I00-I99, I60-I69, I20-I25, I21-I22, J00-J99, J40-J47	temperature and humidity	Yang et al., 2020a
time series	GAM	Korea	2013-2015			/	NAM, CM, RM	RR	ICD-10 A00-R99, I00-I99, J00-J99	long-term temporal trend, day of the week and meteorology	Yoo et al., 2019
time series	logistic regression using generalized estimating equations	USA	1995-2002, follow-up period 3.1 years	19-90	203	lag1, lag2, lag3, lag0-3	ventricular arrhythmic episode days	OR		season, temperature, relative humidity, day of the week, patient, and a recent prior arrhythmia	Dockery et al., 2005

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	GLM	Korea	2003.3-2007.11			lag0, lag1, lag2, lag3	ACM, CM, RM	%ER	ICD-10 A00-R99, I00-I99, J00-J98, S00-T98	weather	Heo et al., 2014
time series	Time-series analysis using Poisson regression	China	2001-2010	all		lag0-7	NAM, diseases of the circulatory system mortality, hypertensive heart diseases mortality, IHD mortality, myocardial infarction mortality, heart failure mortality, cerebrovascular diseases mortality, RM, PM, chronic obstructive pulmonary disease mortality, diseases of the digestive system mortality, diseases of the genitourinary system mortality	%ER	ICD-10 A00-R99, I00-I99, I10-I15, I20-I25, I21-I23, I500, I60-I69, J00-J99, J12-J18, J41-J44, K00-K93, N00-N99	co-constituent	Sun et al., 2019
time series	GAM and GLM	France	1990-1995	all		lag0, lag1, lag2, lag0-1, lag0-2	TM, CM, RM	percent change	ICD-9 460-519, 390-459	temporal variations, temporal trends	Zeghnoun et al., 2001

Study design	Statistical model	Location	Period	Age	Number of events	lag	Outcome	Effect size	ICD	Confounding factors	Reference
time series	GAM	China	2010-2016	all	2557	lag0, lag1, lag2, lag3, lag4, lag0-1, lag0-2, lag0-3, lag0-4	NAM, CM, RM	percent change	ICD-10 A00-R99, I00-I99, J00-J99	long-term seasonal patterns	Zhang et al., 2020
time series	DLNM (distributed lag nonlinear model) with Poisson	USA	2002-2004			lag0, lag1, lag2, lag3, lag0-2	ACM, CM, RM	%ER	ICD-10 A00-R99, I01-I99, J00-J99	time-varying covariates	Zhou et al., 2011

TM means total mortality, DM means daily mortality, CM means cardiovascular mortality, TRM means total respiratory mortality, RM means respiratory mortality, ACM means all causes mortality, NAM means Non-accidental mortality, LCM means lung cancer mortality, LCI means lung cancer incidence, AP means asthma Prevalence, AI means asthma incidence, WP means wheezing prevalence, RSP means respiratory symptoms prevalence, PM means pneumonia mortality, NCM means Natural cause mortality, CRM means cardiorespiratory mortality, CCI means colon cancer incidence, ADC means Alzheimer's disease incidence, CA means Cardiovascular admissions, RA means Respiratory admissions, AA means asthma admissions, ACDA means all cause daily admission, LRS means lower respiratory tract, URS means upper respiratory tract, FEV1 means Forced Expiratory Volume In 1s, FVC means Forced Vital Capacity, PEF means Peak Expiratory Flow, IHD means Ischemic Heart Disease, COPD means Chronic Obstructive Pulmonary Disease, SES means socioeconomic status, BMI means body mass index, OR means odd ratio, RR means relative risk, ER means excess risk, HR means hazard ratio, ERR means excess relative risk, CI means confidence interval, IQR means interquartile range, ICD means International Classification Of Diseases, ETS means environmental tobacco smoke, USA means the United States of America, UK means the United Kingdom.

4.1.2.1 Study design and Region

Studies of short-term health effects of BC usually use time series, case crossover, etc., which are common methods of observational epidemiological studies. The associations between short-term air pollution exposure (usually daily) and acute health effects can be established by these studies. Time series studies are usually based on stochastic process theory and mathematical and statistical methods to study the statistical patterns followed by random data series. Case crossover studies are a self-controlled approach.

The research areas covered Europe, Asia and South America. The highest number is in Europe with 27 studies. And the second is Asia with 8 articles on China, 3 of which are on northern China.

4.1.2.2 Air Exposure Assessment

Population exposure is assessed by measuring BC at one or more centrally located outdoor monitoring stations. The accuracy of estimates of the effects on health eventually depends on how well daily BC levels measured at the central outdoor monitoring site (ambient BC) reflect daily changes in personal exposure to BC (personal BC) in the study area. However, the accuracy of its response to personal exposure needs to be improved, otherwise to biased health effect estimates. Personal exposure data is most accurate and useful to assess air exposure. In recent years, more and more researchers pay attention to personal exposure, such as dressing vest which can real-time monitor of air pollutant concentrations or filling in the daily behavior pattern table.

4.1.2.3 Statistical Analysis

Poisson regression model and logistic regression model are used to analyze the association between pollutants and health outcomes in the short-term exposure. Logistical regression is used to deal with the regression problem in which the dependent variable is classified variable, such as binary variable and binomial distribution. Poisson regression is a regression method used for the count data. In addition, the impact of pollutants on human health might often have a lag effect. The short-term exposure studies the association between pollutant concentrations within a few days and human health effects. Therefore, the lag effect usually needs to be calculated. The lag effect time is usually calculated as the day of exposure (lag0), 1 day before exposure (lag1) to 3 days before exposure (lag3) and the period lags (lag01, lag02, lag03). For example, lag01 would be the 1-day moving average pollutant exposure (the average pollutant concentrations of the current day and 1 day). Air Pollutants often have strong correlation with each other. If they are all included in the model estimation, there may be strong collinearity impact results.

Therefore, it is necessary to properly select single pollutant model, two-pollutant model or multi-pollutant model.

Before estimating the association between pollutants and health effects, it might be better calculated the correlation among various pollutants. They usually have strong correlation, resulting in strong collinearity in the model and affecting the results. Therefore, it is necessary to properly select a single-pollutant model, double-pollutant model or multi-pollutant model. Furthermore, when estimating the association between pollutants and health effects in the model, it is important to control the impact of other factor on the result as much as possible. In short-term exposure studies, the confounding factors that often need to be controlled are age, gender, BMI, temperature and relative humidity. Sometimes, educational levels, economic condition, workday, occupation, seasonality, smoking and cooking are also factors that need to be considered. However, it needs to distinguish the primary and secondary, and selects the suitable confounding factor to adjust the model according to the actual situation such as sample size.

4.1.2.4 Time-series Studies of Short-term Exposure to BC and Healthy Outcomes

For the time-series studies, a meta-analysis was performed. Pooled random effects relative risk (RR) estimates were calculated for mortality (such as total mortality, cardiovascular mortality and respiratory mortality) for which estimates from at least three different studies were available for the same age group and for different cities. Summary random effects estimates were calculated using the “meta” package in R Studio. The standardized effect estimates were pooled using the random-effect model for there was substantial heterogeneity between the included studies. Heterogeneity among the included studies was evaluated by a Chi-square-based Cochrane Q statistic test and I-squared statistic.

In order to calculate pooled estimates and compare estimated effects for BC in each study, pooled effect estimates were expressed RR per 10 μg/m^3 and 95% confidence intervals (CIs). The short-term relationships between air pollution and health effects are characterized by the time lag (in days) between exposure and health events and investigators vary in which lags they study and report. This means that the use of any particular lag would result in the exclusion of many other studies. Therefore, if more than one lag measure was presented, this review selected one for meta-analysis according to the following algorithm: 1) the lag that the author focused on or stated a priori; 2) the lag that was the most statistically significant (positive or negative) and 3) the lag with the largest effect estimate (positive or negative).

There are 5 studies (Ballester et al., 1996; Ballester et al., 2002; Kim et al., 2015; Le Tertre et al., 2002; Sunyer et al., 1996) on total mortality, 9 studies (Ballester et al., 1996; Ballester et al., 2002; Cakmak et al., 2009a; Fischer et al., 2003; Kim et al., 2015; Le Tertre et al., 2002; Stanković et al., 2007; Yoo et al., 2019) on cardiovascular disease mortality, and 8 studies (Ballester et al., 1996; Ballester et al., 2002; Cakmak et al., 2009a; Dab et al., 1996; Kim et al., 2015; Le Tertre et al., 2002; Sunyer et al., 1996; Yoo et al., 2019) on respiratory disease mortality, all of which had complete information and the effect size was RR. The results of the meta-analysis showed a significant positive association of BC with both total mortality and respiratory disease mortality. The respiratory result has high heterogeneity.

4.1.3 Long-term Health Effects

The search results of long-term exposure got 40 articles. See Table 4-2 for details. Through literature review, it is found that long-term exposure to BC is prone to lung and cardiorespiratory adverse effects. Some studies showed that the cognitive function of the elderly and the newborns (due to exposure of the mother during the pregnancy) may be also affected, thus including neurological effects as a possible consequence of BC exposure.

4.1.3.1 Study Design and Country

The long-term health impact of BC is often followed up and investigated in the form of prospective cohort /retrospective cohort /ambispective cohort. The cohort study is a common observational study for long-term exposure without any intervention. It has high reliability and can reveal the objective causal association between air pollutants and human health effects well. In addition, the case-control study is also one of common observational studies.

The research area involves Europe, Asia, North America, Oceanica and South America. The largest number of studies was in Europe, with 28. The second was in Asia. Among it, there were 4 articles on China, but all of which are cities in South China.

Table 4-2 Extracted data of long-term exposure articles

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	GAM	Netherland	1989-2006			CM, RM, TM	RR	ICD-10 I00-R99, J00-R99	potential confounding due to long-term trends, seasonal trends, influenza epidemics, ambient temperature, ambient air pressure, ambient relative humidity, day of the week and holidays	Ischer et al., 2009
cohort	Cox proportional hazard model	USA	2010-2015	⩾18	41869	CeM, coronary heart disease mortality	HR	ICD-9 410.x, 36.x, 431.x, 434.x, 436.0 or ICD-10 I21.x-I22.x,02.x, I60.x, I61.x, I63.x, I64.x	age, race, sex, BMI, NDI, smoking, baseline co-morbidities, medications, SES	Alexeeff et al., 2018
cohort	Cox proportional hazard model	Netherland	1987-1996	55-69	111816	NCM, CM, RM, LCM, other mortality	RR	ICD-9 all, <800, 400-440, 460-519, 162 or ICD-10 all, <V01, I10-I70, J00-J99, C33-C34	age, sex, smoke, socioeconomic status	Beelen et al., 2008
cohort	Cox proportional hazard model	Netherland	1987-1996	55-69	117528	CM, IHD mortality, CeM, heart failure mortality, cardiac dysrhythmia mortality	RR	ICD-9 400-440, 410-414, 430-438, 428, 427 or ICD-10 I10-I70, I20-I25, I60-I69, I50, I44-I49	age, sex, smoke, socioeconomic status	Beelen et al., 2009
case-control	Cox proportional hazard model	Scotland	1970-1979	35-64	15331	ACM, CM, RM	percent change	ICD-9 410-414, 426-429, 434-440, 480-487, 490-496	seasonal effects and local air quality predictors including altitude (A), household density within a 250-m radius (HD), distance to nearest major road (MR), and distance to an urban boundary (UB)	Beverl et al., 2012

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	multivariable logistic regression	Peru	2011-2014	9-19	484	rhinoconjunctivitis quality of life	OR		pediatric/adolescent survey, age, sex, socioeconomic status, Cole BMI classification (normal, overweight, obese), site (Pampas, Villa), baseline FEV1 percent predicted, temperature (˚C) and humidity (%)	Bose et al., 2018
cohort	Cox proportional hazard model	China	2006-2018	18-65 or ⩾18	28793	NAM, RM, cardio-CeM, cancer mortality, other mortality	HR	ICD-10 A00-R99, I00-I99, C00-C97, J00-J99	age, sex, BMI, educational level, occupation, household income, smoke, drink, sports activity, history of chronic diseases	Chen et al., 2021
cohort	logistic regression	Canada	1994-1998, follow-up 1999-2002	45-85	380738	incident diabetes	OR	ICD-9 250, ICD-10	age, sex, household income	Clark et al., 2017
cohort	competing risk hazards regression models	UK	to 2014.12, to 201511	60-64; 40-69	3129+4 400	CM	HR	ICD-9 390-459, ICD-10	age, sex, race, smoke, SES, regional poverty index, History of diagnosis of CD	Dehbi et al., 2017
cohort	logistic regression	Netherland	2006-2013 baseline, 2014-2017 second	18-93	132595 baseline , 65009 second	prevalent chronic bronchitis and cough and sputum symptoms	OR	British Medical Research Council (MRC)	age, sex, educational level, BMI, smoke, smoke	Doiron et al., 2021
cohort	poisson regression models	UK	1982.4-1986.3, 1994.4-1998.3	⩾30	≈4919781	ACM, CM, RM, LCM, CRM other mortality	ERR	ICD-9 390-519, 390-459, 460-59, 162	deprivation and urban/rural classification	Elliott et al., 2007

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	linear regression	Australia	2001-2004	⩾70	4249	blood pressure, cholesterol, triglycerides, C-reactive protein, and total homocysteine	percent change	ICD-9 390-434, 250 or ICD-10 AM I00-I78, AM E10-E14	age, smoking history (never-smokers, former-smokers who had quit 10 years, former-smokers who had quit <10years, current-smokers), smoking intensity among current smokers (# tobacco products /day), education level (completed university, completed high school, completed less than five years of high schools, and completed some primary school or never attended school), BMI, and history of CVD. socio-economic indicator score, prudence score, and treatment for either elevated blood pressure or cholesterol	Er Hoorn et al., 2021
cohort	Cox proportional hazard model	France	1961-1971	25-59	14284	NAM, LCM, cardiopulmonary mortality	RR	ICD-9 <800, 460-519, 390-459	sex, smoke, educational attainment, BMI, occupational exposure	Filleul et al., 2005
cohort	multivariable logistic regression	USA	1999-2002		2093	failed glucose challenge test /normal oral glucose tolerance test, impaired glucose tolerance, and GDM compared with normal glucose tolerance during pregnancy (prevalence)	OR		age, race, educational level, household income, previous pregnancy history of GDM, family history of diabetes, smoke, date of last menstruation, BMI, gestational weight	Fleisch et al., 2014

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	Canada	1994-1998, follow-up 1999-2002	45-85	467994	COPD mortality	RR	ICD-9 490-492, 496, ICD-10 J40-44	age, sex, Previous comorbidities, SES	Gan et al., 2013
cohort	Cox proportional hazard model	Canada	1994-1998, follow-up 1999-2002	45-85	452735	CHD mortality	RR	ICD-9 410-414, 429.2, ICD-10 I20-I25	age, sex, Previous comorbidities, SES	Gan et al., 2011
cohort	logistic regression	England and Wales	1971-2009	?	367658	CM, RM, ACM	OR	ICD	age, sex, social class of individual, area, population density, geographical region	Hansell et al., 2016
cohort	logistic regression	Sweden	1991-1994	45-64	6031	carotid plaques of prevalence	OR	B-scan ultrasonography	age, sex, education level, smoke score, apoB/apoA1 radio, use of lipid lowering drugs, living alone, cardiovascular heredity, diabetes mellitus, waist hip ratio, physical activity, alcohol consumption, median income level in residential area, systolic blood pressure, and being born outside of Sweden	Hasslöf et al., 2020
cohort	logistic regression	China	2007-2014			mental and behavioral disorders mortality, nervous system diseases mortality, digestive system mortality, genitourinary system diseases mortality, cancer mortality, CM, RM	OR	ICD-10 F00-F99, G00-G99, K00-K93, N00-N99, C00-C97, I00-I99, J00-J99	age, sex, weekday/weekend and seasonal effects, and short-term impacts of air quality	Ho et al., 2020

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	Netherland	1986-1994	55-69	4492	cardiopulmonary mortality, non-cardiopulmonary and non-lung cancer mortality, ACM	RR	ICD-9 <800, 400-440, 460-519, 162, not 400-440, not 162, not 460-519	age, sex, smoke, SES, adjustment for diet	Hoek et al., 2002a
cohort	GAM (Generalized additive Poisson regression models)	Netherland	1986-1994	all	≈2000000	CM, COPD mortality, pneumonia mortality	RR	ICD-9 480-486, 490-496, 390-448	long-term and seasonal trend, influenza morbidity counts, ambient temperature and relative humidity, and indicator variables for day of the week and holidays	Hoek et al., 2000
cohort	logistic regression	China	2016-2018		2326	gestational diabetes mellitus prevalence	OR	International Association of Diabetes and Pregnancy Study Groups	maternal age, BMI, income levels, city, relative humidity and temperature	Hu et al., 2021
cohort	Cox proportional hazard model		1985-2005	mean: 41.7-72.5	18963	LCI, adenocarcinomas incidence, squamous-cell carcinomas incidence	HR	ICD-9 162.2-162.9, ICD-10 C34, ICDO3 8140-8384, ICDO3 8050-8084	age, sex, calendar year, marital status, smoking, BMI, employment status, and neighborhood-level socio- economic status	Hvidtfeldt et al., 2021
cohort	Cox proportional hazard model	Denmark	1993-2015	50-64	49564	ACM, CM, RM	HR	ICD-10 I00-I99, J00-J99, C34	age, sex, educational attainment, occupational status, marital status, smoking (status, intensity and duration), ETS, alcohol consumption, BMI, waist circumference, fruit consumption, vegetable consumption, physical activity, neighborhood level SES and road traffic noise at the residence	Hvidtfeldt et al., 2019

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	poisson log-linear model	Thailand	2010-2016		59605	CCI	RR	ICD-10		Jenwitheesuk et al., 2020
cohort	Cox proportional hazard model	Canada	2006.4.1-2014.3.31	<6	1130855	childhood AI	HR	ICD-10 J45	maternal age, child sex, proportion of visible minority in the dissemination area (as a proxy of race/ethnicity), dissemination area median family income (as a proxy of SES status), dissemination area percentage of female aged 25-64 years who completed postsecondary education (as a proxy of SES status), urban/rural status of place of residence and residential greenness exposure	Lavigne et al., 2021
cohort	Cox proportional hazard model	Europe	1992-2004		98058	COPD hospitalization	HR	ICD-9 490-492, 494-496, ICD-10 J40-J44	age, sex, sub-cohort, smoking duration, squared term, BMI, marital status, employment status, educational level, secondary school, area-level annual year income	Liu et al., 2021

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	Sweden	1990-2011		114758	stroke incidence, IHD incidence	HR	ICD-9 410-414, 431-436, ICD-10 I20-I25, I61-I65	sex, calendar year, subcohort (in Stockholm), smoking status (current, former, never smoker), alcohol consumption in Stockholm and Umeå (daily, weekly, seldom, never), physical activity (sedentary, moderate, intermediate, or vigorous in Gothenburg and Umeå and once a month or less/<1 h per week, about once a month/1 h per week, 3 times a week or more/>2 h per week in Stockholm), marital status (single, married or living with partner, no answer), socioeconomic index by occupation (blue-collar, low and intermediate white-collar and self-employed, high-level white-collar and selfemployed, professionals with academic degrees, no answer), education level (primary school or less, up to secondary school or equivalent, university degree or more, no answer), occupation status (gainfully employed, unemployed/not gainfully employed, retired, no answer), and mean neighborhood individual income in persons of working age	Ljungman et al., 2019

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	linear mixed model	USA	2000-2011		1015	HRV	percent change		age, BMI, fasting blood glucose, smoking history (current, former, or never), current use of anti-hypertensive medications (yes/no), room temperature at the time of electrocardiogram measurement, season (indicated using the sine and cosine of the date), mean arterial blood pressure, and moving averages of outdoor temperature corresponding to the pollutant exposure measurement interval of interest (using both a linear and quadratic term)	Mordukhovich et al., 2015
cohort	Cox proportional hazard model	France	1999-2001, follow-up 12 years	$\geqslant 65$	7066	All-cause dementia incidence, ADC, vascular or mixed dementia incidence	HR	*The Diagnostic and Statistical Manual of Mental Disorders, fourth edition*	age, sex, centre, education, APOE genotype, deprivation index, alcohol intake, and smoking habits	Mortamais et al., 2021

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	USA	2001-2007	30-80	101884	IHD mortality	HR	ICD-10 I00-I99, I20-I25, C34, J00-J98	age (divided into 2-year categories between 30 and 79 years of age, 3-year categories between 80 and 88 years, and one category for women ⩾ 89 years); race [non-Hispanic white, other (African American, Hispanic, Asian, Pacific Islander, and Native American), or unknown]; marital status (married/living with partner, not married, and unknown); smoking status (never, former, and current smokers) and pack-years of smoking (continuous variable for former and current smokers); secondhand smoke exposure (none, household exposure, unknown); BMI (16-19, 20-24, 25-29, 30-39, 40-55 kg/m2); lifetime physical activity (tertiles, unknown); alcohol consumption [beer (no/yes/ unknown), wine (no/yes/unknown), liquor (no/yes/unknown)]; average daily dietary intake of fat (tertiles, unknown), fiber (tertiles, unknown), and calories (tertiles, unknown); menopausal status and hormone replacement therapy use combined (premenopausal, peri/postmenopausal and no HT use, peri/postmenopausal and past HT use, peri/ postmenopausal and current use of estrogen, peri/postmenopausal and current use of estrogen plus progestin, and unknown menopausal status or HT use); family history of myocardial infarction (yes/no) or stroke (yes/no); and use of blood pressure medication (low, medium, high, unknown) or aspirin (low, medium, high, unknown)	Ostro et al., 2015a

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	USA	2002.6-2007.7	>30	9208	ACM, Cardiopulmonary mortality, IHD mortality, pulmonary mortality	HR	ICD-10 I00-I99, I20-I25, C34, J00-J98	smoking status, total pack-years, BMI, marital status, alcohol consumption, second-hand smoke exposure at home, dietary fat, dietary fiber, dietary calories, physical activity, menopausal status, hormone replacement therapy use, family history of myocardial infarction or stroke, blood pressure medication and aspirin use, and contextual variables (income, income inequality, education, population size, racial composition, unemployment)	Ostro et al., 2010
case-control	logistic regression	Denmark	1991-2015	1921-1985 30-85 years old		NCM, CM, RM, LCM	RR	ICD-8:000-799, ICD-10:A00-R99, ICD-8 390-459, 460-519, 162.1, ICD-10 I10-I99, J00-J99, C34	sex, age, and calendar time, by match; marital status, educational level, occupational status, income, country of origin and number of children; car ownership, rented dwellings, unemployment, manual professions, education, income, one-parent households, immigrants and criminal records	Raaschou-Nielsen et al., 2020
cohort	Cox proportional hazard model	USA	1982-2004	⩾30	445860	IHD mortality	HR	ICD-9 410-414, ICD-10 I20-I25		Thurston et al., 2016

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	USA	1995-2005, followed up until 2005	28-105	3895	Acute Myocardial Infarction	HR		age (cubic), sex, hospital of admission, development of a Q-wave myocardial infarction (MI), occurrence of atrial fibrillation, cardiogenic shock, and heart failure during hospitalization and a medical history of stroke, heart failure, angina, diabetes, and previous MI	Von Klot et al., 2009
cohort	logistic regression	Japan	2005-2010	6-9	10069	AI	OR	American Society-Division of Lung Disease	sex; grade as a surrogate variable of age; BMI; respiratory symptoms, such as persistent cough, persistent phlegm, wheeze and chest illness; presence of allergic disease, such as pollinosis; allergic reactions to food, such as egg, dairy products (milk) or other foods; feeding during the lactation period; past history of diseases or surgery, such as sinusitis, bronchitis, pneumonia, pertussis, otitis media, and tonsillectomy; smoker in the household; siblings and first-born child; parents' past history of respiratory illnesses, such as asthma, atopy or pollinosis; housing materials; cookware used at home; heating system installed; humidifier/dehumidifier use; presence of mold in house; flooring materials used in living room and own bedroom; presence of pets (cats, birds, dogs, hamsters, animals with and without fur); use of air cleaners; and use of clothes dryers. These variables were measured using a questionnaire	Yamazaki et al., 2014

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	Cox proportional hazard model	China	1998-2011	$\geqslant$65	66820	ACM, CM, RM, IHD mortality, CeM, pneumonia mortality, COPD mortality	HR	ICD-10 A00-R99, I00-I99, I60-I69, I20-I25, J00-J47, J80-J99, J12-J18, J40-J44, I47	age at entry, sex, BMI, smoking status, physical activity, education level and monthly expenses; percentage of participants who are equal to or older than 65 years old, percentages of subjects whose educational level are higher than secondary school and average income per month within each Tertiary Planning Units (TPU) and percentage of smokers were also adjusted on district level	Yang et al., 2018
cohort	Cox proportional hazard model	Scotland	1970-1976, follow-up until 1998	45-64	22011	ACM, CM, RM, IHD mortality, LCM	HR	ICD-9 410-414, 426-429, 434-440, 786.5, 480-487, 490-496, 786.0, 786.2, 162	marital status, BMI, smoking, cholesterol, systolic blood pressure, social class	Yap et al., 2012
cohort	Cox proportional hazard model and logistic regression	Netherland	2008-2015	$\geqslant$30		NAM	HR			Fischer et al., 2020

Study design	Statistical model	Location	Period	Age	Number of events	Outcome	Effect size	ICD	Confounding factors	Reference
cohort	logistic regression	Greece	1992-2013, follow-up until 2014		5057	LCM	RR	ICD-10 O3	age, sex, stage, place of residence and smoking status	Sifaki-Pistolla et al., 2017

NOTES: TM means total mortality, DM means daily mortality, CM means cardiovascular mortality, TRM means total respiratory mortality, RM means respiratory mortality, ACM means all cause mortality, NAM means Non-accidental mortality, LCM means lung cancer mortality, CeM means cerebrovascular disease mortality, LCI means lung cancer incidence, AP means asthma Prevalence, AI means asthma incidence, WP means wheezing prevalence, RSP means respiratory symptoms prevalence, PM means pneumonia mortality, NCM means Natural cause mortality, CRM means cardiorespiratory mortality, CCI means colon cancer incidence, ADC means Alzheimer's disease incidence, CA means Cardiovascular admissions, RA means Respiratory admissions, AA means asthma admissions, ACDA means all cause daily admission, LRS means lower respiratory tract, URS means upper respiratory tract, IHD means Ischemic Heart Disease, GDM means Gestational Diabetes Mellitus, COPD means Chronic Obstructive Pulmonary Disease, SES means socioeconomic status, BMI means body mass index, OR means odd ratio, RR means relative risk, ER means excess risk, HR means hazard ratio, ERR means excess relative risk, CI means confidence interval, IQR means interquartile range, ICD means International Classification Of Diseases, ETS means environmental tobacco smoke USA means the United States of America, UK means the United Kingdom.

4.1.3.2 Air Exposure Assessment

Land-use regression model (LUR) is often used to predict the BC concentrations in the environment and speculate the individual BC exposure concentrations in the epidemiological study. In the calculation of effect estimates in long-term epidemiological studies, contrasts in long-term exposure between persons are used. Consequently, the aim of exposure assessment is to accurately predict spatial variability in outdoor concentrations and further in personal exposure. For BC, within-city variability in concentrations is larger than that for $PM_{2.5}$ owing to the considerable effect of local combustion sources, especially traffic, on concentrations (Hoek et al., 2002b; Nicole et al., 2008). Within-city variability may exceed between-city variability, which underlines the importance of considering small-scale variations in BC in epidemiological studies. Some epidemiological studies on the long-term effects of BC have relied on a crude estimation of exposure: BC concentrations measured at a single outdoor monitoring site have been assumed to reflect exposure within a city or even over a whole county. In others, the monitoring network has been dense enough to allow interpolation of exposures over an urban area. Neither method is able sufficiently to consider small-scale variations in BC concentrations, which may lead to an underestimation of the effects of BC. In contrast, LUR models have proved their efficiency in a number of recent studies (Jones et al., 2020; Tripathy et al., 2019). LUR models are stochastic models that typically use predictor variables obtained through geographic information systems. These rather simple regression models can explain similar proportions of variability in long-term outdoor concentrations as can dispersion models (Hoek et al., 2008).

4.1.3.3 Statistical Analysis

Cohort study refers to the observation of people with some common characteristics for a certain period of time. It requires long-term observation and follow-up. The Cox proportional hazards model is often used for statistical analysis in most of cohort studies. This model is a kind of survival analysis model and is used to analyze the effects of exposure factors on two variables of survival outcome and survival time. It is suitable for the study of the impact of large samples on survival time and survival rate and is often used for multivariate analysis of cohort studies. Logistic regression is one of the most widely used analysis methods in the epidemiological studies. The binary data of mortality and prevalence (or morbidity) can still be used in the articles of long-term exposure types as dependent variables.

For the confounding factors of long-term exposure studies, age, sex, BMI are still important factors which need to be controlled. In addition, educational levels, economic condition, smoke and race are also usually controlled.

4.1.3.4 Cohort Studies of Long-term Exposure to BC and Healthy Outcomes

For long-term exposure studies, the cohort study is one of the most commonly used study designs. There was a meta-analysis about the cohort study of long-term exposure. Pooled random effects relative risk (RR) or hazard ratio (HR) estimates were calculated for mortality (such as cardiovascular mortality, respiratory mortality and lung cancer mortality). For mortality, HR and RR can often be converted to each other.

There are 6 studies (Beelen et al., 2009; Dehbi et al., 2017; Elliott et al., 2007; Hansell et al., 2016; Hvidtfeldt et al., 2019; Yap et al., 2012) on cardiovascular disease mortality, 6 studies (Beelen et al., 2008; Chen et al., 2021; Elliott et al., 2007; Hvidtfeldt et al., 2019; Yang et al., 2018; Yap et al., 2012) on respiratory disease mortality, and 3 studies (Beelen et al., 2008; Elliott et al., 2007; Yap et al., 2012) on lung cancer mortality, all of which had complete information and the effect size was RR, HR or OR. The results of the meta-analysis showed only a significant positive association between BC and lung cancer mortality with low heterogeneity. The reason for the lack of significant association of other outcomes with BC may be related to the small number of relevant studies.

4.2 Experimental Studies

The BC and substances adsorbed on the surface of BC are the main factors influencing the human health effect (Ni et al., 2014). In order to better determine the adverse effects of human exposure to BC, this study analyzed and summarized three aspects with toxicological mechanisms, animal experimental studies and human clinical studies. Among them, animal experiments and human clinical studies were compared with exposure groups and control groups from aspects of BC, DEP and soot in order to better understand the adverse effects of BC and the possible influence mechanism.

4.2.1 The Formation of BC and Toxicity

In the combustion process, BC is formed by nucleation after carbon atoms form aromatic rings at high temperature, and then gradually form small particles, which then polymerize into clusters to form larger particles (Figure 4-3) (Ali et al., 2020). Ultra-fine particles could migrate deep into the alveolar area (Shrestha et al., 2010) and they usually carry highly toxic and even carcinogenic substances, such as polycyclic aromatic hydrocarbons (PAHs) (Koelmans et al., 2006). Particles whose size is larger than 4 μm or smaller than 0.002 μm have less harm to human health, because after inhalation, most of them are intercepted in the mouth and throat area (Hinds, 1982).

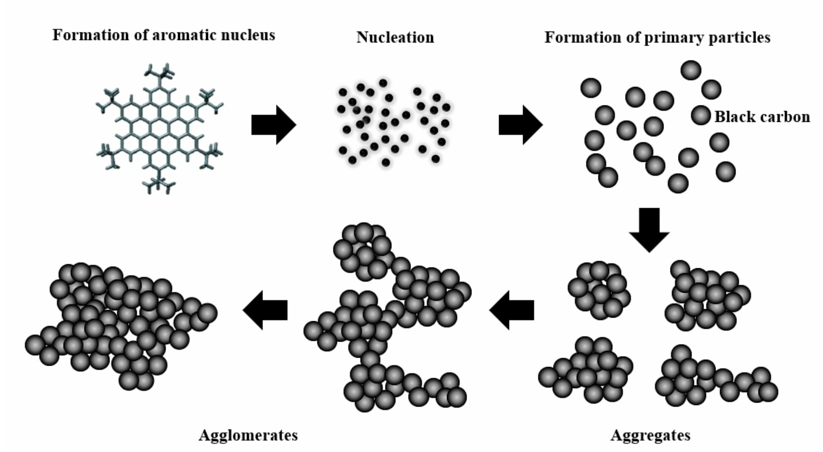


Figure 4-3 Schematic diagram of BC to form particles

The size of BC is smaller, it can reach the alveoli through the human respiratory tract (Marilena Kampa, 1993). Then, it can induce the free radical reaction and causes the consumption of antioxidants and the expression of antioxidant enzymes in the body. When the level of oxidation/antioxidation is unbalanced, oxidative stress (OS) occurs. OS is the main reason for the increase of lung biological index contents through airway inflammation, especially when the body is exposed to PM (Ghio, 2012; Rhoden et al., 2005). The experiment in mice (Gao et al., 2014) showed increased antioxidants and catalases levels in serum after exposure to BC. The levels of OS in vivo were increasing. The frequency of micronucleus in bone marrow with genetic damage increased. Even PM smaller than 100 nm in size can penetrate the alveoli, enter the circulation and has a strong deposition effect (Ching and Kajino, 2018). Then, it could cause lung and cardiorespiratory diseases (Hua et al., 2020). BC enters the human body and has toxic effects on airway cells (such as respiratory epithelial cells) and macrophages. It will cause generation of reactive oxygen species (ROS) and inducepro-inflammatory reactions to the respiratory tract, lungs and cardiovascular system (Marilena Kampa, 1993; Meldrum et al., 2017; Ostro et al., 2015b) and even affect the nervous system (Colicino et al., 2017; Cowell et al., 2015). Among lung cells, macrophages can produce a large number of ROS /reactive nitrogen species (RNS) and pro-inflammatory cytokines such as interleukin (IL)-1β and tumor necrosis factor-α (TNF-α). These factors can cause inflammation or other pathobiological damage, such as increased expression of IL-6, cell adhesion molecule 1 (CAM1), cytoplasmic and inducible nitric oxide synthase and manganese superoxide dismutase (MnSOD) and cytoplasmic phospholipase A2.

Increased levels of ROS /RNS may play a role in the overall response to particulate pollution by activating enzymes and transcription factors. Experiments in mice have shown that BC-stimulated macrophages activate the mitogen-activated protein kinase (MAPK) signaling pathway, which is implicated in inflammation, apoptosis, reproduction, transformation and differentiation processes (Terzano et al., 2010), and regulate the transcription of cytokines and chemokines. The accumulation of these factors in the lungs can lead to the accumulation of inflammatory cells and the production of inflammation (Cheng et al., 2019). After instillation of BC into rat trachea, inflammatory changes were found on lung tissue sections, such as thickening of alveolar septum and bronchiole wall, narrowing of alveolar cavity, infiltration of inflammatory cells in stroma, etc (Gao et al., 2014). In the cellular environment, inflammatory cells can promote the production of ROS and other active substances, which aggravate oxidative damage and promote carcinogenesis. ROS may induce lipid peroxidation and DNA mutation (Gurgueira et al., 2002; Li et al., 2008; Xiao et al., 2003). At this stage, studies (Van et al., 2012) believe that the production of ROS in lung cells is the most important mechanism of carcinogenesis.

In the process of BC formation, a large number of PAHs are often produced. At high temperature, carbon atoms have strong adsorption on PAHs after nucleation (Jonker, 2002). After PAHs enter the body with BC, they can be converted into quinones by bio invertase such as cytochrome P450, epoxide hydrolase and dihydrodiol dehydrogenase in lung (Park et al., 2006; Penning et al., 1999) and then ROS is released by redox cycle (Dellinger et al., 2001). If chlorine (Cl_2) is in the burning process, PAHs can also react with Cl_2 to form highly toxic substances such as dioxins (PCDD) (Johansson et al., 2016) and polychlorinated biphenyls (PCBs) (Roth, 1989). BC can also promote the formation of persistent organic pollutants (POPs) and affect the transfer of POPs (Ni et al., 2014). There are different mixing modes between chemical composition and aerosol particles, including external mixture and internal mixture. Externally mixture is an extreme case of mixing state. It is that the chemical components are separated from each other and the particles are homogeneous mixing of single or the same aerosol type. Internally mixture is another extreme case. It is that the chemical composition is homogeneously mixed with aerosol particles inside the particles, which can also be called volume homogenous mixing. When chemical components are mixed with particles, the particles are wrapped by chemicals in the form of capsules to form a shell core model, which is also a kind of internally mixture (Bond and Bergstrom, 2007; Curci et al., 2019). Particles with different mixing modes will cause different degrees of adverse effects on human health. Moreover, PAHs adsorbed by BC have a wide variety, diverse structures, toxicity and persistence. When they enter the body, they can cause cancer and mutagenesis (Abdel-Shafy and Mansour, 2016) and inhibit immune effect (Armstrong et al., 2004). The process of BC adsorbing toxic substances was shown in Figure 4-4.

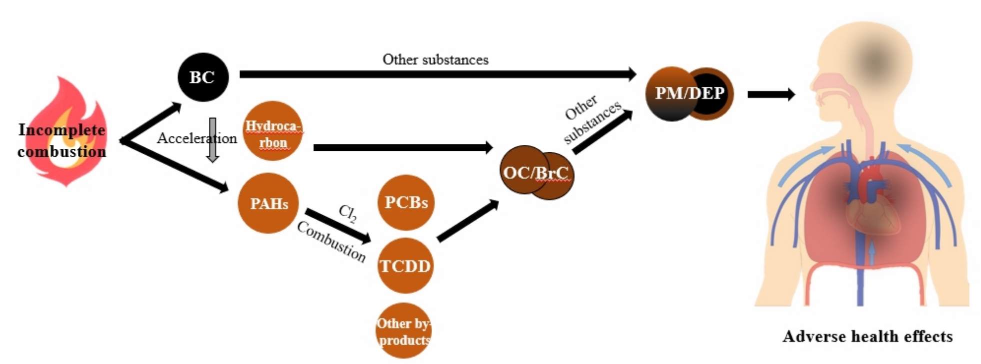


Figure 4-4 The process of BC adsorbing toxic substances

According to the toxicity, carbonaceous aerosols were classified by International Agency for Research on Cancer (IARC). IARC would update the toxicity classification of the carbonaceous aerosols in time with the increase of the experimental evidence. In 2010, BC (1333-86-4) was listed as a group 2B carcinogen by IARC. In 2012, carbon soot (Occupational safety) was listed as a group 1. In 2014, Diesel engine exhaust was listed as a group 1. And in 2016, PM (outdoor air) was listed as in group 1.

4.2.2 Animal Experiments

At the level of animals, experiments could help us to investigate the toxicity of BC and its mechanism, even though there are not many papers on BC. Therefore, we summarized all of the results of animal experiments by or via searching them in the international databases for a better study of toxicity. The databases were PubMed, Embase, The Cochrane Library and web of science. And we searched articles published between 1900 to September 18th, 2021. The searching formula could be showed in Figure 4-5 was the searching process.

There are subtle differences between soot, DEP, BS, EC, BC and others. So, we distinguished between them by listing the methods used to detect pollutants. It can avoid the author's subjective judgment of pollutants. In addition to this, we have listed other important information, such as basic information of the paper, exposed way, outcomes and so on. For the extracted data see Table 4-3.

From the search results, animals exposed to BC, DEP and soot were adversely affected by OS, inflammation, lipid peroxidation, atherosclerosis, changes in heart rate variability, arrhythmia, ST segment depression, changes in vascular function, etc.

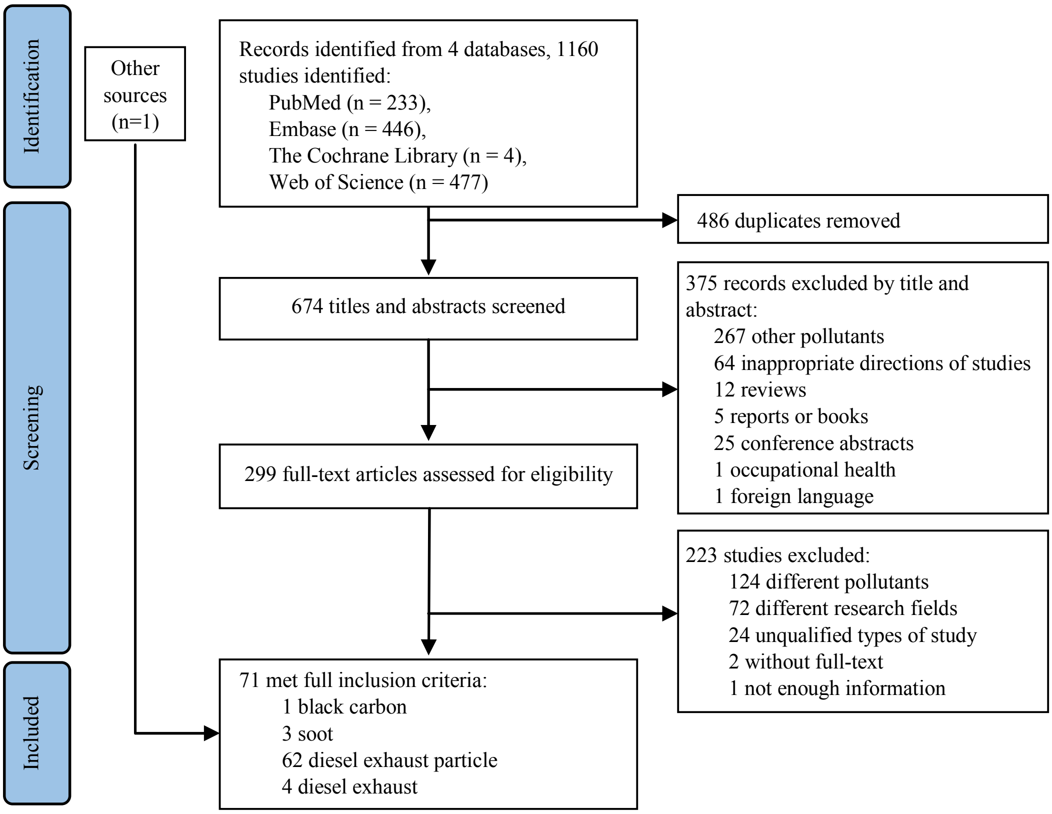


Figure 4-5 Searching process of animal experiment articles

4.2.2.1 BC

Few experiments were directly researching the health impact of being exposed to BC. BC exposure of rats used a continuous whole-body exposure system (ambient particles were homogeneously distributed within each cage in this system). The BC concentrations were monitored with an Aethalometer.

During the experiment, a single variable was often controlled for control experiments: rats were randomly divided into control group and experimental group, in which rats in the control group inhaled air that BC has purified, while rats in the experimental group were exposed to BC. The main ways of BC exposure were inhalation exposure (Chuang et al., 2017).

Through literature review, it is found that BC exposure mainly caused vascular injury and heart disease (Chuang et al., 2017) in rats in pathology, BC exposure mainly caused the increase in blood pressure heart rate and heart rate variability in rats.

Table 4-3 The extracted data from animal experiment articles

Animal type	Sample size	Exposed method	Exposed way	Pollutant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male ICR mice	12 exposure groups of 3 animals each; 4 control groups	To find the appropriate dose for determination of DEPs lung toxicity, the doses of 25, 50 or 100 μg of SRM 2975 were used. ICR mice were randomly separated into four control and twelve exposure groups of 3 animals each. Twelve exposure groups were single intratracheally instilled with 50 μl aqueous suspensions of 25, 50 or 100 μg of SRM 2975 suspended in 0.01 M PBS. The four groups of control were instilled with 50 μl of 0.01 M PBS	intratracheal instillation	DEP		25, 50 or 100 μg of DEP (1 mg/mL, SRM-2975) suspended in 0.01 M PBS	lung inflammation	DEPs produced mild to moderate pulmonary inflammation and injury, as evidenced by infiltration of neutrophils and active AMs, focal alveolitis and particle-laden AMs accumulation	Kae-wamata-wong et al., 2009
Male Sprague-Dawley rats were obtained at the age of 6 weeks old and body weight of around 250 g from Charles River Co., Ltd		Rats were anesthetized with pentobarbital (50 mg/kg body weight) intraperitoneal injection and randomized to be instilled with DEP or equivalent volume of normal saline intra-tracheally (N= 6 for each group). A dose of DEP 250 μg (DEP low dose) or 500 μg (DEP high dose) was suspended in 0.5 ml of normal saline and sonicated for 20 min before use in the DEP-treated group	Rats were randomized to be instilled with DEP intratracheally	DEP		low dose group [(4.5±1.7)× 10^3 mmHg] high dose group [(2.9±2.5)× 10^3 mmHg]	depressed cardiac systolic and diastolic functions after DEP exposure	The cardiac IL-1β gene expression was up-regulated under DEP exposure with a strong trend	Huang et al., 2010

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male TO mice (30-35 g, HsdOla:TO, Harlan, UK)		A Becton Dickinson 24 Gauge cannula was inserted via the mouth into the trachea. Either the DEP sus-pensions (15 μg/mouse) or saline-only were instilled intratracheally (i.t.) (50 μL) via a sterile syringe and followed by an air bolus of 50μL	Rats were instilled with DEP intratracheally via a sterile syringe	DEP		15 μg/mouse	blood pressure/heart rate/heart rate variability	1.DEP administration induced a significant increase of macrophage. 2.DEP caused a significant decrease of SOD activity	Nemmar et al., 2012
eight-week-old male WKY rats	rats were randomly assigned to group 1 (n=4) or group 2 (n=3)	The particle-exposed group 1 was then exposed to 7 days of HEPA-filtered air after a washout, whereas the HEPA-filtered control group 2 was then exposed to 7 days of particles after the washout. The crossover exposure between particles and HEPA-filtered air was repeated 4 times during the experiment. Real-time radio-telemetric data were monitored during the exposure period (particles and HEPA-filtered control)	a continuous whole-body exposure system (ambient particles were homogeneously distributed within each cage in this system)	BC	BC mass concentrations were monitored with an Aethalometer (Magee AE31, USA). Quartz filter samples were used for analysis of EC concentrations with a carbonaceous aerosol analyzer (DRI Model, 2001A Optical Carbon Analyzer, Atmos lytic, USA)	1229.7± 728.0 ng/m^3	blood pressure/heart rate/heart rate variability		Chuang et al., 2017

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
seven-week-old, male, C57BL/6NCrlOri mice were purchased from Orient Bio, Inc. (Seongnam-Si, Korea)	control group (n = 8), treatment 1 group (n=8), treatment 2 group(n=8)	The experimental mice (n= 24) were randomly assigned into three groups (n= 8 per group). DEP was dispersed in 50µL distilled water and were administered in a low (5 mg/kg; n= 8) and a high (15 mg/kg; n= 8) dosage (thus creating the low DEP exposure (DEPL) and high DEP exposure (DEPH) groups) by intratracheal instillation for 7 consecutive days. The control (Ctrl; n= 8) group was treated with distilled water for 7 days	intratracheal instillation	DEP		low dose group (5 mg/kg) high dose group (15 mg/kg)	pulmonary inflammation and anxious behavior	induce inflammatory responses in the lungs by elevating pro-inflammatory cytokine secretion and gene expression	Jeong et al., 2021
swiss albino mice (School of Medicine, University of Sa˜o Paulo, SP, Brazil), 8- to 10-week old	zygotes were allotted in groups of 10	Zygotes obtained from super ovulated mice after IVF were randomly cultured in different DEP concentrations (0,0.2, 2, and 20 mg/cm2) for 5 days and observed for their capacity to attach and develop on a fibronectin matrix until day 8	zygotes were cultured in 20-µL drops of KSOM/AA with the predetermined concentrations of DEP (i.e.,0, 1, 10 and 100l g/mL)	DEP		0,0.2, 2, and 20 mg/cm^2	early embryo development	1. the developmental potential of the zygotes was significantly impaired by the 20ug/cm2 concentration. At a concentration of 2ug/cm2, the develop-mental competence of the blastocysts evaluated by D score was significantly reduce. 2. the exposure of zygotes to DEP, led to the disruption of the normal segregation pattern by a marked decrease in the number of ICM cells	Januario et al., 2010

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
adult male rats weighing 150-170 g were bought from University of Nigeria, Enugu campus	While the control group was made up of thirty rats, the other three groups had 10 rats each making a total of 60	control group: Rats were kept away from black soot and received normal drinking water and food. 4-week group: Rats were exposed to soot for 4 weeks consecutively. 8-week group: Rats were exposed to soot for 8 weeks consecutively.12-week group: Rats were exposed to soot for 12 weeks consecutively	inhalation exposure	soot		soot of Niger Delta in Nigeria	hypothalamic and testicular oxidative stress and apoptosis	1. MDA activity increased significantly following exposure to the soot. 2.alkaline dehydrogenase, alkaline phosphatase and lactate dehydrogenase was a statistically significant decrease. 3. led to the upregulation of caspase-3 expression in the hypothalamus and testis	Onyeso et al., 2020
C57Bl/6J mice (male, 4-week-old)	120	A Becton Dickinson 18 Gauge cannula was then inserted via the mouse mouth into the trachea. DEP suspension (20 µg/50 µL in PBS) or PBS only was intratracheally instilled using a sterile syringe followed by 150µL air bolus	Intratracheal instillation	DEP			the fertility of male	1.compromised the repair of meiotic DSBs and thus the meiotic progression during spermatogenesis. 2. massively altered the testicular gene expression profile including the	Yang et al., 2020b

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
Male TO mice (30-35 g, HsdOla: TO, Harlan, UK)		A Becton Dickinson 24-gauge cannula was inserted via the mouth into the trachea. Either the DEP suspensions (30 mg per mouse) or saline-only were instilled i.t. (40 mL) via a sterile syringe and followed by an air bolus of 50 mL	Intratracheal instillation	DEP		30 mg/mouse	lung inflammation and cardiovascular functions	1.DEP induced a significant increase in the number of leukocytes in whole blood and IL-6 concentration in plasma. 2. low concentrations of DEP (0.1-1µg/mL blood) caused platelet aggregation	Nemmar et al., 2011a
$Tnf^{-/-}$ mice (B6, 129S-Tnftm1Gk1), C57 BL/6J and BALBcJ mice were purchased from Taconic Europe		The study consists of four parts: 1. a single exposure of BALBcJ mice to 20 or 80 mg/m^3 SRM1650. 2. a single dose exposure of C57xCBA mice to 80 mg/m^3 SRM2975. 3. a single dose exposure of $Tnf^{-/-}$ mice and $Tnf^{+/+}$ mice to 20 mg/m^3 SRM2975. 4. four repeated. exposures of $Tnf^{-/-}$ mice and $Tnf^{+/+}$ mice to 20 mg/m^3 SRM2975	inhalation exposure	DEP		low dose group (20 mg/m^3) high dose group (80 mg/m^3)	cytokine expression and neutrophilic inflammation	1.increased the expression level of Mip-2, independently of Tnf status. 2. DEP-induced expression of Mcp-1 and Il-6 occurred in the absence of Tnf	Saber et al., 2006

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
twelve-week-old male ApoE$^{-/-}$ mice		Mice, with either scaffold implantation or hindlimb ischemia, were exposed to either diluted WDE (whole diesel exhaust, containing DEP [DEP] at a concentration of about 1 mg/m^3, as well as all of the gaseous pollutants in the exhaust)	inhalation exposure	DEP		1 mg/m^3	DEP caused cell death	1. increased inflammatory cell infiltration in the tissues and scaffolds. 2.induced vasculogenesis, which was manifested by increased CD31 and α-SMA expression in the scaffolds. 3. resulted in decreased eNOS expression, which may lead to hypoxia by decreasing the vascular production of nitric oxide	Xu et al., 2009
WKY rats		WYK rats were exposed for 4 weeks to DE (0, 50, 150, 500 μg/m^3) by inhalation. DE particles (DEP) were administered intratracheally once (600 μg/mouse) or 8 times (100 μg/mouse) across 28 days to male mice (Trem2$^{+/+}$, Trem2$^{-/-}$, PHOX$^{+/+}$, and PHOX$^{-/-}$)	inhalation exposure	DEP		0, 50, 150, 500 μg/m^3	neuroinflammation	1.TREM2 protein was globally diminished, indicating impaired TREM2 expression. 2. TREM2 regulates the DEP-induced gene expression of Ncf1 and Ncf2, genes that encode two cytosolic components of NADPH oxidase	Greve et al., 2020b

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male C57Bl/6 mice		At 16 weeks of age, animals were randomly divided into room air- and diesel exhaust particle (DEP)-exposed groups for four weeks	inhalation exposure	DEP		15 ng of freshly vortexed DEP per exposure in a bolus of approximately 20μL (3 μg/mL in the plasma)	the function of macrophage mitochondrial	1.DEP Increases Plasma Cytokines and Ceramide. 2. IL-1β TNF-α were measured, revealing a several-fold increase	Gibbs et al., 2019
Female $ApoE^{-/-}$ mice fed a Western diet, and wild-type (C57Bl/6) mice		Female $ApoE^{-/-}$mice and wild-type (C57Bl/6) mice were gavaged with DEP (SRM2975) doses corresponding to mucociliary clearance from inhalation exposure (200 or 1000 ng/day, 3times a week for 3 months and 40, 200 or 1000 ng/day, 3 times a week for 6 months, respectively)	oral administration	DEP		200 or 1000 ng/day, 3 times a week for 3 months and 40, 200 or 1000 ng/day, 3 times a week for 6 months	function of the gut microbiota	1. In $ApoE^{-/-}$ mice, β-diversity was modified by DEP. 2. In C57BL/6 mice, DEP reducedα-diversity (Shannon and Simpson indices) and modifiedβ-diversity	Vanden Brule et al., 2021
C57BL/6Jpun/pun mice were obtained from the Jackson Laboratory		Pregnant dams were DEP treated by gavage (0.2ml suspension in PBS) for five consecutive days at 10.5-15.5 dpc once a day at doses of 31.25, 62.50, 125, 250 and 500 mg/kg/day. A control group received PBS. Another group of mice was administered CB particles suspended in PBS at 500 mg/kg/day. For each group 4-6 dams were exposed	by gavage and inhalation	DEP		31.25, 62.50, 125, 250, and 500 mg/kg/day	DNA deletions after transplacental exposure	1. DEP exposure during gestation resulted in significantly elevated frequencies of DNA deletions in offspring. 2. DEP exposure by the inhalation route showed that DEP induce oxidative DNA damage	Reliene et al., 2005

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
ICR male mice that have been reported to be highly responsive to LPS		The LPS group received 100 μg of LPS dissolved in the identical vehicle. The DEP group received 250 μg of suspended DEPs in the same vehicle. The suspension was sonicated for 3 minutes with an ultrasonic disrupter. The LPS DEP group received the combined treatment of LPS and DEPs	Intratracheal instillation	DEP		949×10^4 $\pm17\times10^4$ EU/mL	lung inflammation	increased local expression of IL-1β, ICAM-1, and chemokines, such as MIP-1α, MCP-1, and KC	Takano et al., 2002
10-12-week-old C57Bl/6 wild-type and ApoE$^{-/-}$mice (C57Bl/6 genetic background, bred at our own facilities)		C57Bl/6 and ApoE$^{-/-}$mice were intranasally treated with DEP (standard reference material 1650b, National Institute of Standards in Technology (NIST, USA) or solvent (PBS) 2 μg DEP was solved in 20 μL PBS and sonicated for 5 min before each administration. Instillation was performed in inhalative anesthesia with isoflurane using a micropipette (Greiner)	intranasal instillation	DEP		1 mg/mL	endothelial progenitor cells and on the associated vascular damage	exposure to DEP reduces the number of circulating EPC and impairs EPC function in two different mouse models, namely C57Bl/6 wild-type and ApoE$^{-/-}$mice	Poss et al., 2013
sixteen-week-old Male WKY rats		The tail was disinfected with ethanol, and 150 μl of vehicle or doses of 0.02 mg or 0.1 mg DEP/kg corresponding to about 8 μg or 44 μg DEP/rat were injected into the tail vein. 48 hours after the systemic administration of DEP, the animals were subjected to blood collection and cell counting and perfusion fixation and tissue sampling	inject into the tail vein	DEP		0.02 mg or 0.1 mg DEP/kg per animal	lung inflammation	increase of monocytes and granulocytes numbers	Nemmar and Inuwa, 2008

Animal type	Sample size	Exposed method	Exposed way	Pollutant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male and female hamsters (Pfd Gold, University of Leuven, Belgium)		The tracheal zone was shaved and disinfected with ethanol (70%), and the trachea was exposed for the intratracheal administration of 120 μL of vehicle or DEPs (5, 50, or 500 μg per animal)	inject into vein	DEP		5, 50, or 500 μg per animal	thrombosis and lung inflammation	DEPs can enhance peripheral vascular thrombosis and that they do so in association with platelet activation	Nemmar et al., 2003
male Wistar rats (Taconic Farms Inc., Germantown, NY), aged 10-12 weeks and initially weighing 258±6 g		A Becton Dickinson 18 Gauge cannula (Franklin Lakes, NJ) was inserted via the mouth into the trachea. DEP suspension (0.5 or 1 mg/kg in 150μL) or vehicle only were instilled (150 μL) via a sterile syringe and followed by an air bolus of 100 μL	Intratracheal instillation	DEP	analyzed the size of DEP used in the present study by transmission electron microscopy	0.5 or 1 mg/kg in 150 μL normal saline	Acute Renal Failure	DEP deposited in the lungs can aggravate experimental acute renal failure by the concomitant administration of cisplatin	Nemmar et al., 2010b
six-week-old female BALB/c mice, weighing 16-19 g	The mice were divided into four groups, each consisting of 10 mice	group A, PBS-exposed group; group B, DEP-exposed group; group C, HDM-exposed group; and group D, HDM and DEP co-exposed group	Intratracheal instillation	DEP		100 μg DEP suspended in 20 μL of 0.05% Tween 80-PBS	allergic rhinitis	The mice exposed only to DEP showed no increase in allergic symptoms	Jung et al., 2021

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
female $ApoE^{-/-}$knock out mice and C57BL/6J $ApoE^{+/+}$mice		The mice were given i.p. injections of DEP suspended in saline in the following concentrations: 0,0.5 and 5 mg/kg bodyweight, 1 h prior to sacrifice by cervical dislocation (n= 8 per group); The doses in the in vitro experiments (10 and 100 μg/mL) were chosen to mimic a circumstance where all particles were translocated from the peritoneum to the circulation (assuming mice weighing 20 g have 1 mL vascular fluid)	internal exposure: intraperitoneal injection and external exposure: inhalation exposure	DEP		internal exposure: 0,0.5 and 5 mg/kg bodyweight external exposure: 10 and 100 μg/mL	lung inflammation and cardiovascular functions	acute systemic exposure to DEP at a modest dose impairs the endothelium-dependent vasorelaxation only in mildly atherosclerotic vessels of $ApoE^{-/-}$mice	Hansen et al., 2007
male TNF-R1-defi-cient mice (C57BL/6-Tnfrsf1atm1 Imx) and wild-type mice (C57BL/6 background)	10 rats or 15 mice	Freshly isolated capillaries were exposed to DEPs at the concentrations indicated for 6 h at room temperature without or with modulators	2 mg of DEPs was suspended in 10 ml PBS buffer	DEP		2mg/10mL PBS	brain inflammation	DEP exposure up-regulates P-glycoprotein in brain capillaries and NADPH	Hartz et al., 2008
Adult male $ApoE^{-/-}$mice (N = 20) and the background strain (C57bl6 mice; N = 16) were purchased from Charles River (Margate, UK)	Adult male $ApoE^{-/-}$ mice (N = 20) and the background strain (C57bl6 mice; N = 16)	$ApoE^{-/-}$mice were fed a Western diet(8 weeks) to induce atherosclerotic plaques, with parallel experiments in normal chow fed wild-type mice. During the last 4 weeks of feeding, mice received twice weekly instillation (oropharyngeal aspiration) of 35 μL DEP (1 mg/mL, SRM-2975) or vehicle (saline)	oropharyngeal aspiration	DEP		DEP (1 mg/mL, SRM-2975)	Atherosclerosis	in the murine apolipoprotein E deficiency model, instillation of DEP increased lesion size, produced more lesions per vessel and generated more buried fibrous caps	Miller et al., 2013

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
adult male Wistar rats (200-250 g; Charles River, Mar-gate, UK;)	at least 4 animals per treatment group	Rats were anesthetised, using 5% isoflurane inhalation (Meriol, Essex, UK) and positioned head-upwards on a board. The vocal chords were visualised by passing apediatric laryngoscope with a plastic cannula into the trachea. Particles (0.5 mg/rat) or vehicle (saline) were then instilled as a 0.5 mL bolus	intratracheal instillation	DEP		DEP (1 mg/mL, SRM-2975)	pulmonary and systemic inflammation	Exposure of rats to DEP induces both pulmonary and systemic inflammation, but does not modify endothelium-dependent vasodilatation	Robertson et al., 2012
female Balb/c mice (Orient Bio, Seongnam, Korea) weighing 16.10 ± 0.52 g	5 experimental groups (n = 5 per group)	The mice in the naive control group received no treatment for the entire experiment. The mice in the vehicle control group received 50 µL saline containing 0.05% (v/v) Tween 80 (Sigma-Aldrich Corp., St.Louis, MO, USA). The mice in the DEP 25, DEP 50 and DEP 100 groups were intratracheally instilled with25 µg, 50 µg and 100 µg DEP	intratracheal instillation	DEP		DEP (1 mg/mL, SRM-2976)	lung inflammation	Conclusions DEP may contribute to neutrophilic lung inflammation pathogenesis by modulating ER stress-mediated CXCL1/KC expression in alveolar macrophages	Kim et al., 2020a
forty-two male Big Blue® (Fischer) rats, approximately 8 weeks of age from Stratagene (La Jolla, USA)	Animals were assigned to seven groups (six ani-mals/ group)	Animals were assigned to seven groups (six animals/group), which were fed with 0,0.2,0.8, 2, 8, 20 or 80 mg DEP/kg Altromin diet prepared by Altromin in Germany	feed	DEP		0,0.2,0.8, 2, 8, 20 or 80 mg DEP/kg	DNA damage in lung	Primary DNA damage was observed in the lung after oral exposure to DEP. The level of DNA strand breaks, bulky DNA adducts and oxidized bases increased especially in the intermediate dose levels	Muller et al., 2004

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
forty-seven male Fischer rats 344, 9 weeks of age, from Taconic, Europe (Ry, Denmark)	The rats were randomly assigned to six groups (n= 8, except the group exposed 6 h to the low dose of DEP that contained 7 rats)	0 mg/kg bodyweight (control),0.064 mg/kg bodyweight (low dose), and 0.64 mg/kg bodyweight (high dose) were exposed to Standard Reference Material 2, 975 at 0.064 or 0.64 mg/kg bodyweight for 6 and 24 h	oral gavage	DEP		0.064 or 0.64 mg/kg body weight	DNA damage, oxidative stress and DNA repair in colon epithelial cells, liver, and lung	The levels of 8-oxodG in the lung, liver, and colon were significantly increased after high dose DEP-exposure at6 and 24h post-exposure; whereas the 8-oxodG level was unaltered for the low dose DEP-exposure at 6 and 24 h compared to the control	Danielsen et al., 2008
male Sprague-Dawley rats		alveolar macrophages were isolated from male Sprague-Dawley rats by bronchoalveolar lavage. alveolar macrophages were incubated with 0, 5, 10, 20, 50 or 100 pg/mL of DEP (2.5×10^5 particles/pg DEP), methanol-washed DEP, or equivalent concentrations of DEP methanol extracts at 37℃ in 5% CO_2	DEP was dissolved in PBS, add to the medium	DEP		0, 5, 10, 20, 50, or 100 pg/mL of DEP	the release of proinflammat-ory cytokines, interhkin-1 (IL-I), and tumor necrosis factor-alpha (TNF-α) by alveolar macrophages (AM)	evidence that DEP enhanced the production of IL-l by AM in vitro suggests that this proinflammatory cytokine may play a role in the pulmonary response to DEP inhalation	Yang et al., 1997

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
40 time-mated, nulliparous, young adult mice (C57BL/6 BomTac, Taconic Europe, Ejby, Denmark)	control group (n=20), treatment group (n=20)	The two groups of mice were exposed to either filtered clean air or approximately 20 mg DEP/m^3 on GDs 7-19 for one hour/day	inhalation exposure	DEP		20 mg/m^3	postnatal development, behavior, genotoxicity and inflammation	In utero exposure to DEP decreased weight gain during lactation the mRNA expression levels were slightly higher in the DEP exposed pups	Hougaard et al., 2008
sixteen-week-old male SHR (Taconic Farms Inc., Germantown, NY, USA)	control group (n=9), treatment 1 group (n=6), treatment 2 group (n=6)	The tail was disinfected with ethanol, and 150µl of vehicle (n= 9) or doses of 0.01 (n= 6) or 0.02 (n= 6) mg DEP/kg corresponding to about 2.8 or 5.6 µg DEP/rat were injected into the tail vein	inject into the tail vein	DEP		0.01,0.02 mg/kg	systolic blood pressure, heart rate and both systemic and pulmonary inflammation in spontaneously hypertensive rats	DEP exposure (0.02 mg/kg) significantly elevated the number of leukocytes in blood, IL-6, tumor necrosis factor alpha and LTB4 concentrations in plasma	Nemmar et al., 2009
male C57BL/6 mice 6-8 weeks old	A minimum of 5 animals were used for each endpoint	Animals were anesthetized using vaporized halothane and suspended on their incisors. The tongue was distended and a bolus of either 50 µL HBSS vehicle or 25 µg DEP in 50 µL HBSS was injected onto the oropharynx	inject onto the oropharynx	DEP		0.5 µg/µL	allergic inflammation	Th2-type cytokines, such as IL-4 and IL-13, and markers of eosinophil chemotaxis, such as CCL11 and CCR3, were increased	Jaspers et al., 2009

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male TO mice (30-35 g, HsdOla:TO, Harlan, UK)		A Becton Dickinson 24 Gauge cannula was inserted via the mouth into the trachea. Either the DEP suspensions (15 µg/mouse) or saline-only were instilled intratracheally (i.t.) (40 µL) via a sterile syringe and followed by an air bolus of 50 µL	intratracheal instillation	DEP		0.1-1µg/mL, 0.25-1 µg/mL	Exacerbation of thrombotic events	1.The direct addition of DEP(0.1-1µg/ml) to untreated mouse blood significantly induced in vitro platelet aggregation. 2.In vitro exposure to DEP (0.25-1µg/ml) led to activated intravascular coagulation, both the APTT and the PT were shortened	Nemmar et al., 2011b
female CD1, C57BL/6, or BALB/c mice (Charles River, Margate, UK)		Mice were exposed once daily to either 80 mg of DEPs dissolved in 40 mL of PBS, after 3 exposures, mice were anesthetized before intranasal infection with 1×10^5 CFU of D39 in 10mL of PBS Control mice were treated with 10mL of PBS only	intratracheal instillation	DEP		0.5 mg/mL DEP	susceptibility to invasive pneumococcal disease	Alveolar macrophages become congested with DEPs, which reduces their phagocytic function and leads to increased production of proinflammatory cytokines	Shears et al., 2020
9-week-old male Sprague-Dawley (Harlan Hsd:SD) rats	control group (n=7), treatment group (n=8)	rats were randomly divided into two groups: vehicle (0.9% saline+0.02% Tween 80; n=7) and DEP (SRM2975,0.2 mg/ml in 0.9% saline+0.02% Tween 80; n=8)	inhalation after aerosolize	DEP		8.75 mg/m^3	cardiac dysfunction	Through the activation of the AHR, exposure to DEP induces ECM remodeling by shifting the collagenous ECM balance toward degradation, leading to loss of collagen, and ultimately causing ventricular dilation and dysfunction	Bradley et al., 2013

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
A total of 32 female BALB/c mice, 6 weeks of age	4 group (n = 8 for each group)	4-week and DEP 8-week groups were exposed to 100 μg/m^3 DEPs for 1 hour a day for 5 days a week. Control mice were exposed to saline solution for durations identical to those in the treated mice	inhalation exposure	DEP		100 μg/m^3	nasal inflammatory	With a 4-week exposure, 173 genes were upregulated and 105 genes were downregulated. With an 8-week exposure, 371 genes were upregulated and 338 genes were downregulated. Interestingly, longer exposure to DEP triggered larger scale differential gene expression than shorter exposure	Kim et al., 2020a
6-week-old female BALB/c mice	control group (n=8), treatment group (n=8)	The DEP-treatment group (n = 8) inhaled 100 μg/m^3 DEPs via an ultrasonic nebulizer for one hour per day, five days a week, for four weeks, with an output of 1 mL/min and 1 to 5 μm particle size. The control group (n = 8) was treated with saline solution under the same conditions as the experimental group	inhalation after aerosolize	DEP		100 μg/m^3	olfaction diseases	Exposure to DEPs decreases olfactory sensitivity in mice. 397 up-regulated genes and 134 down-regulated genes were differentially expressed by DEP exposure in the mouse nasal tissue	Kim et al., 2020b

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
adult male TLR4-deficient ($TLR4^{-/-}$) and female TLR4-heterozygous ($TLR4^{-/-}$; C57BL/6 background)	DEP group, n=9; VEH group, n=7	Beginning on the morning of E2, time-mated females were treated with DEP delivered by oropharyngeal aspiration. Females received 50 µg DEP suspended in 50 µL vehicle (DEP group, n=9) or 50µL vehicle (VEH group, n=7) on E2, E5, E8, E12 and E16	inhalation exposure	DEP		50 µg DEP suspended in 50 µl vehicle	neurodevelopmental disorders	1.DEP exposure increased inflammatory cytokine protein and altered the morphology of microglia. 2.DEP-induced activation of microglia is dependent on TLR4	Bolton et al., 2017
male TO mice (HsdOla: TO, Harlan, UK)		Four weeks following induction of diabetes, the animals were intratracheally instilled (i.t.) with DEP (0.4 mg/kg) or saline, and several cardiovascular endpoints were measured 24 h thereafter	intratracheal instillation	DEP		0.4 mg/kg	diabetes	DEP caused leukocytosis and a significant increase in plasma C-reactive protein and 8-isoprostane concentrations in diabetic mice	Nemmar et al., 2013
pregnant ICR mice obtained from SLC Co. (Shizuoka, Japan)	n=30	DEP suspensions (200 µg/kg body weight) were injected subcutaneously into 15 pregnant mice on gestation days 6, 9, 12, 15 and 18. The total dose of DEPs was adjusted to approximately 1 mg/kg body weight	inject subcutaneously	DEP		1 mg/kg body weight	spatial learning and memory ability reduction	DEP-exposed mice exhibited decreased hippocampal NR2A expression	Yokota et al., 2015

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
pregnant ICR mice obtained from SLC Co. (Shizuoka, Japan)	n=30	DEP suspensions (200 µg/kg body weight) were injected subcutaneously into 15 pregnant mice on gestation days 6, 9, 12, 15 and 18. The total dose of DEPs was adjusted to approximately 1 mg/kg body weight	intratracheal instillation	DEP		1 mg/kg body weight	anxiogenic effects	prenatal DEP exposure increases anxiety like behavior in male offspring later in life, and increases 5-HT levels in the DRN via chronically increased 5-HT neuronal activity	Yokota et al., 2016
male Sprague Dawley rats (2 to 4 mo old)		Cardiomyocytes were treated for 1 h with DEP; diluted to 0.25,0.50, 1.0, and 25 µg/mL; and filtered through 5 µm filter paper to remove aggregates. DEP was added in culture at various concentrations to the apical chamber of the polarized lung epithelial cells	add to culture medium	DEP		Cardiomyocytes: 0.25,0.50, 1.0, and 25µg/mL lung epithelial cell: 1mg/mL	cardiomyocyte and lung function damage	Direct treatment of cardiomyocytes with DEP caused contractile dysfunction and alterations in calcium handling	Gorr et al., 2015
3-week-old wild-type Balb/c mice		Lungs of mice were exposed by pharyngeal aspiration nine times over 3 weeks to DEP at 1.2 or 6.0 mg/kg body weight, HDM at 0.8, 1.2 or 6.0 mg/kg of DEP in combination with HDM, or the same volume (50 µL) of 0.9% sterile saline	intratracheal instillation	DEP		0.8, 1.2 or 6.0 mg/kg of DEP	allergic asthma	1.2 mg/kg of DEP caused no detectable lung inflammation, but 6.0 mg/kg of DEP induced neutrophilic influx	Acciani et al., 2012
adult male Sprague Dawley rats	n=12	In 5 rats (DEP group), DEP dissolved in 0.1 mL PBS was given via endotracheal intubation with the concentration of 100, 200 and 400µg/mL. The lung burden of DEP per rat was 10, 20 and 40 µg at the DEP concentration of 100, 200 and 400 µg/ml, respectively. The same amount of PBS and combined DEP and NAC (5 mmol/L) were given in 4 rats (control group) and in 3 rats (NAC + DEP group)	intratracheal instillation	DEP		100, 200 and 400 µg/mL PBS	cardiovascular impairment	DEP increases APD, spontaneous triggered activity and arrhythmia; DEP increases ROS generation in cardiomyocytes.	Kim et al., 2012

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
female C57BL/6Tac mice	492	mice were exposed to a single dose of collected particles of either 6 µg, 18µg or 54 µg per mouse by intratracheal instillation (6-8 mice per dose per exposure) First cohort was exposed to RME13 and CB, second cohort was exposed to DEP13, third cohort was exposed to DEP9.7, fourth cohort was exposed to DEP17, and fifth cohort was exposed to HVO13. For each exposure cohort there were four vehicle control mice	intratracheal instillation	DEP	The OC and EC of the extracted particles were measured with a thermal-optical carbon analyzer (Sunset Laboratory Inc.), using the EUSAAR_2 protocol	6µg, 18µg or 54µg per mouse	Inflammation, liver and lung damage	1. Exposure to diesel exhaust particulates caused lung damage in mice and the accumulation of particulates in the lungs of mice. 2. Diesel exhaust gas exposure caused inflammation and acute phase reactions in mice. 3. Cause an increase in SAA3 mRNA levels. 4. Cause DNA damage and ROS generation	Bendtsen et al., 2020
adult male Sprague-Dawley rats (250-350 g)		DEP (1.0 µg/ml) was dissolved by thorough sonication in the contracting buffer. For function studies, isolated ventricular myocytes were divided into eight groups. Control (Ctrl): cells were cultured overnight in standard medium; HG: cells were cultured overnight in a media with a high concentration of glucose (25.5 mM) (i.e. diabetic-like media); DEP: cells were cultured overnight with DEP (0.1 µg/mL); HG+DEP: cells were cultured overnight in the presence of both HG (25.5 mM) and DEP (0.1 mg/mL); The next four groups were similar to the initial four groups except that all were treated with antioxidants	add to culture medium	DEP		1.0 µg/mL	cardiomyocyte dysfunction	DEP and high glucose alter the local chemical, mechanical, and/or electrical environment through the activation of ROS generators, potentially including the mitochondria and NADPH oxidases	Zuo et al., 2011

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
adult male and female C57BL/6 mice		Females received 50 µg DEP suspended in 50 µl vehicle (DEP group; n= 5 dams from cohort 1, n= 3 dams from cohort 2) or VEH group; n= 7 dams from cohort 1, n= 5 dams from cohort 2) every 3 days for a total of 6 doses, as a model of intermittent exposure	inhalation exposure	DEP		50 µg DEP suspended in 50 µL vehicle	metabolic and neuroinflam-mation	DEP male offspring mounted an exaggerated peripheral IL-1bresponse to an LPS challenge at postnatal day (P)30, whereas their central IL-1bresponse did not differ from VEH male offspring, which is suggestive of macrophage priming due to prenatal DEP exposure	Bolton et al., 2014
8-week-old B6C3F1 male and female mice (Jackson laboratories; Bar Harbor, ME)	DEP-treated group (n=24) and the controls (n=24)	Mice were exposed at a concentration of 1 mg/m^3 or filtered air for 4 hr/day, (5 days/week) from the beginning of gestation until the first week after birth	inhalation exposure	DEP		1 mg/m^3	locomotor activity and repetitive behaviors increase	exposures to DEP during gestation and early life may impair brain development leading to autism-like symptoms	Rajamani et al., 2013
pregnant female NMRI mice	n=30	Three groups of pregnant mice were exposed to 350-400 µg $DEPs/m^3$ for 2, 4 and 6 h daily in a closed system room	inhalation exposure	DEP		350-400 µg $DEPs/m^3$	anxiety, spatial memory disorders	DEPs exposed mice exhibited decreased hippocampal NR2A and NR3B expression	Ehsanifar et al., 2019
maleTO mice (25-30 g, HsdOla:TO, Harlan, UK)		DEP (0.5 mg/kg in 150 µL saline) was intratracheally (i.t.) instilled every 4th day for 4 weeks (7 i.t. instillation). Four days following the last exposure to either DEP or saline (control), various renal endpoints were measured	intratracheal instillation	DEP		0.5 mg/kg in 150 µl saline	renal oxidative stress, inflammation and DNA damage	The antioxidant calase was significantly decreased in adenine	Nemmar et al., 2016

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
adult male Wistar rats (200-250 g; Charles River, Margate, UK)		DEP (0.5 mg) or an equivalent volume (0.5 mL) of 0.9% saline was administered by intra-tracheal instillation under light anaesthesia an additional group of instilled saline	intratracheal instillation	DEP		1 mg/mL	susceptibility to myocardial ischemia/ reperfusion injury	Diesel exhaust particulate increases vulnerability to ischemia-associated arrhythmia and reperfusion injury. These effects are mediated through activation of pulmonary TRPV1, the sympathetic nervous system and locally generated oxidative stress	Robertson et al., 2014
male ICR mice (6 weeks old, 29-33 g)		ICR mice were divided into six experimental groups which received intratracheal inoculation of vehicle, LPS alone (2.5 mg/kg), organic chemicals in DEP (DEP-OC: 4 mg/kg) extracted with dichloromethane, residual carbonaceous nuclei after the extraction (washed DEP: 4 mg/kg), DEP-OC+LPS, or washed DEP+LPS	intratracheal instillation	DEP		4 mg/kg body weight	lung inflammation	Exposure to washed DEP enhances circulatory level of chemokines during lung inflammation	Arimoto et al., 2007
male ICR mice 6 to 7 weeks of age and weighing 29 to 33 g (Japan Clea Co., Tokyo, Japan)		ICR mice were divided into six experimental groups which received intratracheal inoculation of vehicle, LPS (lipopolysaccharide) alone (2.5 mg/kg), organic chemicals in DEP (DEP-OC: 4 mg/kg) extracted with dichloromethane, residual carbonaceous nuclei after the extraction (washed DEP: 5 mg/kg), DEP-OC+LPS, or washed DEP+LPS	intratracheal instillation	DEP		5 mg/kg	lung inflammation	Both DEP components exacerbate vascular permeability and the increased fibrinogen, and E-selectin levels induced by LPS	Inoue et al., 2006

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
5-week old female C57BL/KsJ-db/db Jcl (db/db) mice and C57BL/KsJ-db/+m Jcl(db/+m) mice	control group (n = 16), treatment group (n=24)	Db/db mice and db/+m mice were randomly divided into four experimental groups: the db/db-vehicle group, the db/db-DEP group, the db/+m-vehicle group, and the db/+m-DEP group. The vehicle groups intratracheally received 100 μL of phosphate-buffered saline at pH 7.4 containing 0.05% Tween-80 every two weeks. The DEP groups intratracheally received 100 μg DEP in the same vehicle every two weeks	intratracheal instillation	DEP		1 μg/μL	fatty change of the liver	Pulmonary exposure to DEP, particulate air pollutants, enhances fatty change in the livers of diabetic obese mice. The enhancement is concomitant with oxidative stress in the liver	Tomaru et al., 2007
female B6C3F1 mice (6-8 week old)		The mice were exposed to 1, 5, or 15 mg DEP/kg of body weight 3 times in a period of 2 weeks, i.e., Monday and Friday of the first week and Wednesday of the second week, or 6 times over 4 weeks. For the AFC experiment, 2 additional doses,0.05 and 0.2 mg/kg, were included. The volume of instillation was 25 μL/10 g of body weight. Control animals received the same volume of sterile saline	inhalation exposure	DEP		1, 5, or 15 mg DEP/kg of body weight	immune response	Exposure to DEP also resulted in a significant decrease in the absolute numbers and the percentages of total spleen cells for total, $CD4^+$ and $CD8^+$ T cells	Yang et al., 2003
male BALB/c ByJ mice		BALB/c ByJ mice were randomly divided into four experimental groups. Two groups received nasal instillations of saline and the other two groups received 3 mg/mL SHE during 5 days per week for 3 weeks. One group in each pair also received 150 μg of DEP in the same instillations 3 days per week	intratracheal instillation	DEP		150 μg/20μL saline	inducing asthma to low doses of allergens	Inhalation of DEP increased neutrophils and decreased total monocytes	De Homde-deu et al., 2021

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
cardiac myocytes of wistar rats		Each concentration of DEPE (5 μg dry DEP was suspended in 5 ml PBS containing 0.05% Tween 80) was dissolved in serum-free D-MEM/F-12 (10% of volume), and incubated in humidified 5% CO_2- 95% air at 37℃ with cells. For experiments of chronic exposure to DEPE, cells were incubated for 24 or 48 h. For experiments of short-time exposure of DEPE, cells were incubated for 1, 2, 4 or 8 h and then medium containing DEPE was replaced by normal serum-free medium and incubated for further 24 h	DEP was dissolved in PBS, add to the medium	DEP		0, 20, 40, 60, 80, 100 μg/mL	cause cardiac myocytes death	Cardiac myocytes showed 50% damage with 50 μg/mL DEPE exposure for 24 h or 4 h and incubated with normal medium for a further 24 h. DEPE-induced cytotoxicity was markedly reduced by SOD activity, catalase activity and MPG levels	Okayama et al., 2006
male BALB/c mice	control group (n = 30), treatment group (n=30)	Male BALB/c mice were divided into two groups: (a) Saline: nasal instillation of saline (n = 30); and (b) DEP: nasal instillation of 30 μg of DEP/10 μL of saline (n = 30). Nasal instillations were performed 5 days a week, over 30 and 60 days	intratracheal instillation	DEP		30 μg of DEP/10 μL of saline	respiratory tract inflammation	A low-dose of DEP over 60 days induces respiratory tract inflammation The expression of Muc5ac mRNA increased with 60 days of DEP exposure	Yoshizaki et al., 2010

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male Brown Norway rats [BN/CrlBR] weighing 200 -250 g		Rats were exposed to either filtered air or DEP (20.62±1.31 mg/m^3) for 4 h/day for 5 consecutive days using a nose-only directed flow exposure unit followed by intratracheal inoculation with 100,000 Listeria at 2 h after the last DEP exposure	inhalation exposure	DEP	DEP concentrations in the exposure unit were monitored by both gravimetric sampling of dust collected on a polycarbonate membrane filter (37mm,0.45µm, Poretics Corporation, Livermore, CA) at a sampling rate of 1 l/min and a Grimm Model 1.108 portable dust monitor (GRIMM Technologies, Inc., Douglasville, GA)	20.62±1.31 mg/m^3	suppression of immune response	DEP inhibited AM production of IL- 1β, TNF-α and IL-12 but enhanced Listeria-induced AM production of IL-10, which has been shown to prolong the survival of intracellular pathogens such as Listeria	Yin et al., 2004

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male Brown Norway (BN) rats [BN/CrlBR] weighing 200-250 g		rats were exposed to either filtered air or DEP (21.2 ± 2.3 mg/m^3) for 4 h/day for 5 consecutive days using a nose-only directed flow exposure unit Brown Norway rats were exposed to filtered air or DEP by inhalation at a dose of 21.2 ± 2.3 mg/m^3, 4 h /day for 5 days, and intratracheally instilled with saline or 100,000 Listeria monocytogenes (Listeria) 7 days after the final DEP exposure	inhalation exposure	DEP	DEP concentrations in the exposure unit were monitored by both gravimetric sampling of dust collected on a polycarbonate membrane filter (37mm,0.45μm, Poretics Corporation, Livermore, CA) at a sampling rate of 1 l/min and a Grimm Model 1.108 portable dust monitor (GRIMM Technologies, Inc., Douglasville, GA)	21.2 ± 2.3 mg/m^3	suppression of immune response	DEP was found to inhibit Listeria-induced production of IL-1band TNF-α, which are responsible for the innate immunity, and IL-12, which initiates the development of T helper (Th)1 responses, but enhance Listeria-induced AM production of IL-10, which prolongs Listeria survival in these phagocytes	Yin et al., 2005
C57 mice (10-week)		C57 mice were received a tail vein injection of DEP at high dose (0.5 μg/g of body weight) or low dose (0.25 μg/g of body weight) (7.5 or 15 μg DEP suspended in 200 μl saline), and sacrificed at 1-h post-injection. The dose was calculated by assuming a 3-day exposure at 125 (DEP-low) or 250 μg/m^3 (DEP-high), and 2% of DEP was translocated to the blood stream. 200 μl saline injection was used as the control	inject into the tail vein	DEP		high dose (0.5 μg/g of body weight) low dose (0.25 μg/g of body weight)	inflammation and vascular effects	Acute systemic DEP exposure caused a significant increase in TNF-α, peripheral neutrophil and band cell counts	Bai and Van Eeden, 2013

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male BALB/cOla Hsd mice (7-8 weeks)	n=18	3 mice for each experimental group were intratracheally instilled, and the experiments were replicated twice, for a total of 6 sham, 6 biomass burning-derived (BB) particles-treated, and 6 DEP-treated mice, Intratracheal instillations with 50 µg of BB or DEP in 100 µL of isotonic saline solution or with 100 µL of isotonic saline solution were achieved by means of a MicroSprayer®Aerosolizer system	intratracheal instillation	DEP		0.5 µg/µL saline	oxidative stress and Inflammation in brain	DEP exposures induced inflammatory pathways in mouse brain and DEP showed strong oxidative stress	Milani et al., 2020
9-week-old C57BL/6J male mice	the mice were randomly divided into 4 exposure groups with 15 mice in each group	A cannula (18 Gauge, Becton Dickinson, USA) was inserted via the mouth into the trachea. The exposed samples PBS, DEP, water-soluble DEP (WS), water-insoluble DEP (WIS) exposure groups with 20 µL each were intratracheally instilled via a sterile syringe and followed by 150 µL air bolus	intratracheal instillation	DEP		2 mg/mL	blood coagulation function	BT, FIB, APTT and PT altered or showed a lower tendency after DEP, WS and WIS exposure	Lei et al., 2021
adult male Sprague-Dawley rats (250-300 g)	control group (n = 4), treatment group (n=12)	In 12 rats (DEP group), DEP dissolved in 0.1 mL PBS was given via endotracheal intubation at the concentrations of 200 µg/mL for 1 h. The lung burden of DEP per rat was 20 µg at the DEP concentrations of 200 µg/mL, respectively. For the control (n=4), the same amounts of PBS were given via endotracheal intubation, respectively	intratracheal instillation	DEP		200 mg/L	retinal thickening	Among the retinal structure, inner plexiform, inner and outer nuclear and rod/cone cell layers were significantly thickened after DEP exposure	Kim et al., 2016

Animal type	Sample size	Exposed method	Exposed way	Pollutant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
female white rats (Rattus novergicus)	control group (n = 10), treatment 1 group (n=12), treatment 2 group (n=12)	control group without soot particulate exposure- (n=10); treatment 1 group (n=2) exposed by soot particulate with the concentration of 532 mg/m^3 an hour each day for 30 days; treatment 2 group exposed by soot particulate with the concentration of 1064 mg/m^3 an hour each day for 30 days- (n=12)	inhalation exposure	soot		high dose (1064 mg/m^3) low dose (532 mg/m^3)	cardiovascular system disruption	The exposure to soot particulates significantly increased MAPK expression in experimental rats	Yutanto, 2020
female rats (Rattus novergicus)	control group (n=10), treatment 1 group (n=12), treatment 2 group (n=12)	consisted of 3 groups: control group (n=10), without soot particulate exposure; treatment 1 group (n=12), exposed by soot particulate with the concentration of 532 mg/m^3 an hour each day for 30 days; treatment 2 group (n=12), exposed by soot particulate with the concentration of 1064 mg/m^3 an hour each day for 30 days	inhalation exposure	soot particulate		high dose (1,064 mg/m^3) low dose (532 mg/m^3)	cardiovascular system disruption	The exposure to soot particles increased VCAM-1 expression significantly in laboratory animals	Leonard and Aminuddin, 2020
male BALB/c ByJ mice (~25 gr, 6 weeks old)		In the same instillation two experimental groups received 150 mg of DEP in each of the three challenges. The experimental groups were: SS, saline-sensitized and saline-challenged; DEP, saline-sensitized and DEP-challenged; AP, AP-sensitized and AP-challenged; and AP-DEP, AP-sensitized and challenged with a mixture of AP and DEP	intratracheal instillation	DEP		1 mg/mL, SRM-2975	asthma	Mice exposed to DEP alone showed increased levels of neutrophils and NKs, reduced numbers of monocytes and alveolar macrophages, and increased levels of CD11+Ly6C-DCs.	De Homdedeu et al., 2020

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
12-week-old male Wistar rats	n= 5-8 in each group	The tail was disinfected with ethanol, and 150µL of vehicle or dose 0.02 mg DEP/kg corresponding to about 4.4 µg DEP/animal were injected into the tail vein of rats. After 6 h, 18 h, 48 h and 168 h after systemically injecting rats with DEP, conducted SBP measurements, blood cell counting, plasma analysis and histopathology of several major organs	injected into the tail vein	DEP		0.02 mg/kg	systolic blood pressure (SBP), systemic inflammation, oxidative status, and morphological alterations in lungs, heart, liver and kidneys in wistar rats	The platelet numbers were significantly decreased 6 h following the systemic administration of DEP. The IL-6 concentrations in plasma were increased at 6 h and 18 h. Similarly, superoxide dismutase activity was significantly increased at 6 and 18 h following DEP exposure	Nemmar et al., 2010a
male ddy mice, (slc: ddy) weighing 39.6-46.0 g		All three substances, DEP, soluble-component and residual particle-component were dispersed at concentrations of 4 and 8 mg/mL using a sonicator (UH-50; SMT, Tokyo, Japan) in ice-cold PBS containing 0.05% Tween 80	intratracheal instillation	DEP		4 mg/mL, 8 mg/mL	cardiovascular function and oxidative stress	Inhalation of the particle component of DEP enhances myocardial oxidative stress, and promote excessive production of cytokines, IL-6 and G-CSF	Yokota et al., 2008

Animal type	Sample size	Exposed method	Exposed way	Pollu-tant	Pollutant measurement	Concentration	Outcome	Finding/exposure-response relationship	Reference
male WKY rats (10-12 weeks of age)		rats (n = 20 /group) were exposed for 5 hr/day, 1 day/week for 16 weeks, to either ozone or DEP or to a combination of ozone + DEP, the desired chamber concentrations were 0.5 ppm ozone and 2.0 mg/m^3 DEP	inhalation exposure	DEP		2.0 mg/m^3	vascular and cardiac impairments	Exposed to DEP alone for 16 weeks, biomarkers of vascular impairments in the aorta elevate, with the loss of phospholipid fatty acids in myocardial mitochondria. There is a possible role of oxidized lipids and protein through LOX-1 and/or RAGE signaling	Kodavanti et al., 2011
male Wistar rats (Charles River Laboratories, France) and male SHR (Janvier, France)	n=22	Rats of both strains were divided into 2 groups. Group 1 received PBS at pH 7.2 containing 0.05% Tween 80, and group 2 received suspended DEP (0.8 mg) in the same vehicle, 3 times a week for 4 weeks. The rats were all 18-20 weeks old at the time of the final DEP administration and were euthanized 4 hours after the last instillation	inhalation after aerosolize	DEP		100 μg/mL	vascular oxidative stress and hypertension	Although in vitro exposure to DEP produces a vascular oxidative stress, repeated in vivo exposures to DEP only impair vascular function in SHR, via an upregulation of p22phox	Labranche et al., 2012

4.2.2.2 Diesel Exhaust Particle

In terms of pollutant sources, most of the pollutants used in animal experimental research were diesel exhaust particles (DEP), DEP were mainly from some standard samples a reference for the chemical and toxicological constitutes of the standardized DEPs such as SRM-2975 (NIST) (Miller et al., 2013) or DEP were collected from standard diesel engines (Yokota et al., 2016).

Through literature, mice are mainly used to do acute, subacute and chronic toxicity tests, rats are used to simulate some disease models such as Cardiovascular diseases (Nemmar et al., 2009), diabetes (Nemmar et al., 2013) and so on, in the aspect of experimental design.

During the experiment, a single variable was often controlled for control experiments: mice (or rats) were randomly divided into control group and experimental group, in which mice (or rats) in the control group inhaled air that DEP has purified, while mice (or rats) in the experimental group were exposed to DEP. The main ways of DEP exposure were intratracheal instillation (Bendtsen et al., 2020), oral gavage (Danielsen et al., 2008), feed (Muller et al., 2004), intraperitoneal injection (Huang et al., 2010), inhalation after aerosolize (Bradley et al., 2013), inhalation exposure (Greve et al., 2020a) and injected into the tail vein (Nemmar and Inuwa, 2008).

Through literature review, it is found that DEP exposure mainly caused inflammatory reaction (Jeong et al., 2021), lung injury (Bendtsen et al., 2020), vascular injury and heart disease (Nemmar et al., 2012) in mice (rats) in pathology, DEP exposure mainly caused the increase of ROS (reactive oxygen species) in the cells (Bendtsen et al., 2020), the level of inflammatory cytokines (Kim et al., 2020a) in mouse (rat) cells, DNA damage (Danielsen et al., 2008) and some changes in mRNA levels after translation (Hougaard et al., 2008). At the same time, some intracellular signal transduction pathways were activated and the expression levels of related proteins were changed (Greve et al., 2020a) in toxicology.

4.2.3 Human Clinical Studies

In addition to animal experiment articles, the human clinical study is the important part of the study of toxicity mechanism. As the same of animal experiment parts, there aren't many studies. We still select the same four authoritative databases to search for human clinical studies. Searching time was 1900 to September 19th, 2021. The searching formula and process were shown in Figure 4-6.

We have listed important information, such as basic information of the paper, exposed way, outcomes and so on. The extracted data see Table 4-4.

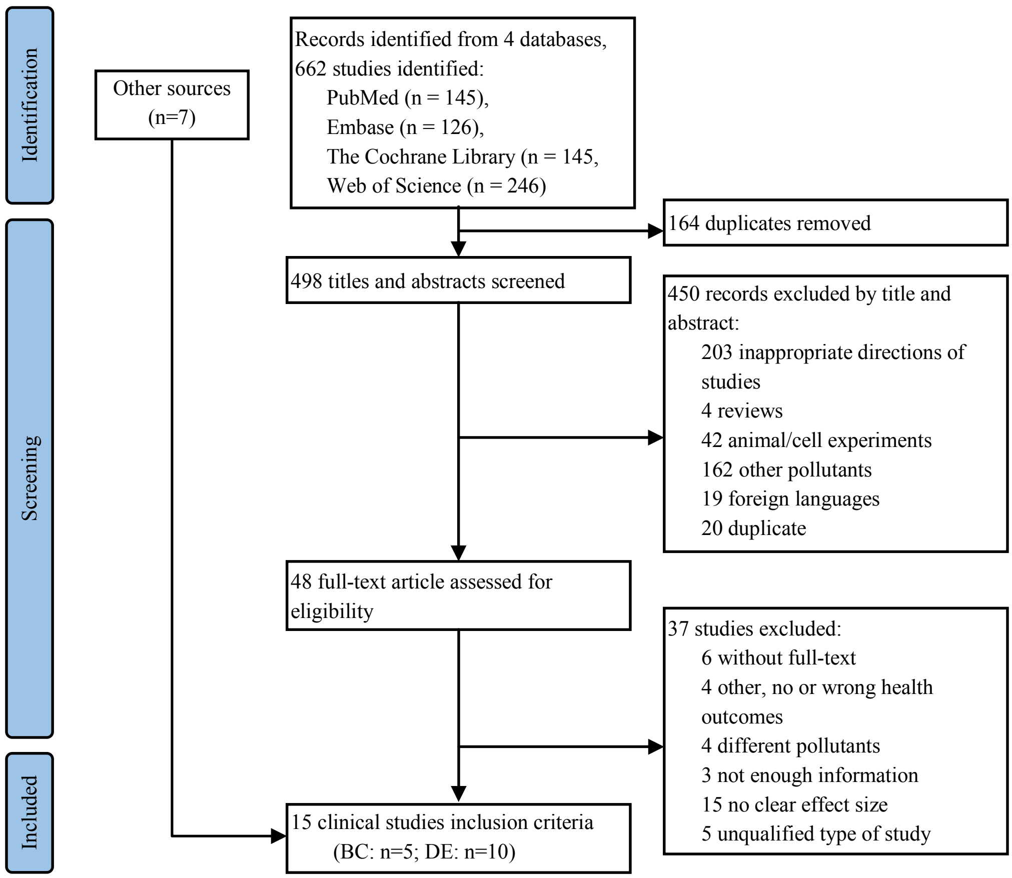


Figure 4-6 Searching process of human clinical studies

BC or any other alternative (optical) methods such as reflection on filters are seldom assessed (or reported) in toxicity studies on ambient air, either in controlled human clinical studies. For this reason, this chapter will also consider exposure to DE or DEP. From the search results, human clinical studies are mainly about cardiopulmonary function and vascular function after exposure to BC, UFP, DEP and DE. And the countries where the subjects are located are all in Europe.

Table 4-4 The extracted data of human clinical studies

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
never-smoker subjects, with stable type 2 diabetes but otherwise healthy (9 men and 10 women)	30-60	19		USA	Inhaled either filtered air (0-10 particles/ cm^3) or EC UFP by mouthpiece, for 2 hours at rest, in a double-blind, randomized, crossover study design. A digital 12-lead electrocardiogram (ECG) was recorded continuously for 48 hours, beginning 1 hour prior to exposure	ultrafine particles consisting of EC exposure for 2 hours	particle number (condensation particle counters, model 3220a; TSI, Inc., St. Paul MN); size distributions (Scanning Mobility Particle Sizer, model 3071; TSI, Inc.)	EC UFP (~107 particles/cm^3, ~50 μg/m^3, count median diameter 32 nm)	Inhalation of EC ultrafine particles alters heart rate and heart rate variability in people with type 2 diabetes. Our findings suggest that effects may occur and persist hours after a single 2-hour exposure	Analysis of 5-minute segments of the ECG during quiet rest showed reduced high-frequency heart rate variability with UFP relative to air exposure (p=0.014), paralleled by non-significant reductions in time-domain heart rate variability parameters. In the analysis of longer durations of the ECG, we found that UFP exposure increased the heart rate relative to air exposure. During the 21 to 45 hours interval after exposure, the average heart rate increased approximately 8 beats per minute with UFP, compared to 5 beats per minute with air (p=0.045). There were no UFP effects on cardiac rhythm or repolarization	Vora et al., 2014

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
self-reported healthy, non-asthmatic, without prescribed medication, non-smoking and non-pregnant participants living in the Copenhagen region	21-71	83	2017/5-2017/11 (without July)	Denmark	The study design was a crossover, repeated measures. On well-characterized real-life DE exposure in humans. In the present study, 29 healthy volunteers were exposed to DE while sitting as passengers in diesel-powered trains. Exposure in electric trains was used as control scenario. Each train scenario consisted of three consecutive days (6 h/day) ending with biomarker samplings	UFP, BC exposure for 6h/d and 3 consecutive days	Nano Tracer and Disc Mini portable devices UFP (Disc Mini and Nano Tracer), BC (Micro Aeth AE51)	UFP: (1.2-1.8) $\times 10^5$ particles/cm^3 BC: 8.3 μg/m^3 $PM_{2.5}$:36 μg/m^3 DE:300 μg/m^3	Exposure to DE inside diesel-powered trains for 3 days was associated with reduced lung function and systemic effects in terms of altered heart rate variability and increased levels of DNA strand breaks in PBMCs compared with electric trains	Exposure to DE was associated with reduced lung function and increased levels of DNA strand breaks in peripheral blood mononuclear cells (PBMCs), whereas there were unaltered levels of oxidatively damaged DNA, soluble cell adhesion molecules, acute phase proteins in blood and urinary excretion of metabolites of polycyclic aromatic hydrocarbons. And the microvascular function was unaltered. An increase in the low frequency of heart rate variability measures was observed, whereas time-domain measures were unaltered	Andersen et al., 2019

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
15 healthy and 15 asthmatics	18-45	30	before 1992	USA	Volunteer subjects were studied in groups of one to four. Downey, California Every subject was exposed on four separate occasions, usually separated by 7-day intervals. The four separate exposures-clean air (control), carbon aerosol at a nominal concentration of 200 μg/m^3, H_2SO_4 aerosol at a nominal concentration of 100 μg/m^3, and acid and carbon combined—were presented in random order under double-blind conditions. Each subject was exposed for 60-min on any given day, with alternating 10-min periods of exercise and rest	carbon aerosol a exposure for 1 hour and 7 consecutive days	ambient particulate carbon carbon aerosol: on-line particle laser spectrometer system (PMS 100-HV CSASP)	carbon aerosol: 200 μg/m^3 H_2SO_4 aerosol: 100 μg/m^3 (acid and carbon)	Coexisting carbon aerosol did not increase respiratory irritancy of H_2SO_4, in most healthy and asthmatic subjects exposed for 1 hour under simulated "worst-case" ambient conditions	Group data showed no more than small equivocal effects of any exposure on any health measure. One individual's responses were consistent with a clinically significant excess airway constriction from H_2SO_4 plus carbon, and 2-3 others showed slight excess responses to the combined pollutants, but all these observations might have reflected chance variations	Anderson et al., 1992

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
no smoking history, no cardiopulmonary disease history	40-75	77		USA	Using a 3-period crossover design, participants were randomized to attend three 2-hour-long exposure sessions separated by 1-week washout periods. Each participant was exposed to high, medium, and low TRAP concentrations in a room near an interstate highway. Particle number concentrations, BC concentrations, and temperature were monitored continuously. Systolic BP (SBP), diastolic BP and heart rate were measured every 10 minutes. Outcomes were analyzed with a linear mixed model	traffic-related air pollution (TRAP): including BC and UFP exposure for 2-3 hours	BC: aethalometer (Magee Scientific; model AE16) at 1-minute resolution PNC (Particle number concentrations): particle counter (TSI, Inc; model 3873; d50= 7 nm) at 1-second resolution	1.low exposure mean particle number and BC concentrations, 2500 particles/cm^3 and 149 ng/m^3. 2.medium exposure mean particle number and BC concentrations, 11,000 particles/cm^3 and 409 ng/m^3. 3.high exposure mean particle number and BC concentrations, 30000 particles/cm^3 and 826 ng/m^3	Reducing indoor concentrations of TRAP was effective in preventing acute increases in systolic blood pressure (SBP)	Using a 3-period crossover design, 77 participants were randomized to attend three 2-hour-long exposure sessions separated by 1-week washout periods. Each participant was exposed to high, medium, and low TRAP concentrations in a room near an interstate highway. Particle number concentrations, BC concentrations, and temperature were monitored continuously. SBP, diastolic BP, and heart rate were measured every 10 minutes. Outcomes were analyzed with a linear mixed model	Hudda et al., 2021

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
31 with mild and 29 with moderate asthma	adults	60	2003-2005 (between November and March) confined to weekdays and avoid pollen seasons and rainy days	UK	In this randomized, crossover study, participants walked for 2 hours (10:30 a.m. to 12:30 p.m.) along the western end of Oxford Street, participants walked about 6 km during each exposure, at a steady pace on predefined paths, resting for 15 minutes every half hour. Exposure sessions, separated by more than 3 weeks, were confined to weekdays between November and March to avoid pollen seasons; rainy days were also avoided. Equal numbers of participants were randomly assigned to each exposure sequence	EC and UFC: including EB exposure for 2 hours	real-time condensation particle counter (Model 3007, TSI)	Oxford Street: Elemental carbon 7.5 μg/m^3(range: 3.9-16). Hyde Park: Elemental carbon: 3 μg/m^3 (range: 0.4-6.7)	The epidemiologic evidence that associates the degree of traffic exposure with lung function in asthma	Walking for 2 hours on Oxford Street induced asymptomatic but consistent reductions in the forced expiratory volume in 1 second (FEV1) (up to 6.1%) and forced vital capacity (FVC) (up to 5.4%) that were significantly larger than the reductions in FEV1 and FVC after exposure in Hyde Park (p=0.04 and p=0.01, respectively, for the overall effect of exposure, and p<0.005 at some time points). These changes were accompanied by increases in biomarkers of neutrophilic inflammation (sputum myeloperoxidase, 4.24 ng per milliliter after exposure in Hyde Park vs. 24.5 ng per milliliter after exposure on Oxford Street; p=0.05) and airway acidification (maximum decrease in pH,0.04% after exposure in Hyde Park and 1.9% after exposure on Oxford Street; p=0.003)	McCreanor et al., 2007

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
130 nurses from two hospitals in Belgium, of which 56 (56%) were assigned at random to this study. healthy adults (92% women). All participants reported to be free of clinical cardiovascular diseases and diabetes	mean age: 40.7	54	2013/4-2013/5	Belgium	A panel study design was used to investigate the association between measures of arterial stiffness and short-term exposure to BC. BC exposure was monitored during one workweek. Functional and structural properties of the carotid artery were examined ultrasonographically on two separate days. The effect of different short-term personal BC exposure windows (1,2,4,6,8,24 and 48 h before the ultrasound examination) on carotid artery stiffness was estimated using mixed models while adjusting for other known correlates of arterial stiffness	BC exposure for 2 hours and 7 consecutive days	micro-aethalo meter the portable Micro Aeth® Model AE51 BC aerosol monitor (Aeth Labs, San Francisco, CA, USA)	BC: 599.8-728.9 ng/m^3	Short-term elevations in personal BC exposure, even within hours, are associated with increased arterial stiffness. This response may reflect a pathway by which air pollution triggers cardiovascular events	Median personal BC exposures within the same day ranged from599.8 to 728.9 ng/m^3 and were associated with carotid arterial stiffness measures. Young's elastic modulus and pulse wave velocity, both measures of stiffness, were positively associated with BC exposure, while the distensibility and compliance coefficient, measures of elasticity, were negatively associated with BC exposure. The strongest associations were observed with BC exposure 8 h before the clinical examination. For each 100 ng/m^3 increase in exposure within this time window, Young's elastic modulus increased by 2.38% (95% Cl: 0.81 to 3.97; p=0.0033), while the distensibility coefficient decreased by 2.27% (95% Cl: -3.62 to -0.92; p=0.0008)	Provost et al., 2016

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
non-smokers, health females	19-46	26		Canada	26 non-smokers were studied under each of three experimental conditions (filtered air with placebo, diesel exhaust with placebo, and diesel exhaust with N-acetylcysteine) using a randomized, double-blind, crossover design, with a 2-week washout between conditions. Methacholine challenge was performed pre-exposure (baseline airway responsiveness) and post-exposure (effect of exposure)	DE exposure for 2 hours every day and for 6 days	real-time concentration of particles nephelometer	DE:300 g/m^3 (of PM smaller than 2.5 microns)	Anti-oxidant (N-acetylcysteine) supplementation protects against increased airway responsiveness associated with DE inhalation and reduces need for supplement bronchodilators in those with baseline airway hyper-responsiveness. Individuals with variants in genes of oxidative stress metabolism when exposed to DE are protected from increases in airway responsiveness if taking anti-oxidant supplementation	Anti-oxidant supplementation reduced baseline airway responsiveness in hyper-responsive individuals by 20% (p=0.001). In hyper-responsive individuals, airway responsiveness increased 42% following DE compared with FA (p=0.03) and this increase was abrogated with anti-oxidant supplementation (diesel exhaust with N-acetylcysteine vs. filtered air with placebo, p=0.85)	Carlsten et al., 2016

Subject	Age	Sam-ple size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
healthy, nonsmoking volunteers (3males and 7 females)	25-31	10		USA	The study was conducted in a double-blind manner, with randomized exposures to DEP or clean air. Before each exposure, baseline spirometry was conducted, baseline measurements of pulse rate, blood pressure and exhaled CO were made, and blood was taken. Subjects were then exposed at rest to DEP or to air for 2 hours in a challenge chamber. After exposure, spirometry was repeated and pulse, blood pressure, and exhaled CO were measured immediately, and a further sample of blood was taken. Clinical measurements were then repeated half-hourly for 4	DEP(Diesel exhaust particles) exposure for 2 hours	dust analyzer (Version 5.30E; Grimm Labortechnik GmbH, Germany); Las-X spectrometer (Particle Measuring Systems, Boulder, CO)	DEP: 200 mg/m^3	The present study confirms that exposure to ambient concentrations of DEP provokes an inflammatory response in the airways of normal subjects that is characterized by an influx of activated neutrophils accompanied by an increase in exhaled CO levels, indicative of oxidant stress	There were no changes in cardiovascular parameters or lung function following exposure to DEP. Levels of exhaled CO were increased ater exposure to DEP and were maximal at 1 h (air: 2.9 ± 0.2 ppm (mean ± SEM); DEP: 4.4 ± 0.3 ppm; $p<0.001$). There was an increase in sputum neutrophils and myeloperoxidase (MPO) at 4 h after DEP exposure as compared with 4 h after air exposure (neutrophils: 41 6 ± 4% versus 32 ± 4%; MPO: 151 ng/ml versus 115 ng/ml, $p<0.01$), but no change in concentrations of inflammatory markers in peripheral blood	Julia A. et al., 1999

Subject	Age	Sam-ple size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
					hours. At 4 hours, methacholine challenge was performed and an induced sputum sample was collected. Subjects returned at 24 hours after exposure and all measurements other than sputum induction were repeated						
nonsmoking, atopic asthmatics with stable disease	22-57	14	in winter	America	Each subject was exposed to DE [particles with a 50% cut-off aerodynamic diameter of 10 mm (PM_{10}) 300 mg/m^3] and air during 1 hour on two separate occasions. Lung function was measured before and immediately after the exposures. Sputum induction was performed 6 hours, and methacholine inhalation test 24 hours, after each exposure	DE exposure for 1hour	infrared-instrument (Foxboro Co., East Bridgewater, MA, USA); chemilumine-scence instrument, (ECO-Physics CLD 700, Boo Instruments, Stockholm, Sweden)	DE:300 g/m^3	This study indicated that short-term exposure to diesel exhaust, equal to high ambient levels of PM, is associated with adverse effects in asthmatic airways, even in the presence of inhaled corticosteroid therapy	All were hyperresponsive to methacholine. Each subject was exposed to DE [particles with a 50% cut-off aerodynamic diameter of 10 mm (PM_{10}) 300 mg/m^3] and air during 1 h on two separate occasions. Lung function was measured before and immediately after the exposures. Sputum induction was performed 6 h, and methacholine inhalation test 24 h, after each exposure	C. Nordenha et al., 2001

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
healthy volunteers (8 females and 7 males)	21-40	15	2000	Sweden	Fifteen healthy subjects were exposed to DE at an average PM concentration of 270 $\mu g/m^3$ and filtered air for 1 h. Bronchoscopy with endobronchial mucosal biopsy sampling and airway lavage was performed 6 h post-exposure	DE, OC (origan carbon) exposure for 1 hour and at least for 3 weeks		DE:300 g/m^3 (OC/TC (Volvo TD45, 4.5 L four cylinders): 94.5%)	DE generated under urban running conditions increased bronchial adhesion molecule expressions, together with the novel finding of bronchoalveolar eosinophilia, which has not been shown after exposure to DE at idling. Variations in airway inflammatory response to DE generated under diverse running condition may be related to differences in exhaust composition	Fifteen healthy subjects were exposed to DE at an average PM concentration of 270 $\mu g/m^3$ and filtered air for 1 h. Bronchoscopy with endobronchial mucosal biopsy sampling and airway lavage was performed 6 h postexposure	Maria Sehlstedt et al., 2010

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
18 healthy male nonsmokers with normal physical, body mass index: 21.7 ± 0.5 kg/m^2	mean age: 22.2 ± 0.5	18		Belgium	Eighteen subjects were exposed to either nonfiltered ambient air (AA) or DE for 12 either nonfiltered ambient air (AA) or DE for 120-min using a randomized, crossover, double-blinded design with the different exposures occurring at least 1 weak apart. Pulmonary hemodynamic parameters were calculated using echocardiography, which was initiated 2 hours after exposure. Ten subjects performed the dobutamine stress protocol under AA and DE conditions. Pulmonary hemodynamic parameters were calculated using echocardiography, which was initiated 2 hours after exposure. Ten subjects performed the dobutamine stress protocol under AA and DE conditions. Eight subjects performed the exercise in acute hypoxia stress protocol	DE exposure for 2 hours	GRIMM Laser Aerosol Spectrometer 1109 (GRIMM Aerosol Technik, Ainring, Germany)	DE：300 μg $PM_{2.5}/m^3$	Acute exposure to DE increased pulmonary vasomotor tone by decreasing the distensibility of pulmonary resistive vessels at high cardiac output	Effects of DE on PVR, on the coefficient of distensibilty of pulmonary vessels, and on right and left ventricular function were evaluated at rest (*n*=18), during dobutamine stress echocardiography (*n*=10), and during exercise stress echocardiography performed in hypoxia (*n*=8). Serum endothelin-1 and fractional exhaled nitric oxide were also measured. At rest, exposure to DE did not affect PVR. During dobutamine stress, the slope of the mean pulmonary artery pressure-cardiac output relationship increased from 2.8-0.5 mmHg·min·L⩾1in AA to 3.9-0.5 mmHg·min·L⩾1 in DE	Wauters et al., 2015

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
all participants were free of cardiometabolic and respiratory disease, and not pregnant	22-33	19	2015	Columbia	In a double-blind, repeated-measures design, participants with EIB completed four visits: FA-placebo (FA-PLA), FA-SAL, DE-PLA, DE-SAL. After the inhalation of either 400 μg of SAL or PLA, participants sat in the exposure chamber for 60-min, breathing either FA or DE. Participants then cycled for 30-min at 50% of peak work rate while breathing FA or DE. Respiratory responses were assessed via spirometry, work of breathing (WOB), fractional use of ventilatory capacity (V˙E/V˙E, CAP), area under the maximal expiratory flow-volume curve (MEFVAUC), and dyspnea during and following cycling	DE exposure for 1 hour	Diesel on-road emissions were simulated with a 5.5-kW diesel engine Diesel on-road emissions were simulated with a 5.5-kW diesel engine	DE: $PM_{2.5}$ = 300 μg/m^3	The use of SAL prior to moderate-intensity exercise when breathing high levels of DE, does not reduce respiratory function or exercise ventilatory responses for up to 60-min following exercise	Bronchodilation in response to SAL and acute cycling was observed, independent of FA/DE exposure. Specifically, FEV1 was increased by 7.7% (confidence interval (Cl): 7.2%-8.2%; $p <$ 0.01) in response to SAL, and MEFVAUC was increased after cycling by 1.1% (0.9%-1.3%; $p = 0.03$). Despite a significant decrease in total WOB by 6.2 J/min (4.7-7.5 J/min; p=0.049) and a reduction in V˙E/˙VE, CAP by 5.8% (5%-6%, $p < 0.01$) in the SAL exposures, no changes were observed in dyspnea. The DE exposure significantly increased V˙E/˙VE, CAP by 2.4% (0.9%-3.9%; $p < 0.01$), but this did not affect dyspnea	Sarah Koch et al., 2021

Subject	Age	Sam-ple size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
allergen sensitized participants (7 males and 7 females, no smoking history)	23-50	14		Canada	Participants completed this randomized, double-blinded, cross-over, controlled exposure study. each participant underwent four exposures (allergen-alone exposure, De and allergen co-exposure, particle- depleted DE (PDDE) and allergen co-exposure, sham exposure) on different order-randomized dates. serum and bronchoalveolar lavage (Bal) were assayed for pattern recognition molecules, cytokines, chemokines and inflammatory mediators	DE, PDDE (particle-depleted DE) exposure for 2 hours	2.5 kW applied to the diesel generator	mean concentration of $PM_{2.5}$ in DE (292.2 (95% Cl 279.5-304.9) $\mu g/m^3$); mean concentration of $PM_{2.5}$ in PDDE [18.9 (14.4-23.4) $\mu g/m^3$]	Short- term co-exposure to aeroallergen and DE alters immune regulatory proteins in lungs; surfactant levels are dependent on particle depletion	Allergen-alone exposure led to accumulation of surfactant protein D (SPD; p=0.02). co-exposure to allergen and De did not elicit the same increase of SPD as did allergen alone; diesel particulate reduction restored allergen- induced SPD accumulation. soluble receptor for advanced glycation end products was higher with particle reduction than without it. in the systemic circulation, there was a transient increase in SPD and club cell protein 16 (cc16) 4 hours after allergen alone. cc16 was augmented by PDDE, but not DE.% eosinophils in Bal (p<0.005), eotaxin-3 (p<0.0001), interleukin 5 (il-5; p<0.0001) and thymus and activation regulated chemokine (p=0.0001) were each increased in Bal by allergen. il-5, SPD and% eosinophils in Bal were correlated with decreased FEV1	Sarah Koch et al., 2021

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
blinded atopic volunteers	19-49	18	2015	British	volunteers were exposed to filtered air or f diesel exhaust in random fashion. 1 hour post-exposure, diluent-controlled segmental allergen challenge was performed; 2 days later, samples from the challenged segments were obtained by bronchoscopic lavage. Samples were analyzed for markers and modifiers of allergic inflammation (eosinophils, Th2 cytokines) and adaptive immune cell activation. Mixed effects models with ordinal contrasts compared effects of single and combined exposures on these end points.	DE expose for 1 hour	Diesel on-road emissions were simulated with a 5.5-kW diesel engine Diesel on-road emissions were simulated with a 5.5-kW diesel engine	DE: 300 mg $PM_{2.5}/m^3$	Inhalation of diesel exhaust at environmentally relevant concentrations augments allergen-induced allergic inflammation in the lower airways of atopic individuals and the GSTT1 genotype enhances this response. Allergic individuals are a susceptible population to the deleterious airway effects of diesel exhaust	Diesel exhaust augmented the allergen-induced increase in airway eosinophils, interleukin 5 (IL-5) and eosinophil cationic protein (ECP) and the GSTT1 null genotype was significantly associated with the augmented IL-5 response. Diesel exhaust alone also augmented markers of non-allergic inflammation and monocyte chemotactic protein (MCP)-1 and suppressed activity of macrophages and myeloid dendritic cells	Carlsten et al., 2016

Subject	Age	Sample size	Period	Study region	Study design	Pollutant	Pollutant measurement	Concentration	Outcome	Exposure-response relationship	Reference
Adults do not pregnant, and free of cardiometabolic and respiratory disease. Study participants were asked to refrain from exercise to avoid EIB or bronchodilation and muscular fatigue	18-35	18	2015/5-2016/7	Vancouver and British Columbia, Canada	Double-blind, counter-balanced, randomized crossover study. On four exposure visits, volunteers inhaled either 400 µg of the β2-agonist salbutamol or placebo before resting for 60-min, followed by a 30-min cycling bout. During rest and cycling, participants inhaled FA or DE. Microvascular (central retinal arteriolar and venular equivalents, CRAE and CRVE, respectively) and macrovascular parameters (blood pressure (BP) and heart rate (HR)) were assessed at baseline (T1), 10 min (T2) and 70 min (T3) after cycling	DE	using 10-min averages; real-time data from the nephelometer (Radiance Research model M903) which provides bscat, then calculated as: PM=bscat *1E6 *0.45. Particle size distribution between 10 and 600 nm with a mode at 100 nm were determined with a TSI Scanning Mobility Particle Scanner (Model 3936; TSI, Shoreview, MN)	DE: 300µg $PM_{2.5}/m^3$ expose for 1 hour	Acute physical activity induces a vasodilatory response in the micro-and macrovasculatur in healthy adults by increasing CRAE and CRVE, and by reducing systolic BP post exercise, despite breathing DE. The DE-associated increase in HR might be indicative of an increased sympathetic response to physical activity while breathing DE	The cycling bout increased CRAE (T2-T1 difference (95th% CI): 4.88 µm (4.73, 5.00 µm), $p < 0.001$; T3-T1 difference: 2.10 µm (1.62, 2.58 µm), p=0.031) and CRVE (T2-T1 difference: 3.78 µm (3.63, 3.92 µm), $p < 0.001$; T3-T1 difference: 3.73 µm (3.63, 3.92 µm), $p < 0.001$). The exposure to DE had no effect on CRAE (FA-DE difference at T2: 0.46 µm (−0.02,0.92 µm); p=0.790; FA-DE difference at T3: 1.76 µm (1.36, 2.16 µm), p=0.213) and CRVE (FA-DE difference at T2: 0.26 µm (-0.35,0.88 µm), p=0.906; FA-DE difference at T3: 0.55 µm (0.05, 1.06 µm), p=0.750). Compared to T1, systolic BP was decreased at T2 by 2.5 mmHg (2.8, 2.3 mmHg, p=0.047), independent of inhaled exposure. Heart rate at T2 was significantly increased by 3 bpm (2, 3 bpm, p=0.025) after the DE-exposure when compared to FA	Chris Carlsten, 2020

4.2.3.1 Study Design

For the human clinical study, the randomized controlled trial (RCT) is the most important study design. It has three principles, namely randomization, control, and blinding. When the subject was human, crossover design and double-blind were usually selected. The difference between the human clinical study and the observational study is whether to intervene with the subjects. Therefore, RCT can obtain the causality and the quantitative relationship between the intervention and health effects. They are more reliable than observational studies.

4.2.3.2 BC

In the search results of human clinical studies, most studies analyzed BC and UFP together. The results showed that BC and UFP might be associated with heart rate and cardiorespiratory functions. However, the contribution of BC and UFP to human health effects couldn't be completely separated.

4.2.3.3 Diesel Exhaust Particle and Diesel Exhaust

Through literature review, it is found that exposure to DE affects respiratory function and arterial stiffness. Leading to systemic inflammation, nervous system and cardiovascular system damage, especially in susceptible people with asthma, coronary heart disease or other diseases.

At present, it is not possible to say definitively whether health effects due to exposure to BC or PM mass are different qualitatively (for example, different health outcomes) and/or quantitatively from each other. This is partly due to the fact that an insufficient number of controlled health studies have been implemented which involve human subjects with simultaneous BC or EC measurements and other PM speciation.

4.3 Good Practice Statement About Carbonaceous Aerosol in WHO AQGs[①]

This review analyzed epidemiological studies and animal studies. However, there are few related studies, especially articles that study BC alone. From this review of the literature concluded, evidence links BC with cardiorespiratory health effects, for short- and long-term exposures. Moreover, in studies that take BC/EC and $PM_{2.5}$/UFP into account simultaneously, associations remained robust for BC/EC.

In 2015, the World Health Assembly adopted a landmark resolution on air quality and health,

① World health Organization. WHO global air quality guidelines[R]. European Union: World health organization, 2021.

recognizing air pollution as a risk factor for non-communicable diseases such as ischemic heart disease, stroke, chronic obstructive pulmonary disease, asthma and cancer and the economic toll they take. These guidelines [such as World health organization global air quality guidelines (WHO AQGs)], taking into account the latest body of evidence on the health impacts of different air pollutants, are a key step in that global response.

As yet, insufficient data are also available to provide recommendations for AQG levels and interim targets for BC/EC and UFP. However, due to health concerns related to these pollutants, actions to enhance further research on their risks and approaches for mitigation are warranted. Therefore, to address concerns about the health and environmental effects of BC/EC and UFP, some good practice statements are summarized in Table 4-5.

Table 4-5 Summary of good practice statements

Pollutants	Good practice statements
BC/EC	1. Make systematic measurements of BC and/or EC. Such measurements should not replace or reduce existing monitoring of those pollutants for which guidelines currently exist. 2. Undertake the production of emission inventories, exposure assessments and source apportionment for BC/EC. 3. Take measures to reduce BC/EC emissions from within the relevant jurisdiction and, where appropriate, develop standards (or targets) for ambient BC/EC concentrations
UFP	1. Quantify ambient UFP in terms of PNC* for a size range with a lower limit of ≤10 nm and no restriction on the upper limit. 2. Expand the common air quality monitoring strategy by integrating UFP monitoring into the existing air quality monitoring. Include size-segregated real-time PNC measurements at selected air monitoring stations in addition to and simultaneously with other airborne pollutants and characteristics of PM. 3. Distinguish between low and high PNC to guide decisions on the priorities of UFP source emission control. Low PNC can be considered $<$1,000 particles/cm^3 (24-hour mean). High PNC can be considered $>$10,000 particles/cm^3 (24-hour mean) or 20,000 particles/cm^3 (1-hour mean). 4. Utilize emerging science and technology to advance approaches to the assessment of exposure to UFP for their application in epidemiological studies and UFP management

* PNC: particle number concentration.

BC is produced by incomplete combustion of fuel or biomass. Because of the large specific surface area and small particle size of BC, it is easily to adsorb toxic substances and could enter the deep respiratory tract through breathing. Therefore, it has the stronger health impacts. Exposure to BC for a short or long term, people will have varying degrees of adverse effects. In this study, the results of systematic review and meta-analysis showed that short-term exposure to BC might be associated with total mortality and respiratory mortality and long-term exposure might be associated with lung cancer mortality. From the aspects of toxicological mechanism, animal experiments and human clinical studies, most animal studies take DEP, but there are few studies on human clinical trials at this stage. The impact of BC on health cannot be ignored and it should be addressed concerns about the health and environmental effects of BC/EC.

5 Review of China's Status on BC/OC Control

Since greenhouse gas emissions such as carbon dioxide and air pollutant emissions have many of the same sources, there can be good synergistic effects of emission reductions both for climate and air pollution. In order to maximise the benefits, however, it is important to perform integrated studies and make holistic plans. During the 14th Five-Year Plan period (2021-2025), China will accelerate the transformation and upgrading of industrial structure and strictly control the construction of high energy-consuming and high-emission projects. The fundamental policies are: (1) construction of a clean and low-carbon energy system, development of non-fossil energy, and reduction of fossil energy consumption; (2) increasing optimization and adjustment of transportation structure, and promoting public to rail, public to water, and multimodal transportation; (3) selection of typical regions and cities to carry out pilot demonstrations of meeting environmental quality standards and carbon emission peaks. At the same time, China will strengthen work coordination and conduct unified policy planning and standard setting, along with unified monitoring and assessment, unified supervision and enforcement, and unified inspection and accountability, in order to provide support and guarantees for achieving the synergistic effect of pollution reduction and carbon reduction.

This section provides a review and summary of BC/OC control related policy and regulations carried out in China.

5.1 Strengthening Environmental and Climate Regulation and Institutional Arrangement

5.1.1 Improving Legal Framework for Improving Air Quality

Since 2013, China has made and revised many laws and regulations on prevention and control of air pollution, including the Environmental Protection Law of the People's Republic of China, the

Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution, the Law of the People’s Republic of China on Environmental Impact Appraisal, the Environmental Protection Tax Law of the People’s Republic of China, the Law of the People’s Republic of China on Desert Prevention and Transformation, and the Energy Conservation Law of the People’s Republic of China. They cover various fields of air pollution control. The Environmental Protection Law of the People’s Republic of China and the Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution are described in more detail below.

The Environmental Protection Law of the People’s Republic of China is a law enacted to protect and improve the environment, prevent and control pollution and other public hazards, safeguard public health, promote ecological progress, and promote sustainable economic and social development. The term “environment” as used in this Law refers to the totality of all natural and artificially transformed natural factors affecting human existence and development, including the atmosphere, water, oceans, land, mineral deposits, forests, grasslands, wetlands, wildlife, natural and human remains, nature reserves, scenic spots, and urban and rural areas. The Environmental Protection Law of the People’s Republic of China was adopted at the 11th Session of the Standing Committee of the 7th National People’s Congress on December 26, 1989, and entered into force as of the date of promulgation. On April 24, 2014, the 8th session of the Standing Committee of the 12th National People’s Congress adopted the Revised Environmental Protection Law, which is described as the “strictest in history.” The new law went into effect on January 1, 2015.

The Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution is a law enacted to protect and improve the environment, prevent and control air pollution, safeguard public health, promote ecological civilization construction, and promote sustainable economic and social development. The Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution was adopted at the 22nd Session of the Standing Committee of the 6th National People’s Congress on September 5, 1987, and entered into force as of June 1, 1988. The law was revised for the second time at the 16th Session of the Standing Committee of the 12th National People’s Congress on August 29, 2015, and became effective as of January 1, 2016. As revised, “The Law of the People’s Republic of China on the Prevention and Control of Atmospheric Pollution” expanded from the original 66 to 129 articles, focusing on the goal of improving the quality of the air environment, strengthening source management and collaborative control, implementing joint prevention and control of air pollution in key regions, actively responding to heavily polluted weather, effectively implementing local government responsibility, increasing the penalties for air environmental violations, and making specific provisions to strengthen prevention work regarding coal, industry, motor vehicles, dust, agriculture, and other sources of air pollution. According to the Deepening Reform of Party and State Institutional Program announced by office of the Central Committee for Institutional Development in March 2018, the Ministry of Environmental Protection was renamed the Ministry of Ecology and Environment, with significant changes in function configuration, internal structure and staffing, and the full implementation of the Air Pollution Prevention and Control Action Plan is an important part of its new function. In accordance with this change, the new names of the relevant institutions were subsequently revised in the Law of the People's Republic of China on Prevention and Control of Air Pollution and was adopted and issued for implementation by the Standing Committee of the 13th National People's Congress at its sixth meeting on October 26, 2018.

The early versions of the law mainly controlled emission of SO_2, NO_x, dust and harmful gases from factories, power plants, motor vehicles and ships. The 2015 and 2018 versions, however, showed a significant change in the country's strategy for air pollution governance, from individual pollutant emission controls to overall air quality improvement, which highlights the regulation of emission sources rather than just focusing on end-of-pipe emissions. Furthermore, the revised versions highlighted regional joint prevention and control, while the co-governance of greenhouse gases and such air pollutants as PM, SO_2, NO_x, ammonia (NH_3) and volatile organic compounds (VOCs), was proposed for the first time. An accountability and performance target evaluation system has been established in China to enforce environmental and climate governance. Major targets have been established at the national level and then disaggregated to provincial and lower levels of government (UNEP, 2019). Thus, local governments of the 31 provinces, municipalities and autonomous regions in China promulgated or revised supporting regulations on environmental protection and prevention of air pollution. Some regions have even framed special legislation for key tasks. For instance, Zhejiang, Anhui and Chongqing have released guidelines for the prevention and control of motor vehicle pollution.

5.1.2 Carbon Peak and Carbon Neutrality Action in China

China amowces carbon-control timeline to the world. In 2020, China released action plam to peak carbon emission before 2030 and carbon neutrality before 2060 in its overall plan for ecological conservation, and promoted the development of a green and low-carbon circular economy in a comprehensive way. The Paris Agreement on Climate Change represents the general direction of the global green and low-carbon transition, which is the minimum action needed to protect the Earth as a home for life, and countries must take decisive steps. Countries should establish a new development concept of innovation, coordination, greenness, openness and sharing, seize the historic opportunity of the new round of scientific and technological revolution and industrial change, promote the "green recovery" of the world economy after the COVID-19 and gather a strong synergy for sustainable development (Xinhua, 2020). As the world's largest carbon emitter, China has clearly proposed a long-term climate goal to control emissions.

Achieving the ambitious goals of carbon emissions and carbon neutrality will not be easy and will require a Herculean effort. In terms of timing, Europe and the United States have a 50-70 years transition period from peak carbon to carbon neutrality, while China only has a 30-year transition period. As the largest developing country, China still has many shortcomings in addressing climate change. China's low-carbon development transition also faces three major challenges: (1) China's manufacturing industry is still in the middle and low end of the international value chain since product energy consumption and material consumption are high, the value-added rate is low, and economic restructuring and industrial upgrading tasks are arduous; (2) the share of coal in Chinese energy consumption is still more than 50% of Chinese energy consumption, the intensity of carbon dioxide emissions per unit of energy is about 30% higher than the world average and the task of optimizing the energy structure is arduous; (3) the energy intensity (energy consumption per unit GDP) is very high, 1.5 times the global average and 2 to 3 times that of OECD countries and the task of establishing a green low-carbon economic system is extremely arduous.

China is actively promoting carbon trading pilot work. In 2011, China launched the carbon market pilot work. Seven provinces and cities—Beijing, Tianjin, Shanghai, Chongqing, Guangdong, Hubei and Shenzhen—have launched local pilot carbon emissions trading systems (ETSs). The seven local pilot carbon markets have started online trading, one after another, since 2013, covering nearly 3,000 key emission industries in more than 20 sectors such as power plants, steel, and cement. The national carbon ETS launched online trading on July 16, 2021. The power generation industry was the first industry to be included, with 2,225 power companies taking the lead. According to the Ministry of Ecology and Environment's calculations, the carbon emissions of the enterprises covered by the first carbon market exceeded 4 billion tonnes of carbon dioxide, meaning that China's carbon emissions trading market will become the world's largest carbon market covering greenhouse gas emissions as soon as it is launched (Xinhua, 2021).

5.1.3 Standards and Action Plans on Air Quality Improvement

Revision of Air Quality Standard. China's ambient air quality standards were first issued in 1982 and subsequently revised in 1996, 2000 and 2012. On February 29, 2012, China's current Ambient Air Quality Standard (GB 3095—2012) was published, incorporating the monitoring indicators of $PM_{2.5}$ and identifying the six basic components in environmental air pollutants—$PM_{2.5}$, PM_{10}, SO_2, NO_2, CO, and O_3 and four other components—total suspenaleel pairticulate (TSP), NO_x, Pb, BaP. It also specified the concentration limits and monitoring requirements of various pollutants. This Standard marked the beginning of the transformation of China's atmospheric environmental management from a target-oriented approach of controlling environmental pollution to a target-oriented approach of improving air quality. The ambient air quality levels, pollution components covered, monitoring methods, etc. were classified in the Standard. The pollutant concentrations are all mass concentrations (Table 5-1 for details). The Standard was implemented in stages and by regions:

- In 2012, Beijing-Tianjin-Hebei, the Yangtze River Delta, the Pearl River Delta and other key regions and municipalities directly under the central government and provincial cities.
- In 2013, 113 key cities and the national environmental protection model city for environmental protection.
- In 2015, all cities at prefectural level and above.
- In 2016, nationwide implementation of the new Standard.

An amendment to the Ambient Air Quality Standard (GB 3095—2012) was released on July 31, 2018 and implemented on September 1, 2018. The modification list is as follows for the contents of Article 3.14:

The original said: "Standard state refers to a temperature of 273 K and pressure of 101.325 kPa. The pollutant concentration in this standard is the concentration in the standard state." This was modified to: "Reference state refers to when the atmospheric temperature is 298.15K and the atmospheric pressure is 1013.25HPa. The concentration of gaseous pollutants such as sulfur dioxide, nitrogen dioxide, carbon monoxide, ozone and nitrogen oxide in this standard is the concentration under the reference state. The concentrations of PM PM_{10} (particle size less than or equal to 10 μm), PM $PM_{2.5}$, TSP and its components lead and Benzo [a] pyrene are the concentrations under atmospheric temperature and pressure at the time of monitoring." The AQI values correspond to different pollution levels (Class I to Class VI) and air quality types ("excellent" to "severe pollution") (Shown as Figure 5-1), which can be used as a general criterion to judge the air quality and its impact on human health (MEE, 2016). A list of Air Pollutant Emissions Standards in China from 2013 to 2020 can be found in Annex Ⅱ.

Table 5-1 Basic particles in ambient air quality standard (GB 3095—2012)

Name of Pollutant	Average Time	Concentration Limits		Unit
		Grade Ⅰ *	Grade Ⅱ *	
SO_2	Annual average	20	60	μg/m^3
	24-hour average	50	150	
	1-hour average	150	500	
NO_2	Annual average	40	40	
	24-hour average	80	80	
	1-hour average	200	200	
CO	24-hour average	4	4	mg/m^3
	1-hour average	10	10	
O_3	Average in the top consecutive 8-hour period	100	160	μg/m^3
	1-hour average	160	200	
PM_{10}	Annual average	40	70	
	24-hour average	50	150	
$PM_{2.5}$	Annual average	15	35	
	24-hour average	35	75	

NOTES: *means some of China's air quality standards Grade I and II are in line with level I and III target values for global air quality guidelines (2005) issued by the World Health Organization (WHO).

Index	Grade	Category	Health Recommendations
0-50	I	Excellent	Outdoor activities are recommended
51-100	II	Good	Outdoor activities are OK
101-150	III	Mild pollution	Susceptible populations should reduce heavy outdoor activities
151-200	IV	Moderate pollution	Susceptible populations would be significantly affected
201-300	V	Heavy pollution	Everyone should reduce outdoor activities
>300	VI	Serious pollution	Try not to stay outdoors

Figure 5-1 AQI classification in Technical Regulations for Ambient Air Quality Index (AQI) (Trial Implementation) (HJ 633—2012)

5.1.4 Air Pollution Prevention and Control Action Plans

China has delineated a clear path for achieving its clean air goals in a series of national policies and plans in the 13th FYP period, mainly including the Action Plan of Air Pollution Control and Prevention (SCPRC, 2013) and the Three-Year Action Plan to Win the Battle for Blue Skies (SCPRC, 2018). The timelines of individual action plans are given in Table 5-2.

Table 5-2 The timelines of Action Plans and the 13th Five-Year Plan

<table>
<tr><th>2013</th><th>2014</th><th>2015</th><th>2016</th><th>2017</th><th>2018</th><th>2019</th><th>2020</th></tr>
<tr><td colspan="5">The Action Plan of Air Pollution Control and Prevention Tier I (Action Ten, 2013-2017)</td><td></td><td></td><td></td></tr>
<tr><td></td><td></td><td></td><td colspan="5">The 13th Five-Year Plan</td></tr>
<tr><td></td><td></td><td></td><td></td><td></td><td colspan="3">The Action Plan of Air Pollution Control and Prevention Tier II (Three-year Action Plan to Win the Battle for Blue Skies, 2018-2020)</td></tr>
</table>

The Air Pollution Prevention Action Plan (Action Ten, 2013-2017), also known as the Ten Articles of Air Quality, was formally released by the State Council of China in September 2013 against the backdrop of pressing air pollution in China. The plan lays out the roadmap for air pollution control by setting concrete pollutant reduction targets in 2017 and 2020 with a focus on three key regions: The Beijing-Tianjin-Hebei region, the Yangtze River Delta (YRD) and the Pearl River Delta (PRD). The Action Plan is the first plan at national level that sets air quality targets for China. The Chinese government gives considerable attention to the prevention and control of air pollution, taking it as an important measure to improve people's livelihood, a concrete action to build an ecological civilization, and an effective tool to maintain economic growth, adjust economic structure and promote reform and upgrading of the Chinese economy.

According to the Action Ten, by 2017 the concentration of PM_{10}, in cities at and above the prefecture level should be reduced by more than 10% from 2012 levels. The concentration of $PM_{2.5}$ in the Beijing-Tianjin-Hebei region, the Yangtze River Delta and the Pearl River Delta should go down by 25%, 20% and 15%, respectively, compared to 2013 levels, with the average annual $PM_{2.5}$ concentrations in Beijing being kept at about 60 μg/m^3. To achieve the above goals, ten specific measures were identified:

(1) Intensify comprehensive control measures and reduce pollutant emission. This will comprehensively improve small coal-fired boilers and accelerate the construction of

desulfurization, denitration, and dust removal renovation projects in key industries. There will be comprehensive control of urban dust and cooking fume pollution. Phasing out of yellow-labelled[①] and old vehicles will be accelerated. Public transportation will be vigorously developed, the use of new-energy vehicles[②] promoted, and improvement of fuel quality accelerated.

(2) Adjust and optimise industrial structure, promote the transformation and upgrading of the economy. Strictly control new production capacity in industries with high energy consumption and high emissions. Speed up the elimination of backward production capacity with outdated technology and resolutely stop construction of illegal projects under construction in industries with severe overcapacity.

(3) Accelerate the technical transformation of enterprises, improve scientific and technological innovation ability. This will vigorously develop the circular economy, foster and strengthen energy conservation and environmental protection industries, and promote the innovative development and industrialised application of major environmental protection technologies, equipment and products.

(4) Speed up the adjustment of energy structure, increase the supply of clean energy. By 2017, coal's share of total energy consumption shall fall below 65%. The Beijing-Tianjin-Hebei region, the Yangtze River Delta, the Pearl River Delta and other regions will strive to achieve negative growth in total coal consumption.

(5) Investment projects must adhere strictly to principles of energy conservation and environmental protection. Barriers to beginning projects will be strengthened. Construction industry projects in ecologically fragile or environmentally sensitive areas will be strictly limited.

(6) Give play to the role of market mechanism, improve environmental economic policies. The central government set up special funds to implement the policy of replacing subsidies with awards. It will adjust and improve policies on prices and taxes and encourage governmental and non-governmental investments in the prevention and control of air pollution.

(7) Amplify the system of laws and regulations with strict supervision and management in accordance with the law. The state regularly publishes air quality rankings of key cities, and has established a mandatory disclosure system for environmental information on heavily polluting

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① "Yellow-Labelled Vehicle" is an alias for high pollution emission vehicles, which are gasoline vehicles that do not meet the National I emission standard, or diesel vehicles that do not meet the National Ⅲ emission standard. As they are affixed with a yellow environmental label, they are called yellow-labelled vehicles.

② New energy vehicles refer to those that use natural gas, electricity, or methanol as fuel, and to hybrid vehicles.

enterprises. This will improve our ability to monitor the environment and step up enforcement of environmental laws.

(8) Establish regional coordination mechanisms, carrying out overall regional environmental governance. In the Beijing-Tianjin-Hebei region and the Yangtze River Delta region, a cooperative mechanism for air pollution prevention and control has been established. The State Council has signed target responsibility letters with provincial governments, conducted annual assessment and strictly investigated responsibility.

(9) Establish a monitoring and early warning emergency system. Establish and launch a perfect emergency response plan in a timely manner to properly deal with heavy pollution conditions.

(10) Clarify the responsibilities of all parties to mobilise the whole population to participate and to improve air quality.

Effectiveness of Action Ten. In 2017, China successfully achieved the target of the Action Ten: the average concentration of $PM_{2.5}$ in 74 major cities was 48 μg/m^3, down 33.3% from 72 μg/m^3 in 2013.The average $PM_{2.5}$ concentrations in the Beijing-Tianjin-Hebei region, the Yangtze River Delta and the Pearl River Delta was 64.6 μg/m^3, 44.7 μg/m^3 and 34.8 μg/m^3, respectively, down 39.2%, 33.3% and 26.0%, respectively, compared with 2013. The average concentration of $PM_{2.5}$ in Beijing was 57 μg/m^3, down 36.0% from the average annual level in 2013 and meeting the target of 60 μg/m^3 set in the Action Ten.

The Three-Year Action Plan to Win the Blue-Sky Defense War (Three-year Action Plan, 2018-2020). To speed up the improvement of environmental air quality and win the battle for blue skies, a “Three-year Action Plan to Win the Battle for Blue Skies” (hereinafter referred to as the Action Plan) was released by the State Council of China on June 27, 2018. The Action Plan aims to cut total emissions of SO_2 and NO_x by more than 15% by 2020 compared to 2015 levels. The concentration of $PM_{2.5}$ in cities at and above the prefectural level fell by more than 18% compared to 2015, the number of days with good air quality reached 80% and the number of days with heavy pollution should dropby more than 25% in 2020. Provinces that have completed the 13th Five-Year Plan ahead of schedule should maintain and consolidate their achievements. Further, the plan states that provinces that have not yet reached objectives must ensure that they fully meet the binding targets set for the 13th Five-Year Plan. The goal of improving Beijing’s ambient air quality should be further improved on the basis of the 13th Five-Year Plan. The Action Plan sets out six tasks and measures to achieve the above targets of air quality:

(1) Adjust and optimize industrial structure and promote the development of green industry. The plan will improve the distribution of industries, strictly control the production capacity of industries with "too high levels, " strengthen the comprehensive control of enterprises with "disorderly and polluting" industries, deepen the treatment of industrial pollution, and vigorously foster green and environmental protection industries.

(2) Speed up the adjustment of energy structure and build a clean low carbon efficient energy system. It will effectively promote clean heating in the northern region, continue to control total coal consumption in key regions, comprehensively improve coal-fired boilers, improve energy efficiency and accelerate the development of clean and new energy sources.

(3) Actively adjust the structure of transportation and develop a green transportation system. It will substantially increase the proportion of freight carried by railways, upgrade the structure of vehicles and vessels, upgrade the quality of oil products and strengthen the prevention and control of pollution from mobile sources.

(4) Optimise adjustment of land use structure and promote non-point source pollution control. It will carry out windbreak, sand-fixation, and afforestation projects, advance the comprehensive improvement of open pit mines, strengthen the comprehensive treatment of dust and strengthen the comprehensive utilization of straw and the control of ammonia emissions.

(5) Action implementation of major projects, greatly reduce pollutant emission. It will carry out major campaigns in autumn and winter in key areas, fight a tough battle against pollution caused by diesel trucks, carry out special campaigns to control industrial furnaces and kilns, and carry out special campaigns to control volatile organic compounds.

(6) Strengthen means to prevent spread of pollution zones, effectively cope with heavy pollution weather. It will establish and improve regional cooperation mechanisms for the prevention and control of air pollution, strengthen the coordination of heavy pollution weather emergencies and consolidate emergency mitigation measures.

The Action Plan calls for speeding up the improvement of relevant policies to provide a strong guarantee for air pollution control. It will improve the system of laws, regulations and standards, expand investment and financing channels, and increase economic policy support. It will improve the environmental monitoring and monitoring network, strengthen the foundation of science and technology, step up environmental law enforcement, and carry out in-depth inspections of environmental protection. It will strengthen organizational leadership, clearly implement the responsibilities of all parties, strictly assess and hold people accountable, make environmental information more public, so as to build a national pattern of action.

Effectiveness of Action Plan 2018-2020. In 2020, which was the closing year of the 13th Five-Year Plan in China, the Action Plan was completed with remarkable results, with both the concentration of $PM_{2.5}$ and the numbers of good and excellent days exceeding the 13th Five-Year Plan targets and tasks ahead of schedule. In 2020, the average ratio of good and excellent days in China was 87%, increasing 7.7 percentage points between 2018 and 2020. The average concentration of $PM_{2.5}$, PM_{10}, SO_2, NO_2 and CO in China were 33 μg/m^3, 56 μg/m^3, 24 μg/m^3, 10 μg/m^3 and 1.3 mg/m^3, down 15.4%, 21.1%, 28.6%, 17.2% and 13.3% compared with 2018, respectively. The average ratio of good and excellent days in Beijing-Tianjin-Hebei and surrounding areas was 63.5%, up 13 percentage points since 2018; $PM_{2.5}$, PM_{10}, SO_2, NO_2 and CO concentration were 51 μg/m^3, 87 μg/m^3, 12 μg/m^3, 35 μg/m^3 and 1.7 mg/m^3, down 15%, 20.2%, 40%, 18.6% and 22.7% between 2018 and 2020, respectively . The proportion of good and excellent days in Beijing was 75.4%, up 13.2 percentage points compared with 2018; $PM_{2.5}$ concentration was 38 μg/m^3, down 25.5% since 2018. The average number of good and excellent days in 41 cities in the Yangtze River Delta region was 85.2%, up 11.1 percentage points since 2018; $PM_{2.5}$ concentration was 35 μg/m^3, down 20.5% between 2018 and 2020. The average number of good and excellent days in 11 cities in the Fenwei Plain was 70.6%, up 16.3 percentage points since 2018; $PM_{2.5}$ concentration was 48 μg/m^3, down 17.2% from 2018 to 2020.

In January 2020 WHO declared a global health emergency because of COVID-19 that has been uncontrollably spreading all over the world (WHO, 2020). The COVID-19 pandemic has put much of the world into lockdown; as one unintended upside to that response, air quality has been widely reported to have improved worldwide (Liu et al., 2021). It is estimated that the change rates of $PM_{2.5}$, NO_2 and SO_2 before and during the lockdown in the Chinese megacities ranged from −49.9% to −78.2% (average: −59.3%± 9.4%), −55.4% to −32.3% (average: −43.0% ± 9.7%), and −21.1% to 11.9% (average: −10.9% ± 15.4%), respectively (Gao et al., 2021).

Causes of Severe Air Pollution and Solutions. In recent years, China has launched key R&D programs, including the Study on Causes and Control Technology of Air Pollution, the Clean Air Research Program and the Scientific Research Project for Public Welfare, with a focus on research in combined atmospheric pollution and R&D in controlling technology. Established in 2017, the National Center for Air Pollution Prevention and Control organised nearly 2,000 scientists and researchers to find the causes of severe air pollution and provide solutions. Various follow-up research programs in different cities allow fast transformation from research to management practices.

One city, one policy. In 2017, the then Ministry of Environmental Protection sent a team of 28 experts and 500 researchers from universities and institutes to the "2+26" city cluster (Air pollution transmission channel cities, see Table 5-3) where they carried out research on "One City, One Policy" and provided on-site technical guidance. The team established a working model integrating research, yield, application, feedback and improvements. They proposed comprehensive solutions targeting the characteristics of local air pollution to serve the local government in air pollution prevention and control.

Table 5-3 The range of "2+26" city cluster

No.	Provinces	Cities
1	Beijing	
2	Tianjin	
3	Hebei	Shijiazhuang
4		Tangshan
5		Langfang
6		Baoding
7		Cangzhou
8		Hengshui
9		Xingtai
10		Handan
11	Shanxi	Taiyuan
12		Yangquan
13		Changzhi
14		Jincheng
15	Shandong	Jinan
16		Zibo
17		Jining
18		Dezhou
19		Liaocheng
20		Binzhou
21		Heze

No.	Provinces	Cities
22	Henan	Zhengzhou
23		Kaifeng
24		Anyang
25		Hebi
26		Xinxiang
27		Jiaozuo
28		Puyang

5.2 Sector Specific Plans, Policies, and Regulations in China

5.2.1 Upgrading Industrial Standards and Companies

China has been pushing ahead with pollutant emissions control of industrial enterprises. Since 2013, 39 standards for key industries such as cement and petrochemical industries and transportation were released or revised (See Annex Ⅱ for details). Upgrading and renovation programs for pollution control facilities in key industries such as steel, cement, and plate glass were pushed forward. It was estimated that strengthening industrial emission standards led to reductions of 7.01 Tg (43% of total abatements) of SO_2, 4.77 Tg (60%) of NO_x and 1.42 Tg (41%) of primary $PM_{2.5}$ emissions at the national level.

Strengthen industrial emission standards. Ultra-low emission and energy-saving transformation of coal-fired power plants was launched in 2014. By the end of 2018, the capacity of China's coal-powered generators with ultra-low-emissions reached more than 810 million kilowatts, accounting for over 80 percent of the country's total installed capacity of coal-power generating units, which resulted in China having the world's largest clean coal-fired power generation system (Figure 5-2)[①]. China's coal consumption exceeds 4 billion tonnes, which accounted for more than 50% of the world's coal consumption in 2019 (Energy Research Institute National Development and Reform Commission, 2019), 51.8% of which was burnt in power plants (Department of Energy Statistics, National Bureau of Statistics, 2020).

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① In 2014, China introduced an ultra-low emissions (ULE) standards policy for renovating coal-fired power-generating units to limit SO_2, NO_x, and particulate matter (PM) emissions to 35, 50, and 10 mg/m^3, respectively. The ULE standard policy had ambitious levels (surpassing those of all other countries) and implementation timeline.

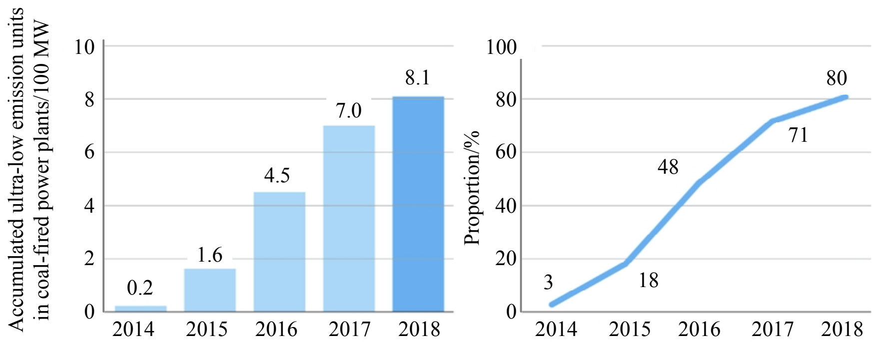

Figure 5-2 Proportion of ultra-low emission units in coal-fired power plants in China from 2014 to 2018(source: Chinese Academy of Environmental Planning)

Meanwhile, China has launched VOCs control in the petrochemical industry, starting with the promotion of LDAR technology (leak detection and repair) and oil and gas recovery in gas stations. Online monitoring devices have been installed in more than 8,000 gas-relevant enterprises with critical pollution sources. Based on online data of pollution sources, the Ministry of Ecology and Environment has organized remote supervision of heavy polluters suspected of excessive emissions or discharges, with local environment authorities responsible for follow-up inspections.

5.2.2 Industrial Restructuring

Enterprises are forced to transform, upgrade and speed up efforts to shut down backward production facilities, address overcapacity and improve the industrial structure. The share of the tertiary industry (provision of services) has grown year-on-year. In 2015, the tertiary industry contributed over 50% of the nation's GDP for the first time; in 2018, the figure was 52.2%, up 5.3% from 2013.

Phase out outdated industrial capacity. From 2013 to 2017, China slashed production of steel by 200 million tonnes, cement by 250 million tonnes, plate glass by 110 million weight cases, and coal-fired units by 25 MW, and barred the production of 140 million tonnes of nonconforming steel to upgrade traditional industries and achieve environmental, economic and social benefits. Consequently, during the period 2013-2017, this structure-focused measure led to the reduction of 2.08 Tg, 1.23 Tg, and 0.69 Tg in SO_2, NO_x, and primary $PM_{2.5}$ emissions, respectively.

Upgrades on industrial boilers. Industrial coal boilers are major sources of emissions, especially SO_2, because of their considerable coal consumption. From 2013 to 2017, more than 200,000 small coal boilers (≤7 MW) were shut down and all small coal boilers in urban areas were phased out. Large operating boilers were extensively equipped with SO_2 and particulate control devices after enforcement of the new emission standard (i.e., GB 13271—2014) in 2014. Consequently, considerable emission abatements were obtained: abatements of 5.54 Tg (34% of all abatements) in SO_2 and 0.71 Tg (20% of all abatements) in primary $PM_{2.5}$ emissions.

Phase out small and polluting factories. Small and polluting factories, i.e., those companies that do not comply with industrial policies and plans, do not have government approvals, and fail to reach emission standards, are mostly small scale, widely spread, hidden, and with no pollution control facilities, making supervision extremely difficult. Complaints about these companies were continuously received. Driven by tightened emission standards, this measure aimed to replace small and highly polluting factories with large facilities equipped with clean production technologies and advanced pollution control equipment, with a focus on northern China. In 2017, China launched a campaign to investigate and crack down on such companies, resulting in that 62,000 were investigated and shut down in the “2+26” city cluster in Beijing-Tianjin-Hebei and neighboring areas. This measure yielded 10%, 3%, and 9% of regional abatements in SO_2, NO_x, and $PM_{2.5}$ emissions, respectively.

5.2.3 Optimizing Energy Structure

In 2013, China put forward the concept of “total coal consumption control” in the Action Plan of Air Pollution Control and Prevention (2013-2017) and set targets for coal consumption controls in key areas. From 2013 to 2018, the proportion of coal consumption in primary energy dropped from 67% to 59%, curbing the trend of rapid growth of coal consumption as shown in Figure5-3.

Setting Coal Quality Standards. China has exerted tight controls over the distribution and burning of low quality coal with high sulfur and ash content. In 2014, China released the Interim Measures for Quality Control of Commodity Coal, bringing inferior coal with ultra-high ash and sulfur under control. In 2017, standards for bulk coal for household use and briquettes were issued, introducing mandatory requirements for coal quality in production, processing, storage, transportation, sales, and burning (Table 5-4).

Eliminating Small Boilers. In 2013, China launched a campaign to shut down coal-fired boilers with steam capacity of 10 tonnes per hour or below in urban built-up areas. As of 2018, more than 230,000 boilers were removed. Meanwhile, no new boilers under 20 tonnes per hour are allowed in those areas and no new boilers under 10 tonnes per hour, in principle, can be built in other areas.

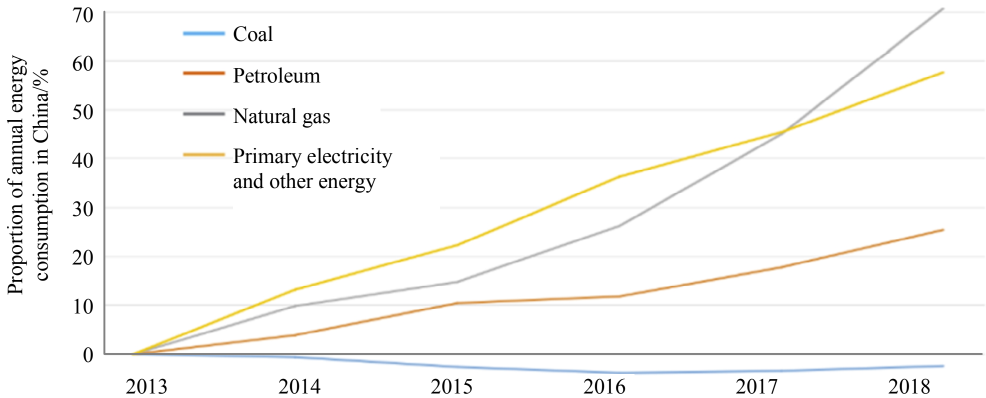

Figure 5-3 China's annual energy consumption since 2013
(source: Chinese Academy of Environmental Planning)

Table 5-4 Quality standard for bulk coal for household use

Items	anthracite coal No.1	anthracite coal No.2	bituminous coal No.1	bituminous coal No.2
Volatile matter	⩽12%	⩽12%	⩽37%	⩽37%
Ash	⩽16%	⩽30%	⩽16%	⩽25%
Sulfur	⩽0.5%	⩽1%	⩽0.5%	⩽1%
Mercury	⩽0.25 μg/g			
Arsenic	⩽20 μg/g			
Phosphorus	⩽ 0.1%			
Chlorine	⩽0.15%			
Fluorine	⩽200 μg/g			

Promote clean fuels for heating in the residential sector. The residential sector is a notable contributor to $PM_{2.5}$ pollution in China, especially in northern China during the heating season. To resolve this issue, advanced stoves and clean coal were promoted nationwide from 2013 to 2016 by providing subsidies to individual households. In the winter of 2017, a clean heating program was launched in northern China featuring natural gas, electric energy, solar energy, geothermal energy and biomass energy replacing coal. The government helped fuel the project with financial support, pricing policy, and energy supply. The central budget devoted more than 20 billion yuan to push through bulk coal control for more than 100 million households. The clean heating area increased by 1,550 km^2. The substitution of coal with natural gas and electricity was further promoted, affecting 6 million households nationwide, among which 4.8 million households were located in the BTH (Beijing-Tianjin-Hebei) and surrounding regions.

This measure has reduced 0.14 Tg of SO_2 (11% of total abatements) and 0.1 Tg (20%) of primary $PM_{2.5}$ emissions, respectively.

Clean Energy Development. Since 2013, the installed capacity of wind power and photovoltaic power in China has maintained high growth (Figure 5-4). By the end of 2018, the installed capacity of hydropower, wind power and photovoltaic power was 35,000 megawatts, 18,000 megawatts and 17,000 megawatts, respectively. The non-fossil energy power capacity grew to about 40% of the total installed capacity. The proportion of clean energy in the primary energy mix increased from 15.5% to 22.1% (MEE, 2019).

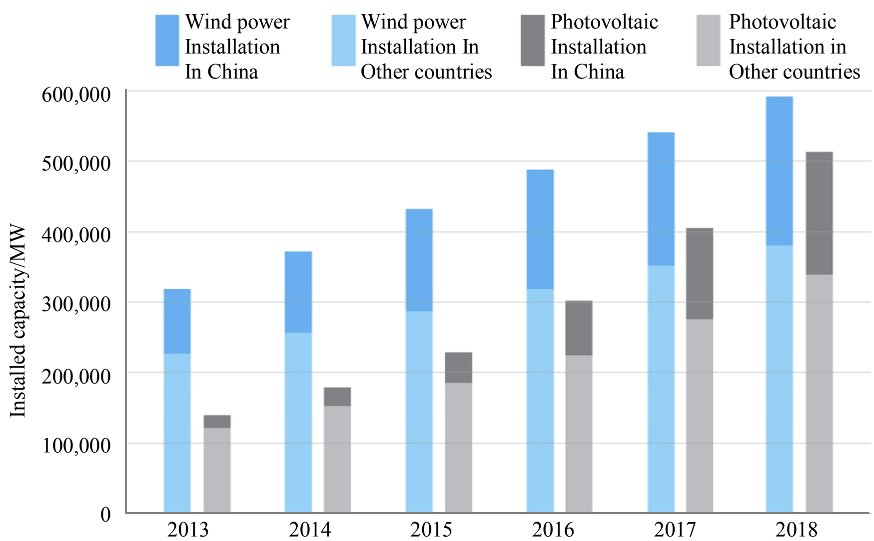

Figure 5-4　Installed capacity of wind and photovoltaic power generation in China and other countries since 2013(source: Chinese Academy of Environmental Planning)

5.2.4　Pollution Control of “Fuel, Road, Vehicle”

Strengthen vehicle emission standards. This measure was a prominent contributor to NO_x abatements (1.06 Tg, 13% of all national abatements). This shift was mainly a result of fleet turnover triggered by the strengthened emission standards in the transportation sector (i.e., China 4 and China 5 emission standards implemented from 2013 to, 2017) and the forced elimination of old vehicles (Figure 5-5). In less than two decades, China has raised the standards for vehicle emissions from National I to National V. China implemented the National I Vehicle Emission Standard in 2001; now, the National V standard has been implemented fully. In 2016, China unveiled the Limit Value and Measurement Method of Pollutant Emission from Light Vehicles (the sixth stage in China). Compared with National I, individual car pollutant emission levels were down by more than 90% under the National VI standard, with emissions of PM per 100 km of diesel cars reduced from 293 grams to 1.5 grams.

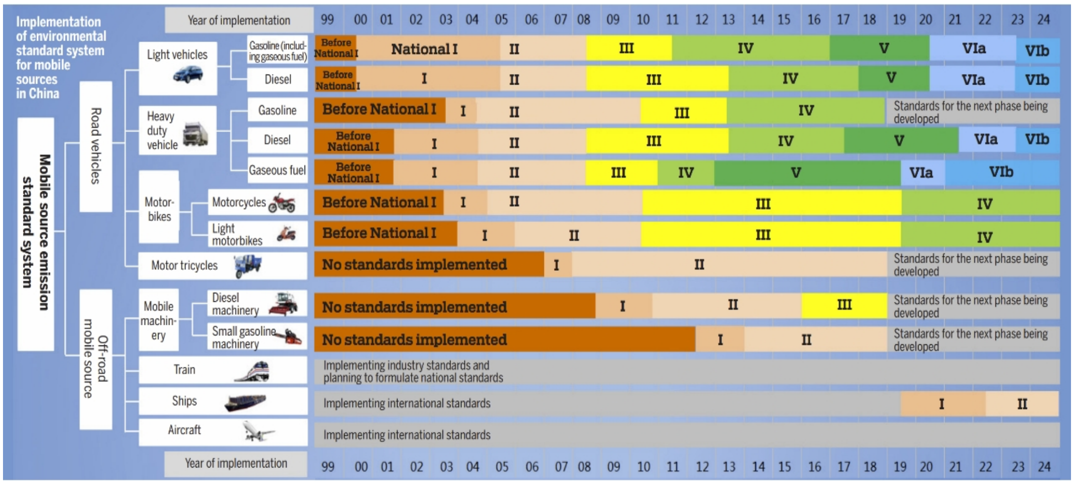

Figure 5-5 Environment standard implementation process for vehicles in China (source: Chinese Research Academy of Environmental Sciences)

Push for Low-Sulfur Fuel. When the National V standard was implemented for vehicle fuel in 2017, sulfur content was down to 10 ppm from 150 ppm for gasoline and 350 ppm for diesel in 2013. In 2018, China unified the standards for vehicle diesel, general diesel and some types of marine diesel. On January 1, 2019, the National VI standard for vehicle gasoline and diesel came into force.

Restructuring Transportation. Compared with more developed countries, China for long had an irrational freight transport structure featuring a high proportion of highway transportation. Chinese highways handle 2.5 times the cargo handled by railways, but per unit pollutant discharge is 13 times that of railways. In September 2018, China issued the Three-Year Action Plan for Promoting Transportation Restructuring (2018-2020), which clarified the targets of adjusting the transport structure and vigorously developing railway transport. On May 1, 2017, Tianjin Port stopped accepting truck-delivered coal or bulk cargo, and all cargo must be transported by train. Consequently, the number of coal trucks entering Yanqing district of Beijing daily is down by half to 3,000-4,000 trips.

New Energy Vehicles. At the end of 2012, only 17,000 new energy vehicles were on the road in China. In 2018, over 1.2 million such vehicles were manufactured and sold in China, accounting for more than half the world's total, bringing the total number of new energy vehicles on the road to 2.61 million (Figure 5-6).

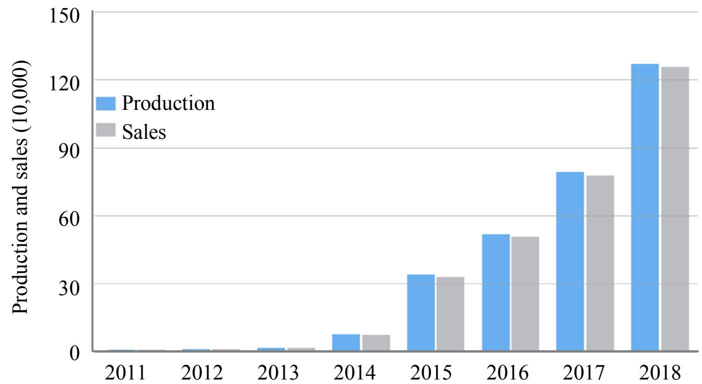

Figure 5-6 Annual production and sales of new energy vehicles in China 2011-2018 (source: China Association of Automobile Manufacturers, Chinese Research Academy of Environmental Sciences)

Eliminate "Old" and "Yellow Label" vehicles. "Yellow label" cars are gasoline vehicles that do not meet National I emission standards and diesel vehicles that do not meet National III emission standards. "Old cars" are vehicles that do not meet National IV emission standards. Since 2013, more than 20 million old and "yellow label" vehicles have been phased out of the Chinese market.

In-Use Motor Vehicle Supervision System. Local authorities monitor in-use vehicles across China to precisely locate vehicles with excessive emissions through a combination of remote sensing, regular detection, remote online supervision, road inspection and big data analysis. China has built 851 remote sensing monitoring sites and 6,140 environmental emission inspection institutions for vehicles. The data at national, provincial and city levels are connected.

Ship Emission Control Zone. In December 2015, China set up ship emission control zones in the waters of the Pearl River Delta, Yangtze River Delta and Bohai Rim (Beijing-Tianjin-Hebei), the first time the country put forward requirements on pollution discharge for ships. In 2018, the coverage was extended to all sea areas 12 nautical miles off the coast, and the trunk lines of the Yangtze River and the Xijiang River. There are strict rules on sulfur content of marine fuel, NO_x control, control of electricity usage for ships going ashore, and VOCs emission.

5.2.5 Treatment of Non-Point Source Pollution

For years, the government has reinforced construction of an ecological security barrier in the sand control zone of northern China, pushing forward forest and grassland protection, wind-proofing, and sand-fixing projects and improvement of the Sanbei shelterbelt (North, Northeast, and Northwest China). As a result, desertification and stony desertification have been effectively curbed and the forest coverage rate has improved, with a marked reduction of dust storms.

China has made great contributions to the world's afforestation effort. A study by Boston University posted on February 11, 2019 on the website of *Nature* said that by reviewing remote sensing data collected from 2000 to 2017 by NASA satellites, researchers made a surprising discovery (Chen et al., 2019): Total global green area was bucking the trend and had increased by 5%, the size of the Amazon rainforest. China is a top contributor, accounting for 10.5% of global afforestation (Figure 5-7).

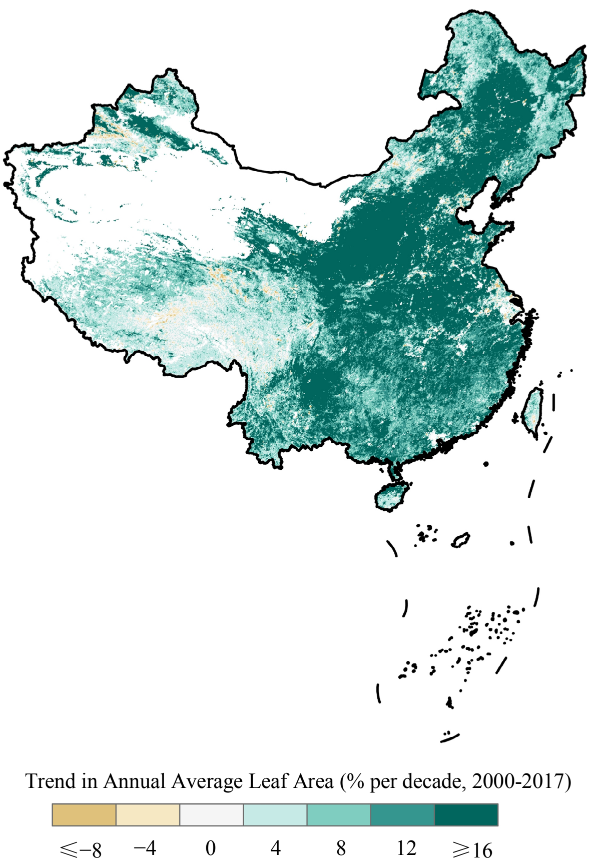

Figure 5-7 Data analysis of China's afforestation efforts on NASA's official website

With large-scale afforestation, the occurrence of dust storms has been reduced markedly. Over the past 60 years in northern China, the average number of days with sand and dust storms has dropped 0.46 days/10 years, and the average number of days with dust-storms in spring (March-May) has dropped 1.67 days/10 years (Figure 5-8).

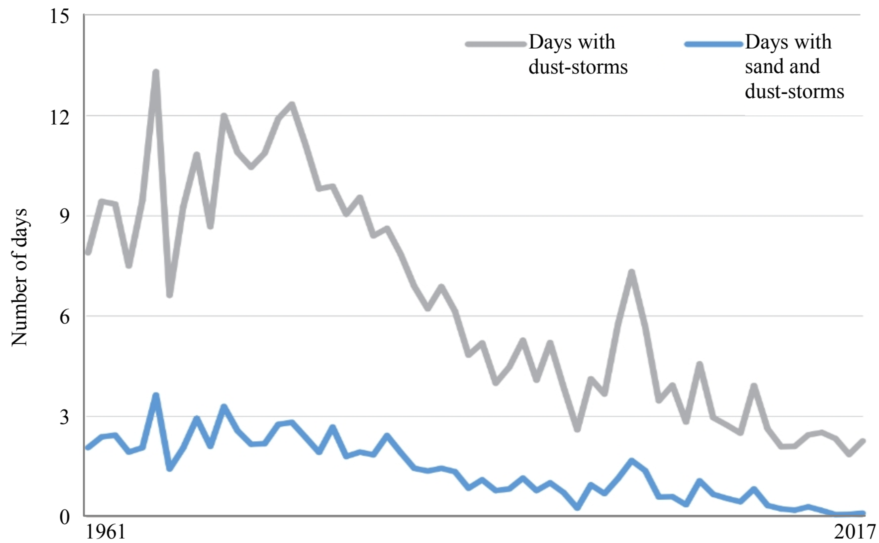

Figure 5-8 Sand and dust activities from 1961 to 2017
(source: National Climate Center of China)

Strengthening comprehensive utilization of straw. In recent years, the comprehensive utilization rate of straw has been greatly improved through feed, fertilizer and energy, and the control of open burning of straw has been strengthened through grid-based supervision. Since 2013, open burning of straw has been basically controlled in northern and eastern China.

Strengthening comprehensive management of dust. Large construction sites should basically achieve "six hundred percent"① with regard to site perimeter fencing, material piling coverage, wet excavation, road hardening, vehicle washing and enclosing the load of vehicles transporting dirt; vigorously promote mechanised road cleaning and cleaning operations; and gradually increase the rate of mechanised cleaning of driveways in urban built-up areas. In pilot cities (e.g., Tangshan City, Hebei Province), innovative road dust mobile monitoring and control means have been installed in cabs with vehicle-mounted mobile road dust monitoring systems to monitor the

① "six hundred percent" means 100% fencing around the site, 100% coverage of material stacking, 100% flushing of incoming and outgoing vehicles, 100% hardening of construction site ground, 100% wet work on the demolition site, and 100% sealing of the loads of vehicles transporting dirt.

amount of dust accumulation in urban streets in real time. The monitoring data are presented on a GIS map, forming a trajectory map of the cleanliness of urban roads, which visually reflects the level of road dust pollution (Figure 5-9).

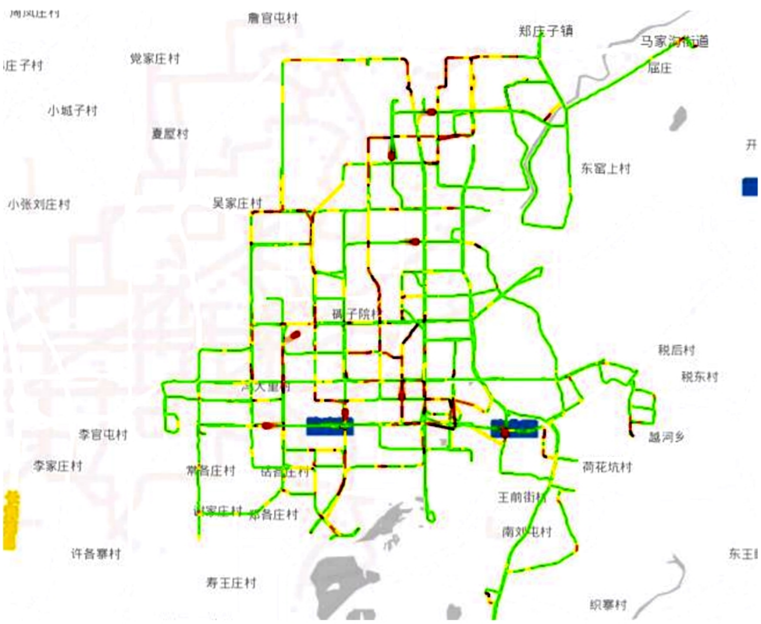

Figure 5-9 GIS map of street cleanliness in tangshan city

(source: National Joint Center for Air Pollution Prevention and Control)

5.2.6 Gaps and Needs to Optimise Policy Making

As relatively new fields, the emissions, impact, and control policy for BC/OC and its co-benefits remain very challenging tasks, especially regarding policy making and planning. The challenges or constraints faced include:

Practical studies are needed: One of the reasons for the difficulties in the BC/OC emission inventory is the lack of practical studies. First, most previous related studies have focused on air pollutant emissions or chemical reactions other than air quality management methods. As a result, the study results are rarely used by any governments for emissions inventories or air quality management activities. Second, the aim of many related studies is simply to publish findings and report theoretical implications and not to support governments in any form. Therefore, the findings are rarely understood and duplicated by governments. Third, some studies developed indicators and approaches for emission inventories or chemical reactions, but they are too theoretical and complicated to be understood and used by governments.

Narrow the uncertainties of BC/OC emissions: Adequate knowledge of BC emissions is the basis for insight into air quality/climate/health effects and control measures. However, the BC/OC emission inventories in China have been developed by different institutions and are often inconsistent in total amounts or in specific sectors due to the significant differences in emission factors chosen, activity levels recognised and methodologies applied. These inconsistent emission results often lead to misunderstandings and confusion, and add difficulty when trying to identify major emission sources and formulate pertinent policy. For this reason, a more comprehensive and convincing BC/OC emission inventory for China should be developed to provide scientific data support to formulate and implement policies by incorporating new emission factors derived from new measurements, by investigating activity levels and by improving methodology. In addition, it is also necessary to present an emission inventory of other pollutants (PM, $PM_{2.5}$, NO_x, nmVOCs, CO, SO_2 and methane) for air quality and climate modeling purposes. Such inventories could be constructed by integrating and improving the inventories developed by other institutions; however, all inventories need to be consistent. According to available literature, the residential and mobile source sectors are among the largest emission sources and with high uncertainties. These sectors are experiencing rapid changes due to China's economic development, increasing standard of living and strengthening of specific environmental efforts. Intensive studies are necessary to narrow knowledge gaps in emissions estimates in these sectors.

Links between scientists and governments need to be established: The links between scientists and governments are extremely important to apply BC research in policy making, because the ultimate aim of these studies is to support governments in policy making and planning, which in turn will promote improved regional air quality and decrease the negative effect on climate change and human health. Little communication currently exists between Chinese scientists and governments, and the governments are not involved when a study is designed and conducted. Many options are available to establish such links between scientists who conduct studies in this field and governments. These options include: (1) governmental participation in the studies from beginning to end, which will enable government officials to understand the valuation procedures and approaches and to use the results or methods in their work and policy making process; (2) a communication channel should be set up so that scientists understand the needs and requirements of governments before the design and implementation of their studies, so that scientists will be able to provide results, approaches and information needed by governments for policy making and planning; (3) government supported studies, not only studies financially supported by scientific funds or the Ministry of Science and Technology, should be encouraged to ensure the utilization of research results.

Knowledge and awareness of government officials need to be enhanced: Currently, BC is steadily growing and contributing increasingly to urban and regional air pollution and health problems without a restrictive target to control the problem. Serious photochemical smog and haze have emerged in some areas, air pollution from local and regional sources reacts and contributes to a complex pollution situation. In addition, the international focus on climate change and air quality is increasing, so China, which is a key emitter of pollutants, is under pressure to take measures to reduce BC emissions. However, it is widely recognised that the lack of understanding and awareness on the negative impact of BC on climate change and air quality is one of the main reasons that governments ignore this issue. Undoubtedly, the knowledge and awareness of government officials will play a decisive role in this aspect because the decisions and policies for controlling BC will be made by those officials. The promotion of more stringent BC emission targets in policy making, planning and actions should be made parallel to efforts to enhance the knowledge and awareness on the roles of BC in air quality and climate change. The targets will include government officials and the public. This can be realised through training and workshops, particularly direct towards participation of government officials in the related workshops or seminars. This proposed way forward will focus on enhancing knowledge and awareness, with emphasis on the balance of trade-offs between air pollution and economic development.

Lack of experience: We are proposing pioneering work in BC emissions, effect evaluation and policy making. The lack of adequate knowledge of the negative impacts of BC/OC emissions has become a major bottleneck. At present, China is taking the first step to develop strategies for controlling SO_2 and NO_x. Measures to reduce BC/OC emissions have not started. It has become an urgent task for the Chinese government to establish scientifically reasonable policies to reduce BC/OC emissions.

Legislation: In response to the major needs of "scientific, precise and legal pollution control, " China urgently needs to build a modern governance system for air pollution prevention and control. Under the leadership of advanced science and technology in air pollution prevention and control, a regional legal and regulatory system for air pollution prevention and control will be formed the development and implementation of four major structural adjustment mechanisms for energy, industry, transportation and land use will be accelerated to give full play to the potential of structural adjustment in air pollutant reduction. At the same time, it will take the opportunity to achieve China's carbon peak and carbon neutrality targets, achieve a "win-win" situation in terms of air quality improvement and climate change response and promote changes in China's participation in the global governance system. On this basis, through the "double leverage" of air quality improvement and national carbon peak and carbon neutrality, it promotes the exploration and practice of green development model in key regions in China.

5.3 Highlights for Emission, Impact, and Control Policy for BC/OC

China's air pollution control efforts accelerated in 2013 and have incessantly broken new ground since then. The measures contained non-electricity industry treatment, clean heating in rural areas, coal-fired boiler remediation, mobile source emission control and VOC treatment, etc. It is valuable and useful to evaluate the effect of these measures on emission, health, climate change and air quality, which also can provide a reference for the 14th Five-Year Plan and even medium and long-term Chinese air quality improvement.

- Updated emission inventory for the change of energy structure. Taking residential heating as an example, where residential coal consumption continues to decrease and is replaced by natural gas or electricity, emissions from residential combustion sources should be updated based on constantly updated activity data and emission factors.

- Updated emission inventory for transportation restructuring. China's emission standards for vehicles continue to tighten, strictly controlling the number of heavy-duty diesel trucks and significantly increasing production and sales of new energy vehicles since 2013. Emission inventories for the transportation sector need to be updated with the latest transportation restructuring data.

- Multi-effect evaluation. With the obvious improvement of air quality in China and further reduction of BC/OC concentrations, there is an urgent need to track and assess its multi-effect on health, air quality and climate in order to provide guidance for Chinese medium and long-term air environment management.

6 Review of Norway's Status on BC/OC Control

This section provides a review and summary of BC/OC control related policy and regulations carried out in Norway.

6.1 Scope

The scope of this section is primarily Norway, while EU and global climate conventions are also discussed. Norway is not a member of the EU but is a member of the European Economic Area (EEA), and as such Norway has implemented several of the EU Directives relevant to climate and air pollution. We have not covered legislation within Norway or the EU completely but highlighted some aspects that we find most relevant in this context. Relevant EU policy and regulations are also presented in Annex Ⅲ.

The pollutants covered here will mainly be PM (BC/OC, PM_{10}, and $PM_{2.5}$).

Norway and EU quick facts:

-There are approximately 5.4 million residents in Norway.

-The number of residents in the EU is approximately 447 million.

-Norway is not an EU member but has special agreements though EEA.

-Norway has an area of 385,207 km².

-The Norwegian coastline is over 2,650 km long.

-GDP for Norway is $350 billion.

-The per capita GDP is $64,856.

Particulate matter (including BC) is covered, for example, in the following international bodies in which Norway participates:

- Arctic Council

- Climate and Clean Air Coalition (CCAC)
- Convention on Long-range Transboundary Air Pollution (UNECE-CLRTAP)
- Intergovernmental Panel on Climate Change (IPCC)

6.2 Introduction to Norwegian Emissions of BC/OC

The Norwegian emissions of BC amounted to 2,841 tonnes in 2019, a total reduction of 41% since 1990 and of 6% since 2018 (Norwegian Environment Agency[①] (NEA), 2021a). Figure 6-1 shows the trends per sector in BC emissions from 1990 to 2019. In 2019, the most important source of emissions was "other combustion, " contributing 37% of total emissions (Figure 6-2). From this category, 75% of emissions in 2019 originated from residential stationary plants, primarily due to wood combustion in private households. From 1990 to 2019, emissions from residential stationary plants have been reduced by 28%.

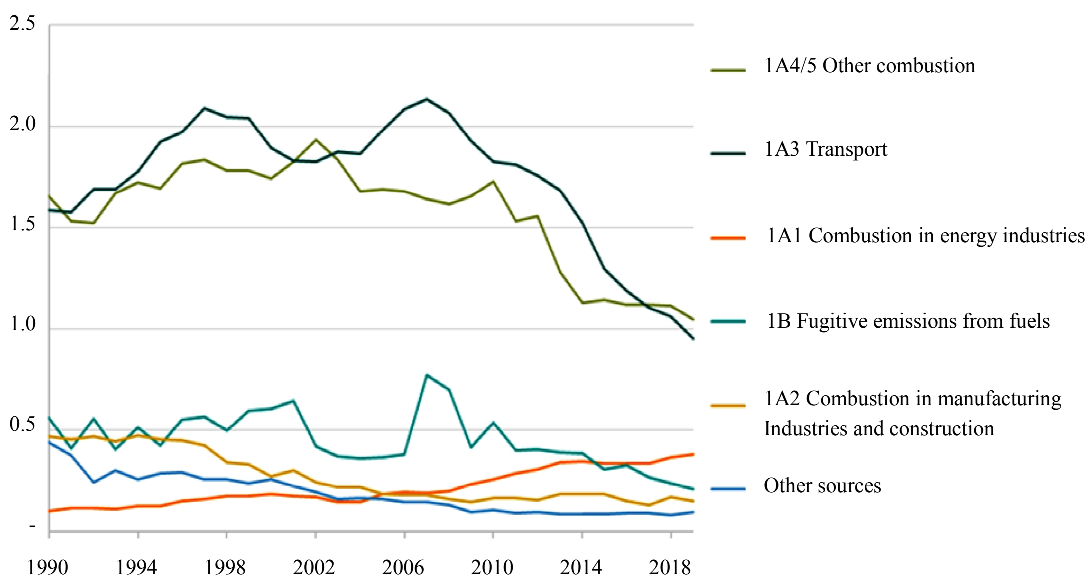

Figure 6-1 Trends in BC emissions, 1990-2019. 1,000 tonnes (Norwegian Environment Agency, 2021a)

① NEA for short.

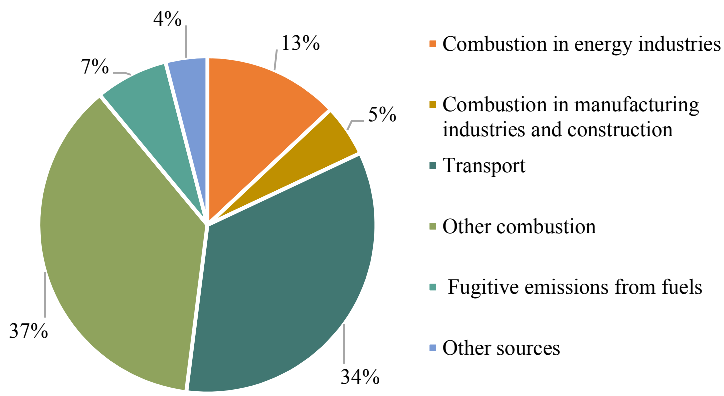

Figure 6-2 Distribution of BC emissions between emission sources, 2019 (source: Norwegian Environment Agency, 2021a)

In 2019, the second most important source of emissions was transport. It contributed 34% of total BC emissions. The greatest share of emissions within the transport sector, 57%, stems from coastal navigation. That is followed by light-duty vehicles, passenger cars and heavy-duty vehicles and buses, contributing 11%, 12% and 10%, respectively. From 1990 to 2019, emissions from navigation increased by 15%, while emissions from passenger cars increased by 28%. However, emissions from light- and heavy-duty vehicles have been reduced by 51% and 86%, respectively, from 1990 to 2019, leading to an overall reduction of about 30% from the transport sector.

Emissions of OC in Norway amounted to 9,537 tonnes in 2019 and are primarily from residential wood burning.

Nationally reported emissions to UNECE (CLRTAP) were also spatially distributed to the EMEP (The co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe) grid system in 2015 (CEIP, 2020). Downscaling of nationally reported emissions are required every four years for all pollutants in each emissions sector. Figure 6-3 shows an example of downscaled BC emissions for the emissions category for off-road sources from 2015 nationally reported data.

Norway has developed many fine scale bottom-up emission models for BC/OC and PM fractions (Table 6-1). These models cover all the main emission sources for PM and generate data at very high spatial and temporal resolutions. Data from these models are mainly used in dispersion modeling for local air quality mapping. However, some of the data from these models are also used to present climate and air pollution information for local municipalities and in national reporting (for example to CLRTAP).

A detailed presentation of each specific model is shown in Annex Ⅳ.

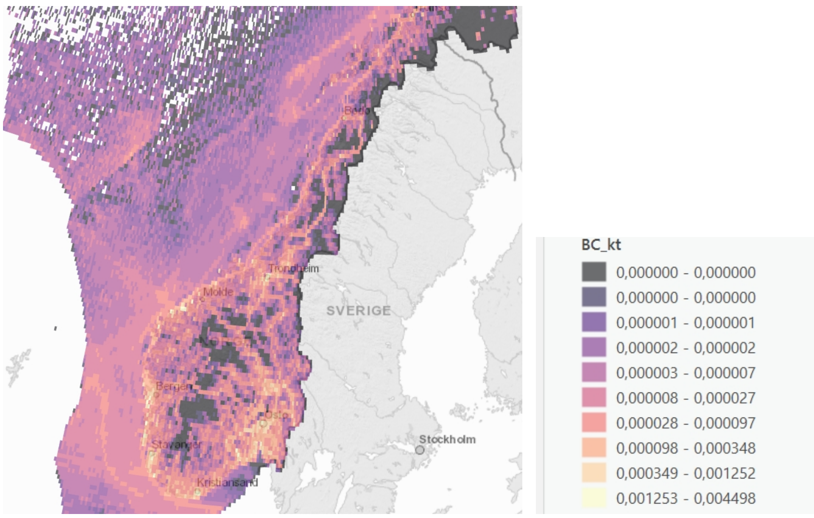

Figure 6-3 Spatial distribution of BC emissions (kt/year) from off-road sources, 2015. Spatial resolution is at the EMEP grid, approximately 8 km x 8 km.

Table 6-1 Overview of fine scale bottom-up emission models for local air pollution in Norway

Emissions Source	Model/tool	Components	Spatial resolution	Temporal resolution
Wood burning	MetVed-2	BC, OC, + all major air pollutants	250m	Hourly (with regional time variations)
Shipping	Havbase/UtAgg	BC, OC, + all major air pollutants	250m, 1,000m, 8km	Hourly
Road traffic, exhaust	NERVE-H	PM, + all major air pollutants	Road links >50m	Hourly (with general time variations)
Industry	Tilde	Soot, + all major air pollutants	Point sources	Hourly (with general time variations)
Road traffic, non-exhaust	NORTRIP	PM	Road links >50m	Annual
Airports*	Flybase	PM, + all major air pollutants	Airport site (~5km^2)	Hourly
Construction sites*	EmSite	PM, + all major air pollutants	Construction site (1km^2)	Hourly

*Model under development, available at the end of 2021.

6.3 Introduction to Norwegian Policies and Regulations

The polluter pays principle is a cornerstone of the Norwegian policy framework for air pollution and climate change. Norway does not have regulations, policies or national targets directed specifically to reduce BC and OC emissions, but air pollution and climate change legislation targeting other components has contributed to large reductions in BC and OC emissions since 1990.

In 2017, the eight member states of the Arctic Council adopted a collective goal to reduce BC emissions by 25%-33% below 2013 levels by 2025.[①] There are large uncertainties in the emission data, but a 2021 assessment[②], shows that the collective goal probably will be attained. however, the collective goal does not, require each country to have a national target for reductions in BC emissions.

Policies and regulations affecting BC/OC are presented below.

6.3.1 Air Quality

Both Norway and the EU have ratified protocols under the Convention on Long-range Transboundary Air Pollution (LRTAP, the Air Convention). The Air Convention was adopted in 1979 and celebrates its 40th anniversary in 2019. Within the Air Convention framework, a number of task forces, centres, and International Cooperative Programmes provide research, scientific assessments and dialogue on the common knowledge base on air quality issues. The most important program for Norway is the co-operative program for monitoring and evaluation of the long-range transmission of air pollutants in Europe (EMEP).[③] The program includes four task forces and five centres.

The Air Convention has been extended by eight Protocols. Notably, the original Gothenburg protocol was agreed to in November 1999 and formed the basis for the original National Emission reduction Commitments Directive (NEC), Directive 2001/81/EC. The protocol was revised in 2012 and the reduction commitments established for 2020 for the EU and its Member States have been transposed into EU law by a new NEC Directive, Directive 2016/2284/EU. The Directive sets 2020 and 2030 emission reduction commitments for NO_x, NMVOCs, SO_2, NH_3

① https://oaarchive.arctic-council.org/handle/11374/1936, last access 25 May, 2022.

② https://oaarchive.arctic-council.org/handle/11374/2610, last access 25 May, 2022.

③ https://www.emep.int, last access 25, May, 2022.

and $PM_{2.5}$.

The revised Gothenburg Protocol recognises the human health and climate co-benefits of reducing BC and ground-level ozone (Mark W. Frampton et.al., 2006). An objective is that Parties should, in implementing measures to achieve their national targets for PM, give priority, to the extent they consider appropriate, to emission reduction measures that also significantly reduce BC.

Norway and the EU continue to work closely with the Air Convention to encourage ratification and implementation of the revised Protocol by the broadest range of parties, and to pursue further work on key areas such as BC and intercontinental transport of air pollution.

The EU Ambient Air Quality Directives[①] (2004/107/EC and 2008/50/EC) set the guidelines, regulations, and laws for managing air quality in Europe. In addition to the limit values set out in the directive, Norway has established its own more ambitious guidelines and laws, while still adhering to the EU regulations. The country-specific guidelines and laws for air quality management in Norway are given in Chapter 7 of the National Pollution Regulations Regarding Ambient Air Quality.[②] Norway has various levels of air quality standards (Table 6-2):

- **EU Air Quality directive limit values**: legally binding values for the entire EU; if not met, mitigation measures must be performed to hold the values under these limits. Air quality plans should be developed for areas where concentrations of pollutants in ambient air exceed or are in danger of exceeding, the limit values. These plans are reported back to the EU.

- **National Legal limit values**: legally binding values specific for Norway; if not met, mitigation measures must be performed to hold the values under these limits. Air quality plans should be developed for areas where concentrations of pollutants in ambient air exceed or are in danger of exceeding, the limit values.

- **National Goals**: the government's long-term goals for local air quality in Norway. These goals are set to the same levels as the air quality criteria.

- **Air Quality Criteria**: health-based goals for air quality that establish safe air quality when levels are under the criteria.

① https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0050&from=EN, last access 27 May, 2022.

② https://lovdata.no/dokument/SF/forskrift/2004-06-01-931/KAPITTEL_3#KAPITTEL_3, last access 27 May, 2022.

The values for these various standards for PM are shown in Table 6-2.

Table 6-2 EU and Norwegian limit values, goals, and criteria for PM

		EU Air Quality Directives limit values	National legal limit value	Proposed national legal limit value*	National goals	Air quality criteria
PM_{10}	Daily	50 μg/m^3 (max 35 exceedances)	50 μg/m^3 (max 30 exceedances)	50 μg/m^3 (max 15 exceedances)	n/a	30 μg/m^3
	Annual	40 μg/m^3	25 μg/m^3	22 μg/m^3	20 μg/m^3	20 μg/m^3
$PM_{2.5}$	Daily	n/a	n/a	n/a	n/a	15 μg/m^3
	Annual	25 μg/m^3	15 μg/m^3	12 μg/m^3	8 μg/m^3	8 μg/m^3

*In addition, there have been proposed changes to the National legal limit values, which took effect January 2022 .

In addition, Norway has air pollution classes that are used in forecasting of air quality; the classes for PM are shown in Table 6-3.

Table 6-3 Norwegian pollution classes used in air quality forecasting of PM

Classes	Level	Health risk	PM_{10} Day/ (μg/m^3)	$PM_{2.5}$ Day/ (μg/m^3)	PM_{10} Hour*/ (μg/m^3)	$PM_{2.5}$ Hour*/ (μg/m^3)
	Low	Low	<30	<15	<60	<30
	Moderate	Moderate	30-50	15-25	60-120	30-50
	High	Considerable	50-150	25-75	120-400	50-150
	Very high	Serious	>150	>75	>400	>150

* The pollution class for PM_{10} and $PM_{2.5}$ are primarily given for the daily mean (the mean concentration in a 24-hour period). The equivalent pollution class for hourly mean (the mean concentration in one hour) is a mathematical conversion based on statistics. As an example: When the hourly mean for PM is in the yellow pollution class, it is most likely that the day will also be yellow.

According to Norway's National Pollution Regulations, $PM_{2.5}$ from background stations must be analyzed for OC/EC and BC measurements have been previously taken for specific research projects.

If a municipality exceeds the legally binding limit values for local air quality, the regulations require production of an Air Quality Plan for the municipality. The plans must show how the proposed mitigation will allow for air quality for the entire municipality to be under the limit values. These plans normally include detailed emission calculations and dispersion modelling to quantify the air quality situation, they are based on the following general parameters:

- Description of meteorology and local variations
- Localization of housing and other planning areas
- Localization of emission sources

These action plans have primarily addressed PM_{10} and NO_2 concentrations, as other pollutants have been less problematic in recent years in Norwegian municipalities. Sources of these PM_{10} concentrations are normally road dust and re-suspension. In addition, the national goals for $PM_{2.5}$ (Table 6-2) are listed as national Environmental Indicator number 4.4.2, in which the goal currently has a "good" status with "positive" development.[①]

6.3.2 Climate Change

Climate change policy will in many cases also contribute to the reduction of BC and OC emissions. Climate change and emission of greenhouse gases have been concerns of Norwegian policy since the late 1980s and remedial measures have broad political backing. The polluter pays principle is a cornerstone of the policy framework on climate change. CO_2 taxes on emissions were introduced in 1991 as a cost-effective policy to limit emission of greenhouse gases. The EU emission trading scheme covers around 50% of Norway's greenhouse gas emissions. Together with the Norwegian CO_2 tax, economic measures cover almost 80% of total greenhouse gas emissions.

When developing its climate policy, Norway also addresses drivers of climate change other than reduction of the greenhouse gases included in Annex A to the Kyoto Protocol. Measures covering certain sources of CO_2 emissions may also affect BC emissions and other short-lived climate forcers.

In June 2017, the Norwegian Parliament adopted the Climate Change Act. The purpose of the Act is to promote Norway's long-term transformation to become a low-emission society by 2050. Norway's climate targets for 2030 and 2050 are established by law in the Act.

① Norwegian Environmental Indicators: https://www.environment.no/, last access 27 May, 2022.

In its White Paper on the 2030 climate strategy[①] from 2017 the government states that it will promote the use of cost-effective mitigation measures to meet the 2030 commitment (see below). If the CO_2 tax is not considered to be an adequate or appropriate instrument, other instruments that provide equally strong incentives to reduce emissions will be considered, including direct regulation under the Pollution Control Act and voluntary agreements.

In February 2020, Norway submitted an enhanced climate target under the Paris Agreement, number 2 in the following list of national mitigation targets:

(1) Under the Kyoto Protocol, Norway will reduce global greenhouse gas emissions by the equivalent of 30% of its own 1990 emissions by 2020.

(2) Under the Paris Agreement, Norway has undertaken a commitment to reduce emissions by at least 50%, and up to 55%, by 2030 compared to 1990 levels (increased from the previous target of "at least 40%").

(3) Norway will be climate neutral by 2030. This means that from 2030, Norway must achieve emission reduction abroad equivalent to remaining Norwegian greenhouse gas emissions. Possible mechanisms are the EU emissions trading system, international cooperation on emissions reductions, emissions trading and project based co-operations.

(4) Norway has established by law a target of becoming a low-emission society by 2050. In quantitative terms this means reducing emissions by 90%-95% compared to 1990 levels.

Norway and Iceland are cooperating with EU to fulfil their emission reduction targets of an at least 40% reduction of GHG emissions by 2030 compared to 1990 levels. This cooperation means Norway and Iceland will participate in the three pillars of the EU climate legislation towards 2030: Manufacturing plants and power stations in Norway and Iceland have been part of one of the EU pillars—the EU emissions trading system (EU ETS)—since 2008. The EU ETS entails no national targets. It puts a limit to the overall emissions from covered installations, which is reduced each year, reducing emissions by 43% in 2030 compared to 1990. Within this limit, companies can buy and sell emission allowances as needed. Through the climate cooperation with EU and Iceland, Norway will be part of the other two EU pillars: the Effort Sharing Regulation (ESR) and the Regulation on Land-use, land-use change and forestry (LULUCF). Through the ESR, Norway will have yearly emissions budgets and a commitment to reduce emissions not covered by the EU emission trading system (so-called non-ETS emissions)

① https://www.regjeringen.no/en/dokumenter/meld.-st.-41-20162017/id2557401/, last access 27 May, 2022.

by 40% in 2030 compared to 2005. In the government's political platform, it is stated that the government intends to reduce domestic non-ETS emissions by 45% in 2030 compared to 2005 levels.

Therefore Norway will be well prepared when the EU strengthens its climate legislation. Norway also intends to cooperate with the EU on the enhanced 2030 climate targets. Norway may have to meet stricter requirements under EU climate legislation as rules are tightened to meet the enhanced 2030 target the EU has adopted.

In 2021, the Norwegian government submitted a white paper with a Climate Action Plan for 2021-2030. The plan shows how Norway can attain its national climate targets by implementing measures and instruments. One of the main instruments proposed is to gradually increase the CO_2 tax from around 590 NOK/tonne CO_2 today to 2000 NOK/tonne CO_2 in 2030. The plan was approved by Parliament with some amendments. The main elements of the climate plan had support from a majority of the parliament. Still, elements of the plan, such as the gradual increase of the CO_2 tax, are expected to be debated in the yearly budgets in the years to come.

6.4 Sector Specific Plans, Policies, and Regulations in Norway

Policies and regulations that affect BC/OC emissions for specific emission sectors in Norway are presented below. This section includes several sectors, but with a focus on transport and residential combustion, because these are the two main emission sectors for BC/OC that are prioritised in this project.

6.4.1 Transport

The EU has adopted stringent emission standards for various vehicle categories, which also apply in Norway (see Figure 6-4 that shows light-duty vehicles). The Euro 6 emission standards include PM and have been active for nearly 10 years, with even stricter Euro 7 expected to be proposed later in 2022.

Norway has set out ambitious national goals for rapid transition to electric mobility through incentives such as tax breaks and reduced toll fares.

From 2018, cash payment was introduced for scrapping motorcycles, mopeds, campers and caravans. Scrapyards get operational support for receiving these vehicles. Cash payment for scrapping passenger cars has been in place since 1978.

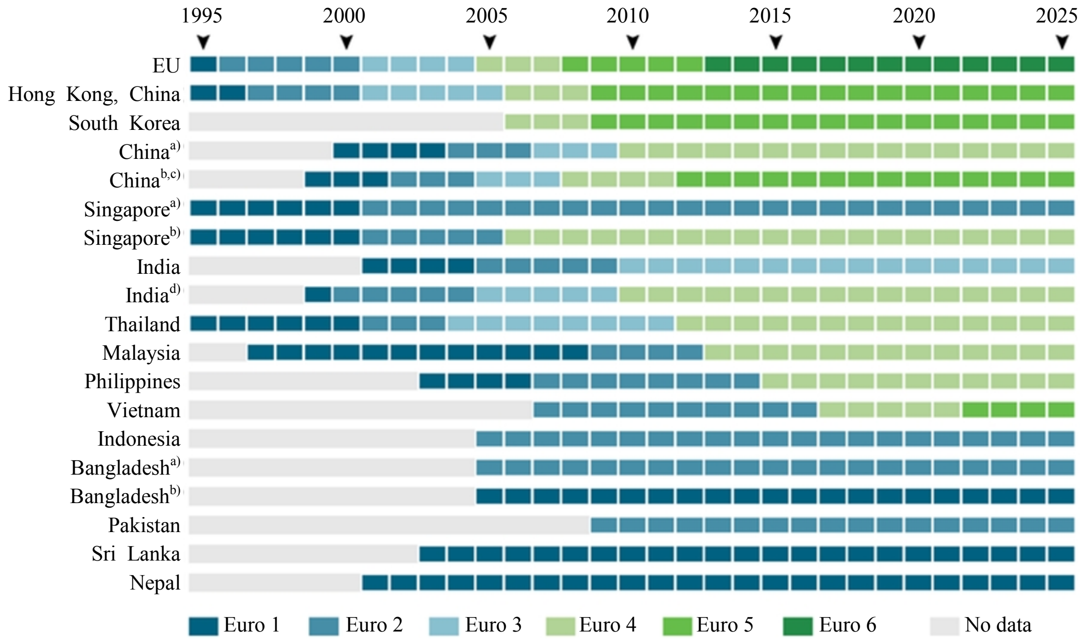

Figure 6-4 EU vehicle emission standards for light-duty vehicles in comparison to other cities and countries (Source: EEA, 2016). a) gasoline, b) diesel, c) entire country, d) Delhi, Mumbai, Kolkata, Chennai, Hyderabad, Bangalore, Lucknow, Kanpur, Agra, Surat, Ahmedabad, Pune and Sholapur

Since the 1990s, electric battery vehicles have been exempted from registration tax, benefitted from free parking, and been exempted from tolls, etc. The sale of electric vehicles has increased considerably over the last six years (Figure 6-5), emissions have decreased and air quality has improved (especially NO_2 concentrations). The Norwegian policy for electric battery vehicles is further described below.

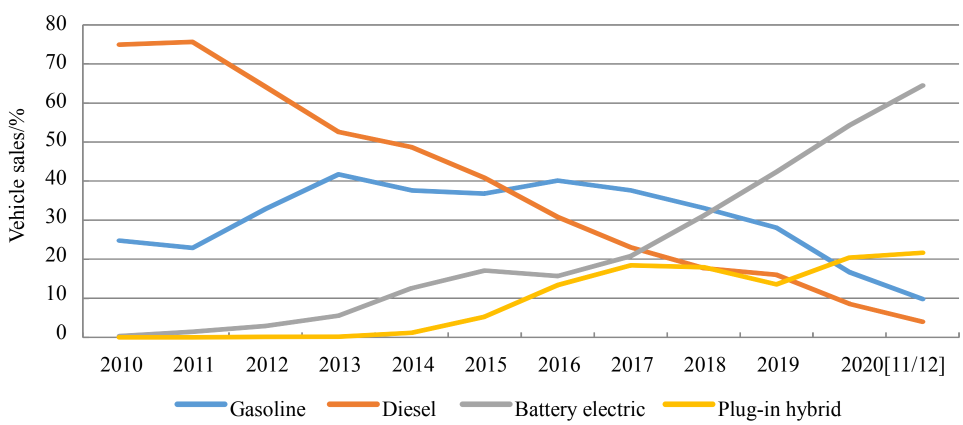

Figure 6-5 Vehicle sales (%) in Norway from 2010 to 2020

To reduce emissions through cleaner diesel fuels and vehicles, Norway aims for a rapid transition to zero and low emission transportation. This will reduce emissions of BC in addition to CO_2 emissions. Norway has the highest sales rate of electric vehicles in the world. In 2020, 141, 412 new passenger vehicles were sold, of which the share of zero emission cars amounted to 54% (OFV, 2021). This represents an increase of 20% from 2017. Sales have increased considerably because Norway has for several years had in place numerous measures to promote the uptake of zero emission vehicles.

The zero VAT rate for the supply and import of electric vehicles was adopted in 2001. An important element is also the general tax on CO_2 that covers the use of most transportation modes. The use of fossil fuels in vehicles is charged with a CO_2 tax, which favors the use of electric vehicles. Furthermore, fossil fuels for road traffic are charged a road use duty. This duty is intended to cover the externalities from road traffic, apart from CO_2, such as local air pollution, accidents and noise. Electric vehicles contribute to such effects but do not pay a road use duty. In addition to the reduced taxes, zero emission cars can have benefits related to usage such as free public parking, free charging at public charging stations, access to bus lanes and reduced fees on ferries. Currently, zero emission vehicles are not charged at all toll roads, the Norwegian parliament has established a maximum toll of 50% of the toll that fossil fuel vehicles pay. However, local governments can reduce or remove the user benefits if the zero emission cars are causing difficulties and inefficiencies. For instance, if the electric vehicles are filling the bus lanes and causing queues for public transport.

ENOVA SF is owned by the Norwegian Ministry of Climate and Environment. Enova makes financial contributions to individuals and businesses in order to start using the newest and most climate friendly technologies. ENOVA has support schemes for establishment of charging stations and zero emission technologies in the transport sector. In addition to these schemes, a zero-emission fund for commercial transport was established in 2019, with a budget of 1 billion NOK for 2020. The Government has set ambitious targets for emission from new vehicles in 2025 and 2030. In its White paper on the National Transport Plan for 2018-2029,[①] the government established several new targets:

- All new passenger cars and light vans sold in 2025 shall be zero-emission vehicles.
- All new urban buses sold in 2025 shall be zero emitters or use biogas.

① https://www.regjeringen.no/en/aktuelt/a-national-transport-plan-for-better-and-safer-daily-travel/id2548623/, last access 27 May, 2022.

- By 2030, all new heavy-duty vehicles, 75% of new long-distance coaches and 50% of new trucks shall be zero emission vehicles.
- The distribution of freight in the largest urban centres shall have almost zero emissions by 2030.

A regulatory framework for public procurement entered into force in January 2018. Requirements for emission are set out for vehicles, as well an obligation to adopt best practice on environment for overall public procurement policy. An increasing number of local and national authorities and agencies require, through public procurement, that mobile and stationary engines at construction sites do not use fossil fuels. The Norwegian Environment Agency manages a support scheme called *Klimasats*, supporting local initiatives on climate change mitigation, such as fossil free construction sites and more environmentally friendly ferries. ENOVA and Innovation Norway also offer economic support.

The Parliament adopted the White Paper on National Transport Plan 2018-2029 in June 2017. In June 2017, the government also issued a White Paper on climate policies, [①] in which the government set a working target of a cut of 35%-40% in greenhouse gas emissions from the transport sector by 2030, compared with 2005. Further, the government said in the White Paper that it would build upon current policy instruments to stimulate use of zero emission vehicles, and by that contribute to reaching the targets for zero emission vehicles in the National Transport Plan 2018-2029. Depending on market development, the government will consider necessary changes in policy measures. The government also stated that it will facilitate making zero emission cars competitive, and that economic measures should support this. A new White Paper with a Climate Action Plan 2020-2033 was submitted to the Parliament in January 2021. The plan states that the government will reduce emissions from the transport sector by half by 2030, using the policy instruments presented in the climate action plan. The targets for zero-emission vehicles set out in the National Transport Plan 2018-2029 are to be maintained, and the government will design the policy instruments described in the climate action plan so that they can be achieved. The government has also recently launched the National Transport Plan for 2022-2033 (NTP, 2021), which has the general goal for the entire transport sector of an effective, environmentally-friendly and safe transport system in 2050.

Norway has an action plan for fossil free public transport by 2025 and an action plan for infrastructure for alternative fuels. Furthermore, an action plan on fossil-free construction sites in

①https://www.regjeringen.no/en/dokumenter/meld.-st.-41-20162017/id2557401/, last access 27 May, 2027.

the transport sector is being developed.

The Norwegian government launched its action plan for green shipping in 2019, in which a key element is wider use of alternative fuels.① The Norwegian government's ambition for its domestic shipping and fishing vessels is to reduce emissions by 50% by 2030 and promote the development of zero- and low-emission solutions for all vessel categories. Norway's action plan for green shipping describes the status of the fleet and how Norway will work to speed up the pace of this transition.

6.4.2 Residential

Norway had 2.3 million households in 2016 according to Statistics Norway (SSB). About half of those, 1.2 million, heated their homes by burning wood; 60% of the households burnt the wood in stoves with new technology.② Since 1998, enclosed wood heaters must be approved for sale and use in Norway according to Norwegian standard NS 3058. The stoves and fireplaces must meet the emissions requirements described in NS 3059 (10 g PM/kg dry wood). Eco-design regulation [COMMISSION REGULATION (EU) 2015/1185] entered into force on January 1, 2018, with new emission limits applicable from 2022 (5 g PM/kg dry wood). A side effect of the legislation targeting PM is reduced BC emissions.

The Nordic Swan Ecolabel was established in 1989 by the Nordic Council of Ministers as a voluntary ecolabelling scheme for the Nordic countries Denmark, Finland, Iceland, Norway and Sweden. It is an effective tool to help companies that want to go ahead with sustainable solutions and thereby enable consumers and professional buyers to choose the environmentally best goods and services. The Nordic Swan Ecolabel scheme is in accordance with the Norwegian standard NS 3058/59, but with a stricter emission limit of 4 g PM/kg dry wood .

Some cities like Oslo and Bergen have exchange programs and give subsidies to residents who exchange their old wood stoves. Burn right campaigns are being organized all over the country in cooperation with the fire brigades. ENOVA SF has programs for energy efficiency, including enhanced home heating efficiency.

A measure that combines accelerated exchange of wood stoves for the best wood stoves on the market and exchange of old wood stoves for electric heating was included in the report *Klimakur*

① https://www.regjeringen.no/contentassets/2ccd2f4e14d44bc88c93ac4effe78b2f/the-governments-action-plan-for-green-shipping.pdf, last access 27 May, 2027.

②https://www.ssb.no/natur-og-miljo/artikler-og-publikasjoner/mindre-ved-brennes-i-gamle-ovner, last access 27 May, 2027.

2030 (Klimakur 2030, 2020). With the current design of this measure it is estimated that GHG-emissions can be reduced by 0.51 million CO_2 equivalents primarily due to methane reductions.

Unintended rebound effects are possible when advocating for the use of modern appliances. In Oslo, Norway, a follow-up study that investigated the results of a stove exchange program found that despite relatively rapid change-out of appliances, emission reductions did not follow. Although the new stoves emitted less per unit of wood burned, the study found that users used their stoves more and thereby increased their wood use, thus negating emission reductions. This case emphasises the importance of advocating, when possible, for the adoption of the cleanest technology possible in change-out programs. Similarly, energy efficiency in buildings is a critical element in achieving emission reduction for change-out programs, in order to reduce energy required to heat homes.

The view of wood as a renewable alternative to fossil fuels can present a roadblock for action to reduce BC emissions from wood burning appliances. Individuals who are motivated to reduce their own carbon footprints may opt for solid fuel combustion appliances and inadvertently contribute to poor air quality and short-term climate impacts from resulting BC and other SLCF emissions. Therefore, educational and awareness-raising campaigns should endeavor to inform the general population on the air quality and climate consequences of solid fuel combustion.

The building code is the main legal instrument for improving energy efficiency. Norway introduced energy requirements for buildings in 1949. They have been revised and made stricter several times, most recently in 2016. The new and stricter requirements (passive house level) entered into force on January 1, 2016. The 2016 requirements were tightened such that dwellings became 26% more energy efficient and office buildings 38% more energy efficient compared to previous requirements. Energy performance certificates are mandatory for buildings that are to be sold or rented out. Among other things, the new energy requirements specify that installation of fossil fuel heating installations are not permitted. For specifications on the building code energy requirements, see Norway's Seventh National Communication under the UNFCCC (2018).[①]

In June 2018, the Norwegian government introduced a prohibition on the use of mineral oil for permanent heating of buildings from January 1, 2020. Schemes to support households to phase out the use of mineral oil for heating of buildings have been in place for several years; the use of

① https://www.regjeringen.no/contentassets/52d65a62e2474bafa21f4476380cffda/t-1563e.pdf, last access 27 May, 2022.

mineral oil for heating of buildings has been regulated through different measures such as CO_2-tax, mineral oil tax, standards in the building code and support schemes from Enova and municipalities.

6.4.3 Industry

The Pollution Control Act sets out a general prohibition against pollution from industrial activities unless a permit from the authorities has been issued. Specific emission limit values are set on a case-by-case basis in the individual permits, based on best available techniques (BAT) as defined nationally or in the EU.[①, ②] In addition, local factors are considered when the emission limits are set (Ambient air quality guidelines). Emission limits can be set for soot, total OC, dust, and/or PM emissions, and, more rarely, for BC.

Emissions to air of these components are reported to NEA annually, and the figures are published on NEA's website The Norwegian PRTR.[③]

Flaring of natural gas in the oil and gas industry is only permitted when necessary for safety reasons. Implementation of the flare gas recovery system is BAT for new field development projects in the oil and gas industry. For large-scale modification projects on existing installations, flare gas recovery is evaluated as part of the Plan for Development and Operation of petroleum deposits (PDO). Firms/oil companies operating on the Norwegian Continental Shelf are encouraged to engage in forums and programs. Examples of such important programs and forums are the Climate and Clean Air Coalition, the Global Methane Initiative, Global Gas Flaring Reduction Partnership (GGFR) and the Oil and Gas Climate Initiative (OGCI).

6.4.4 Agriculture

Burning of agricultural residues gives emissions of a large range of standard combustion products. Emissions of NO_x, CO, NH_3, NMVOC, SO_2, particles, and the heavy metals Pb, Cd, Hg, As, Cu, and Cr, and benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, and indeno(1,2,3-cd)pyrene (PAH-4) and dioxins are included in the Norwegian inventory.

Emissions from the burning of crop residues are being calculated in accordance with a Tier 1 approach (EEA, 2019):

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① EUR-Lex - 32010L0075 - EN - EUR-Lex (europa.eu), last access 27 May, 2027.

② EUR-Lex - 32016D0902 - EN - EUR-Lex (europa.eu), last access 27 May, 2027.

③ https://www.norskeutslipp.no/en/Frontpage/, last access 27 May, 2027.

$$E_{Pollutant} = AR_{residue\_burnt} \times EF_{Pollutant}$$

Where: $E_{Pollutant}$ = emission (E) of pollutant

$AR_{residue\_burnt}$ = activity rate (AR), mass of residue burnt (dry matter)

$EF_{Pollutant}$ = emission factor (EF) for pollutant

The annual amount of crop residue burned on the fields is calculated based on crop production data for cereals and rapeseed from Statistics Norway and estimates of the fraction burned are made by the Norwegian Crop Research Institute and Statistics Norway. The fraction of crop residue burned on fields was updated in 2012 by the Norwegian Agricultural Authorities.[①] This reduced the fraction for 2011 from 7.5% to 4%. For cereals, a water content of 15% is used (Statistics Norway). The activity data are consistent with the data used in the estimations of N_2O from crop residues.

Agricultural waste burning is estimated to occur on about 80 million m² annually and almost exclusively during the spring season. This is because it is a condition for grants to alter tillage in the Regional environmental subsidy in agriculture (RMP), i.e., grants to take environmental measures on one's own farm or rental land. It is also prohibited to burn agricultural waste in many municipalities. Therefore, agricultural waste burning is on a clear downward trend in Norway.

Table 6-4 The emission factors used for PM are shown in the table below

Component	Emission factor	Unit	Data source
TSP	5.8	kg/tonnes crop residue (d.m.)	EEA, 2019
PM_{10}	5.7	kg/tonnes crop residue (d.m.)	EEA, 2019
$PM_{2.5}$	5.4	kg/tonnes crop residue (d.m.)	EEA, 2019
BC	13	% of $PM_{2.5}$	GAINS model (IIASA)

6.4.5 Wildfires

The Norwegian Directorate for Civil Protection (DSB) is the national contact towards the European Union Civil Protection Mechanism, under which countries can ask for bilateral aid, e.g., as Sweden did to fight the great forest fires of 2014 and 2018.

During the hot summer in 2018, DSB devised an ambitious media strategy with educational

① Johan Kollerud, Norwegian Agricultural Agency, unpublished material, 2012.

campaigns to increase awareness on how to help prevent forest fires, such as promoting a prohibition on barbecues (that several municipalities enforced). According to a questionnaire by DSB, 72% of the population took preventive measures as a consequence of the high fire risk that summer. Following the hot and dry summer of 2018, with its high number of forest fire incidents, DSB issued an evaluation report (in Norwegian) ① with lessons learned and recommendations for the future.

The Norwegian Meteorological Institute has developed a wildfire hazard index for 100 locations in Norway. Monitoring is also done using aircraft, drones and satellites to detect wildfires at an early stage.

6.5 Methodology for Emission Inventories Development and Reporting

6.5.1 Air pollution

There are currently two international bodies, to which official Norwegian inventory estimates of BC are reported: UNECE-CLRTAP and the Arctic Council. In both forums the reporting of national BC emissions data is not mandatory, but rather encouraged. Despite the absence of a mandatory reporting obligation, a relatively high level of reporting has been achieved in recent years. As of 2018, 41 of the 51 CLRTAP Parties, 26 of 28 EU Member States and all eight Arctic Council Member States (plus 10 of 13 Observer States) had submitted estimates for national total black emissions to some extent during recent reporting cycles (EU, 2019a; EU, 2019b).

Reporting of emissions under CLRTAP should follow the EMEP/EEA air pollutant emission inventory guidebook (EEA, 2019). Norway established a methodology for estimation of BC and OC emissions in 2013 (SSB, 2013), well before guidelines were available under CLRTAP. Emissions are estimated from specific emission factors for different fuels used by different types of machines in different sectors. In general, emissions are estimated by emissions (E) = activity level (A) × emission factor (EF), but different approaches are used to estimate emissions of BC/OC depending on the information available. Activity data are available from Statistics Norway, but emission factors for BC and OC applicable to Norwegian technologies and circumstances are not readily available. Furthermore, to ensure consistency of BC and OC emissions with emissions of $PM_{2.5,}$ the emission factors for BC and OC are estimated as shares of

① https://www.dsb.no/globalassets/dokumenter/rapporter/skogbrannsesongen_2018.pdf.

the $PM_{2.5}$ emission factors, except for wood burning and flaring.

Wood combustion in the residential sector is the largest source of PM in Norway, thus it was necessary to have emission factors that adequately reflected the emissions from this source and measurements were requested. The amount of EC and OC and the TSP emitted from Norwegian wood stoves were measured (Seljeskog et al., 2013). In a follow up study, the effect of maintenance on particulate emissions from residential woodstoves was measured (SINTEF, 2016).

Flaring of natural gas is only performed for safety reasons in Norway. The emission factors applied are taken from the literature (McEwen and Johnson, 2012). For point sources where emissions of TSP are reported to the Norwegian Environment Agency, emission of BC and OC are estimated based on fractions of the $PM_{2.5}$ emission. For sources where emissions of BC and OC are assumed to occur and information on BC and OC were not available in the literature, emissions are estimated by using a default method (SSB, 2013).

6.5.2 Climate

Reporting of emissions to the United Nations Framework Convention on Climate Change (UNFCCC) follows the 2006 IPCC Guidelines for National Greenhouse Gas Inventories and the reporting guidelines adopted by the Conference of the Parties [①,②]. In addition to the Kyotogases, Norway also reports emission of the air pollutants SO_2, NO_x, NMVOC, NH_3 and CO.

In 2019, the IPCC authorised its Task Force on National Greenhouse Gas Inventories to produce an IPCC Methodology Report on short-lived climate forcers, including BC. Norway was instrumental to this decision. The overall objective is to fill gaps in existing methodologies and to develop and disseminate an internationally approved, globally applicable methodological guidance based on existing methodologies. The work is now well underway, Norway is contributing with funds and participation in expert meetings. The process creates a platform for paying more attention to BC in future IPCC assessments and in climate negotiations under the UNFCCC.

IPCC's sixed assessment report (AR6, IPCC, 2021) has a Chapter 6 dedicated to SLCFs. Even though the IPCC has been taking the climate impact of SLCFs into account also on previous assessments, this is the first time that these pollutants have been addressed so thoroughly. Both

① http://www.ipcc-nggip.iges.or.jp/public/2006gl/, last accessed 27 May, 2022.

② Decision 247CP.19, https://unfccc.int/resource/docs/2013/cop19/eng/10a03.pdf, last accessed 27 May, 2022.

emissions, comparisons with long-lived climate gases and mitigation options are assessed in an holistic way. Many co-benefits between climate, air quality and health are pointed out and the opportunities of short-term mitigation options in combination with long-lived greenhouse gases. The summary for policy makers, Figure SPM.2, show the assessed contributions to observed warming in 2010-2019 relative to 1850-1900 for all climate forcers including BC/OC aerosols.

Assumed reductions in anthropogenic aerosol emissions lead to a net warming (primarily due to the assumed reduction of sulphur in these scenarios), while reductions in methane and other ozone precursor emissions lead to a net cooling. Because of the short lifetime of both methane and aerosols, these climate effects partially counterbalance each other and reductions in methane emissions also contribute to improved air quality by reducing global surface ozone. Reductions in GHG emissions is shown to lead to air quality improvements. In the near term, however, targeted measures to reduce air pollutants are needed in addition to the GHG measures in many polluted regions in order to achieve air quality guidelines specified by the World Health Organization .

6.5.3 Verification of Emission Inventories

As stated in the background papers on Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories[①], due to the amount of data and the number of institutions involved in the emission inventory compilation, errors and inconsistencies (e.g., across sources or in time) may easily occur. There is also always some uncertainty in the judgement of representativity of emission factors used in the inventory. Therefore, checking and verification procedures are indispensable elements of the Quality Assurance/Quality Control (QA/QC) system of the inventory management.

Checking is part of the validation of the inventory, which involves ensuring that the inventory has been compiled correctly, i.e., calculations have been done correctly and in line with guidelines and reporting instructions. Thus, validation refers to internal checks of the consistency of the inventory. Verification, on the other hand, refers to activities using external data that help to establish reliability for the intended applications of the inventor, for example, external methods to check the truth of the inventory including comparisons with reference calculations, estimates made by other bodies, atmospheric concentrations or external review.

During past years, several quality assurance and quality control procedures for the preparation of

① https://www.ipcc-nggip.iges.or.jp/public/gp/gpg-bgp.html, last accessed 27 May, 2022.

the national emission inventory have been established in Norway. Statistics Norway made its first emission inventory for some gases in 1983 for the calculation year 1973. The emission estimation methodologies and the QA/QC procedures have been developed continuously since then. Norway has implemented a formal quality assurance/quality control plan, which covers the reporting of long-range transboundary air pollution as well as greenhouse gases. A detailed description of this is presented in Annex V in the National Inventory Report for Norway (Norwegian Environment Agency et al., 2021b).

In general, the final inventory data provided by Statistics Norway are checked and verified by the Norwegian Environment Agency. Two verification studies, considered particularly relevant for OC/BC emissions, are briefly described below:

(1) In 2006, the Nordic Council of Ministers initiated a new project that was finalised in 2010. In this study, the Nordic PM emission inventories were compared, and for the most important sources—residential wood burning and road transport—a quality analysis was carried out based on PM measurements conducted and models used in the Nordic countries. The objective was to increase the quality of the national PM inventories. The ratio between the reported emissions of PM_{10} and $PM_{2.5}$ was calculated for each country. The completeness of the inventories was assessed and it was found that the PM emission inventories generally were complete and that the sources reported as "Not Estimated" were expected to have only minor contributions to the total PM emissions. The variability of emission factors for residential wood combustion was discussed and it was shown that the emission factors can vary by several orders of magnitude. For residential wood combustion, differences can probably be attributed to whether the emission measurements are carried out in the hot flue gas, either in-stack or out-stack, or if the measurements are carried out after the semi-volatile compounds have condensed.

(2) In 2017, a project, financed by the Nordic Council of Ministers, went through the emission factors for Short Lived Climate Pollutants (SLCP) emissions from residential wood combustion in the Nordic countries. The overall objective of this project was to improve the Nordic emission inventories of SLCPs (Kindbom et al., 2018). The project included comparisons of emission factors for EC, OC, $PM_{2.5}$, CH_4 and non-methane volatile organic compounds (NMVOC). Emission factors were developed for standard combustion conditions, as well as for "bad" combustion conditions. Emission measurements were conducted on residential wood burning appliances, boilers and stoves representative of the Nordic countries. Generally the older technologies exhibited higher emission levels than more modern types of equipment. Results from measurements showed, e.g., that the modern stoves were sensitive to moist fuel, where

emissions of, for example, $PM_{2.5}$ and OC increased in the order of 5-8 times compared to when fired with standard fuel. To improve the national emission inventories of SLCPs, the large sensitivity to operational conditions (moist fuel and part load) needs to be taken into consideration in national emission inventories, where “real life” emissions are estimated. In order for national emission inventory results to be comparable, a harmonisation of emission factor levels is needed, unless there are real differences between the countries.

Annual review of individual inventories of each Annex I Party to the UNFCCC became mandatory in a 2003 decision 19/CP.8).[①] The UNFCCC Annex I inventory review guidelines, revised in 2014 (decision 13/CP.20), ensure that the Conference of the Parties (COP) is provided with an objective, consistent, transparent, thorough, and comprehensive technical assessment of the quantitative and qualitative inventory information submitted annually by Annex I Parties. Annual review ensures that adequate consideration is given to recalculations and emission trends over time.

The review of greenhouse gas (GHG) inventories comprises two stages. Each stage complements the previous one.

(1) Initial assessment by the Secretariat: a standardised set of data comparisons mainly based on the common reporting format (CRF) data, aiming to determine that each Annex I Party has submitted a consistent, complete, and timely annual inventory in the correct format, including the national inventory report (NIR) and the CRF tables, and to identify issues for further consideration during the review of individual inventories. Status reports for each Party are published at this stage while assessment reports are available to Parties and expert review teams (ERTs).

(2) Review of individual annual inventories by ERTs: ERTs examine the data, methodologies and procedures used in preparing the national inventory. ERTs are required to pay particular attention to key categories (categories with significant influences of the country’s inventory), areas of the inventory where issues have been identified and recommendations made in previous reviews or stages of the reviews, progress in the implementation of the planned improvements and where recalculations or other changes have been reported by the Annex I Party. This is the most detailed review stage. Individual review reports are published for each Party.

Three operational approaches may be used during the second stage of the technical review, namely desk reviews, centralised reviews or in-country reviews. Review reports are prepared

① https://unfccc.int/process-and-meetings/transparency-and-reporting/reporting-and-review-under-the-convention/greenhouse-gas- inventories-annex-i-parties/review-process, last accessed 27. May 2022.

and published on the Secretariat web site.[①]

6.5.4 Uncertainties

The Norwegian Research Institute SINTEF performed an expert evaluation of measurement uncertainty of emission factors from Norwegian wood stoves for particles (TSP and PMT), EC and OC. The PM uncertainty was in general found to be about 7%. The uncertainty for EC was found to be higher for newer stoves (45%) than for older stoves (27%), which could be because the test measurements resulted in larger corrections in EC measurements from newer stoves than from the older ones.

The uncertainty for EC and OC in the SINTEF study may be mainly because the Norwegian measurement standard (NS 3058 and NS, 3059) overestimated the amount of carbonaceous particles on the filters analysed. This resulted in the laser instruments, used to measure EC and OC, not giving trustworthy results in most cases. The filters were then analysed with use of thermal-optical methods (NIOSH protocol 5040, 1999), it was necessary to apply corrections to the results. The resulting emission factors represent an improvement in relation to previously used factors for PM, but there is still a need for more research and improvements to the Norwegian Standard for emission measurements of EC and OC.

6.6 Integrated Analysis of Climate and Air Pollution

In 2013, NEA, on behalf of the Ministry of the Environment, published an integrated assessment of the short- and long-term climate, health and environmental impacts of mitigation measures for Norwegian emissions of short-lived climate forcers (SLCFs) (Norwegian Environment Agency, 2013). Emissions of SLCFs might either warm or cool the atmosphere. As the global climate impacts of SLCFs depend on where the emissions take place. NEA modelled the global climate effects of Norwegian emissions. A new climate metric, GTP10-Norway, was adopted to assess the short-term climate impacts of measures and the cost-efficiency (cost per unit of CO_2equivalent) of measures in a short-term perspective. The study found that Norway's BC emissions have approximately 1.5 times higher climate impacts per tonne of emitted BC than the global average, mostly due to Norway's proximity to the Arctic and to BC deposition on snow and ice (Hodnebrog et al., 2013). The results also showed the importance of reducing Norway's

① https://unfccc.int/process-and-meetings/transparency-and-reporting/reporting-and-review-under-the-convention/greenhouse-gas-inventories-annex-i-parties/inventory-review-reports-2020, last accessed 27. May 2022.

BC emissions due to the health impacts of $PM_{2.5}$ in cities and that the inclusion of the health benefits of the assessed measures substantially reduced their socio-economic cost. Finally, the assessment pointed out that it is important to assess the net climate effect of emission measures. The 2013 study did not assess greenhouse gas measures, but, given the high short-term climate effect of CO_2, hypothesised that such measures could be as efficient in reducing SLCFs as targeted SLCF-measures. Therefore, integrated assessments of both short- and long-lived climate pollutants could be useful in giving a complete picture of the net climate effect, as well as health impacts, of the measures.

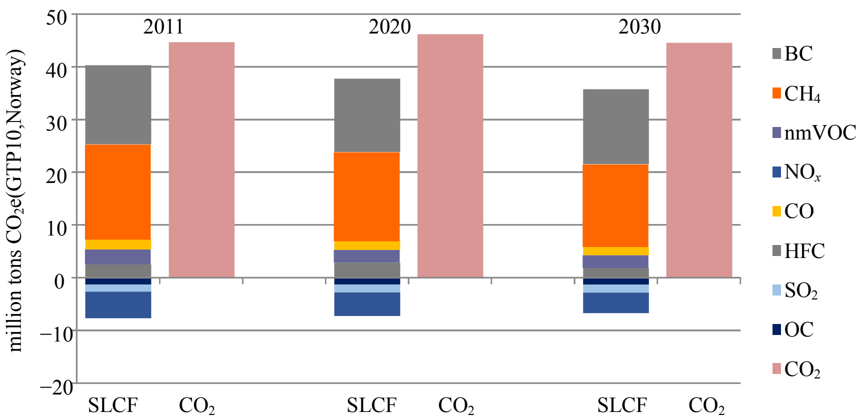

Figure 6-6 Global climate effect of Norwegian SLCF emissions compared to that of Norwegian CO_2 emissions in 2011, 2020, and 2030. (Source: Norwegian Environment Agency, 2013)

To acknowledge this, the Norwegian Environment Agency has subsequently performed several integrated studies including of both SLCFs and greenhouse gases. In a later study, the Norwegian Environment Agency highlighted that BC could be reduced by 33% and methane by 16% from 2013 to 2025 if the analyzed measures are implemented, compared to respectively 28% and 9% without such measures (Norwegian Environment Agency, 2019). The main measures to reduce BC are *accelerated introduction of new stoves* and *best stoves and pellet burners,* and *electrification of ferries and passenger ships*. Due to methane emission reductions, measures to reduce emissions from wood burning by replacing high emitting stoves with low emitting stoves are now regarded as climate measures as well as health measures. New road transport measures do not result in significant reductions of BC emissions because measures already included in climate and environment policies are expected to reduce BC emissions to almost zero by 2030. This could be different in other countries with a different emission profile.

In 2020, the Norwegian Environment Agency and other agencies published a mitigation analysis of a range of measures that could be used to achieve Norway's 2030 climate target for non-ETS emissions and in the land use, land-use change and forestry (LULUCF) sector (Klimakur 2030, 2020). The Agency subsequently analyzed the short-term climate impacts and the health and environmental co-benefits of these measures. The results show that many of the measures will have substantial climate impacts both in the long term and short term and will also provide health and other co-benefits (Norwegian Environment Agency, 2021c).

Integrated assessments of climate measures are important to highlight measures that have important short-term effects but fail to contribute substantially to long-term climate goals. This type of analysis assists in compiling a portfolio of measures that contribute both to reducing the short-term rate of warming as well as safeguarding the long-term perspectives of the Paris Agreement. In Norway, health benefits have traditionally been included in greenhouse gas analyses. The short-term climate effects of those measures have become increasingly a part of their regular evaluation.

Integrated climate and air quality studies have also had an impact on Norway's international climate policy. For example, Norway included a description of BC and OC in its Seventh National Communication to the United Nations Framework Convention on Climate Change (UNFCCC) (Norwegian Ministry of Climate and Environment, 2018) and Norway's submission to the 2018 UNFCCC Talanoa Dialogue called for the application of a multiple-benefit methodology. Interest in performing integrated analysis of climate and air pollution has been increasing in many countries. Thus Norway has played an active role in, e.g., the Climate and Clean Air Coalition (CCAC), and has published results together with, e.g., Tsinghua University (UNEP, 2019).

6.7 Health Effects[①]

The Norwegian Institute of Public Health (NIPH) has found that BC is often used as an indicator in population exposure studies. In general, it is found that short-term exposure has an association with mortality and hospital visits for persons with asthma and heart and lung diseases. For long-term exposure to BC there is a correlation to death and respiratory problems in addition to reduced lung function. The risk estimates for BC in regard to mortality and sickness are higher

① Source to this section is NIPH: https://www.fhi.no/nettpub/luftkvalitet/temakapitler/svevestov/, last accessed 27. May 2022 than for $PM_{2.5}$ and PM_{10}, for both short- and long-term exposure.

Combustion particles are measured with various methods and are described in population exposure studies as BS, BC or EC. It is common in such studies to use the BS to EC conversion of 10 μg/m^3 BS equals 1.1 μg/m^3 EC. In such studies, WHO recommends that BC is a better indicator for harmful components, especially for traffic, than the total mass of $PM_{2.5}$ or PM_{10}.

However, there exist few experimental studies on the effect of $PM_{2.5}$ compared to BC/EC or in which the composition of the particles is characterized. The few studies that do exist show that BC/EC does not in itself result in toxicological effects, but that the effects of $PM_{2.5}$ are rather attributed to various organic compounds or metals bound to BC/EC.

Mills et al. (2011) performed a study on the effect of diesel exhaust on the heart and respiratory system, in which the health effects were limited with the use of particle filters. Exposure of healthy volunteers to two hours of ultrafine carbon particles in an ambient concentration (10 μg/m^3) showed changes in the surface markers on blood cells, something that can represent an early stage of an inflammatory reaction (Framton et al., 2006). Further studies of ultrafine particles (EC 50 μg/m^3 over two hours) show effects to blood circulation (inhibited vasodilation) in healthy individuals (Shah et al., 2008).

NIPH has compiled a figure that shows the various health impacts from PM (Figure 6-7).

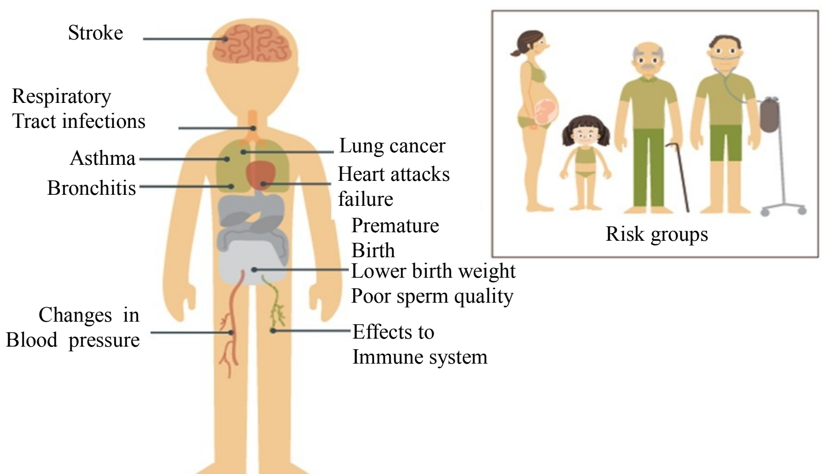

Figure 6-7 Overview of health effects from PM. (Source: NIPH)

6.8 Effectiveness of Policies

Norway has 24 national environmental goals, several of them dealing with climate and air pollution. The goals are detailed on the Environment Norway web pages.[1] In order to see if the country is on track to attain these goals, Norway has developed environmental indicators. The environmental indicators for climate and air pollution are related to status for and trends in emissions of GHG and concentrations of harmful components.

6.8.1 Emission Trends and Projections

Air pollution and climate policies combined have resulted in large reductions of BC and OC emissions in Norway. The BC emission trends per sector are shown in Figure 6-1. The emissions of BC amounted to 2,841 tonnes in 2019, a total reduction of 41% since 1990 and of 6% since 2018. In 2019, the most important source of emissions was "other combustion" (NFR 1A4 and 1A5), contributing to 37% of the total emissions. From this category, 75% of emissions originated in 2019 from residential stationary plants, primarily due to wood combustion in private households. From 1990 to 2019, emissions from residential stationary plants were reduced by 28%.

In 2019, the second most important source of emissions was transport. It contributed to 34% of the total BC emissions. The greatest share of emissions within the transport sector, 57%, stems from navigation with 57% of the emissions. That is followed by light-duty vehicles, passenger cars and heavy-duty vehicles and buses, contributing to 11%, 12% and 10%, respectively. From 1990 to 2019, emissions from navigation increased by 15%, while emissions from passenger cars increased by 28%. Emissions from light and heavy-duty vehicles have been reduced by 51% and 86%, respectively, since 1990.

Combustion in the energy industries, which in 2019 accounted for 13% of total BC emissions, increased by 269% since 1990 due to increased production. The greatest sources of emissions within this category are public electricity and heat production, and manufacture of solid fuels and other energy industries, which contributed 43% and 56%, respectively, to the sector emissions in 2019. From 1990 to 2019, BC emissions from these sub-sectors increased by 2914% and 134%, respectively (NEA, 2021a).

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① www.environment.no

Flaring is acknowledged as an important source of BC emissions by many researchers, e.g., in the EU-funded Action on BC in the Arctic (Annex Ⅲ). There is little flaring per unit of oil and gas produced in Norway compared to other oil producing countries. The Norwegian Oil and Gas Association estimated that the global average amounts of flare gas per produced unit were 12 times higher than in Norway in 2017. The amounts of flared gas per year decreased by 43% from 1990 to 2019, and BC emissions from flaring decreased by 45% in the same time period. Although amounts of flare gas and emissions have fluctuated, there seems to be a consistent downwards trend in recent years (Figure 6-8 and Figure 6-9).

Figure 6-8 Flare gas amounts (PJ) in Norway 1990-2019 (including Hammerfest LNG)
(Source: Footprint database with reported data from the operators)

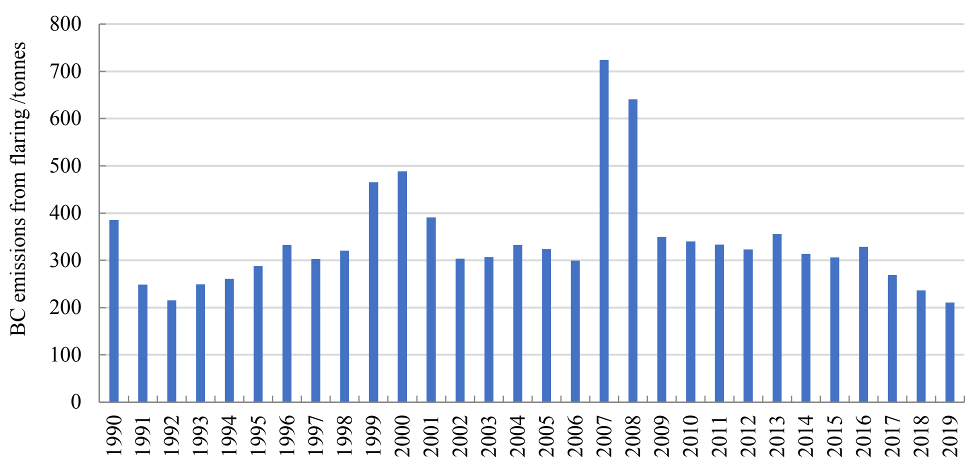


Figure 6-9 BC emissions from flaring in Norway 1990-2019 (tonnes)
(Source: Footprint database with reported data form the operators)

The prohibition of flaring except for safety reasons, together with the CO_2 tax and the EU ETS has resulted in improvements in technology and emission-reducing measures, e.g., flare gas recovery (closed flare system). In the late 1990s, several measures were implemented in Norway to reduce continuous flaring through development and use of new technologies (Pederstad et al., 2015). These measures were considered economically profitable for businesses. A large portion of the remaining reduction potential was then associated with limiting flaring at start-up and shutdown of installations/facilities and pressure relief of equipment during maintenance and breakdowns. Low oil prices and limited investment activity resulted in fewer measures implemented during the early 2000s. In recent years, however, there has been renewed attention to flaring and its emissions, both politically and within industry.

Norwegian policies and measures are expected to also reduce emissions of BC in the future. According to projections in 2018, when the most important aggregated (GNFR[①]) sector was "Other Stationary Combustion" (mainly residential wood burning) emissions of BC are expected to be reduced further towards 2030 (Figure 6-10).

The second and third most important GNFR sectors, "Shipping" and "Road Transport" are also projected to be further reduced towards 2030.

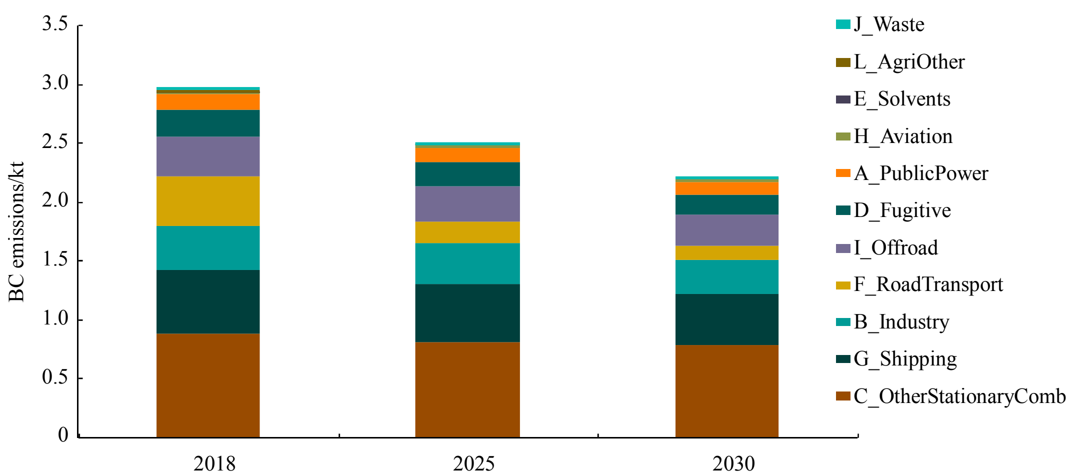


Figure 6-10 Historical emissions of BC in Norway 2018, and projections for 2025 and 2030 (Source: Norwegian Environment Agency and Statistics Norway)

① Gridded Nomenclature for Reporting (GNFR) is the most aggregated format used to report emissions to the Convention on Long-range Transboundary Air Pollution (CLRTAP).

6.8.2 Health Impact Trends

The EEA has estimated that in 2018 there were 1,400 premature deaths in Norway from $PM_{2.5}$ exposure (Table 6-5). Premature deaths from $PM_{2.5}$ overall in Europe have been greatly reduced in recent years (Figure 6-11). In Norway, the relative reduction in premature deaths from $PM_{2.5}$ in 2019 compared to 2009 is between 10% and 20% (EEA, 2018).

Table 6-5 Premature deaths from air pollution exposure in Norway compared to the EU, 2018 (Source: EEA, 2018)

Country	Population ($\times 10^3$)	Annual mean ($PM_{2.5}$)	Premature Deaths ($PM_{2.5}$)
Norway	5,296	6.40	1,400
EU-28	507,558	13.20	379,000

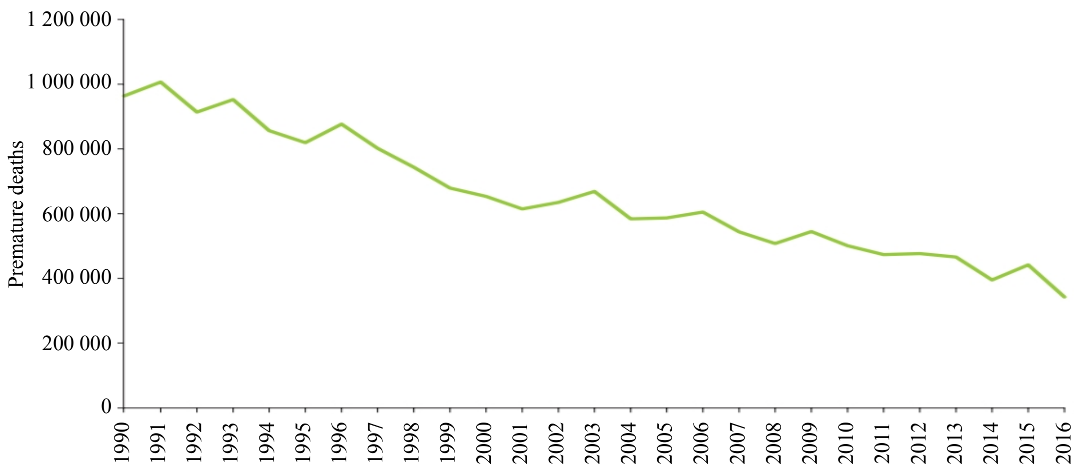


Figure 6-11 Trends in premature deaths from $PM_{2.5}$ in Europe, 1990-2016 (Source: EEA, 2018)

7 How to Reduce BC/OC Emission in China in the Future

China currently focuses its air pollution control efforts on $PM_{2.5}$ and O_3 pollution and is yet to formulate specific emission control policy targeting BC/OC emissions. Since 2013, the Chinese government has issued a series of policies including the *Action Plan on Prevention and Control of Air Pollution* (2013-2017) (The State Council of the People's Republic of China, 2013) and the *Three-Year Action Plan to Fight Air Pollution* (2018-2020) (The State Council of the People's Republic of China, 2018) for the purpose of improving ambient air quality, especially reducing the concentration of $PM_{2.5}$ and implemented a number of measures including elimination of outdated production capacity, energy transformation for clean heating, in-depth control of pollution in key industrial sectors, and prevention and control of mobile source pollution. Though not specifically designed for BC/OC emission reduction, these measures still have a positive effect on the reduction of BC/OC emissions as important $PM_{2.5}$ precursors. To provide a better understanding of China's efforts on controlling BC/OC emissions, this section summarizes China's air pollution control policies and measures for key BC/OC emitting sectors such as coal-fired boilers, residential bulk coal, and motor vehicles. To be noted, we've reviewed the latest policy implementation progress till 2020, the base year of the scenario analysis is 2018 considering the project target assignment and the availability of relevant historical emissions and other basic data.

7.1 Air Pollution in Northern China

7.1.1 Overall Air Quality Sees Improvement

7.1.1.1 National Air Quality

Since 2013, with the implementation of the *Action Plan on Prevention and Control of Air Pollution* and the *Three-Year Action Plan to Fight Air Pollution*, China has achieved positive results in air pollution prevention and control, with the overall ambient air quality seeing substantial improvement. Several studies (Hammer, et al., 2021; Shi, Song et al., 2021; He, Pan et al., 2020; Le, Wang et al., 2020) suggested that the decrease in anthropogenic activity level had led to positive effects in improving air quality of China during the COVID-19 lockdown period in 2020. The annual average measured $PM_{2.5}$ concentration of 337 cities at or above prefecture level (hereinafter referred to as the "337 cities") in 2020 was 33 μg/m^3, down by 29.1% from 2015; at the same time, the annual average measured concentrations of SO_2, CO, PM_{10} and NO_2 dropped by 10%-56%; compared with the same meteorological conditions in previous years, the peak concentration, pollution intensity, duration and scale of impact on days of heavy pollution were substantially reduced; the number of cities meeting air quality standards in 2020 was nearly three times that of 2015. Unlike the other five criteria air pollutants, O_3 concentration

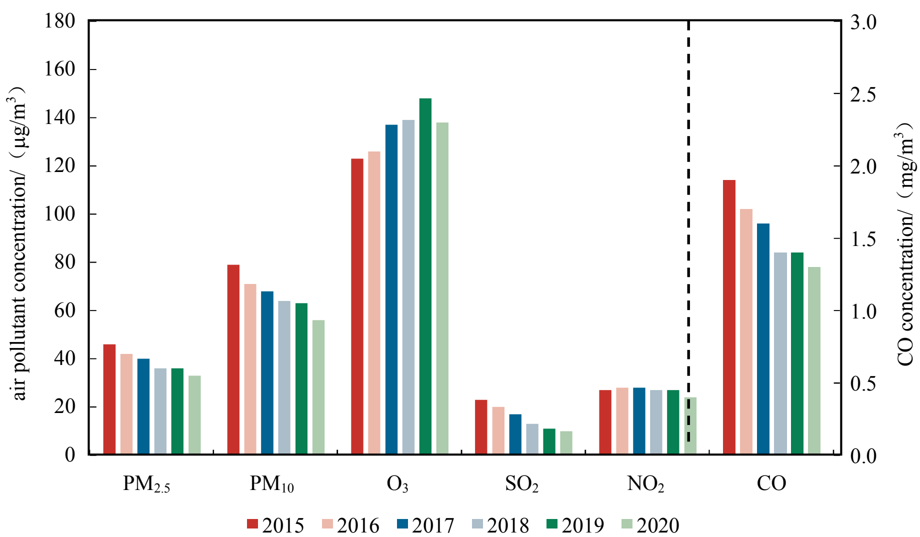


Figure 7-1 Changes in national annual average measured concentrations of major air pollutants from 2015-2020 in the 337 cities

increased. The national average measured 90th percentile of the maximum daily 8-hour average O_3 concentration (the official metric for evaluating annual O_3 pollution status in China) increased by 12.2% from 2015 to 2020. Non-optimized reduction rate between NO_x (nitrogen oxides) and VOC (volatile organic compound) emissions, as well as the weakened aerosol uptake of hydroperoxy radicals due to $PM_{2.5}$ reduction have been proposed as reasons for the increment in O_3 concentrations.

7.1.1.2 Air quality in the northern part of China

The northern part of China is one of the most severely polluted areas in China, where the industrial structure is dominated by heavy industry, energy structure is based on coal, transportation mainly relies on highways. In recent years, with the intensified implementation of the *Action Plan on The northern part of China Prevention and Control of Air Pollution* and the *Three-Year Action Plan to Fight Air Pollution*, this part of the country has experienced industrial structure transformation and upgrading as well as continuous adjustment and optimization of its energy structure, transportation structure and land use structure, with the overall air quality improved. From 2015 to 2020, in Beijing-Tianjin-Hebei and the surrounding areas (hereinafter referred to as the "2+26 cities" region), the annual average measured concentrations of various pollutants, except O_3, showed a downward trend (Figure 7-2). In 2020, the concentrations of SO_2, PM_{10}, $PM_{2.5}$ and CO dropped by. 35%-70% compared to 2015, and the concentration of $PM_{2.5}$ was down by 35%. The annual average

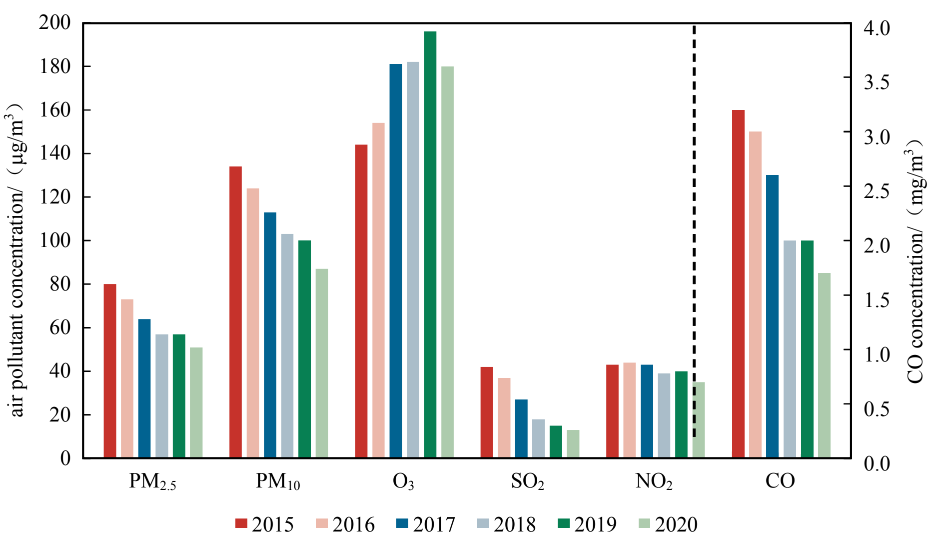


Figure 7-2 Changes in measured concentrations of major air pollutants in the "2+26 cities" region in 2015-2020

measured concentration of $PM_{2.5}$ in Beijing dropped by more than 50%, a remarkable progress. Starting from 2016, all of the 2+26 cities met the national annual standards for SO_2 and CO concentration (that is, 60 μg/m^3 for SO_2 and 4 mg/m^3 for CO; national air quality standard issued by the Ministry of Ecology and Environment, GB 3095—2012)

The number of days with measured good air quality (air quality index (AQI) less or equal to 100) in the "2+26 cities" region shows a slight increasing trend (Figure 7-3). The ratio of clean days increased from 53.6% in 2015 to 63.5% in 2020. Based on observation-based statistics, the number of days with heavy pollution (AQI＞200) shows a decreasing trend. The ratio of days of heavy pollution or above (see Table 7-1 for the definition) was down from 8.9% in 2015 to 3.5% in 2020.

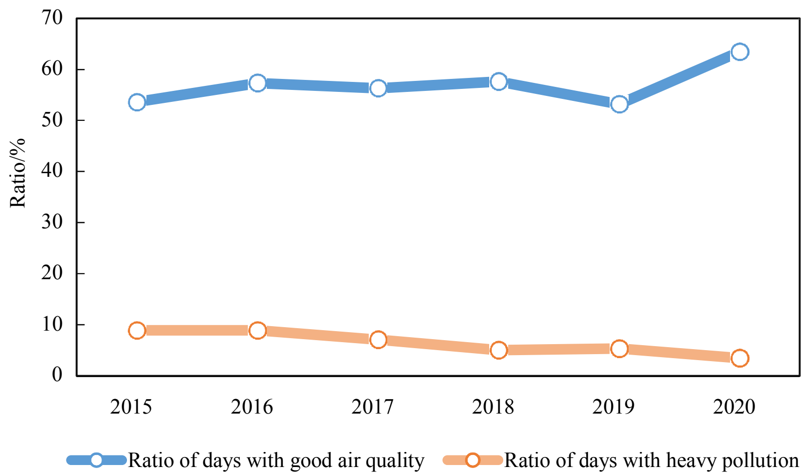


Figure 7-3 Changes in ratio of clean days and in ratio of days with heavy pollution or above in the "2+26 cities" region in 2015-2020. Days with good air quality refers to days with air quality index (AQI) lower or equal to 100; days with heavy pollution refers to days with AQI greater than 200

7.1.2 Air Pollution Remains Challenging

With the successive implementation of a series of measures, the compound air pollution characterized by high concentration of $PM_{2.5}$ in North China has been substantially improved, but the region is still facing huge air pollution challenges. In 2020, the average annual $PM_{2.5}$ concentration in the "2+26 cities" region was 51μg/m^3, which was still at a high level, equivalent to 1.46 times the national air quality standard (GB 3095—2012, 35μg/m^3) and 1.57 times the national average annual $PM_{2.5}$ concentration in the 337 cities across the country (Figure

7-4), keeping this area the most severely polluted area by particulates in China. In addition, this area is still one of the regions in China with frequent occurrence of heavy-pollution episodes. The problem of heavy pollution characterized by high concentrations of particulates in autumn and winter is still prominent. The $PM_{2.5}$ concentration in autumn and winter is about twice that in spring and summer. At the same time, due to the large consumption of fossil energy in the area, the control of greenhouse gas emissions is facing tremendous pressure. It will be an important challenge for the northern part of China to coordinate the improvement of air quality and the control of greenhouse gases in the management of the atmospheric environment.

Table 7-1 Annual limits for major air pollutant in the national air quality standard (GB 3095—2012)

Air pollutants		Limit requirements	
		For Class I area	For Class II area
SO_2/(μg/m^3)	Annual average	20	60
NO_2/(μg/m^3)	Annual average	40	40
$PM_{2.5}$/(μg/m^3)	Annual average	15	35
PM_{10}/(μg/m^3)	Annual average	40	70
O_3/(μg/m^3)	90 percentile of daily maximum 8-hour average	100	160
CO/(mg/m^3)	95 percentile of daily average	4	4

Note: Class I area refers to nature reserves, scenic spots and other areas that need special protection; Class II area refers to residential areas, mixed commercial and residential areas, cultural areas, industrial areas and rural areas.

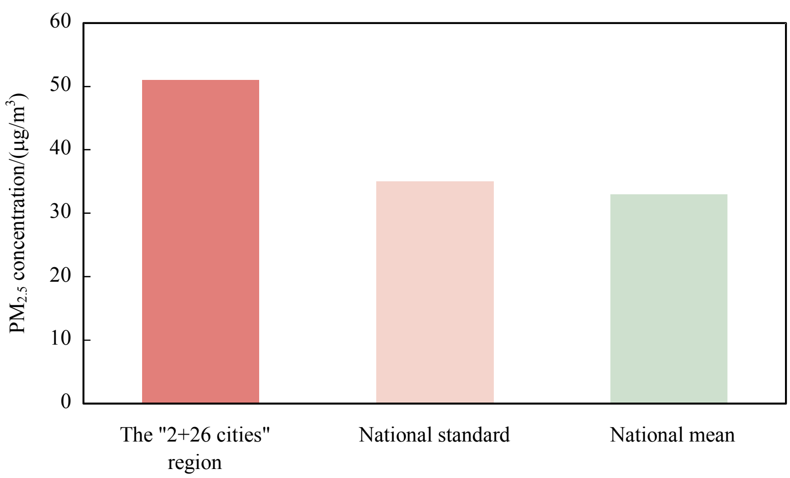


Figure 7-4 Average annual $PM_{2.5}$ concentration in the “2+26 cities” region in 2020

7.2 Review of Major Relevant Polices on Prevention and Control of Air Pollution

In order to improve the ambient air quality, the Chinese government has successively promulgated and implemented two important policies, the *Action Plan on Prevention and Control of Air Pollution* and the *Three-Year Action Plan to Fight Air Pollution* since 2013, and implemented supporting policies including elimination of outdated production capacity, in-depth control of pollution in key industrial sectors (e.g. thermal power, iron and steel, cement), control of coal-fired boilers, clean heating and prevention and control of mobile source pollution. The policies involve the industrial, residential, transportation and many other sectors. This series of measures directly promoted the continuous transformation of China's energy structure, with the proportion of coal in primary energy consumption steadily declining from 67.4% in 2010 to 56.8% in 2020, while primary electricity and energy other than coal, oil and natural gas increasing from 9.4% in 2010 to 15.9% in 2020 (NBS data) (Figure 7-5).

Table 7-2 Summary of major air pollution control measures implemented since 2013

Air pollution control measures	Description
Elimination of outdated production capacity	Include overcapacity industry capacity control, elimination of backward capacity and other measures
Scattered pollution enterprise governance	Refer to the rectification of "scattered pollution" enterprises with incomplete licenses, illegal construction, illegal operation, environmental pollution and non-compliance with local industrial layout planning
Ultra low emission transformation of coal-fired power plant	Refer to the ultra-low emission transformation of coal-fired power plants to meet the new emission limit standards
Upgrading of industrial source end of pipe control	Refer to the upgrading and transformation of end of pipe control measures such as desulfurization, denitration and dust removal in key industries such as steel, cement and glass
Renovation of coal-fired and gas-fired boilers	Include the elimination of small coal-fired boilers, efficient desulfurization of coal-fired boilers, upgrading and transformation of dust collectors, and low nitrogen combustion transformation of gas-fired boilers
Civil fuel cleaning	Include coal washing and processing, clean treatment of bulk coal, coal to gas, coal to electricity and other measures
Traffic structure optimization and emission control	Include measures to control the total number of motor vehicles, eliminate yellow-label vehicles and old vehicles, and renovate non road machinery
Comprehensive control of dust sources	Include construction site dust pollution control, road dust pollution control, port and wharf dust pollution control and other measures

Air pollution control measures	Description
VOCs control	Refer to VOCs control work in industries involving VOCs emissions
Comprehensive management of agricultural resources	Include the management of livestock and poultry breeding industry and the improvement of comprehensive utilization rate of chemical fertilizer

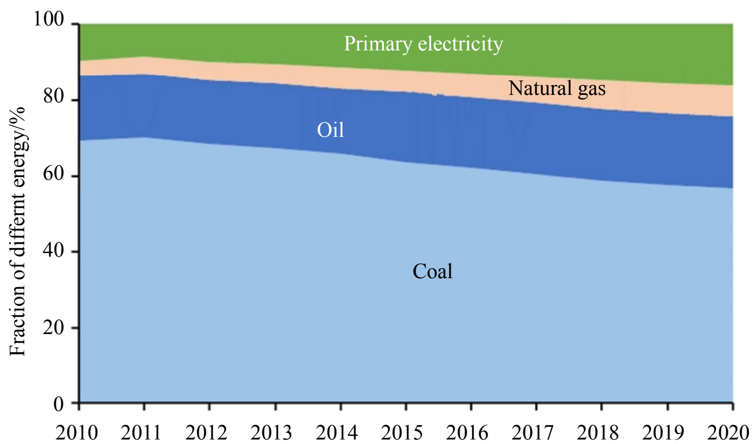


Figure 7-5 Changes in proportion of energy sources in energy consumption in China 2010-2020

As shown in the following, to better understand the implementation progress of BC control related policies in northern China, we've reviewed the air pollution control policies targeting boilers, residential coal and vehicles, which are closely connected to BC emission control at the same time.

7.2.1 Elimination of Small Coal-fired Boilers

7.2.1.1 Policy Implementation Progress

In 2013, China fully initiated the elimination of small coal-fired boilers with a capacity at or below 10 ton/hour in built-up areas of cities at or above prefecture level. By 2018, a total of more than 230,000 small coal-fired boilers (widely used across the country to provide heat steam to various small industrial and residential facilities) had been shut down nationwide. At the same time, the Chinese government banned the construction of new coal-fired boilers with a capacity at or below 20t/h in built-up areas of cities at or above prefecture level. In other areas, no new coal-fired boilers at or below 10t/h shall be built in principle. In the “2+26 cities” region, more than 50,000 small coal-fired boilers were eliminated. The elimination of small coal-fired boilers effectively improved the overall efficiency of boilers, because newly-built larger boilers generally have higher combustion efficiency and because advanced end-of-pipe control measures could be applied to larger boilers. The elimination of small coal-fired boilers thus helped reducing BC and OC emissions as well as CO_2 emissions, thereby helping to cope with climate change.

7.2.1.2 Case Introduction of Typical Provinces

Shandong is an important energy consuming province in China, where the energy consumption structure is dominated by coal and the proportion of high pollution and high consumption industries is relatively high. In order to control the air pollutant emissions from boilers, Shandong promulgated a series of local programs, primarily including the "*Implementation scheme of comprehensive improvement project for energy conservation and environmental protection of coal-fired boilers in Shandong Province*" in 2015 and the "*Guidance on strictly controlling total coal consumption and promoting clean and efficient utilization*" in 2019.

Objectives:

Based on the *"Implementation scheme of comprehensive improvement project for energy conservation and environmental protection of coal-fired boilers in Shandong Province"* in 2015, the objectives of the program mainly included:

"By 2018, promote high-efficiency boilers with the scale of 30,000 t/h and increase the market share of high-efficiency coal-fired boilers to 40%. Complete the task of eliminating backward coal-fired boilers assigned by the state council. Complete the energy-saving transformation of coal-fired boilers with the scale of 20,000 t/h. Accelerate the research and development of high-efficiency boilers, cultivate a number of backbone enterprises, increase the average operation efficiency of coal-fired industrial boilers by 5 percentage points on the basis of 2013, and form an annual energy-saving capacity of 4.5 million tons of standard coal. By implementing all the control measures, reduce 112,500 tons of soot, 144,000 tons of sulfur dioxide and 27,000 tons of nitrogen oxides."

Based on the *"Guidance on strictly controlling total coal consumption and promoting clean and efficient utilization"* in 2019, the objectives of the program relevant to boiler governance mainly were:

"In about five years, the province's coal consumption will strive for a reduction of 50 million tons. The seven cities involved in "2+26" cities strive to basically eliminate coal-fired boilers with a steam capacity of less than 35 t /h by the end of 2019 and achieve total elimination by the end of 2020."

Main measures:

Based on the *"Implementation scheme of comprehensive improvement project for energy conservation and environmental protection of coal-fired boilers in Shandong Province"* in 2015, the main measures implemented in Shandong for boiler pollution control during 2015-2018 mainly included:

(1) Promote the application of high efficiency boilers. Vigorously promote the high-efficiency boiler products that have entered the national announcement catalogue and the promotion catalogue of key energy-saving technologies, products and equipment in Shandong Province. For new reconstruction and expansion of fixed asset investment projects and government procurement projects, priority shall be given to products listed in the promotion catalogue of high-efficiency boilers or with energy efficiency grade of grade 1.

(2) Eliminate outdated coal-fired boilers. Outdated old boilers that fail to pass the energy efficiency test and have no transformation value shall be eliminated according to law. Except for those necessary to be reserved, all coal-fired boilers of 10 t/h and below will be eliminated in urban built-up areas. Strictly control obsolete boilers to re-enter the market and prevent outdated boilers from being moved to rural or remote areas for continued use.

(3) Transform inefficient coal-fired boilers. Strengthen the energy-saving transformation of boiler combustion equipment, auxiliary equipment and supporting facilities, focusing on the transformation of coal-fired boilers with more than 10 t/h in urban built-up areas and within the coverage of thermal pipe network and the cogeneration or central heating in industrial parks and industrial concentrated areas. Briquette or clean coal transformation shall be carried out in other areas not covered by the heating and gas supply network. Implement the industrial green power plan, promote the application of solar industrial boilers and actively use new energy to auxiliary heat the boiler feed water. Clean coal, biomass and natural gas are used to replace raw coal to reduce pollutant emission.

(4) Strengthen boiler operation management. Strengthen the energy efficiency test of boilers. Promote the standardized management of safety, energy conservation and environmental protection of boiler system, carry out standard pilot demonstration and promote the construction of 50 benchmark boiler rooms. Encourage online energy-saving monitoring and diagnosis of boilers. Strengthen the management of water treatment and fuel in boiler installation and operation, and

improve boiler efficiency. Improve the level of boiler operators, and organize the training of boiler energy-saving operation skills in process operation procedures and post operation procedures.

(5) Improve boiler pollution control level. According to the requirements of comprehensively renovating small coal-fired boilers, in principle, no new coal-fired boilers shall be built in the built-up areas of cities above the prefecture level. Newly produced and installed coal-fired boilers shall be equipped with efficient pollution control facilities. Improve the pollution control level of in-service coal-fired boilers to achieve fully up to standard emission. Coal fired boilers of 20 t/h and above shall be equipped with on-line monitoring devices and networked with local environmental protection departments.

(6) Promote the industrialization of high-efficiency boilers. Strengthen the research and development of basic, cutting-edge and common key technologies for boiler energy conservation and environmental protection, overcome key technologies such as efficient combustion, efficient waste heat utilization, automatic control and pollution control, and increase support for the promotion and application of scientific and technological achievements.

(7) Promote the optimization and adjustment of fuel structure. Strengthen the quality management of coal, realize the utilization of coal by quality and classification, and promote clean coal combustion. Promote the use of washed coal. Coal-fired boilers shall not directly burn raw coal with high sulfur and high ash. Domestic coal and other small coal-fired facilities shall give priority to the use of briquette with low sulfur and low ash and added sulfur fixing agent. Build coal storage and distribution base, carry out pilot demonstration of centralized coal blending and logistics supply and improve coal washing and processing capacity.

Based on the "*Guidance on strictly controlling total coal consumption and promoting clean and efficient utilization*" in 2019, the main measures implemented in Shandong for boiler pollution control during 2019-2020 mainly included:

(1) Improve the calorific value of coal and gradually increase the calorific value standard of coal from 3700-4300 kcal to about 5000 kcal in 3-5 years.

(2) Strengthen the transformation of old equipment, promote new coal water slurry and pulverized coal boilers, and improve energy efficiency.

(3) The seven cities in Shandong involved in "2+26" cities strive to basically eliminate coal-fired boilers with a steam capacity of less than 35 t/h by the end of 2019 and achieve total elimination by the end of 2020.

7.2.2 Residential Clean Heating

7.2.2.1 Policy Implementation Progress

In 2017, China launched the clean heating program for the northern part of the country in winter, replacing the coal used by residents for heating with natural gas, electricity, solar energy, geothermal energy and other energy sources, accompanied by safeguarding measures in finance, price and energy supply. By the end of 2020, the project had completed the bulk coal substitution in 25 million households, the pilot cities had achieved full coverage of the Beijing-Tianjin-Hebei and its surrounding areas and the Fenwei Plain (Chinanews, 2021). Residential sources, especially rural residential sources, are one of the most important sources of BC and OC emissions. The residential clean heating policies and measures substantially reduced BC and OC emissions in northern China. During the 14th Five-Year Plan period, the Chinese government will continue to promote clean heating in the northern part of the country so as to improve people's living standards and regional air quality.

7.2.2.2 Case Introduction of Typical Provinces

In order to implement the national plan for clean heating in winter in northern China published in 2017, as one of most air-polluted northern provinces in China, Hebei province released the "*Implementation Scheme of Clean Heating Project in Winter for Hebei Province*" with many solid measures in the requirement in 2018.

Objectives:

Base on the "*Implementation Scheme of Clean Heating Project in Winter for Hebei Province*" (https://www.sohu.com/a/243282930_760848), the objectives of clean heating campaign in Hebei province in 2018 were as follow.

(1) For build-up area in key cities and counties: focus on promoting clean heating in the main urban areas of Xingtai, Handan and Hengshui, as well as the urban-rural junction and plain counties of Shijiazhuang, Baoding and Hengshui.

(2) For rural area: increase central heating capacity and eliminate a number of coal-fired heating boilers.

Main measures:

(1) Promote the development of central heating. Promote the commissioning and commencement

of cogeneration projects. Improve the heating capacity of industrial waste heat. Accelerate the development and utilization of geothermal energy. Accelerate the construction of biomass (waste) thermal power. Transform and upgrade coal-fired central heating stations and accelerate the construction of heating pipe network.

(2) Optimize the overall layout of clean heating. Orderly carry out the work of replacing coal with gas to ensure sufficient gas source. Strictly control all kinds of new coal to gas projects, in principle, no new rural gas to replace coal and urban coal-fired boiler to gas projects will be arranged. Pay close attention to the comprehensive "look back" work, comprehensively "look back" on project construction, operation management and gas source implementation, find project loopholes, rectify the problems found, eliminate potential safety hazards and improve management services.

(3) Strive to promote the implementation of electricity instead of coal. In combination with the basic conditions of power security and power grid, it is planned to arrange the implementation of Hebei South Power Grid in Shijiazhuang, Baoding, Handan, Xingtai, Cangzhou and Hengshui, and Hebei North Power Grid in Tangshan, Qinhuangdao and Chengde. All cities shall promptly organize relevant counties (cities and districts) to implement tasks, put forward work plans such as village identification, household identification and equipment selection and connect with the power grid company in advance. Accelerate the construction and transformation of supporting power grids and strengthen overall guarantee in the stable operation of power grids.

(4) Carry out renewable energy pilot projects. Select areas with good conditions, pilot promote, demonstrate and drive and promote renewable energy heating according to local conditions.

(5) Promote clean coal in a transitional way. In rural areas where clean heating has not been implemented for the time being, continue to promote the use of clean briquette, blue charcoal and high-quality coal.

7.2.3 Prevention and Control of Mobile Source Pollution

7.2.3.1 Policy Implementation Progress

In the past decades, China have been upgrading emission standard of on-road vehicles from National I to National V. In 2001, China implemented the National I emission standard for motor vehicles and has now fully switched to National V emission standard. In December 2016, China issued the *Limits and Measurement Methods for Emissions from Light-Duty Vehicles (China VI)*. At present in northern China, Beijing Municipality, Tianjin Municipality, Hebei Province, Shandong Province, Shanxi Province and Henan Province have taken the lead in implementing the National VI standard (for new cars only). Compared with the previous National I emission standard, the National VI standard has reduced the emissions of pollutants per vehicle by more than 90%. Among them, the emissions of particulates from diesel vehicles per 100km have been reduced from 293g to 1.5g.

In China, "yellow-label vehicles" refer to gasoline vehicles that do not meet the National I emission standard and diesel vehicles that do not meet the National III emission standard. "Old vehicles" are vehicles that fail to meet the National IV emission standard. Since 2013, China has eliminated more than 20 million yellow-label vehicles and old vehicles. The proportion of vehicles meeting National III and higher emission standards increased from 68.4% in 2013 to 92.1% in 2018.

While phasing out yellow-label vehicles and old vehicles, China vigorously promoted new energy vehicles (that is, battery electric vehicle, plug-in hybrid electric vehicle and fuel cell electric vehicle) and promoted the optimization of vehicle types. By the end of 2012, China had only 17,000 new energy vehicles. In 2019, China produced and sold 1.2 million new energy vehicles, with both production and sales accounting for more than 50% of the world's totals; the new-energy vehicle ownership reached 3.81 million, of which 81% are battery electric vehicles (Figure 7-6).

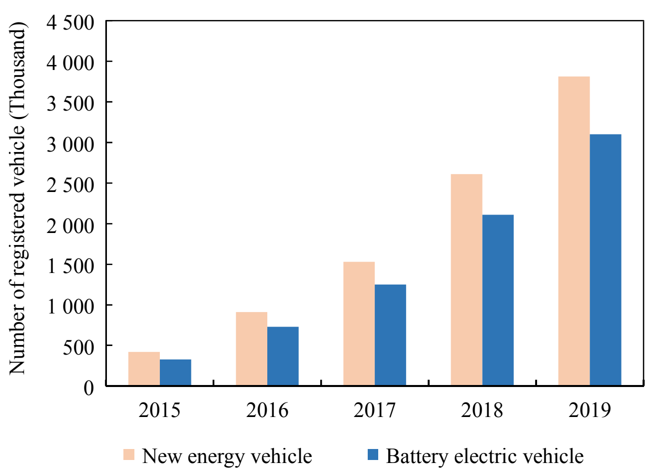


Figure 7-6 Number of registered new energy vehicles and battery electric vehicles in China by years 2015-2019

Mobile sources, especially diesel vehicles, have a greater contribution to BC emissions than gasoline vehicles. Among them, old vehicles significantly surpass new vehicles in terms of emission factor and have become a source of BC pollution that has attracted international attention. The Chinese government's measures to tighten motor vehicle emission standard and phase out old and yellow-label vehicles are expected to directly reduce related BC emissions. At the same time, by vigorously promoting new energy vehicles and supplemented with clean electricity (e.g., renewable electricity), the country can also achieve synergistic effect in pollution reduction and carbon reduction.

7.2.3.2 Case Introduction of Typical Provinces

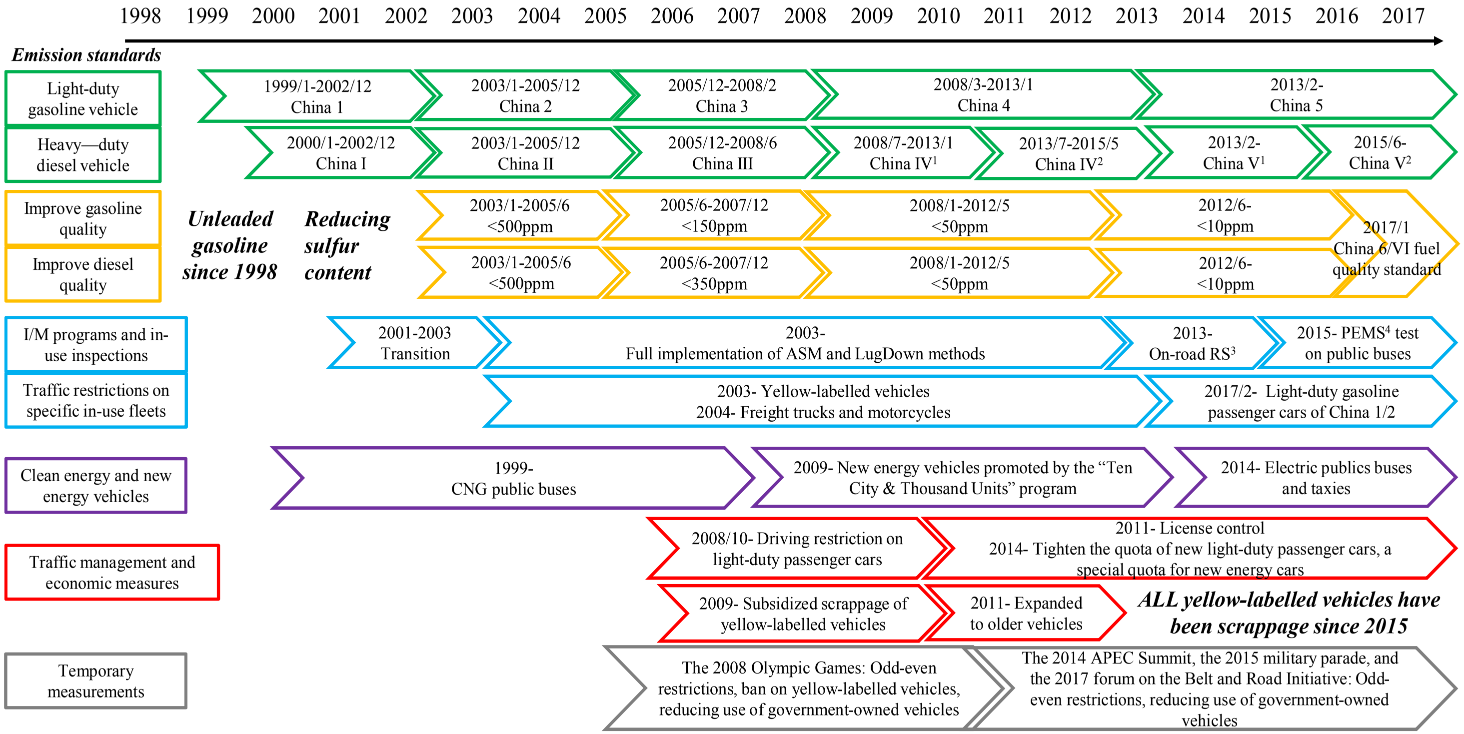
The pollution prevention and control of vehicles have long been a critical task in Beijing's air pollution control. Focusing on new vehicles, in-use vehicles and fuel quality, Beijing has implemented a series of local emission standards and comprehensive control measures; as well as strengthened traffic management and economic incentives continuously.

(1) Stricter emission standards and in-use vehicle retrofitting

Beijing's motor vehicle emission standards have always led the country. In 1999, Beijing became the first city in China to implement the national Ⅰ emission standard for light gasoline vehicles. In 2004, 2005 and 2008, Beijing took the lead in implementing the "National Ⅱ", "National Ⅲ" and "National Ⅳ" standards for motor vehicle emissions respectively. In 2013, Beijing continued to take the lead in implementing the "Beijing Ⅴ" emission standard equivalent to the European phase Ⅴ standard, further narrowing the gap with the motor vehicle emission control level of developed countries. In 2019, Beijing has officially issued the notice on Beijing's early implementation of National Ⅵ emission standards, which required to implement National Ⅵ emission standards in stages for difference vehicle types. In the past decade, Beijing has achieved leapfrog development in new vehicle emission control and led the technological progress of energy conservation and environmental protection in automobile industry through standards.

(2) Phasing out old and high-polluting vehicles

In Beijing, yellow-labeled vehicles were first restricted within the Second Ring Road in 2003. During the 2008 Olympic Games period, they were banned in the whole city. In 2010, the scope of the restricted area for yellow-labeled vehicles was extended to within the Sixth Ring Road and then to the whole city in December 2015. For light-duty gasoline vehicles, starting from 2017, those of China National I and National II emission standards have been restricted within the Fifth Ring Road of Beijing; for heavy-duty diesel trucks, those of China National III emission standard or less have been restricted within the Sixth Ring Road starting from 2017 (United Nations Environment Programme, 2019).



[1] Only implemented for public fleets; [2] for freight trucks and long-distance coaches; [3] remote sensing test; [4] portable emission measurement system

Figure 7-7 Review of main emissions control measures for motor vehicles in Beijing, 1998-2017

(source: Former Beijing Municipal Environmental Protection Bureau, Tsinghua University, UNEP, 2019)

(3) Upgrading fuel quality

The quality of oil products is also improved simultaneously in Beijing. In 1997, Beijing took the lead in using unleaded gasoline throughout the country. In 2004, it formulated and implemented the local standard for the stage II of vehicle fuel that was stricter than the national standard. Since then, Beijing's oil standard has continued to lead the country by one or two stages. At present, Beijing has implemented the "Beijing VI" oil standard and some indicators have been stricter than European standards (BeijingDaily, 2019).

(4) Developing new energy vehicles

In 1999, Beijing introduced compressed natural gas (CNG) to the bus fleet and gradually promoted clean fuels and new energy buses. Among the 2,306 buses that were updated in 2016, 1,368 were electric vehicles, accounting for 59% (United Nations Environment Programme, 2019).

(5) Integrating a comprehensive "vehicle, oil and road" treatment framework

Over the past 20 years, by formulating and implementing a series of strict local standards for the emission management of new and in-service vehicles and the quality of oil products, adopting comprehensive treatment measures and continuously strengthening traffic control and economic incentives, Beijing has gradually developed and formed an integrated vehicle emission control framework including "vehicle, oil and road". More important, a large-scale public transport system has been built to allow gradual formation of a green and low-carbon in-city travel habit by the people.

Although the number of vehicles increased three-folds in Beijing during the last two decades, the total pollutants emissions decreased remarkably. Based on the study report of UNEP (United Nations Environment Programme, 2019), compared with 1998, CO, THC(total hydrocarbon), NO_x and $PM_{2.5}$ emissions from the transportation sector in 2017 were reduced by nearly 89%, 64%, 55% and 81% respectively.

Our review of China's policies on air pollution prevention and control helps to identify key measures that have a positive effect on reducing primary $PM_{2.5}$, especially on reducing BC/OC emissions. The review would also help to identify the shortcomings of existing measures in BC/OC control. The experiences and shortcomings summarized could further provide inspiration and a basis for future studies on BC/OC control, as well as helping the design of BC/OC control strategies by policymakers.

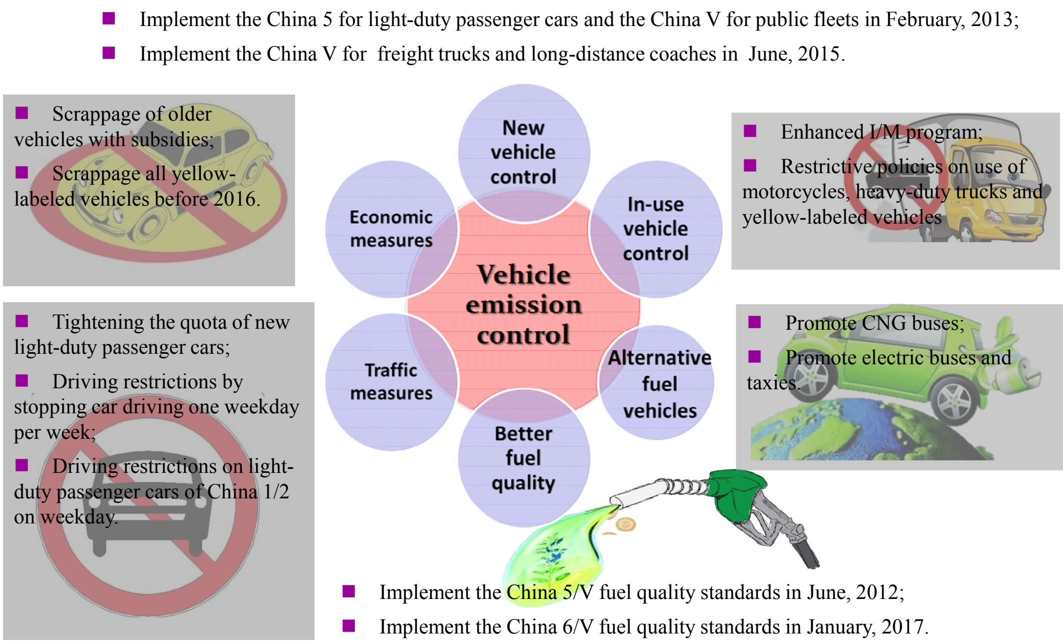


Figure 7-8 Vehicle-fuel-road integrated control system (source: Former Beijing Municipal Environmental Protection Bureau, Tsinghua University, UNEP, 2019)

References

[1] Aamaas, B. and Berntsen, T. K., 2021. Methodologies for the presentation of climate impact in both the short and long term, Cicero report 2021:01 and the Norwegian Environment Agency (NEA) report M-1719. In Norwegian but with an English summary. https://www.miljodirektoratet.no/publikasjoner/2020/juni-2020/metodikk-for-framstilling-av-klimaeffekt-pa-kort-og-lang-sikt/(Accessed 24. June, 2022)

[2] Abdel-Shafy, H.I., and Mansour, M.S.M. 2016. A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. Egyptian Journal of Petroleum 25, 107-123.

[3] Acciani, T., Brandt, E., Pastura, P., et al., 2012. Low dose diesel exhaust particle exposure exacerbates allergic asthma in young mice. European Respiratory Journal 40.

[4] Agency, U.S.E.P. 2012. Report to Congress on Black Carbon.

[5] Alexeeff, S.E., Roy, A., Shan, J., et al., 2018. High-resolution mapping of traffic related air pollution with Google street view cars and incidence of cardiovascular events within neighborhoods in Oakland, CA. Environmental Health 17, 13.

[6] Ali, M.U., Siyi, L., Yousaf, B., et al., 2020. Emission sources and full spectrum of health impacts of BC associated polycyclic aromatic hydrocarbons PAHs in urban environment: A review. Critical Reviews in Environmental Science and Technology, 1-40.

[7] Amann, M., Bertok, I., Borken-Kleefeld, J, et al., 2011. Cost-effective control of air quality and greenhouse gases in Europe: Modeling and policy applications, Environ. Modell. Softw., 26, 1489-1501, https://doi.org/10.1016/j.envsoft.2011.07.012.

[8] Amann, M., Cofala, J., Heyes, C, et al., 2004. The Regional Air Pollution Information and Simulation (RAINS) model February 2004, in: Agency, E.E. (Ed.).

[9] An, Z., Huang, R. J., Zhang, R., et al., 2019. Severe haze in northern China: A synergy of anthropogenic emissions and atmospheric processes. Proceedings of the National Academy of Sciences of the United States of America, 116(18), 8657-8666. https://doi.org/10.1073/pnas.1900125116

[10] Analitis, A., Katsouyanni, K., Dimakopoulou, K., et al., 2006. Short-term effects of ambient particles on cardiovascular and respiratory mortality. Epidemiology 17, 230-233.

[11] Andersen, M.H.G., Frederiksen, M., Saber, A.T., et al., 2019. Health effects of exposure to diesel exhaust in diesel-powered trains. Particle and Fibre Toxicology 16.

[12] Anderson, H.R., Bremner, S.A., Atkinson, R.W., et al., 2001. Particulate matter and daily mortality and hospital admissions in the west midlands conurbation of the United Kingdom: associations with fine and coarse particles, black smoke and sulphate. Occupational and Environmental Medicine 58, 504-510.

[13] Anderson, K.R., Avol, E.L., Edwards, S.A., et al., 1992. Controlled exposures of volunteers to respirable carbon and sulfuric acid aerosols. Journal of the Air & Waste Management Association 42, 770 - 776.

[14] Andreae, M.O. and Merlet, P., 2001. Emission of trace gases and aerosols from biomass burning. Global Biogeochemical Cycles 15, 955-966.

[15] Andreae, M.O., and Gelencsér, A. 2006. Black carbon or brown carbon? The nature of light-absorbing carbonaceous aerosols. Atmospheric Chemistry and Physics 6, 3131-3148.

[16] Appel, K. W., Bhave, P. V, Gilliland, A. B., et al., 2008. Evaluation of the community multiscale air quality (CMAQ) model version 4 . 5 : Sensitivities impacting model performance ; Part II — PM. 42, 6057-6066. https://doi.org/10.1016/j.atmosenv.2008.03.036

[17] Appel, K. W., Gilliland, A. B., Sarwar, G., et al., 2007. Evaluation of the Community Multiscale Air Quality (CMAQ) model version 4.5: Sensitivities impacting model performance. Part I-Ozone. Atmospheric Environment, 41(40), 9603-9615. https://doi.org/10.1016/j.atmosenv.2007.08.044.

[18] Arimoto, T., Takano, H., Inoue, K., et al., 2007. Pulmonary exposure to diesel exhaust particle components enhances circulatory chemokines during lung inflammation. International Journal of Immunopathology and Pharmacology 20, 197-201.

[19] Armstrong, B., Hutchinson, E., Unwin, J., et al., 2004. Lung cancer risk after exposure to polycyclic aromatic hydrocarbons: a review and meta-analysis. Environ Health Perspect 112, 970-978.

[20] Atkinson, R.W., Analitis, A., Samoli, E., et al., 2016. Short-term exposure to traffic-related air pollution and daily mortality in London, UK. Journal of Exposure Science and Environmental Epidemiology 26, 125-132.

[21] Bahner M A, Weitz K A, Zapata A, et al., 2007. Use of BC and organic carbon inventories for projections and mitigation analysis[C]//Proceedings of the 16th annual international emission inventory conference emission inventories: integration analysis and communications, Raleigh.

[22] Bahner, M. A., Weitz, K. A., Zapata, A., et al., 2007. Use of black carbon and organic carbon inventories for projections and mitigation analysis. In Proceedings of the 16th annual international emission inventory conference emission inventories: integration analysis and communications, Raleigh.

[23] Bai C., Li Y., Jiang Z., 2016. The BC research status of marine vessels and the impact on the arctic (in Chinese). China shipping (second half), 16 (12): 152-154.

[24] Bai, N., and Van Eeden, S.F. 2013. Systemic and vascular effects of circulating diesel exhaust PM. Inhalation Toxicology 25, 725-734.

[25] Ballester, F., Corella, D., PerezHoyos, S., et al., 1996. Air pollution and mortality in Valencia, Spain: A study using the APHEA methodology. Journal of Epidemiology and Community Health 50, 527-533.

[26] Ballester, F., Saez, M., Perez-Hoyos, S., et al., 2002. The EMECAM project: a multicentre study on air pollution and mortality in Spain: combined results for particulates and for sulfur dioxide. Occupational and Environmental Medicine 59, 300-308.

[27] Ban-Weiss, G.A., Cao, L., et al., 2012. Dependence of climate forcing and response on the altitude of BC aerosols. Clim Dyn 38, 897-911 https://doi.org/10.1007/s00382-011-1052-y.

[28] Beelen, R., Hoek, G., Br, v.d., et al., 2008. Long-term effects of traffic-related air pollution on mortality in a Dutch cohort NLCS-AIR study. Environmental Health Perspectives 116, 196-202.

[29] Beelen, R., Hoek, G., Houthuijs, D., et al., 2009. The joint association of air pollution and noise from road traffic with cardiovascular mortality in a cohort study. Occupational and Environmental Medicine 66, 243-250.

[30] BeijingDaily. 2019. "The United Nations recommended the experience of Beijing in atmospheric Governance: this is not accidental." from https://baijiahao.baidu.com/s?id=1629853516063598794&wfr=spider&for=pc.

[31] Bendtsen, K.M., Gren, L., Malmborg, V.B., et al., 2020. Particle characterization and toxicity in C57BL/6 mice following instillation of five different diesel exhaust particles designed to differ in physicochemical properties. Part Fibre Toxicol 17, 38.

[32] Beverl, I.J., Carder, M., et al., 2014. Associations between short/medium-term variations in black smoke air pollution and mortality in the Glasgow conurbation, UK. Environment International 62, 126-132.

[33] Beverl, I.J., Cohen, G.R., et al., 2012. A Comparison of Short-term and Long-term Air Pollution Exposure Associations with Mortality in Two Cohorts in Scotland. Environmental Health Perspectives 120, 1280-1285.

[34] Binkowski, Francis S., Roselle, S. J., 2003. Models-3 Community Multiscale Air Quality (CMAQ) model aerosol component 1. Model description. Journal of Geophysical Research D: Atmospheres, 108(6). https://doi.org/10.1029/2001jd001409.

[35] Bo, Z., Dan, T., Meng, L., et al., 2018. Trends in China's anthropogenic emissions since 2010 as the consequence of clean air actions. Atmospheric Chemistry & Physics Discussions, 1-27.

[36] Boezen, H.M., Vonk, J.M., van der Zee, S.C., et al., 2005. Susceptibility to air pollution in elderly males and females. European Respiratory Journal 25, 1018-1024.

[37] Bolton, J.L., Auten, R.L., and Bilbo, S.D. 2014. Prenatal air pollution exposure induces sexually dimorphic fetal programming of metabolic and neuroinflammatory outcomes in adult offspring. Brain, Behavior, and Immunity 37, 30-44.

[38] Bolton, J.L., Marinero, S., Hassanzadeh, T., et al., 2017. Gestational exposure to air pollution alters cortical volume, microglial morphology, and microglia-neuron interactions in a sex-specific manner.

Frontiers in Synaptic Neuroscience 9.

[39] Bond T.C, Doherty SJ, Fahey DW et al., 2013. Bounding the role of BC in the climate system: A scientific assessment. Journal of Geophysical Research: Atmospheres. 118:5380-552. DOI: 10.1002/jgrd.50171.

[40] Bond, T. C. and Bergstrom, R. W., 2006. Light absorption by carbonaceous particles: an investigative review. Aerosol Science and Technology, 40, 27-67.

[41] Bond, T.C., and Bergstrom, R.W. 2007. Light Absorption by Carbonaceous Particles: An Investigative Review. Aerosol Science and Technology 40, 27-67.

[42] Bond, T.C., Bhardwaj E., Dong, R, et al., 2007. Historical emissions of black and organic carbon aerosol from energy-relatedcombustion:1850-2000. Global Biogeochemical Cycles, vol.21, GB2018.

[43] Bond, T.C., Doherty, S.J., Fahey, D.W., et al., 2013. Bounding the role of BC in the climate system: A scientific assessment. Journal of Geophysical Research: Atmospheres 118, 5380-5552.

[44] Bond, T.C., Streets D.G., Yarber K.F, et al., 2004. A technology-based global inventory of black and organic carbon emissions from combustion. Journal of Geophysical Research, v.109, n.D14.

[45] Bose, S., Romero, K., Psoter, K.J., et al., 2018. Association of traffic air pollution and rhinitis quality of life in Peruvian children with asthma. Plos One 13, 13.

[46] Boucher, O., Randall, D., Artaxo, P., et al., 2013.Clouds and Aerosols: In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

[47] Bourgeois, Q. and Bey, 2011. I.: Pollution transport efficiency toward the Arctic: Sensitivity to aerosol scavenging and source regions, J. Geophys. Res.-Atmos., 116, D08213, doi:10.1029/2010JD015096.

[48] Bouwman, A.F., Lee, D.S., Asman, W.A.H, et al., 1997. A global high-resolution emission inventory for ammonia. Global Biogeochemical Cycles 11, 561-587.

[49] Boylan, J. W., & Russell, A. G. 2006. PM and light extinction model performance metrics, goals, and criteria for three-dimensional air quality models. Atmospheric Environment, 40(26), 4946-4959. https://doi.org/10.1016/j.atmosenv.2005.09.087

[50] Bradley, J.M., Cryar, K.A., El Hajj, M.C., et al., 2013. Exposure to diesel exhaust particulates induces cardiac dysfunction and remodeling. Journal of Applied Physiology 115, 1099-1106.

[51] Brasseur, G. P., & Jacob, D. J. 2017. Modeling of atmospheric chemistry. Cambridge University Press.

[52] Bravo, M.A., Fuentes, M., Zhang, Y., et al., 2012. Comparison of exposure estimation methods for air pollutants: ambient monitoring data and regional air quality simulation. Environ. Res. 116, 1e10. http://dx.doi.org/ 10.1016/j.envres.2012.04.008.

[53] Bremner, S.A., Anderson, H.R., Atkinson, R.W., et al., 1999. Short-term associations between outdoor air pollution and mortality in London 1992-4. Occup Environ Med 56, 237-244.

[54] Browse, J., Carslaw, K. S., Arnold, S. R., et al., 2012. The scavenging processes controlling the seasonal cycle in Arctic sulphate and BC aerosol, Atmos. Chem. Phys., 12, 6775-6798, doi:10.5194/acp-12-6775-2012.

[55] Brune, W.H., Baier, B.C., Thomas, J., et al., 2016. Ozone production chemistry in the presence of urban plumes. Faraday Discuss. http://dx.doi.org/10.1039/C5FD00204D.

[56] C. Nordenha, J.P., M-C. Ledin, J-O. Levin, T. Sandstro¨m, E. A¨ delroth. 2001. Diesel exhaust enhances airway responsiveness in asthmatic subjects. European Respiratory Journal 17, 909-915.

[57] Cakmak, S., Dales, R.E., and Blanco, C. 2009a. Components of Particulate Air Pollution and Mortality in Chile. International Journal of Occupational and Environmental Health 15, 152-158.

[58] Cakmak, S., Dales, R.E., Gultekin, T., et al., 2009b. Components of Particulate Air Pollution and Emergency Department Visits in Chile. Archives of Environmental & Occupational Health 64, 148-155.

[59] Cao G.L., Zhang X.Y., Wang, Y.Q, et al., 2007 Inventory of Black Carbon Emission from China, Advances in Climate Change Research, vol 3 (Suppl.), 75-81.

[60] Cao J, Tie X, Xu B, et al., 2010. Measuring and modeling BC (BC) contamination in the SE Tibetan Plateau[J]. Journal of Atmospheric Chemistry, 67(1): 45.

[61] Cao, G., Zhang, X., Gong, S, et al., 2011. Emission inventories of primary particles and pollutant gases for China. Chinese Science Bulletin 56, 781-788.

[62] Cao, G., Zhang, X., Zheng, F., et al., 2006. Inventory of BC and organic carbon emissions from China. Atmospheric Environment 40, 6516-6527.

[63] Cao, J.J., Xu, H.M., Xu, Q., et al., 2012. Fine Particulate Matter Constituents and Cardiopulmonary Mortality in a Heavily Polluted Chinese City. Environmental Health Perspectives 120, 373-378.

[64] Carder, M., McNamee, R., Beverl, et al., 2008. Interacting effects of particulate pollution and cold temperature on cardiorespiratory mortality in Scotland. Occupational and Environmental Medicine 65, 197-204.

[65] Carlsten, C., Blomberg, A., Pui, M., et al., 2016. Diesel exhaust augments allergen-induced lower airway inflammation in allergic individuals: A controlled human exposure study. Thorax 71, 35-44.

[66] Cassee, F.R., Héroux, M.-E., Gerlofs-Nijland, M.E., et al., 2013. Particulate matter beyond mass: recent health evidence on the role of fractions, chemical constituents and sources of emission. Inhalation Toxicology 25, 802-812.

[67] Center on Emissions Inventories and Projections (CEIP). 2020. https://www.ceip.at/the-emep-grid/grid-definiton, last accessed 27. May 2022

[68] Chai, T., & Draxler, R. R. 2014. Root mean square error (RMSE) or mean absolute error (MAE)?- Arguments against avoiding RMSE in the literature. Geoscientific Model Development, 7(3), 1247-1250. https://doi.org/10.5194/gmd-7-1247-2014

[69] Chang Y, Deng C, Cao F, et al., Assessment of carbonaceous aerosols in Shanghai, China-Part 1:

long-term evolution, seasonal variations, and meteorological effects[J]. Atmospheric Chemistry and Physics, 2017, 17(16): 9945-9964.

[70] Chang, J. C., & Hanna, S. R. 2004. Air quality model performance evaluation. Meteorology and Atmospheric Physics, 87(1-3), 167-196. https://doi.org/10.1007/s00703-003-0070-7

[71] Chang, Y., Deng, C., Cao, F., et al., 2017. Assessment of carbonaceous aerosols in Shanghai, China-Part 1: Long-term evolution, seasonal variations, and meteorological effects. Atmospheric Chemistry and Physics, 17(16), 9945-9964. https://doi.org/10.5194/acp-17-9945-2017

[72] Chen Z, Chen D, Wen W, et al., 2019. Evaluating the “2+26” regional strategy for air quality improvement during two air pollution alerts in Beijing: variations in $PM_{2.5}$ concentrations, source apportionment, and the relative contribution of local emission and regional transport[J]. Atmospheric Chemistry and Physics, 19(10): 6879-6891.

[73] Chen, Dan, Liu, Z., Fast, J., et al., 2016. Simulations of sulfate-nitrate-ammonium (SNA) aerosols during the extreme haze events over northern China in October 2014. Atmospheric Chemistry and Physics, 16(16), 10707-10724. https://doi.org/10.5194/acp-16-10707-2016

[74] Chen, J., Wu, Y., Song, M, et al., 2018. The residential coal consumption: Disparity in urban-rural China. Resources Conservation & Recycling 130, 60-69.

[75] Chen, Y., Chen, R., Chen, Y., et al., 2021. The prospective effects of long-term exposure to ambient $PM_{2.5}$ and constituents on mortality in rural East China. Chemosphere 280, 130740.

[76] Chen, Y., Sheng, G., Bi, X, et al., 2005. Emission factors for carbonaceous particles and polycyclic aromatic hydrocarbons from residential coal combustion in China. Environmental Science & Technology 39, 1861-1867.

[77] Chen, Y., Tian, C., Feng, Y, et al., 2015a. Measurements of emission factors of $PM_{2.5}$, OC, EC, and BC for household stoves of coal combustion in China. Atmospheric Environment 109, 190-196.

[78] Chen, Y., Zhi, G., Feng, Y, et al., 2006. Measurements of emission factors for primary carbonaceous particles from residential raw‐coal combustion in China. Geophysical Research Letters 33, 382-385.

[79] Chen, Y., Zhi, G., Feng, Y, et al., 2009a. Measurements of Black and Organic Carbon Emission Factors for Household Coal Combustion in China: Implication for Emission Reduction. Environmental Science & Technology 43, 9495.

[80] Chen, Y., Zhi, G., Feng, Y, et al., 2015b. Increase in polycyclic aromatic hydrocarbon (PAH) emissions due to briquetting: A challenge to the coal briquetting policy. Environmental Pollution 204, 58-63.

[81] Chen, Y.J., Jiang X. H., Zhi G. R, et al., 2009b. Black carbon emissions from residential coal combustion and reduction strategy (in Chinese). Sci China Ser D-Earth Sci 39, 1554-1559.

[82] Chen, Z., Chen, D., Zhao, C., et al., 2020. Influence of meteorological conditions on $PM_{2.5}$ concentrations across China : A review of methodology and mechanism. Environment International, 139(February), 105558. https://doi.org/10.1016/j.envint.2020.105558

[83] Cheng M, Zhi G, Tang W et al., 2017. Air pollutant emission from the underestimated households' coal consumption source in China. Science of the Total Environment. 580:641-50.

[84] Cheng, J., Su, J., Cui, T., et al., 2019. Dominant role of emission reduction in $PM_{2.5}$ air quality improvement in Beijing during 2013-2017: A model-based decomposition analysis. Atmospheric Chemistry and Physics, 19(9), 6125-6146. https://doi.org/10.5194/acp-19-6125-2019.

[85] Cheng, Z., Chu, H., Wang, S., et al., 2019. TAK1 knock-down in macrophage alleviate lung inflammation induced by BC and aged BC. Environmental Pollution 253, 507-515.

[86] Cheng, Z., Wang, S., Fu, X., et al., 2014. Impact of biomass burning on haze pollution in the Yangtze River delta, China: A case study in summer 2011. Atmospheric Chemistry and Physics, 14, 4573-4585

[87] ChinaNews. 2021. MEE: Winning the Blue Sky Battle. from https://www.sohu.com/a/452745029_270669.

[88] Ching, J., and Kajino, M. 2018. Aerosol mixing state matters for particles deposition in human respiratory system. Sci Rep 8, 8864.

[89] Choi, M. W., Lee, et al., 2019. Comparison of $PM_{2.5}$ chemical components over East Asia simulated by the WRF-Chem and WRF/CMAQ models: On the models' prediction inconsistency. Atmosphere, 10(10). https://doi.org/10.3390/atmos10100618.

[90] Choi, Y., Kanaya, Y., Park, S. M., et al., 2020. Regional variability in BC and carbon monoxide ratio from long-term observations over East Asia: Assessment of representativeness for BC (BC) and carbon monoxide (CO) emission inventories. Atmospheric Chemistry and Physics, 20(1), 83-98. https://doi.org/10.5194/acp-20-83-2020

[91] Chris Carlsten, A.B., Mandy Pui, Thomas Sandstrom, et al., 2020. Vascular effects of physical activity are not modified by short-term inhaled diesel exhaust: results of a controlled human exposure study. Environmental research 183.

[92] Chuang, H.C., Lin, Y.J., Chou, C.C.K., et al., 2017. Alterations in cardiovascular function by PM in rats using a crossover design. Environmental Pollution 231, 812-820.

[93] Chuang, P. Y. 2003. Measurement of the timescale of hygroscopic growth for atmospheric aerosols. Journal of Geophysical Research: Atmospheres, 108(D9).

[94] Clark, C., Sbihi, H., Tamburic, L., et al., 2017. Association of Long-Term Exposure to Transportation Noise and Traffic-Related Air Pollution with the Incidence of Diabetes: A Prospective Cohort Study. Environmental Health Perspectives 125, 10.

[95] Colicino, E., Wilson, A., Frisardi, M.C., et al., 2017. Telomere Length, Long-Term Black Carbon Exposure, and Cognitive Function in a Cohort of Older Men: The VA Normative Aging Study. Environ Health Perspect 125, 76-81.

[96] Cong, Z. Y., Kang, S. C., Zhang, Y. L., et al., 2015. New insights into trace element wet deposition in the Himalayas: Amounts, seasonal patterns, and implications. Environemental Science and Pollution Research International, 22(4), 2735-2744. https://doi.org/10.1007/s11356-014-3496-1.

[97] Cooke W.F. and Wilson J.N., 1996. A global BC aerosol model. Journal of Geophysical Research, vol. 101, n. d14, 19395-19409

[98] Cooke W.F., Liousse C, et al., 1999. Construction of a 1°×1° fossil fuel emission data set for carbonaceous aerosol and implementation and radiative impact in the ECHAM4 mode. Journal of Geophysical Research, vol.104, n.d18:22137-22162.

[99] Cowell, W.J., Bellinger, D.C., Coull, B.A., et al., 2015. Associations between Prenatal Exposure to Black Carbon and Memory Domains in Urban Children: Modification by Sex and Prenatal Stress. PLoS One 10, e0142492.

[100] Crippa, M., Solazzo, E., Huang, G, et al., 2019. High resolution temporal profiles in the Emissions Database for Global Atmospheric Research (EDGAR), Nature Scientific Data, doi:10.1038/s41597-020-0462-2.

[101] Curci, G., Alyuz, U., Baro, R., et al., 2019. Modelling BC absorption of solar radiation: combining external and internal mixing assumptions. Atmos Chem Phys 19, 181-204.

[102] Dab, W., Medina, S., Quenel, P., et al., 1996. Short term respiratory health effects of ambient air pollution: Results of the APHEA project in Paris. Journal of Epidemiology and Community Health 50, S42-S46.

[103] Danielsen, P.H., Risom, L., Wallin, H., et al., 2008. DNA damage in rats after a single oral exposure to diesel exhaust particles. Mutation Research - Fundamental and Molecular Mechanisms of Mutagenesis 637, 49-55.

[104] De Homdedeu, M., Cruz, M., Sanchez-Díez, S., et al., 2020. The immunomodulatory effects of diesel exhaust particles in asthma. Environmental Pollution 263.

[105] De Homdedeu, M., Cruz, M.J., Sánchez-Díez, S., et al., 2021. Role of diesel exhaust particles in the induction of allergic asthma to low doses of soybean. Environmental Research 196.

[106] Dehbi, H.M., Blangiardo, M., Gulliver, J., et al., 2017. Air pollution and cardiovascular mortality with over 25 years follow-up: A combined analysis of two British cohorts. Environment International 99, 275-281.

[107] Dellinger, B., Pryor, W.A., Cueto, R., et al., 2001. Role of Free Radicals in the Toxicity of Airborne Fine Particulate Matter. Chemical Research in Toxicology 14, 1371.

[108] Department of Energy Statistics, National Bureau of Statistics. 2020. China energy statistical yearbook. Beijing: China Statistics Press.

[109] Development Technical Guidelines (in Chinese). 2015, Beijing China.

[110] Di, Q., Kloog, I., Koutrakis, P., et al., 2016. Assessing $PM_{2.5}$ exposures with high spatiotemporal resolution across the continental United States. Environ. Sci. Technol. 50, 4712e4721. http://dx.doi.org/10.1021/ acs.est.5b06121.

[111] Ding, A. J., Huang, X., Nie, W., et al., 2016. Enhanced haze pollution by BC in megacities in China. Geophysical Research Letters, 43(6), 2873-2879. https://doi.org/10.1002/2016GL067745

[112] Dockery, D.W., Luttmann-Gibson, H., Rich, D.Q., et al., 2005. Association of air pollution with increased

incidence of ventricular tachyarrhythmias recorded by implanted cardioverter defibrillators. Environ Health Perspect 113, 670-674.

[113] Doiron, D., Bourbeau, J., de Hoogh, K., et al., 2021. Ambient air pollution exposure and chronic bronchitis in the Lifelines cohort. Thorax.

[114] Dou T.-F. & Xiao C.-D. 2016, An overview of BC deposition and its radiative forcing over the Arctic, Advances in Climate Change Research. 7(3), 115-122, https://doi.org/10.1016/j.accre.2016.10.003.

[115] EEA. 2016. Adoption of the EU Euro emissions standards for road vehicles in Asian countries https://www.eea.europa.eu/data-and-maps/figures/number-of-international-environmental-agreements-adopted-1, last accessed 27. May 2022.

[116] EEA. 2018. Norway - Air Pollution country fact sheet. https://www.eea.europa.eu/themes/air/country-fact-sheets/2021-country-fact-sheets/norway, last accessed 27. May 2022.

[117] EEA. 2019. EMEP/EEA air pollutant emission inventory guidebook. https://www.eea.europa.eu/themes/air/air-pollution-sources-1/emep-eea-air-pollutant-emission-inventory-guidebook/emep, last accessed 27. May 2022.

[118] Egger, M. 1997. Bias in meta-analysis detected by a simple, graphical test. J Bmj British Medical Journal 316.

[119] Ehsanifar, M., Jafari, A.J., Nikzad, H., et al., 2019. Prenatal exposure to diesel exhaust particles causes anxiety, spatial memory disorders with alters expression of hippocampal pro-inflammatory cytokines and NMDA receptor subunits in adult male mice offspring. Ecotoxicology and Environmental Safety 176, 34-41.

[120] El-Harbawi, M. 2013. Air quality modelling, simulation, and computational methods: A review. Environmental Reviews, 21(3), 149-179. https://doi.org/10.1139/er-2012-0056.

[121] Elliott, P., Shaddick, G., Wakefield, J.C., et al., 2007. Long-term associations of outdoor air pollution with mortality in Great Britain. Thorax 62, 1088-1094.

[122] Emery, C., Liu, Z., Russell, A. G., et al., 2017. Recommendations on statistics and benchmarks to assess photochemical model performance. Journal of the Air and Waste Management Association, 67(5), 582-598. https://doi.org/10.1080/10962247.2016.1265027.

[123] Energy Research Institute National Development and Reform Commission. 2019. Key Coal Consuming Industries Thirteenth Five-Year Coal Cap Mid-Term Evaluation and Later-Term Outlook. http://www.nrdc.cn/Public/uploads/2019-06-24/5d10818b26b5f.pdf, last accessed 27. May 2022.

[124] Er Hoorn, S., Murray, K., Nedkoff, L., et al., 2021. Long-term exposure to outdoor air pollution and risk factors for cardiovascular disease within a cohort of older men in Perth. Plos One 16, 18.

[125] EU, 2019b. EU-funded Action on Black Carbon in the Arctic, 2019. Review of Reporting Systems for National Black Carbon Emissions Inventories: EU Action on Black Carbon in the Arctic - Technical Report 2. April 2019. iv + 72pp. http://www.amap.no/documents/doc/review-of-reporting-systems-for-

national-black-carbon-emissions-inventories/1780, last accessed 27. May 2022.

[126] EU. 2019a. EU-funded Action on Black Carbon in the Arctic, 2019. Review of observation capacities and data availability for Black Carbon in the Arctic region: EU Action on Black Carbon in the Arctic - Technical Report 1. December 2019. iv + 35pp. https://www.amap.no/documents/doc/review-of-observation-capacities-and-data-availability-for-black-carbon-in-the-arctic-region/3058, last accessed 27. May 2022.

[127] Fan, Q., Lan, J., Liu, Y., et al., Process analysis of regional aerosol pollution during spring in the Pearl River Delta region, China.

[128] Fan, S. M, et al., 2012. Inferring ice formation processes from global-scale BC profiles observed in the remote atmosphere and model simulations. J. Geophys. Res. Atmos. 117, D23205, https://doi.org/10.1029/2012JD018126. 11. .

[129] Feng, Jialiang, Zhong, M., et al., 2014. Concentrations, seasonal and diurnal variations of BC in $PM_{2.5}$ in Shanghai, China. Atmospheric Research, 147-148, 1-9. https://doi.org/10.1016/j.atmosres.2014.04.018.

[130] Filleul, L., Le Tertre, A., Baldi, I., et al., 2004. Difference in the relation between daily mortality and air pollution among elderly band all-ages populations in southwestern France. Environmental Research 94, 249-253.

[131] Filleul, L., Rondeau, V., entorren, S., et al., 2005. Twenty five year mortality and air pollution: results from the French PAARC survey. Occupational and Environmental Medicine 62, 453-460.

[132] Fischer, P., Hoek, G., Brunekreef, B., et al., 2003. Air pollution and mortality in the Netherlands: are the elderly more at risk? European Respiratory Journal 21, 34S-38S.

[133] Fischer, P.H., Marra, M., Ameling, C.B., et al., 2020. Particulate air pollution from different sources and mortality in 7.5 million adults - The Dutch Environmental Longitudinal Study DUELS. Science of the Total Environment 705, 7.

[134] Fleisch, A.F., Gold, D.R., Rifas-Shiman, S.L., et al., 2014. Air Pollution Exposure and Abnormal Glucose Tolerance during Pregnancy: The Project Viva Cohort. Environmental Health Perspectives 122, 378-383.

[135] Fu J.F., Qi M, et al., 2018. Emissions and spatial distribution of Anthropogenic BC of China in 2012. Science & Technology Review, vol.36, n2.

[136] Fu, L., Hao, J., He, D, et al., 2001. Assessment of vehicular pollution in China. Journal of the Air & Waste Management Association 51, 658-668.

[137] Gan, W.Q., FitzGerald, J.M., Carlsten, C., et al., 2013. Associations of Ambient Air Pollution with Chronic Obstructive Pulmonary Disease Hospitalization and Mortality. American Journal of Respiratory and Critical Care Medicine 187, 721-727.

[138] Gan, W.Q., Koehoorn, M., Davies, H.W., et al., 2011. Long-Term Exposure to Traffic-Related Air Pollution and the Risk of Coronary Heart Disease Hospitalization and Mortality. Environmental Health Perspectives 119, 501-507.

[139] Gao, B., Yuan, H., Geng, L. et al, 2016. Analysis on rural household energy consumption in Hebei

province, in: Kim, Y.H. (Ed.), Proceedings of the 2016 5th International Conference on Energy and Environmental Protection, pp. 501-504.

[140] Gao, C., Li, S., Liu, M., et al., 2021. Impact of the COVID-19 pandemic on air pollution in Chinese megacities from the perspective of traffic volume and meteorological factors. The Science of the total environment, 773, 145545. https://pubmed.ncbi.nlm.nih.gov/33940731/. .

[141] Gao, M., Carmichael, G. R., Wang, Y., et al., 2016. Modeling study of the 2010 regional haze event in the North China Plain. Atmospheric Chemistry and Physics, 16(3), 1673-1691. https://doi.org/10.5194/acp-16-1673-2016.

[142] Gao, X., Shang, J., Yang, J.L., et al., 2014. Comparison of genetic damage in mice exposed to BC and ozone-oxidized BC. Beijing Da Xue Xue Bao Yi Xue Ban 46, 400-404.

[143] Geng, F.H., Hua, J., Mu, Z., et al., 2013. Differentiating the associations of BC and fine particle with daily mortality in a Chinese city. Environmental Research 120, 27-32.

[144] Gentile, D.A., Morphew, T., Elliott, J., et al., 2020. Asthma prevalence and control among schoolchildren residing near outdoor air pollution sites. Journal of Asthma, 11.

[145] Ghio, A.J., M. S. Carraway, and M. C. Madden. 2012. Composition of Air Pollution Particles and Oxidative Stress in Cells, Tissues, and Living Systems. Journal of Toxicology and Environmental Health 15, 1-21.

[146] Gibbs, J.L., Dallon, B.W., Lewis, J.B., et al., 2019. Diesel exhaust particle exposure compromises alveolar macrophage mitochondrial bioenergetics. International Journal of Molecular Sciences 20.

[147] Gittins, M., McNamee, R., Carder, M., et al., 2013. Has the short-term effect of black smoke exposure on pneumonia mortality been underestimated because hospitalisation is ignored: findings from a case-crossover study. Environmental Health 12, 8.

[148] Gong, T.Y., Sun, Z.B., Zhang, X.L., et al., 2019. Associations of BC and $PM_{2.5}$ with daily cardiovascular mortality in Beijing, China. Atmospheric Environment 214, 8.

[149] Gong, W., Zhang, T., Zhu, Z., et al., 2015. Characteristics of $PM_{1.0}$, $PM_{2.5}$, and PM_{10}, and their relation to BC in Wuhan, Central China. Atmosphere, 6(9), 1377-1387.

[150] Gorr, M.W., Youtz, D.J., Eichenseer, C.M., et al., 2015. In vitro PM exposure causes direct and lung-mediated indirect effects on cardiomyocyte function. American Journal of Physiology - Heart and Circulatory Physiology 309, H53-H62.

[151] Granier, C., Bessagnet, B., Bond, T, et al., 2011. Evolution of anthropogenic and biomass burning emissions of air pollutants at global and regional scales during the 1980-2010 period. Climate Change 109, 163-190, doi:10.1007/s10584-011-0154-1.

[152] Greve, H.J., Mumaw, C.L., Messenger, E.J., et al., 2020a. Diesel exhaust impairs TREM2 to dysregulate neuroinflammation. Journal of Neuroinflammation 17.

[153] Greve, H.J., Mumaw, C.L., Messenger, E.J., et al., 2020b. Diesel exhaust impairs TREM2 to dysregulate

neuroinflammation. Journal of Neuroinflammation 17.

[154] Grytting, V. S., M. Refsnes, J. Øvrevik, M. S. Halle, J. Schönenberger, R. van der Lelij, B. Snilsberg, T. Skuland, R. Blom and M. Låg 2021. "Respirable stone particles differ in their ability to induce cytotoxicity and pro-inflammatory responses in cell models of the human airways." Particle and Fibre Toxicology 18(1): 18.

[155] Grytting, V. S., P. Chand, M. Lag, J. Ovrevik and M. Refsnes 2022. "The pro-inflammatory effects of combined exposure to diesel exhaust particles and mineral particles in human bronchial epithelial cells." Part Fibre Toxicol 19(1): 14.

[156] Gu, Y., Wong, T. W., Law, C. K., et al., 2018. Impacts of sectoral emissions in China and the implications: Air quality, public health, crop production, and economic costs. Environmental Research Letters, 13(8). https://doi.org/10.1088/1748-9326/aad138.

[157] Guan P.B. Shi H.D, et al., 2017. Study on BC aerosol simulation of climate effect in China (in Chinese). Journal of Environmental Engineering Technology, vol.7, n.4:418-423.

[158] Gurgueira, S.A., Lawrence, J., Coull, B., et al., 2002. Rapid increases in the steady-state concentration of reactive oxygen species in the lungs and heart after particulate air pollution inhalation. Environmental health perspectives 110, págs. 749-755.

[159] Hammer, M. S., A. van Donkelaar, R. V. Martin, et al., 2021) "Effects of COVID-19 lockdowns on fine PM concentrations." Science Advances 7(26): eabg7670.

[160] Han, C., Liu, Y., Ma, J., et al., 2012. Key role of organic carbon in the sunlight-enhanced atmospheric aging of soot by O2. PNAS, 109, 21250-21255.

[161] Han, X., Zhang, M., Gao, J., et al., 2014. Modeling analysis of the seasonal characteristics of haze formation in Beijing. Atmospheric Chemistry and Physics, 14(18), 10231-10248. .

[162] Han, X., Zhang, M., Han, Z., et al., 2011. Simulation of aerosol direct radiative forcing with RAMS-CMAQ in East Asia. Atmospheric Environment, 45(36), 6576-6592. https://doi.org/10.1016/j.atmosenv.2011.08.006.

[163] Hansell, A., Ghosh, R.E., Blangiardo, M., et al., 2016. Historic air pollution exposure and long-term mortality risks in England and Wales: prospective longitudinal cohort study. Thorax 71, 330-338.

[164] Hansen, C.S., Sheykhzade, M., Møller, P., et al., 2007. Diesel exhaust particles induce endothelial dysfunction in apoE-/- mice. Toxicology and Applied Pharmacology 219, 24-32.

[165] Hartz, A.M.S., Bauer, B., Block, M.L., et al., 2008. Diesel exhaust particles induce oxidative stress, proinflammatory signaling, and P-glycoprotein up-regulation at the blood-brain barrier. FASEB Journal 22, 2723-2733.

[166] Hasslöf, H., Molnár, P., Andersson, E.M., et al., 2020. Long-term exposure to air pollution and atherosclerosis in the carotid arteries in the Malmö diet and cancer cohort. Environmental Research 191.

[167] He, C., Li, Q., Liou, K. N., et al., Microphysics-based BC aging in a global CTM: constraints from HIPPO

observations and implications for global BC budget. Atmos. Chem. Phys. 16, 3077-3098 2016.

[168] He, G., Y. Pan and T. Tanaka 2020. "The short-term impacts of COVID-19 lockdown on urban air pollution in China." Nature Sustainability 3(12): 1005-1011.

[169] Heo, J., Schauer, J.J., Yi, O., et al., 2014. Fine Particle Air Pollution and Mortality Importance of Specific Sources and Chemical Species. Epidemiology 25, 379-388.

[170] Hiltermann, T.J.N., Stolk, J., van der Zee, S.C., et al., 1998. Asthma severity and susceptibility to air pollution. European Respiratory Journal 11, 686-693.

[171] Hinds. 1982. Aerosol technology: properties, behavior, and measurement of airborne particles Aerosol technology: properties, behavior, and measurement of airborne particles.

[172] Ho, H.C., Wong, M.S., and Chan, T.C. 2020. Spatially differentiating the effects of long-term air pollution on specific causes of death from cardiovascular and respiratory mortality in Hong Kong: a territory-wide register-based study. Air Quality Atmosphere and Health 13, 721-730.

[173] Hodnebrog, Ø., Aamaas, B., Berntsen, T.K., et al.,. 2013. Klimaeffekt av norske utslipp av kortlevde klimadrivere, CICERO report to the Norwegian Climate and Pollution Agency (KLIF).

[174] Hodnebrog, Ø., Myhre, G., & Samset B. H. 2014. How shorter BC lifetime alters its climate effect. Nat. Commun. 5, 5065.

[175] Hoek, G., Beelen, R., de Hoogh, K., et al., 2008. A review of land-use regression models to assess spatial variation of outdoor air pollution. Atmospheric Environment 42, 7561-7578.

[176] Hoek, G., Brunekreef, B., Goldbohm, S., et al., 2002a. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. Lancet 360, 1203-1209.

[177] Hoek, G., Brunekreef, B., Verhoeff, A., et al., 2000. Daily mortality and air pollution in the Netherlands. Journal of the Air & Waste Management Association 50, 1380-1389.

[178] Hoek, G., Meliefste, K., Cyrys, J., et al., 2002b. Spatial variability of fine particle concentrations in three European areas. 36, 4077-4088.

[179] Hoesly, R. M., Smith, S. J., Feng, L., et al., 2018. Historical (1750–2014) anthropogenic emissions of reactive gases and aerosols from the Community Emissions Data System (CEDS). *Geoscientific Model Development*, *11*(1), 369-408.

[180] Hoesly, R. M., Smith, S. J., Feng, L, et al., 2018. Historical (1750-2014) anthropogenic emissions of reactive gases and aerosols from the Community Emissions Data System (CEDS), Geosci. Model Dev., 11, 369-408, https://doi.org/10.5194/gmd-11-369-2018.

[181] Hoesly, R. M., Smith, S. J., Feng, L., et al., 2018. Historical (1750-2014) anthropogenic emissions of reactive gases and aerosols from the Community Emissions Data System (CEDS). Geosci. Model Dev., 11(1), 369-408. https://doi.org/10.5194/gmd-11-369-2018.

[182] Hong, H., Shuncheng, L., Junji, C., et al., 2005. Characterization of indoor/outdoor organic and elemental carbon in $PM_{2.5}$ during summer in Guangzhou City. Acta Scientiae Circumstantiae 25, 1242-1249.

[183] Hougaard, K.S., Jensen, K.A., Nordly, P., et al., 2008. Effects of prenatal exposure to diesel exhaust particles on postnatal development, behavior, genotoxicity and inflammation in mice. Particle and Fibre Toxicology 5.

[184] Hu, J., Chen, J., Ying, Q., et al., 2016. One-Year Simulation of Ozone and Particulate Matter in China Using WRF/CMAQ Modeling System. Atmospheric Chemistry and Physics Discussions, April, 1-31. https://doi.org/10.5194/acp-2016-148.

[185] Hu, Q.S., Wang, D., Yue, D.L., et al., 2021. Association of ambient particle pollution with gestational diabetes mellitus and fasting blood glucose levels in pregnant women from two Chinese birth cohorts. Science of the Total Environment 762, 10.

[186] Hua, Y., Liu, F., Wu, H., et al., 2020. Effects of alcohol addition to traditional fuels on soot formation: A review. International Journal of Engine Research.

[187] Huang Y H, Shen Z, Chen H, et al., 2014. Quantification of Global Primary Emissions of $PM_{2.5}$, PM_{10} and TSP from Combustion and Industrial Process Sources［J］. Environmental Science & Technology, 48(23):13834－13843.

[188] Huang Y.W., Zhao Y., et al., 2017. Research on the emission inventory of air pollutants from motor vehicles based on different methods--A case study of Jiangsu Province (in Chinese). China Sciencepaper, n3.

[189] Huang, C.H., Lin, L.Y., Tsai, M.S., et al., 2010. Acute cardiac dysfunction after short-term diesel exhaust particles exposure. Toxicology Letters 192, 349-355.

[190] Huang, L., Zhu, Y., Zhai, H., et al., 2021. Recommendations on benchmarks for numerical air quality model applications in China - Part 1: $PM_{2.5}$ and chemical species. Atmospheric Chemistry and Physics, 21(4), 2725-2743. https://doi.org/10.5194/acp-21-2725-2021.

[191] Huang, R. J., Zhang, Y., Bozzetti, C., et al., 2015. High secondary aerosol contribution to particulate pollution during haze events in China. Nature, 514(7521), 218-222. https://doi.org/10.1038/nature13774.

[192] Hudda, N., Eliasziw, M., Hersey, S.O., et al., 2021. Effect of Reducing Ambient Traffic-Related Air Pollution on Blood Pressure: a Randomized Crossover Trial. Hypertension dallas, tex: 1979 77, 823 - 832.

[193] Huebert, B.J., Bates, T., Russell, P.B., et al., 2003. An overview of ACE-Asia: Strategies for quantifying the relationships between Asian aerosols and their climatic impacts. Journal of Geophysical Research: Atmospheres 108.

[194] Hussein, T., C. Johansson, H. Karlsson and H.-C. Hansson 2008. “Factors affecting non-tailpipe aerosol particle emissions from paved roads: On-road measurements in Stockholm, Sweden.” Atmospheric Environment 42(4): 688-702.

[195] Hvidtfeldt, U.A., Severi, G., Andersen, Z.J., et al., 2021. Long-term low-level ambient air pollution exposure and risk of lung cancer - A pooled analysis of 7 European cohorts. Environment International 146.

[196] Hvidtfeldt, U.A., Sorensen, M., Geels, C., et al., 2019. Long-term residential exposure to $PM_{2.5}$, PM_{10}, BC,

NO2, and ozone and mortality in a Danish cohort. Environment International 123, 265-272.

[197] Ikeda, K., Tanimoto, H., Sugita, T., et al., 2017. Tagged tracer simulations of BC in the Arctic: transport, source contributions, and budget. Atmospheric Chemistry and Physics, 17(17), 10515-10533.

[198] Inoue, K.I., Takano, H., Sakurai, M., et al., 2006. Pulmonary exposure to diesel exhaust particles enhances coagulatory disturbance with endothelial damage and systemic inflammation related to lung inflammation. Experimental Biology and Medicine 231, 1626-1632.

[199] IPCC, 2021: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S. L. Connors, C., et al., Cambridge University Press. In Press.

[200] IPCC: Climate change: The IPCC scientific assessment, edited by: Houghton, J. T., Jenkins, G. J., and Ephraums, J. J., Cambridge.

[201] Ischer, P., Ameling, C., Marra, M., et al., 2009. Absence of trends in relative risk estimates for the association between Black Smoke and daily mortality over a 34 years period in The Netherlands. Atmospheric Environment 43, 481-485.

[202] Ito, K., Mathes, R., Ross, Z., et al., 2011. Fine Particulate Matter Constituents Associated with Cardiovascular Hospitalizations and Mortality in New York City. Environmental Health Perspectives 119, 467-473.

[203] J.A.Ogren, and R.J.Charlson. 1983. Elemental carbon in the atmosphere: cycle and lifetime. J Tellus B.

[204] Jacob, D. J. (1999. Introduction to atmospheric chemistry. Princeton University Press.

[205] Jacob, D.J., Crawford, J.H., Kleb, M.M, et al., 2003. Transport and chemical evolution over the Pacific (TRACE-P) aircraft mission: Design, execution, and first results. Journal of Geophysical Research: Atmospheres 108, D003276.

[206] Januario, D., Perin, P.M., Maluf, M., et al., 2010. Biological Effects and Dose-Response Assessment of Diesel Exhaust Particles on In Vitro Early Embryo Development in Mice. Toxicological Sciences 117, 200-208.

[207] Jaspers, I., Sheridan, P.A., Zhang, W., et al., 2009. Exacerbation of allergic inflammation in mice exposed to diesel exhaust particles prior to viral infection. Part Fibre Toxicol 6, 22.

[208] Jensen, K.A., Suo-Anttila, J.M., and Blevins, L.G. 2007. Measurement of Soot Morphology, Chemistry, and Optical Properties in the Visible and near-Infrared Spectrum in the Flame Zone and Overfire Region of Large Jp-8 Pool Fires. Combustion Science and Technology 179, 2453-2487.

[209] Jenwitheesuk, K., Peansukwech, U., and Jenwitheesuk, K. 2020. Accumulated ambient air pollution and colon cancer incidence in Thailand. Scientific Reports 10, 12.

[210] Jeong, S., Lee, J.H., Ha, J.H., et al., 2021. An exploratory study of the relationships between diesel engine exhaust particle inhalation, pulmonary inflammation and anxious behavior. International Journal of Environmental Research and Public Health 18, 1-14.

[211] Ji, D., Gao, W., Maenhaut, W., et al., 2019. Impact of air pollution control measures and regional transport on carbonaceous aerosols in fine PM in urban Beijing, China: insights gained from long-term measurement. Atmospheric Chemistry and Physics, 19(13), 8569-8590.

[212] Ji, D., Gao, W., Maenhaut, W., et al., 2019. Impact of air pollution control measures and regional transport on carbonaceous aerosols in fine PM in urban Beijing, China: Insights gained from long-term measurement. Atmospheric Chemistry and Physics, 19(13), 8569-8590. https://doi.org/10.5194/acp-19- 8569-2019.

[213] Jia, X. 2014. The relationship between indoor and outdoor PM and the health risk assessment based on chemical components and source apportionment Nankai University.

[214] Jiafeng, F., Meng, Q., Qian, L., et al., 2018. The emission estimation and spatial distribution of BC aerosol. Science & Technology Review 036, 38-46.

[215] Johansson, K.O., Dillstrom, T., Monti, M., et al., 2016. Formation and emission of large furans and oxygenated hydrocarbons from flames. Proc Natl Acad Sci U S A 113, 8374-8379.

[216] Jones, R.R., Hoek, G., Fisher, J.A., et al., 2020. Land use regression models for ultrafine particles, fine particles, and BC in Southern California. Science of The Total Environment 699, 134234.

[217] Jonker, M.T.O., & Koelmans, A. A. 2002. Sorption of Polycyclic Aromatic Hydrocarbons and Polychlorinated Biphenyls to Soot and Soot-like Materials in the Aqueous Environment: Mechanistic Considerations. Environmental Science & Technology 36, 3725-3734.

[218] Julia A. Nightingale, R.M., Paul Cullinan, Louise E. Donnelly, et al., 1999. Airway inflammation after controlled exposure to diesel exhaust particulates. American journal of respiratory and critical care medicine.

[219] Jung, H.J., Ko, Y.K., Shim, W.S., et al., 2021. Diesel exhaust particles increase nasal symptoms and IL-17A in house dust mite-induced allergic mice. Scientific reports 11, 16300.

[220] Junker, C. and Liousse, C., 2008. A global emission inventory of carbonaceous aerosol from historic records of fossil fuel and biofuel consumption for the period 1860-1997. Atmospheric Chemistry and Physics, 8, 1195-1207.

[221] Kaewamatawong, T., Shimada, A., Morita, T., et al., 2009. Acute and Subacute Pulmonary Effects of Diesel Exhaust Particles in Mice: Pathological Changes and Translocation Pathways to the Circulation. Thai Journal of Veterinary Medicine 39, 311-318.

[222] Kaiser JW, Heil A, Andreae MO et al., 2012. Biomass burning emissions estimated with a global fire assimilation system based on observed fire radiative power. Biogeosciences. 9:527-54, doi:10.5194/bg-9-527-2012.

[223] Kanaya, Y., Yamaji, K., Miyakawa, T, et al., 2020. Rapid reduction in BC emissions from China: evidence from 2009-2019 observations on Fukue Island, Japan, Atmos. Chem. Phys., 20, 6339-6356, https://doi.org/10.5194/acp-20-6339-2020.

[224] Kanaya, Y., Yamaji, K., Miyakawa, T., et al., 2019. Rapid reduction of BC emissions from China:

evidence from 2009-2019 observations on Fukue Island, Japan. Atmospheric Chemistry and Physics Discussions, x, 1-28. https://doi.org/10.5194/acp-2019-1054.

[225] Karin Kindbom., Ingrid Mawdsley., Ole-Kenneth Nielsen., et al., 2018. Emission factors for SLCP emissions from residential wood combustion in the Nordic countries: Improved emission inventories of Short Lived Climate Pollutants (SLCP). Report for the Nordic Council of Ministers, 2018. p. 76. http://norden.diva-portal.org/smash/record.jsf?pid=diva2%3A1174670&dswid=-3179. .

[226] Karroum, K., Lin, Y., Chiang, Y. Y., et al., 2020. A Review of Air Quality Modeling. Mapan - Journal of Metrology Society of India. https://doi.org/10.1007/s12647-020-00371-8.

[227] Katich, J. M., Samset, B. H., Bui, T. P., et al., 2018. Strong contrast in remote BC aerosol loadings between the Atlantic and Pacific basins. Journal of Geophysical Research: Atmospheres, 123, 13, 386- 13, 395. https://doi.org/10.1029/2018JD029206.

[228] Kemball-Cook, S., Yarwood, G., Johnson, J., et al., 2015. Evaluating NO_x emission inventories for regulatory air quality modeling using satellite and air quality model data. Atmospheric Environment, 117, 1-8. https://doi.org/10.1016/j.atmosenv.2015.07.002.

[229] Kim, D.I., Song, M.K., Kim, H.I., et al., 2020a. Diesel Exhaust Particulates Induce Neutrophilic Lung Inflammation by Modulating Endoplasmic Reticulum Stress-Mediated CXCL1/KC Expression in Alveolar Macrophages. Molecules 25, 16.

[230] Kim, H.S., Kim, B.G., Park, S., et al., 2020a. Gene Expression Analysis to Investigate Biological Networks Underlying Nasal Inflammatory Dysfunctions Induced by Diesel Exhaust Particles Using an In Vivo System. Annals of Otology, Rhinology and Laryngology 129, 245-255.

[231] Kim, J.B., Kim, C., Choi, E., et al., 2012. Particulate air pollution induces arrhythmia via oxidative stress and calcium calmodulin kinase II activation. Toxicology and Applied Pharmacology 259, 66-73.

[232] Kim, S., Park, H., Park, H., et al., 2016. The acute respiratory exposure by intratracheal instillation of Sprague-Dawley rats with diesel PM induces retinal thickening. Cutaneous and Ocular Toxicology 35, 275-280.

[233] Kim, S.J., Kim, N., Park, S.H., et al., 2020b. Genomic approach to explore altered signaling networks of olfaction in response to diesel exhaust particles in mice. Scientific reports 10, 16972.

[234] Kim, S.Y., Dutton, S.J., Sheppard, L., et al., 2015. The short-term association of selected components of fine PM and mortality in the Denver Aerosol Sources and Health DASH study. Environmental Health 14, 11.

[235] Kindbom K, Mawdsley I, Nielsen O K, et al., 2018.Emission factors for SLCP emissions from residential wood combustion in the Nordic countries: Improved emission inventories of Short Lived Climate Pollutants (SLCP)[J]. .

[236] Kipling, Z., Stier, P., Schwarz, J. P., et al., 2013. Constraints on aerosol processes in climate models from vertically-resolved aircraft observations of BC. Atmos. Chem. Phys. 13, 5969-5986 .

[237] Klimakur 2030, 31 January 2020. Tiltak og virkemidler mot 2030 [Mitigation analysis for Norway, 2021-2030: possible measures and policy instruments]. Report M-1625 | 2020. https://www.miljodirektoratet.no/publikasjoner/2021/mars-2021/mitigation-analysis-for-norway-20212030-short-term-climate-impacts-and-co-benefits/. In Norwegian.

[238] Klimont, Z., Cofala, J et al., 2009. Projections of SO2, NO_x and carbonaceous aerosols emissions in Asia. Tellus 61B:602-617.

[239] Klimont, Z., Kupiainen, K., Heyes, C, et al., 2017. Global anthropogenic emissions of PM including BC, Atmos. Chem. Phys., 17, 8681-8723, https://doi.org/10.5194/acp-17- 8681-2017.

[240] Klimont, Z., Kupiainen, K., Heyes, C., et al., 2017. Global anthropogenic emissions of PM including BC. Atmos Chem Phys 17, 8681-8723.

[241] Kodavanti, U.P., Thomas, R., Ledbetter, A.D., et al., 2011. Vascular and cardiac impairments in rats inhaling ozone and diesel exhaust particles. Environmental Health Perspectives 119, 312-318.

[242] Koelmans, A.A., Jonker, M.T., Cornelissen, G., et al., 2006. Black carbon: the reverse of its dark side. Chemosphere 63, 365-377.

[243] Krecl P., Christer J., Targino, et al., 2017.. Trends in BC and size-resolved particle number concentrations and vehicle emission factors under real-world conditions. Atmospheric Environment, 165(sep.), 155-168.

[244] Kupiainen, K., R. Ritola, A. Stojiljkovic, L. Pirjola, A. Malinen and J. Niemi 2016. "Contribution of mineral dust sources to street side ambient and suspension PM_{10} samples." Atmospheric Environment 147: 178-189.

[245] Kurokawa, J., & Ohara, T. 2019. Long-term historical trends in air pollutant emissions in Asia: Regional Emission inventory in ASia (REAS) version 3.1. Atmospheric Chemistry and Physics, December, 1-51. https://doi.org/10.5194/acp-2019-1122.

[246] Kurokawa, J., and Ohara, T. 2019. Long-term historical trends in air pollutant emissions in Asia: Regional Emission inventory in ASia REAS version 3.1. Atmospheric Chemistry and Physics.

[247] Kurokawa, J., Ohara, T., Morikawa, T, et al., 2013. Emissions of air pollutants and greenhouse gases over Asian regions during 2000-2008: Regional emission inventory in ASia (REAS) version 2. Atmospheric Chemistry and Physics 13, 11019-11058.

[248] Kwon, H.S., Ryu, M.H., and Carlsten, C. 2020. Ultrafine particles: unique physicochemical properties relevant to health and disease. Exp Mol Med 52, 318-328.

[249] Labranche, N., El Khattabi, C., Dewachter, L., et al., 2012. Vascular oxidative stress induced by diesel exhaust microparticles: Synergism with hypertension. Journal of Cardiovascular Pharmacology 60, 530-537.

[250] Lack, D.A., Moosmüller, H., Mcmeeking, G.R., et al., 2014. Characterizing elemental, equivalent black, and refractory BC aerosol particles: a review of techniques, their limitations and uncertainties. Analytical and Bioanalytical Chemistry 406, 99-122.

[251] Lamarque J.-F., Bond T.C., Eyring et al., 2010. Historical (1850-2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols: methodology and application. Atmospheric Chemistry and Physics, vol.10:7017-7039.

[252] Laskin, A., Laskin, J., and Nizkorodov, S.A. 2015. Chemistry of atmospheric brown carbon. Chem Rev 115, 4335-4382.

[253] Lavigne, E., Talarico, R., van Donkelaar, A., et al., 2021. Fine PM concentration and composition and the incidence of childhood asthma. Environment International 152, 7.

[254] Le Tertre, A., Quenel, P., Eilstein, D., et al., 2002. Short-term effects of air pollution on mortality in nine French cities: A quantitative summary. Archives of Environmental Health 57, 311-319.

[255] Le, T., Y. Wang, L. Liu, et al., 2020. "Unexpected air pollution with marked emission reductions during the COVID-19 outbreak in China." Science 369(6504): 702-706.

[256] Lee, Y. H., Lamarque, J. F., Flanner, M. G., et al., 2013. Evaluation of preindustrial to present-day BC and its albedo forcing from Atmospheric Chemistry and Climate Model Intercomparison Project (ACCMIP). Atmos. Chem. Phys. 13, 2607-2634.

[257] Lee, Y. H., Lamarque, J. F., Flanner, M. G, et al., 2012. Evaluation of preindustrial to present-day BC and its albedo forcing from ACCMIP (Atmospheric Chemistry and Climate Model Intercomparison Project). Atmospheric Chemistry and Physics, 12, 21713-21778.

[258] Lei, J., Li, Z., Huang, X., et al., 2021. The acute effect of diesel exhaust particles and different fractions exposure on blood coagulation function in mice. International Journal of Environmental Research and Public Health 18.

[259] Lei, Y., Zhang, Q., He, K.B, et al., 2011. Primary anthropogenic aerosol emission trends for China, 1990-2005. Atmos. Chem. Phys. 11, 931-954.

[260] Lei, Y., Zhang, Q., He, K.B., et al., 2011. Primary anthropogenic aerosol emission trends for China, 1990-2005. Atmos Chem Phys 11, 931-954.

[261] Leonard, E., and Aminuddin, M. 2020. The effect of soot particulate towards vascular cell adhesion molecule-1 Vcam-1 expression in the mechanism of cardiovascular system disruption. Medico-Legal Update 20, 830-835.

[262] Li, M., Zhang, Q., Kurokawa, J. I., et al., 2017. MIX: a mosaic Asian anthropogenic emission inventory under the international collaboration framework of the MICS-Asia and HTAP. Atmospheric Chemistry and Physics, 17(2), 935-963.

[263] Li, G., Sun, J., Dai, A., 2016a. Village differences in rural household energy consumption within the Loess hilly region of China. Energy Sustainability and Society 6, 33.

[264] LI, J. L., ZHANG, M. G., GAO, Y., et al., 2016. Model analysis of secondary organic aerosol over China with a regional air quality modeling system (RAMS-CMAQ). Atmospheric and Oceanic Science Letters, 9(6), 443-450. https://doi.org/10.1080/16742834.2016.1233798.

[265] Li, K., Jacob, D. J., Liao, H., et al., 2019a). Anthropogenic drivers of 2013-2017 trends in summer surface ozone in China. Proceedings of the National Academy of Sciences of the United States of America, 116(2), 422-427. https://doi.org/10.1073/pnas.1812168116.

[266] Li, K., Jacob, D. J., Liao, H., et al., 2019b). A two-pollutant strategy for improving ozone and particulate air quality in China. Nature Geoscience, 12(11), 906-910. https://doi.org/10.1038/s41561-019-0464-x.

[267] Li, K., Liao, H., Zhu, J., et al., 2016. Implications of RCP emissions on future $PM_{2.5}$ air quality and direct radiative forcing over China. Journal of Geophysical Research: Atmospheres, 121(21), 12-985. .

[268] Li, M., Liu, H., Geng, G, et al., 2017a. Anthropogenic emission inventories in China: a review. National Science Review 4, 834-866.

[269] Li, M., Zhang, Q., Kurokawa, J.I, et al., 2017b. MIX: a mosaic Asian anthropogenic emission inventory under the international collaboration framework of the MICS-Asia and HTAP. Atmos. Chem. Phys. 17, 935-963.

[270] Li, N., Bian Y.H., et al., 2017. Development and verification of BC and organic carbon anthropogenic emission inventory in Guangdong Province (in Chinese), Acta Scientiae Circumstantiae, vol37, n2.

[271] Li, N., Xia, T., and Nel, A.E. 2008. The role of oxidative stress in ambient PM-induced lung diseases and its implications in the toxicity of engineered nanoparticles. Free Radic Biol Med 44, 1689-1699.

[272] Li, Q., Li, X., Jiang, J, et al., 2016b. Semi-coke briquettes: towards reducing emissions of primary $PM_{2.5}$, particulate carbon, and carbon monoxide from household coal combustion in China. Scientific Reports 6, 19306.

[273] Li, X., Lin, C., Wang, Y, et al., 2015. Analysis of rural household energy consumption and renewable energy systems in Zhangziying town of Beijing. Ecological Modelling 318, 184-193.

[274] Liao, J., Jin, A.Z., Chafe, Z.A, et al., 2017. The impact of household cooking and heating with solid fuels on ambient $PM_{2.5}$ in peri-urban Beijing. Atmospheric Environment 165, 62-72.

[275] Lin, Jian, An, J., Qu, Y., et al., 2016. Local and distant source contributions to secondary organic aerosol in the Beijing urban area in summer. Atmospheric Environment, 124, 176-185. https://doi.org/10.1016/j.atmosenv.2015.08.098.

[276] Lipsett, M., and Campleman, S. 1999. Occupational exposure to diesel exhaust and lung cancer: a meta-analysis. Am J Public Health 89, 1009-1017.

[277] Liu M, Peng X, Meng Z, et al., Spatial characteristics and determinants of in-traffic BC in Shanghai, China: Combination of mobile monitoring and land use regression model[J]. Science of the Total Environment, 2019, 658: 51-61.

[278] Liu, F., Wang, M., & Zheng, M. 2021. Effects of COVID-19 lockdown on global air quality and health. The Science of the total environment, 755(Pt 1), 142533. https://doi.org/10.1016/j.scitotenv.2020.142533. .

[279] Liu, J., Fan, S., Horowitz, L. W., Levy, H. 2011. Evaluation of factors controlling long - range transport of BC to the Arctic. Journal of Geophysical Research: Atmospheres, 116(D4).

[280] Liu, J., Mauzerall, D.L., Chen, Q, et al., 2016. Air pollutant emissions from Chinese households: A major and underappreciated ambient pollution source. Proceedings of the National Academy of Sciences 113, 7756-7761.

[281] Liu, Jiu meng, Wang, P. fei, et al., 2020. Integration of field observation and air quality modeling to characterize Beijing aerosol in different seasons. Chemosphere, 242, 125195. https://doi.org/10.1016/j.chemosphere.2019.125195.

[282] Liu, M., & Matsui, H. 2021. Improved simulations of global BC distributions by modifying wet scavenging processes in convective and mixed‐phase clouds. Journal of Geophysical Research: Atmospheres, 126, e2020JD033890. https://doi.org/10.1029/2020JD033890.

[283] Liu, S., Jorgensen, J.T., Ljungman, P., et al., 2021. Long-term exposure to low-level air pollution and incidence of chronic obstructive pulmonary disease: The ELAPSE project. Environment International 146, 8.

[284] Liu, W., Spaargaren, G., Heerink, N, et al., 2013. Energy consumption practices of rural households in north China: Basic characteristics and potential for low carbon development. Energy Policy 55, 128-138.

[285] Liu, Y., Zhang, Y.X., J., W.Y., 2007. Measurement of emission factors of carbonaceous aerosols from residential coal combustion (in Chinese). Acta Scientiae Circumstantiae 27, 1409-1416.

[286] Ljungman, P.L.S., Andersson, N., Stockfelt, L., et al., 2019. Long-Term Exposure to Particulate Air Pollution, Black Carbon, and Their Source Components in Relation to Ischemic Heart Disease and Stroke. Environmental Health Perspectives 127, 11.

[287] Long, C.M., Nascarella, M.A., and Valberg, P.A. 2013. Carbon black vs. BC and other airborne materials containing elemental carbon: Physical and chemical distinctions. Environmental Pollution 181.

[288] Lu, Z., Zhang, Q., and Streets, D.G. 2011. Sulfur dioxide and primary carbonaceous aerosol emissions in China and India, 1996-2010. Atmos Chem Phys 11, 9839-9864.

[289] Lu, Z., Zhang, Q., Streets, D.G., 2011. Sulfur dioxide and primary carbonaceous aerosol emissions in China and India, 1996-2010. Atmos. Chem. Phys. 11, 9839-9864.

[290] Lund MT, Berntsen TK, Samset BH 2017) Sensitivity of BC concentrations and climate impact to aging and scavenging in OsloCTM2-M7. Atmos. Chem. Phys. 17(9):6003-6022.

[291] Lund, M. T., Myhre, G., Haslerud, A. S., et al., 2018a) Concentrations and radiative forcing of anthropogenic aerosols from 1750 to 2014 simulated with the Oslo CTM3 and CEDS emission inventory. Geosci. Model Dev. 11(12):4909-4931.

[292] Lund, M. T., Samset, B. H., Skeie, R. B., et al., 2018b. Short Black Carbon lifetime inferred from a global set of aircraft observations. Npj Climate and Atmospheric Science, 1(1), 1-8.

[293] Lund, M. T. and Berntsen, T. 2012. Parameterization of black car bon aging in the OsloCTM2 and implications for regional transport to the Arctic, Atmos. Chem. Phys., 12, 6999-7014, doi:10.5194/acp-12-6999-2012.

[294] Lund, M. T., Aamaas, B., Stjern, C. W., et al., 2020. A continued role of short-lived climate forcers under

the Shared Socioeconomic Pathways. Earth System Dynamics, 11(4), 977-993.

[295] Luping, T., Guihong, L., Ni, L., et al., 2020. Research progress of determination methods for environmental BC. environmental chemistry v.39, 170-182.

[296] Schaap, M., Cuvelier, C., Hendriks, C., et al., 2015. Performance of European chemistry transport models as function of horizontal resolution. Atmospheric Environment, 112, 90-105.

[297] Ma, Q., Cai, S., Wang, S., et al., 2017. Impacts of coal burning on ambient $PM_{2.5}$ pollution in China. Atmospheric Chemistry and Physics, 17(7), 4477-4491. .

[298] Ma, Q., Cai, S., Wang, S., et al., 2017. Impacts of coal burning on ambient $PM_{2.5}$ pollution in China. Atmospheric Chemistry & Physics.

[299] Ma, S., Zhang, X., Gao, C., et al., 2019. Multi-model simulations of springtime dust storms in East Asia : Implications of an evaluation of four commonly used air quality. Geoscientific Model Development, April, 1-35.

[300] Ma, W., Zhou, X., Renwick, A., 2019. Impact of off-farm income on household energy expenditures in China: Implications for rural energy transition. Energy Policy 127, 248-258.

[301] Mailroom, U. S. E. P. A. 2010. Peer Review of Source Apportionment Tools in CAMx and CMAQ.

[302] Mar, T.F., Norris, G.A., Koenig, J.Q., et al., 2000. Associations between air pollution and mortality in Phoenix, 1995-1997. Environmental Health Perspectives 108, 347-353.

[303] Maria Sehlstedt, A.F.B., Christoffer Boman, Anders Blomberg, Thomas Sandström, et al., 2010. Airway inflammatory response to diesel exhaust generated at urban cycle running conditions. Inhalation Toxicology 22, 1144-1150.

[304] Marilena Kampa, E.C. 1993. Human health effects of air pollution. Environmental Pollution 151, 362-367.

[305] Mark W. Frampton, Judith C. Stewart, Günter Oberdörster., et al.,2006. Inhalation of ultrafine particles alters blood leukocyte expression of adhesion molecules in humans. Environ Health Perspect; 114: 51--8.

[306] Mark W. Frampton., Judith C. Stewart., Günter Oberdörster., et al., 2006. Inhalation of ultrafine particles alters blood leukocyte expression of adhesion molecules in humans.Environ Health Perspect; 114: 51¬-8.

[307] McCreanor, J., Cullinan, P., Nieuwenhuijsen, M.J., et al., 2007. Respiratory effects of exposure to diesel traffic in persons with asthma. New England journal of medicine 357, 2348 - 2358.

[308] McEwen, J. D. N. & M. R. Johnson. 2012. Black Carbon Particulate Matter Emission Factors for Buoyancy Driven Associated Gas Flares. Journal of the Air & Waste Management Association, 62(3): 307-321.

[309] MEE. 2016.Technical Regulation on Ambient Air Quality Index (on trial), HJ 633-2012. https://www.mee.gov.cn/ywgz/fgbz/bz/ bzwb/jcffbz/201203/t20120302_224166.shtml.

[310] MEE. 2019. China air quality improvement report.

[311] MEE. 2015. The guidelines for the development of inventories of emissions from non-road mobile sources (in Chinese). Beijing China.

[312] Meldrum, K., Guo, C., Marczylo, E.L., et al., 2017. Mechanistic insight into the impact of nanomaterials on asthma and allergic airway disease. Part Fibre Toxicol 14, 45.

[313] Milani, C., Farina, F., Botto, L., et al., 2020. Systemic exposure to air pollution induces oxidative stress and inflammation in mouse brain, contributing to neurodegeneration onset. International Journal of Molecular Sciences 21.

[314] Miller, M.R., McLean, S.G., Duffin, R., et al., 2013. Diesel exhaust particulate increases the size and complexity of lesions in atherosclerotic mice. Particle and Fibre Toxicology 10.

[315] Ming, L., Jin, L., Li, J., et al., 2017. $PM_{2.5}$ in the Yangtze River Delta, China: Chemical compositions, seasonal variations, and regional pollution events. Environmental Pollution, 223, 200-212. https://doi.org/10.1016/j.envpol.2017.01.013.

[316] Ministry of Ecology and Environment of People's Republic of China, 2014. Technical Guidelines for the Development of Atmospheric Fine Particulate Matter Primary Source Emission Inventory (Trial), Beijing.

[317] Ministry of Housing and Urban-Rural Development of the People Republic of China, 2009. Guidance on expanding the pilot of rural dangerous housing renovation, in: Ministry of Housing and Urban-Rural Development of the PRC (Ed.), Beijing.

[318] Moazami, T. N., B. Hilt, K. Soras, K. V. H. Svendsen, et al., 2022. Short-term exposure to stone minerals used in asphalt affect lung function and promote pulmonary inflammation among healthy adults. Scand J Work Environ Health 48(5): 410-418.

[319] Moosmüller, H., Chakrabarty, R.K., and Arnott, W.P. 2009. Aerosol light absorption and its measurement: A review. JOURNAL OF QUANTITATIVE SPECTROSCOPY AND RADIATIVE TRANSFER 110, 844-878.

[320] Mordukhovich, I., Coull, B., Kloog, I., et al., 2015. Exposure to sub-chronic and long-term particulate air pollution and heart rate variability in an elderly cohort: the Normative Aging Study. Environmental Health 14, 10.

[321] Mortamais, M., Gutierrez, L.A., de Hoogh, K., et al., 2021. Long-term exposure to ambient air pollution and risk of dementia: Results of the prospective Three-City Study. Environ Int 148, 106376.

[322] Muller, A.K., Farombi, E.O., Moller, P., et al., 2004. DNA damage in lung after oral exposure to diesel exhaust particles in Big Blue R rats. Mutation Research-Fundamental and Molecular Mechanisms of Mutagenesis 550, 123-132.

[323] Myhre, G., Samset, B. H., Schulz, M., et al., 2013) Radiative forcing of the direct aerosol effect from AeroCom Phase II simulations. Atmos. Chem. Phys. 13(4):1853-1877.

[324] Myhre, G., Aas, W., Cherian, R., et al., 2017) Multi-model simulations of aerosol and ozone radiative forcing due to anthropogenic emission changes during the period 1990-2015. Atmos. Chem. Phys. 17(4):2709-2720.

[325] Myhre G., Samset B. H., Schulz M., et al., 2013. Radiative forcing of the direct aerosol effect from

AeroCom Phase II simulations, Atmos. Chem. Phys. 13(4), 1853-1877, 10.5194/acp-13-1853-2013.

[326] Nemmar, A., Al-Salam, S., Zia, S., et al., 2010a. Time-course effects of systemically administered diesel exhaust particles in rats. Toxicology Letters 194, 58-65.

[327] Nemmar, A., Al-Salam, S., Zia, S., et al., 2010b. Diesel Exhaust Particles in the Lung Aggravate Experimental Acute Renal Failure. Toxicological Sciences 113, 267-277.

[328] Nemmar, A., Al-Salam, S., Zia, S., et al., 2011a. Contrasting actions of diesel exhaust particles on the pulmonary and cardiovascular systems and the effects of thymoquinone. British Journal of Pharmacology 164, 1871-1882.

[329] Nemmar, A., and Inuwa, I.M. 2008. Diesel exhaust particles in blood trigger systemic and pulmonary morphological alterations. Toxicology Letters 176, 20-30.

[330] Nemmar, A., Dhanasekaran, S., Yasin, J., et al., 2009. Evaluation of the direct systemic and cardiopulmonary effects of diesel particles in spontaneously hypertensive rats. Toxicology 262, 50-56.

[331] Nemmar, A., Hoet, P.H.M., Dinsdale, D., et al., 2003. Diesel exhaust particles in lung acutely enhance experimental peripheral thrombosis. Circulation 107, 1202-1208.

[332] Nemmar, A., Karaca, T., Beegam, S., et al., 2016. Prolonged pulmonary exposure to diesel exhaust particles exacerbates renal oxidative stress, inflammation and DNA damage in mice with adenine-induced chronic renal failure. Cellular Physiology and Biochemistry 38, 1703-1713.

[333] Nemmar, A., Subramaniyan, D., Yasin, J., et al., 2013. Impact of experimental type 1 diabetes mellitus on systemic and coagulation vulnerability in mice acutely exposed to diesel exhaust particles. Particle and Fibre Toxicology 10, 10.

[334] Nemmar, A., Subramaniyan, D., Zia, S., et al., 2012. Airway resistance, inflammation and oxidative stress following exposure to diesel exhaust particle in angiotensin II-induced hypertension in mice. Toxicology 292, 162-168.

[335] Nemmar, A., Zia, S., Subramaniyan, D., et al., 2011b. Exacerbation of thrombotic events by diesel exhaust particle in mouse model of hypertension. Toxicology 285, 39-45.

[336] Ni, M., Huang, J., Lu, S, et al., 2014. A review on BC emissions, worldwide and in China. Chemosphere 107, 83-93.

[337] Ni, M., Huang, J., Lu, S., et al., 2014. A review on BC emissions, worldwide and in China. Chemosphere 107, 83-93.

[338] Nicholas L Mills 1., Mark R Miller, Andrew J Lucking., Jon Beveridge., et al.,2011. Combustion-derived nanoparticulate induces the adverse vascular effects of diesel exhaust inhalation. Eur Heart J, 32: 2660-71.

[339] Nicole A. H. Janssen, Kees Meliefste, Oliver Fuchs, et al., 2008. High and low volume sampling of PM at sites with different traffic profiles in the Netherlands and Germany: Results from the HEPMEAP study - ScienceDirect. 42, 1110-1120.

[340] NILU. 2019. Atmos. Chem. Phys., 19, 10217-10237.

[341] Nitter, T. B., B. Hilt, K. V. H. Svendsen, M. Buhagen and R. B. Jorgensen 2021. Association between exposure to different stone aggregates from asphalt and blood coagulability: A human exposure chamber study. Sci Total Environ 778: 146309.

[342] Norwegian Coastal Authority. 2021. AIS gridded emissions. https://utagg.kystverket.no/index.html, last accessed 27. May 2022.

[343] Norwegian Environment Agency. 2013. Summary of proposed action plan for Norwegian emissions of short lived climate forcers. https://www.miljodirektoratet.no/globalassets/publikasjoner/m135/m135.pdf, M135/2014, last accessed 27. May 2022.

[344] Norwegian Environment Agency. 2019. Short-term climate effect and health effects of measures to reduce emissions of climate forcers in Norway. https://www.miljodirektoratet.no/publikasjoner/2019/mars-2019/klimaeffekt-pa-kort-sikt-og-helseeffekter-av-tiltak-for-a-redusere-utslipp-av-klimadrivere-i-norge/, M1215/2018 last accessed 27. May 2022. In Norwegian but with a summary in English.

[345] Norwegian Environment Agency. 2021a. Informative Inventory Report (IIR) 2021, Norway. https://www.miljodirektoratet.no/publikasjoner/2021/mars-2021/informative-inventory-report--iir-2021.-norway/, last accessed 27. May 2022.

[346] Norwegian Environment Agency. 2021b. Norway. 2021 National Inventory Report (NIR) https://unfccc.int/documents/273425, last accessed 27. May 2022.

[347] Norwegian Environment Agency. 2021c. Mitigation analysis for Norway 2021-2030: short-term climate impacts and co-benefits. https://www.miljodirektoratet.no/publikasjoner/2021/mars-2021/mitigation-analysis-for-norway-20212030-short-term-climate-impacts-and-co-benefits/, last accessed 27. May 2022. .

[348] Norwegian Ministry of Climate and Environment. 2018. Norway's Seventh National Communication Under the Framework Convention on Climate Change. https://unfccc.int/files/national_reports/annex_i_natcom/submitted_natcom/application/pdf/529371_norway-nc7-br3-1-nc7_-_br3_-_final.pdf, last accessed 27. May 2022.

[349] Odman, M.T., Hu, Y., Chang, M.E., et al., 2007. Chapter 2.14 Forecasting ozone and $PM_{2.5}$ in southeastern U.S. Dev. Environ. Sci. 6, 220e229. http://dx.doi.org/ 10.1016/S1474-8177(07)06214-6.

[350] OFV. 2021. Bilåret 2020. https://ofv.no/aktuelt/2021/bil%C3%A5ret-2020, last accessed April 2022.

[351] Ohara T, Akimoto H, Kurokawa J, et al., 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980? 2020[J].

[352] Ohara, T., Akimoto, H., Kurokawa, J, et al., 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980-2020. Atmos. Chem. Phys. 7, 4419-4444.

[353] Okayama, Y., Kuwahara, M., Suzuki, A., et al., 2006. Role of reactive oxygen species on diesel exhaust particle-induced cytotoxicity in rat cardiac myocytes. Journal of Toxicology and Environmental Health - Part A: Current Issues 69, 1699-1710.

[354] Onyeso, G., Bohr, E.L., Nkpaa, K.W., et al., 2020. Black soot exposure induced hypothalamic and

testicular oxidative stress and apoptosis in male rats. Andrologia 52.

[355] O'Rourke, P. R, Smith, S. J., Mott, A, et al., 2021. CEDS v-2021-02-05 Emission Data 1975-2019 (Version Feb-05-2021. Zenodo. http://doi.org/10.5281/zenodo.4509372.

[356] Ostro, B., Feng, W.Y., Broadwin, R., et al., 2007. The effects of components of fine particulate air pollution on mortality in California: Results from CALFINE. Environmental Health Perspectives 115, 13-19.

[357] Ostro, B., Hu, J.L., Goldberg, D., et al., 2015a. Associations of Mortality with Long-Term Exposures to Fine and Ultrafine Particles, Species and Sources: Results from the California Teachers Study Cohort. Environmental Health Perspectives 123, 549-556.

[358] Ostro, B., Lipsett, M., Reynolds, P., et al., 2010. Long-Term Exposure to Constituents of Fine Particulate Air Pollution and Mortality: Results from the California Teachers Study. Environmental Health Perspectives 118, 363-369.

[359] Ostro, B., Tobias, A., Karanasiou, A., et al., 2015b. The risks of acute exposure to BC in Southern Europe: results from the MED-PARTICLES project. Occup Environ Med 72, 123-129.

[360] Pace, W.B.K.B.T.G. 2002. Methods for Improving Global Inventories of Black Carbon and Organic Carbon Particulates. Journal of Geophysical Research.

[361] Park, J.H., Troxel, A.B., Harvey, R.G., et al., 2006. Polycyclic aromatic hydrocarbon PAH o-quinones produced by the aldo-keto-reductases AKRs generate abasic sites, oxidized pyrimidines, and 8-oxo-dGuo via reactive oxygen species. Chemical Research in Toxicology 19, 719.

[362] Peacock, J.L., Anderson, H.R., Bremner, S.A., et al., 2011. Outdoor air pollution and respiratory health in patients with COPD. Thorax 66, 591-596.

[363] Pederstad, A., Smith, J.D., Jackson, R., et al.,2015. Assessment of flare strategies, techniques for reduction of flaring and associated emissions, emission factors and methods for determination of emissions to air from flaring. Report number 312|2014. Norwegian Environment Agency. https://www.carbonlimits.no/wp-content/uploads/2015/06/Assessment-of-flare-strategies-techniques-for-reduction-of-flaring-and-associated-emissions-emission.pdf.

[364] Pelegov, D. V. and Eremenko, G. A., 2021. State support of electric vehicle industry in China: the driving forces, actions and perspectives. Economy of Region, 16, (3), 921-934.

[365] Peng, L., Zhang, Q., Yao, Z, et al., 2019. Underreported coal in statistics: A survey-based solid fuel consumption and emission inventory for the rural residential sector in China. Applied Energy 235, 1169-1182.

[366] Penner, J.E., Eddleman, H., Novakov, T., 1993. Towards the development of a global inventory for BC emissions. Atmos. Environ. Part A, vol.27, 1277-1295.

[367] Penning, T.M., Burczynski, M.E., Hung, C.F., et al., 1999. Dihydrodiol dehydrogenases and polycyclic aromatic hydrocarbon activation: generation of reactive and redox active o-quinones. Chemrestoxicol 12, 1.

[368] Petersen, A. K., Brasseur, G. P., Bouarar, I., et al., 2019. Ensemble forecasts of air quality in eastern China-Part 2: Evaluation of the MarcoPolo-Panda prediction system, version 1. Geoscientific Model Development, 12(3), 1241-1266. https://doi.org/10.5194/gmd-12-1241-2019.

[369] Petzold, A., Ogren, J. A., Fiebig, M, et al., 2013. Recommendations for reporting "black carbon" measurements. Atmospheric Chemistry and Physics, 13, (16), 8365-8379.

[370] Petzold, A., Ogren, J.A., Fiebig, M., et al., 2013. Recommendations for reporting "black carbon" measurements. J Atmospheric Chemistry Physics 13, 8365-8379.

[371] Poss, J., Lorenz, D., Werner, C., et al., 2013. Diesel Exhaust Particles Impair Endothelial Progenitor Cells, Compromise Endothelial Integrity, Reduce Neoangiogenesis, and Increase Atherogenesis in Mice. Cardiovascular Toxicology 13, 290-300.

[372] Prescott, G.J., Cohen, G.R., Elton, R.A., et al., 1998. Urban air pollution and cardiopulmonary ill health: a 14.5 year time series study. Occupational and Environmental Medicine 55, 697-704.

[373] Provost, E.B., Louwies, T., Cox, B., et al., 2016. Short-term fluctuations in personal BC exposure are associated with rapid changes in carotid arterial stiffening. Environment International 88, 228-234.

[374] Qi L & Wang S. 2019. Sources of BC in the atmosphere and in snow in the Arctic. Science of The Total Environment 691:442-454.

[375] Qi M, Jiang L, Liu Y, et al., 2018. Analysis of the characteristics and sources of carbonaceous aerosols in $PM_{2.5}$ in the Beijing, Tianjin, and Langfang region, China[J]. International journal of environmental research and public health, 15(7): 1483.

[376] Qi, J., Zheng, B., Li, M, et al., 2017. A high-resolution air pollutants emission inventory in 2013 for the Beijing-Tianjin-Hebei region, China. Atmospheric Environment 170, 156-168.

[377] Qiao, X., Ying, Q., Li, X., et al., 2018. Source apportionment of $PM_{2.5}$ for 25 Chinese provincial capitals and municipalities using a source-oriented Community Multiscale Air Quality model. Science of the Total Environment, 612, 462-471. https://doi.org/10.1016/j.scitotenv.2017.08.272.

[378] Qin SG, T.J., Wen Yp, 2001. Black carbon and its importance in climate change studies. Meteorological Monthly 2711, 3-7.

[379] Qin Y, Xie S D. 2011. Historical estimation of carbonaceous aerosol emissions from biomass open burning in China for the period 1990-2005[J]. Environmental Pollution, 159(12): 3316-3323.

[380] Qin, W., Zhang, Y., Chen, J., et al., 2019. Variation, sources and historical trend of BC in Beijing, China based on ground observation and MERRA-2 reanalysis data. Environmental Pollution 245, 853-863.

[381] Qin, Y., and Xie, S.D., 2011. Spatial and temporal variation of anthropogenic BC emissions in China for the period 1980-2009. Atmos. Chem. Phys. Discuss., vol.11: 32877-32920.

[382] Qiu, Y., Liao, H., Zhang, R., et al., 2017. Simulated impacts of direct radiative effects of scattering and absorbing aerosols on surface layer aerosol concentrations in China during a heavily polluted event in February 2014. Journal of Geophysical Research: Atmospheres, 122(11), 5955-5975.

[383] Quaas J, Boucher O, Lohmann U. 2006. Constraining the total aerosol indirect effect in the LMDZ and ECHAM4 GCMs using MODIS satellite data. Atmos. Chem. Phys. 6(4):947-955.

[384] Quincey, P., Butterfield, D., Green, D., et al., 2011. Black Smoke and Black Carbon: Further investigation of the relationship between these ambient air metrics. Atmospheric Environment 45, 3528-3534.

[385] Raaschou-Nielsen, O., Thorsteinson, E., Antonsen, S., et al., 2020. Long-term exposure to air pollution and mortality in the Danish population a nationwide study. Eclinicalmedicine 28, 10.

[386] Rajamani, K.T., Doherty-Lyons, S., Bolden, C., et al., 2013. Prenatal and Early-Life Exposure to High-Level Diesel Exhaust Particles Leads to Increased Locomotor Activity and Repetitive Behaviors in Mice. Autism Research 6, 248-257.

[387] Reisinger, P., Wonaschütz, A., Hitzenberger, R., et al., 2008. Intercomparison of Measurement Techniques for Black or Elemental Carbon Under Urban Background Conditions in Wintertime: Influence of Biomass Combustion. Environmental Science & Technology 42, 884-889.

[388] Reliene, R., Hlavacova, A., Mahadevan, B., et al., 2005. Diesel exhaust particles cause increased levels of DNA deletions after transplacental exposure in mice. Mutation Research-Fundamental and Molecular Mechanisms of Mutagenesis 570, 245-252.

[389] Rhoden, C.R., Wellenius, G.A., Ghelfi, E., et al., 2005. PM-induced cardiac oxidative stress and dysfunction are mediated by autonomic stimulation. Biochimica Et Biophysica Acta General Subjects 1725, 305-313.

[390] Rivas, I., Donaire - Gonzalez, D., Bouso, L., et al., 2016. Spatiotemporally resolved BC concentration, schoolchildren's exposure and dose in Barcelona. Wiley-Blackwell Online Open 26.

[391] Robertson, S., Gray, G.A., Duffin, R., et al., 2012. Diesel exhaust particulate induces pulmonary and systemic inflammation in rats without impairing endothelial function ex vivo or in vivo. Particle and Fibre Toxicology 9, 13.

[392] Robertson, S., Thomson, A.L., Carter, R., et al., 2014. Pulmonary diesel particulate increases susceptibility to myocardial ischemia/reperfusion injury via activation of sensory TRPV1 and β1 adrenoreceptors. Particle and Fibre Toxicology 11, 1-10.

[393] Roth, L.S.G.Z.J.B.H.B.W. 1989. Carbonaceous particles in fly ash -a source for the de-novo-synthesis of organochlorocompounds. Chemosphere 19, 283-290.

[394] Ru, M., Tao, S., Smith, K, et al., 2015. Direct energy consumption associated emissions by rural-to-urban migrants in Beijing. Environ. Sci. Technol. 49, 13708-13715.

[395] Saber, A.T., Jacobsen, N.R., Bornholdt, J., et al., 2006. Cytokine expression in mice exposed to diesel exhaust particles by inhalation. Role of tumor necrosis factor. Particle and Fibre Toxicology 3.

[396] Samset B. H. 2018a. How cleaner air changes the climate, Science. 360(6385), 148-150, 10.1126/science.aat1723.

[397] Samset, B. H., Myhre, G., Herber, A., et al., 2014) Modelled BC radiative forcing and atmospheric lifetime

in AeroCom Phase II constrained by aircraft observations. Atmos. Chem. Phys. 14(22):12465-12477.

[398] Samset, B. H., Sand, M., Smith, C. J., et al., 2018. Climate Impacts From a Removal of Anthropogenic Aerosol Emissions. Geophysical Research Letters 45(2):1020-1029.

[399] Samset, B. H., Stjern, C. W., Andrews, E., et al., 2018. Aerosol absorption: Progress towards global and regional constraints. Current climate change reports, 4(2), 65-83.

[400] Samset, B. H. & Myhre, G. 2015. Climate response to externally mixed BC as a function of altitude. J. Geophys. Res. Atmos. 120, 2913-2927.

[401] Sandu, A., & Zhang, L. 2008. Discrete second order adjoints in atmospheric chemical transport modeling. Journal of Computational Physics, 227(12), 5949-5983. https://doi.org/10.1016/j.jcp.2008.02.011.

[402] Sarah Koch, Joseph F. Welch, Raymond Tran a, Andrew H. Ramsook, Andy Hung, Christopher Carlsten, Jordan A. Guenette, Michael S. Koehle 2021. Ventilatory responses to constant load exercise following the inhalation of a short-acting ß2-agonist in a laboratory-controlled diesel exhaust exposure study in individuals with exercise-induced bronchoconstriction. Environment international 146.

[403] Schwarz, J. P., Spackman, J. R., Gao, R. S., et al., Global-scale BC profiles observed in the remote atmosphere and compared to models. Geophys. Res. Lett. 37, L18812 2010).

[404] SCPRC. 2013. Notice of the General Office of the State Council on Issuing the Air Pollution Prevention and Control Action Plan. State Council of the People's Republic of China website. http://www.gov.cn/gongbao/content/2013/content_2496394.htm [Chinese], last accessed 10. September 2013.

[405] SCPRC. 2018. Notice of the State Council on Issuing the Three-Year Action Plan for Winning the Blue Sky Defense Battle. State Council of the People's Republic of China website. http://www.gov.cn/zhengce/content/2018-07/03/content_5303158.htm, last accessed 29. August 2019. In Chinese.

[406] Seinfeld, J., 2008. Atmospheric science: Black carbon and brown clouds. Nature Geoscience 1, 15-16.

[407] Seljeskog., Morten., Goile., et al.,2013. Particle emission factors for woodstove firing in Norway. Trondheim, SINTEF Energi AS.

[408] Shah, AP., Pietropaoli, AP., Frasier, LM., et al.,2008. Effect of inhaled carbon ultrafine particles on reactive hyperemia in healthy human subjects. Environ Health Perspect, 116: 375-80.

[409] Shan, M., Wang, P., Li, J, et al., 2015. Energy and environment in Chinese rural buildings: Situations, challenges, and intervention strategies. Build Environ 91, 271-282.

[410] Shears, R.K., Jacques, L.C., Naylor, G., et al., 2020. Exposure to diesel exhaust particles increases susceptibility to invasive pneumococcal disease. Journal of Allergy and Clinical Immunology 145, 1272-1284.e1276.

[411] Shen, G., Xue, M., Chen, Y, et al., 2014. Comparison of carbonaceous PM emission factors among different solid fuels burned in residential stoves. Atmospheric Environment 89, 337-345.

[412] Shen, G., Yang, Y., Wang, W, et al., 2010. Emission factors of PM and elemental carbon for crop residues and coals burned in typical household stoves in China. Environmental Science & Technology 44,

7157-7162.

[413] Shi, Z., C. Song, B. Liu, et al., 2021. "Abrupt but smaller than expected changes in surface air quality attributable to COVID-19 lockdowns." Science Advances 7(3): eabd6696.

[414] Shi, Z., Li, J., Huang, L., et al., 2017. Source apportionment of fine PM in China in 2013 using a source-oriented chemical transport model. Science of the Total Environment, 601-602, 1476-1487. https://doi.org/10.1016/j.scitotenv.2017.06.019.

[415] Shimadera, H., Kojima, T., & Kondo, A. 2016. Evaluation of Air Quality Model Performance for Simulating Long-Range Transport and Local Pollution of $PM_{2.5}$ in Japan. Advances in Meteorology, 2016. https://doi.org/10.1155/2016/5694251.

[416] Shine, K. P., Fuglestvedt, J. S., Hailemariam, K., et al., 2005. Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. Climatic Change, 68(3), 281-302.

[417] Shrestha, G., Traina, S., and Swanston, C. 2010. Black Carbon's Properties and Role in the Environment: A Comprehensive Review. Sustainability 2, 294-320.

[418] Sifaki-Pistolla, D., Lionis, C., Koinis, F., et al., 2017. Lung cancer and annual mean exposure to outdoor air pollution in Crete, Greece. European Journal of Cancer Prevention 26, S208-S214.

[419] Silver, B., Conibear, L., Reddington, C., et al., 2020. Pollutant emission reductions deliver decreased $PM_{2.5}$-caused mortality across China during 2015-2017. Atmospheric Chemistry and Physics Discussions, 2017(February), 1-18. https://doi.org/10.5194/acp-2019-1141.

[420] Simon, H., Baker, K. R., Akhtar, F., et al., 2013. A Direct sensitivity approach to predict hourly ozone resulting from compliance with the National Ambient Air Quality Standard. Environmental science & technology, 47(5), 2304-2313.

[421] Simon, H., Baker, K.R., Akhtar, F., et al., 2013. A direct sensitivity approach to predict hourly ozone resulting from compliance with the national ambient air quality standard. Environ. Sci. Technol. 47, 2304-2313. http://dx.doi.org/10.1021/es303674e.

[422] SINTEF. 2016. Effect of maintenance on particulate emissions from residential woodstoves. M518/2016. .

[423] Sinton, J.E. and Fridley, D.G., 2000. What goes up: Recent trends in China's energy consumption. Energy Policy 28, 671-687.

[424] Skuland, T. S., M. Refsnes, P. Magnusson, et al., 2017. "Proinflammatory effects of diesel exhaust particles from moderate blend concentrations of 1st and 2nd generation biodiesel in BEAS-2B bronchial epithelial cells-The FuelHealth project." Environ Toxicol Pharmacol 52: 138-142.

[425] Skuland, T., V. S. Grytting, M. Lag, et al., 2022. Road tunnel-derived coarse, fine and ultrafine PM: physical and chemical characterization and pro-inflammatory responses in human bronchial epithelial cells. Part Fibre Toxicol 19(1): 45.

[426] Solomon, S., Qin, D., Manning, M, et al., 2007: The Physical Science Basis - Contribution of Working

Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press: Cambridge, UK, New York, U.S.

[427] Song, W.W., He, K.B., Lei, Y., 2012. Black carbon emissions from on-road vehicles in China, 1990-2030. Atmospheric Environment, vol.51:320-328.

[428] Søvde, O. A., Prather, M. J., Isaksen, I. S. A., et al., 2012) The chemical transport model Oslo CTM3. Geosci. Model Dev. 5(6):1441-1469.

[429] Spira-Cohen, A., Chen, L.C., Kendall, M., et al., 2011. Personal Exposures to Traffic-Related Air Pollution and Acute Respiratory Health among Bronx Schoolchildren with Asthma. Environmental Health Perspectives 119, 559-565.

[430] Stanković, A., Nikić, D., Nikolić, M., et al., 2007. Short-term effects of air pollution on cardiovascular mortality in elderly in Niš, Serbia. Central European Journal of Public Health 15, 95-98.

[431] Statistics Norway (SSB). 2013. Emissions of Black carbon and Organic carbon in Norway 1990-2011. https://www.ssb.no/en/natur-og-miljo/artikler-og-publikasjoner/emissions-of-black-carbon-and-organic-carbon-in-norway-1990-2011.

[432] Stjern, C. W., Samset, B. H., Myhre, G., et al., 2017) Rapid Adjustments Cause Weak Surface Temperature Response to Increased Black Carbon Concentrations. Journal of Geophysical Research: Atmospheres 122(21):11, 462-411, 481.

[433] Streets D G, Bond T C, Carmichael G R, et al., 2003. An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. Journal of Geophysical Research: Atmospheres, 108(D21).

[434] Streets, D.G., Bond, T. C., Carmichael, G. R., et al., 2003. An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. Journal of Geophysical Research: Atmospheres 108.

[435] Streets, D.G., Bond, T.C., Carmichael, G.R, et al., 2003. An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. Journal of Geophysical Research Atmospheres 108, 8809.

[436] Streets, D.G., Gupta, S., Waldhoff, S.T, et al., 2001. Black carbon emissions in China. Atmospheric Environment 35, 4281-4296.

[437] Streets, D.G., Zhang, Q., Wang, L, et al., 2006. Revisiting China's CO emissions after the transport and chemical evolution over the Pacific (TRACE-P) mission: Synthesis of inventories, atmospheric modeling, and observations. Journal of Geophysical Research: Atmospheres 111, D07118.

[438] Sun, J., Zhi, G., Jin, W, et al., 2018. Emission factors of organic carbon and elemental carbon for residential coal and biomass fuels in China- A new database for 39 fuel-stove combinations. Atmospheric Environment 190, 241-248.

[439] Sun, S.Z., Cao, W.N., Pun, V.C., et al., 2019. Respirable Particulate Constituents and Risk of Cause-Specific Mortality in the Hong Kong Population. Environmental Science & Technology 53, 9810-9817.

[440] Sun, S.Z., Qiu, H., Ho, K.F., et al., 2016. Chemical components of respirable PM associated with

emergency hospital admissions for type 2 diabetes mellitus in Hong Kong. Environment International 97, 93-99.

[441] Sunyer, J., Castellsague, J., Saez, M., et al., 1996. Air pollution and mortality in Barcelona. Journal of Epidemiology and Community Health 50, S76-S80.

[442] Syrakov, D., Prodanova, M., Georgieva, E., et al., 2016. Simulation of European air quality by WRF-CMAQ models using AQMEII-2 infrastructure. Journal of Computational and Applied Mathematics, 293, 232-245. https://doi.org/10.1016/j.cam.2015.01.032.

[443] Takano, H., Yanagisawa, R., Ichinose, T., et al., 2002. Diesel exhaust particles enhance lung injury related to bacterial endotoxin through expression of proinflammatory cytokines, chemokines, and intercellular adhesion molecule-1. American Journal of Respiratory and Critical Care Medicine 165, 1329-1335.

[444] Takemura T & Suzuki K 2019) Weak global warming mitigation by reducing BC emissions. Scientific Reports 9(1):4419.

[445] Tan, J., Fu, J. S., Carmichael, G. R., et al., 2019. Why models perform differently on PM over East Asia? & ndash; A multi-model intercomparison study for MICS-Asia III. Atmospheric Chemistry and Physics Discussions, 3(July), 1-36. https://doi.org/10.5194/acp-2019-392.

[446] Tang, G., Zhu, X., Xin, J., et al., 2017. Modelling study of boundary-layer ozone over northern China - Part II: Responses to emission reductions during the Beijing Olympics. Atmospheric Research, 193, 83-93. https://doi.org/10.1016/j.atmosres.2017.02.014.

[447] Tao, J., Zhang, L., Cao, J., et al., 2017. Source apportionment of $PM_{2.5}$ at urban and suburban areas of the Pearl River Delta region, south China - With emphasis on ship emissions. Science of the Total Environment, 574, 1559-1570. https://doi.org/10.1016/j.scitotenv.2016.08.175.

[448] Terzano, C., Stefano, F.D., Conti, V., et al., 2010. Air pollution ultrafine particles: Toxicity beyond the lung. European review for medical pharmacological ences 14, 809-821.

[449] The State Council of the People's Republic of China. 2013. "Action Plan on Prevention and Control of Air Pollution." from http://www.gov.cn/zhengce/content/2013-09/13/content_4561.htm.

[450] The State Council of the People's Republic of China. 2018. Three-Year Action Plan to Fight Air Pollution. from http://www.gov.cn/zhengce/content/2018-07/03/content_5303158.htm.

[451] Thompson, R.J., Li, J., Weyant, C.L, et al., 2019. Field emission measurements of solid fuel stoves in Yunnan, China demonstrate dominant causes of uncertainty in household emission inventories. Environmental Science & Technology 53, 3323-3330.

[452] Thurston, G.D., Burnett, R.T., Turner, M.C., et al., 2016. Ischemic Heart Disease Mortality and Long-Term Exposure to Source-Related Components of US Fine Particle Air Pollution. Environmental Health Perspectives 124, 785-794.

[453] Tian, J., Ni, H., Cao, J, et al., 2017. Characteristics of carbonaceous particles from residential coal combustion and agricultural biomass burning in China. Atmospheric Pollution Research 8, 521-527.

[454] Tomaru, M., Takano, H., Inoue, K.I., et al., 2007. Pulmonary exposure to diesel exhaust particles enhances fatty change of the liver in obese diabetic mice. International Journal of Molecular Medicine 19, 17-22.

[455] Tong, D., Cheng, J., Liu, Y, et al., 2020. Dynamic projection of anthropogenic emissions in China: methodology and 2015-2050 emission pathways under a range of socio-economic, climate policy, and pollution control scenarios. Atmos. Chem. Phys. 20, 5729-5757.

[456] Tran, L.K., Quang, T.N., Hue, N.T., et al., 2018. Exploratory assessment of outdoor and indoor airborne BC in different locations of Hanoi, Vietnam. Sci Total Environ 642, 1233-1241.

[457] Travis, K.R., Jacob, D.J., Fisher, J.A., et al., 2016. Why do models overestimate surface ozone in the Southeast United States? Atmos. Chem. Phys. 16, 13561e13577. http://dx.doi.org/10.5194/acp-16-13561-2016.

[458] Tripathy, S., Tunno, B.J., Michanowicz, D.R., et al., 2019. Hybrid land use regression modeling for estimating spatio-temporal exposures to $PM_{2.5}$, BC, and metal components across a metropolitan area of complex terrain and industrial sources. 673, 54-63.

[459] Ueda, K., Yamagami, M., Ikemori, F., et al., 2016. Associations Between Fine Particulate Matter Components and Daily Mortality in Nagoya, Japan. Journal of Epidemiology 26, 249-257.

[460] UNEP and WMO, 2011. Integrated assessment of BC and tropospheric ozone: summary for decision makers.

[461] UNEP, & WMO. 2011. Integrated assessment of BC and tropospheric ozone. fossil fuels.

[462] UNEP, 2015. The Climate and Environmental Benefits of Controlling SLCPs in P.R. China. United Nations Environment Programme (UNEP), Nairobi, Kenya.

[463] United Nations Environment Program (UNEP). 2019. Synergizing action on the environment and climate: good practice in China and around the globe. https://www.ccacoalition.org/en/resources/synergizing-action-environment-and-climate-good-practice-china-and-around-globe. Also available in Chinese.

[464] United Nations Environment Programme 2019. A Review of 20 Years' Air Pollution Control in Beijing. Nairobi, Kenya.

[465] USEPA, 2012. Report to congress on BC (EPA-450/R-12-001). EPA, US.

[466] USEPA. 40 CFR Part 50. https://www3.epa.gov/ttn/amtic/40cfr50.html [2006-10-17]/2020-11-13.

[467] Van den Brule, S., Rappe, M., Ambroise, J., et al., 2021. Diesel exhaust particles alter the profile and function of the gut microbiota upon subchronic oral administration in mice. Particle and Fibre Toxicology 18.

[468] Van, B.D., Hullmann, M., and Schins, R.P. 2012. Toxicology of ambient PM. EXS 101, 165.

[469] Vignati, E., Karl, M., Krol, M, et al., 2010. Sources of uncertainties in modelling BC at the global scale. Atmospheric Chemistry and Physics. 10, 2595-2611.

[470] Von Klot, S., Gryparis, A., Tonne, C., et al., 2009. Elemental Carbon Exposure at Residence and Survival After Acute Myocardial Infarction. Epidemiology 20, 547-554.

[471] Vora, R., Zareba, W., Utell, M.J., et al., 2014. Inhalation of ultrafine carbon particles alters heart rate and

heart rate variability in people with type 2 diabetes. Particle and fibre toxicology 11, 31.

[472] Wang, Jia, Li, X., Zhang, W., et al., 2016. Secondary $PM_{2.5}$ in Zhengzhou, China: Chemical species based on three years of observations. Aerosol and Air Quality Research, 16(1), 91-104. https://doi.org/10.4209/aaqr.2015.01.0007.

[473] Wang, L., Jang, C., Zhang, Y., et al., 2010. Assessment of air quality benefits from national air pollution control policies in China. Part II: Evaluation of air quality predictions and air quality benefits assessment. Atmospheric Environment, 44(28), 3449-3457. https://doi.org/10.1016/j.atmosenv.2010.05.058.

[474] Wang, M. Y., Yim, S. H. L., Wong, D. C., et al., 2019b. Source contributions of surface ozone in China using an adjoint sensitivity analysis. Science of the Total Environment, 662, 385-392. https://doi.org/10.1016/j.scitotenv.2019.01.116.

[475] Wang, Peng, Chen, Y., Hu, J., et al., 2019a). Source apportionment of summertime ozone in China using a source-oriented chemical transport model. Atmospheric Environment, 211(May), 79-90. https://doi.org/10.1016/j.atmosenv.2019.05.006.

[476] Wang, Q, et al., Global budget and radiative forcing of BC aerosol: constraints from pole-to-pole (HIPPO) observations across the Pacific. J. Geophys. Res. Atmos. 119, 195-206 2014).

[477] Wang, R. and Jiang, Z., 2017. Energy consumption in China's rural areas: A study based on the village energy survey. Journal of Cleaner Production 143, 452-461.

[478] Wang, R., Tao, S., Wang, W, et al., 2012. Black carbon emissions in China from 1949 to 2050. Environmental Science & Technology 46, 7595-7603.

[479] Wang, R., Tao, S., Wang, W., et al., 2012. Black carbon emissions in China from 1949 to 2050. Environmental Science and Technology, 46(14), 7595-7603. https://doi.org/10.1021/es3003684.

[480] Wang, R., Tao, S., Wang, W., et al., 2012. Black carbon emissions in China from 1949 to 2050. Environmental Science and Technology. 46, 7595−7603.

[481] Wang, W.D., Liu, C., Ying, Z.K., et al., 2019. Particulate air pollution and ischemic stroke hospitalization: How the associations vary by constituents in Shanghai, China. Science of the Total Environment 695, 8.

[482] Wang, X., Li, K., Li, H, et al., 2017. Research on China's rural household energy consumption - Household investigation of typical counties in 8 economic zones. Renewable and Sustainable Energy Reviews 68, 28-32.

[483] Wang, Y., Zhang, Q., Jiang, J., et al., 2014. Key Role of Nitrate in Phase Transitions of Urban Particles: Implications of Important Reactive Surfaces for Secondary Aerosol Formation. Journal of Geophysical Research: Atmospheres, 425-440. https://doi.org/10.1002/2013JD021426.Received.

[484] Wauters, A., Vicenzi, M., De Becker, B., et al., 2015. At high cardiac output, diesel exhaust exposure increases pulmonary vascular resistance and decreases distensibility of pulmonary resistive vessels. American journal of physiology Heart and circulatory physiology 309, H2137 - 2144.

[485] Wilker, E.H., Mostofsky, E., Fossa, A., et al., 2018. Ambient Pollutants and Spontaneous Intracerebral

Hemorrhage in Greater Boston. stroke 49, 2764-2766.

[486] Winijkul, E. and Bond, T.C., 2016. Emissions from residential combustion considering end-uses and spatial constraints: Part II, emission reduction scenarios. Atmospheric Environment 124, 1-11.

[487] World Health Organization, 2020. WHO Director-General's statement on IHR Emergency Committee on Novel Coronavirus 2019-nCoV). https://www.who.int/dg/speeches/detail/who-director-general-s-statement-on-ihr-emergency-committee-on-novel-coronavirus-2019-ncov), last accessed 12 Aug 2020.

[488] Xiao, G.G., Wang, M., Li, N., et al., 2003. Use of Proteomics to Demonstrate a Hierarchical Oxidative Stress Response to Diesel Exhaust Particle Chemicals in a Macrophage Cell Line. Journal of Biological Chemistry 278, 50781-50790.

[489] Xiao, H., Li, J., Li, Z, et al., 2017. Energy consumption in rural Beijing: Current situation and major influence factors - based on the survey of 1866 rural househoulds in Beijing (In Chinese). Journal of China Agricultural Resources and Regional Planning 38, 127-137.

[490] Xie, M., Zhu, K., Wang, T., et al., 2014. Application of photochemical indicators to evaluate ozone nonlinear chemistry and pollution control countermeasure in China. Atmospheric environment, 99, 466-473.

[491] Xinhua, 2020. Xi Jinping's speech at the general debate of the 75th session of the UN General Assembly, 2020. http://www.qstheory.cn/yaowen/2020-09/22/c_1126527766.htm, last accessed 27. May 2022.

[492] Xinhua, 2021. China's national carbon market starts online trading, 2021. http://english.www.gov.cn/statecouncil/ministries/202107/16/content_WS60f0ee0dc6d0df57f98dd171.html, last accessed 27. May 2022.

[493] Xu, H., Ren, Y., Zhang, W., et al., 2021. Updated Global Black Carbon Emissions from 1960 to 2017: Improvements, Trends, and Drivers. Environmental science & technology 55, 7869-7879.

[494] Xu, J., Zhang, J., Liu, J., et al., 2019. Influence of cloud microphysical processes on BC wet removal, global distributions, and radiative forcing. Atmospheric Chemistry and Physics, 19(3), 1587-1603.

[495] Xu, J.W., Ma, J.L., et al., 2018. Study on BC Emission characteristics of Motor Vehicles in Guangdong Province (in Chinese). Transport Energy Conservation & Environmental Protection, n5.

[496] Xu, X.H., Kherada, N., Hong, X.R., et al., 2009. Diesel exhaust exposure induces angiogenesis. Toxicology Letters 191, 57-68.

[497] Yamaji, K., Chatani, S., Itahashi, S., et al., 2020. Model inter-comparison for $PM_{2.5}$ Components over urban areas in Japan in the J-STREAM framework. Atmosphere, 11(3), 1-25. https://doi.org/10.3390/atmos11030222.

[498] Yamaji, K., Li, J., Uno, I., et al., 2010. Impact of open crop residual burning on air quality over Central Eastern China during the Mount Tai experiment 2006 (MTX2006. Atmospheric Chemistry and Physics, 10(15), 7353-7368.

[499] Yamazaki, S., Shima, M., Nakadate, T., et al., 2014. Association between traffic-related air pollution and

development of asthma in school children: Cohort study in Japan. Journal of Exposure Science and Environmental Epidemiology 24, 372-379.

[500] Yan, H., Wu, Y, et al., 2014. Emission characteristics and concentrations of vehicular BC in a typical freeway traffic environment of Beijing (in Chinese). Acta Scientiae Circumstantiae, vol36, n8.

[501] Yang, H., Chen, J., Wen, J., et al., 2016. Composition and sources of $PM_{2.5}$ around the heating periods of 2013 and 2014 in Beijing: Implications for efficient mitigation measures. Atmospheric Environment, 124, 378-386. https://doi.org/10.1016/j.atmosenv.2015.05.015.

[502] Yang, H.M., Butterworth, L., Munson, A.E., et al., 2003. Respiratory exposure to diesel exhaust particles decreases the spleen IgM response to a T cell-dependent antigen in female B6C3F1 mice. Toxicological Sciences 71, 207-216.

[503] Yang, H.M., Ma, J.Y.C., Castranova, V., et al., 1997. Effects of diesel exhaust particles on the release of interleukin-1 and tumor necrosis factor-alpha from rat alveolar macrophages. Experimental Lung Research 23, 269-284.

[504] Yang, J., Kang, S., Ji, Z., et al., 2018. Modeling the origin of anthropogenic BC and its climatic effect over the Tibetan Plateau and surrounding regions. Journal of Geophysical Research: Atmospheres, 123(2), 671-692. .

[505] Yang, J., Zhou, M.G., Li, M.M., et al., 2020a. Fine PM constituents and cause-specific mortality in China: A nationwide modelling study. Environment International 143, 7.

[506] Yang, W., Xu, Y., Pan, H., et al., 2020b. Chronic exposure to diesel exhaust PM impairs meiotic progression during spermatogenesis in a mouse model. Ecotoxicology and Environmental Safety 202.

[507] Yang, Y., Russell, L. M., Lou, S., et al., 2017. Dust-wind interactions can intensify aerosol pollution over eastern China. Nature Communications, 8(May), 1-8. https://doi.org/10.1038/ncomms15333.

[508] Yang, Y., Tang, R., Qiu, H., et al., 2018. Long term exposure to air pollution and mortality in an elderly cohort in Hong Kong. Environment International 117, 99-106.

[509] Yap, C., Beverl, I.J., et al., 2012. Association between long-term exposure to air pollution and specific causes of mortality in Scotland. Occupational and Environmental Medicine 69, 916-924.

[510] Yin, X.J., Dong, C.C., Ma, J.Y.C., et al., 2005. Sustained effect of inhaled diesel exhaust particles on T-lymphocyte-mediated immune responses against Listeria monocytogenes. Toxicological Sciences 88, 73-81.

[511] Yin, X.J.J., Dong, C.C., Ma, J.Y.C., et al., 2004. Suppression of cell-mediated immune responses to Listeria infection by repeated exposure to diesel exhaust particles in brown Norway rats. Toxicological Sciences 77, 263-271.

[512] Yokota, S., Oshio, S., and Takeda, K. 2016. In utero exposure to diesel exhaust particles induces anxiogenic effects on male offspring via chronic activation of serotonergic neuron in dorsal raphe nucleus. Journal of Toxicological Sciences 41, 583-593.

[513] Yokota, S., Sato, A., Umezawa, M., et al., 2015. In utero exposure of mice to diesel exhaust particles affects spatial learning and memory with reduced N-methyl-D-aspartate receptor expression in the hippocampus of male offspring. Neurotoxicology 50, 108-115.

[514] Yokota, S., Seki, T., Naito, Y., et al., 2008. Tracheal instillation of diesel exhaust particles component causes blood and pulmonary neutrophilia and enhances myocardial oxidative stress in mice. Journal of Toxicological Sciences 33, 609-620.

[515] Yoo, S.E., Park, J.S., Lee, S.H., et al., 2019. Comparison of Short-Term Associations between $PM_{2.5}$ Components and Mortality across Six Major Cities in South Korea. International Journal of Environmental Research and Public Health 16, 13.

[516] Yoshizaki, K., Brito, J.M., Toledo, A.C., et al., 2010. Subchronic effects of nasally instilled diesel exhaust particulates on the nasal and airway epithelia in mice. Inhalation Toxicology 22, 610-617.

[517] Yun, Y.F., 2020. Study on ship BC emission in North pole (in Chinese). China marine safety, n10, 40-42.

[518] Yutanto, D. 2020. The effect of soot particulate towards MAPK expression in the mechanism of cardiovascular system disruption. Medico-Legal Update 20, 267-272.

[519] Zeghnoun, A., Czernichow, P., Beaudeau, P., et al., 2001. Short-term effects of air pollution on mortality in the cities of Rouen and Le Havre, France, 1990-1995. Archives of Environmental Health 56, 327-335.

[520] Zhang, B., Wang, Y., & Hao, J. 2015. Simulating aerosol–radiation–cloud feedbacks on meteorology and air quality over eastern China under severe haze conditionsin winter. Atmospheric Chemistry and Physics, 15(5), 2387-2404.

[521] Zhang, C., Zheng, J., Zang, P, et al., 2015. Rural household energy consumption structure and its influencing factors in Beijing-Tianjin-Hebei region (in Chinese). Chinese Agricultural Science Bulletin 31, 258-262.

[522] Zhang, H., Qi, C., Bing, X., et al., 2014. $PM_{2.5}$ and Tropospheric Ozone in China and Pollutant Emission Controlling Integrated Analyses. Advances in Climate Change Research, 10, 289-296.

[523] Zhang, Hanyu, Cheng, S., Yao, S., et al., 2019. Multiple perspectives for modeling regional $PM_{2.5}$ transport across cities in the Beijing-Tianjin-Hebei region during haze episodes. Atmospheric Environment, 212(May), 22-35. https://doi.org/10.1016/j.atmosenv.2019.05.031.

[524] Zhang, Hongliang, Chen, G., Hu, J., et al., 2014. Evaluation of a seven-year air quality simulation using the Weather Research and Forecasting (WRF)/Community Multiscale Air Quality (CMAQ) models in the eastern United States. Science of the Total Environment, 473-474, 275-285. https://doi.org/10.1016/j.scitotenv.2013.11.121.

[525] Zhang, K., O'Donnell, D., Kazil, J., et al., The global aerosol-climate model ECHAM-HAM, version 2: sensitivity to improvements in process representations. Atmos. Chem. Phys. 12, 8911-8949 2012. .

[526] Zhang, M. 2004. Modeling of organic carbon aerosol distributions over east asia in the springtime. China Particuology, 2(5), 192-195. https://doi.org/10.1016/s1672-2515(07)60057-8.

[527] Zhang, M. G., Xu, Y. F., Zhang, R. J., et al., 2005. Emission and concentration distribution of BC aerosol in East Asia during springtime. Acta Geophysica Sinica, 48(1), 46-51.

[528] Zhang, N., Qin, Y., Xie, S. 2013. Spatial distribution of BC emissions in China. Atmospheric Science, Chinese Science Bulletin, 58, 3830-3839.

[529] Zhang, N., Qin, Y., Xie, S., 2013. Spatial distribution of BC emissions in China. Chinese Science Bulletin 58, 3830-3839.

[530] Zhang, Q., Streets, D. G., Carmichael, G. R., et al., 2009. Asian emissions in 2006 for the NASA INTEX-B mission. Atmospheric Chemistry and Physics, 9(14), 5131-5153.

[531] Zhang, Q., Streets, D.G., Carmichael, G.R, et al., 2009. Asian emissions in 2006 for the NASA INTEX-B mission. Atmospheric Chemistry & Physics 9, 5131-5153.

[532] Zhang, Q., Streets, D.G., Carmichael, G.R., et al., 2009. Asian emissions in 2006 for the NASA INTEX-B mission. Atmospheric Chemistry & Physics Discussions 9, 5131-5153.

[533] Zhang, Q., Zheng, Y., Tong, D., et al., 2019. Drivers of improved $PM_{2.5}$ air quality in China from 2013 to 2017. Proceedings of the National Academy of Sciences of the United States of America, 116(49), 24463-24469. https://doi.org/10.1073/pnas.1907956116.

[534] Zhang, Y., Jiang, J., Ye, J, et al., 2014. Analysis of rural life energy consumption in Beijing-Tianjin-Hebei and suggestion on coal reduction and replace (In Chinese). Energy of China 56, 39-43.

[535] Zhang, Y., Schauer, J.J., Zhang, Y, et al., 2008. Characteristics of particulate carbon emissions from real-world Chinese coal combustion. Environmental Science & Technology 42, 5068-5073.

[536] Zhang, Y., Zhang, X.L., Fan, X.G., et al., 2020. Modifying effects of temperature on human mortality related to BC particulates in Beijing, China. Atmospheric Environment 243, 9.

[537] Zhang, Y., Zhi, G., Guo, S, et al., 2020. Algorithm developed for dynamic quantification of coal consumption for and emission from rural winter heating. Science of the Total Environment. 737:doi: 10.1016/j.scitotenv.2020.139762.

[538] Zhang, Yang, Zhang, X., Wang, L., et al., 2016. Application of WRF/Chem over East Asia: Part I. Model evaluation and intercomparison with MM5/CMAQ. Atmospheric Environment, 124, 285-300. https://doi.org/10.1016/j.atmosenv.2015.07.022.

[539] Zhang, Yanjun, Cai, J., Wang, S., et al., 2017a). Review of receptor-based source apportionment research of fine PM and its challenges in China. Science of the Total Environment, 586, 917-929. https://doi.org/10.1016/j.scitotenv.2017.02.071.

[540] Zhang, Z., Wang, W., Cheng, M., et al., 2017b). The contribution of residential coal combustion to $PM_{2.5}$ pollution over China's Beijing-Tianjin-Hebei region in winter. Atmospheric Environment, 159, 147-161. https://doi.org/10.1016/j.atmosenv.2017.03.054.

[541] Zhao X., 2019. Top-down estimate of regional BC emissions using ground and satellite observations for the Yangtze River Delta Region. Nanjing: Nanjing University; .

[542] Zhao, B., Wang, P., Ma, J.Z, et al., 2012. A high-resolution emission inventory of primary pollutants for the Huabei region, China. Atmos. Chem. Phys. 12, 481-501.

[543] Zhao, B., Wang, S., Donahue, N. M., et al., 2016. Quantifying the effect of organic aerosol aging and intermediate-volatility emissions on regional-scale aerosol pollution in China. Scientific Reports, 6(April), 1-10. https://doi.org/10.1038/srep28815.

[544] Zhao, W., Xu, Q., Li, L, et al., 2015. Estimation of air pollutant emissions from coal burning in the semi-rural areas of Beijing plain (in Chinese). Research of Environmental Sciences 28, 869-876.

[545] Zhao, Y., Nielsen, C.P., Lei, Y, et al., 2011. Quantifying the uncertainties of a bottom-up emission inventory of anthropogenic atmospheric pollutants in China. Atmos. Chem. Phys. 11, 2295-2308.

[546] Zhao, Y., Nielsen, C.P., Lei, Y., et al., 2011. Quantifying the uncertainties of a bottom-up emission inventory of anthropogenic atmospheric pollutants in China. ATMOSPHERIC CHEMISTRY AND PHYSICS 11.

[547] Zhao, Y., Zhang, J., and Nielsen, C.P. 2012. The effects of recent control policies on trends in emissions of anthropogenic atmospheric pollutants and CO_2 in China. Atmospheric Chemistry & Physics Discussions 12, 24985-25036.

[548] Zhao, Y., Zhang, J., Nielsen, C.P., 2013. The effects of recent control policies on trends in emissions of anthropogenic atmospheric pollutants and CO_2 in China. Atmos. Chem. Phys. 13, 487-508.

[549] Zheng X., 2016. Emisison characteristics of BC and polycyclic aromatic hydrocarbon from heavy-duty diesel vehicles based on PEMS method (Ph.D. dissertation, in Chinese). Tsinghua University, Beijing, China.

[550] Zheng, H., Cai, S., Wang, S., et al., 2019. Development of a unit-based industrial emission inventory in the Beijing-Tianjin-Hebei region and resulting improvement in air quality modeling. Atmospheric Chemistry and Physics, 19(6), 3447-3462. https://doi.org/10.5194/acp-19-3447-2019.

[551] Zhi G. R., 2008. Measurement methods of BC and its emission factors for household coal combustion, Guangzhou Institute of Geochemistry. Chinese Academy of Sciences, Guangzhou.

[552] Zhi Guorui, C.J., Yang Junchao 2015. Origin, properties, measurement and emission estimation of brown carbon aerosols. Research of Environmental Sciences 28, 18.

[553] Zhi, G., Chen, Y., Feng, Y, et al., 2008. Emission Characteristics of Carbonaceous Particles from Various Residential Coal-Stoves in China. Environmental Science & Technology 42, 3310.

[554] Zhi, G., Peng, C., Chen, Y, et al., 2009. Deployment of coal-briquettes and improved stoves: possibly an option for both environment and climate. Environmental Science & Technology. 43, 5586-5591.

[555] Zhi, G., Yang, J., Zhang, T, et al., 2015. Rural household coal use survey, emission estimation and policy implications (in Chinese). Research of Environmental Sciences 28, 1179-1185.

[556] Zhi, G., Zhang, Y., Sun, J, et al., 2017. Village energy survey reveals missing rural raw coal in northern China: Significance in science and policy. Environmental Pollution 233, 705-712.

[557] Zhou S.L., Wei W.D, et al., 2018. Cross-regional BC transfer and management strategy: based on MRIO and WRF/Chem models(in Chinese). Journal of Environmental Economics, n4.

[558] Zhou, G., Xu, J., Xie, Y., et al., 2017. Numerical air quality forecasting over eastern China: An operational application of WRF-Chem. Atmospheric Environment, 153, 94-108. https://doi.org/10.1016/j.atmosenv.2017.01.020.

[559] Zhou, J.A., Ito, K., Lall, R., et al., 2011. Time-Series Analysis of Mortality Effects of Fine Particulate Matter Components in Detroit and Seattle. Environmental Health Perspectives 119, 461-466.

[560] Zhu, J., Liao, H., Mao, Y., et al., 2017. Interannual variation, decadal trend, and future change in ozone outflow from East Asia. Atmospheric Chemistry and Physics, 17(5), 3729-3747.

[561] Zhu, X., Yun, X., Meng, W, et al., 2019. Stacked Use and Transition Trends of Rural Household Energy in Mainland China. Environmental Science & Technology 53, 521-529.

[562] Zhuang, B. L., Wang, T. J., Liu, J., et al., 2014. Continuous measurement of BC aerosol in urban Nanjing of Yangtze River Delta, China. Atmospheric Environment, 89, 415-424.

[563] Zuo, L., Youtz, D.J., and Wold, L.E. 2011. Particulate Matter Exposure Exacerbates High Glucose-Induced Cardiomyocyte Dysfunction through ROS Generation. Plos One 6, 8.

Annex

Annex I: Emission related

Table 1 the ratios of BC and OC from PM found in the literature

Mobile source category		Fuel type	China 2 and before		China 3		China 4		China 5	
			BC	OC	BC	OC	BC	OC	BC	OC
Vehicle		Diesel	0.51	0.32	0.51	0.32	0.66	0.21	0.70	0.21
		Gasoline	0.27	0.58	0.27	0.58	0.27	0.58	0.27	0.58
Off-Road	construction machinery	Diesel	0.31	0.44	0.41	0.29	/	/	/	/
	agricultural machinery	Diesel	0.31	0.44	0.41	0.29	/	/	/	/

Table 2 PM_{10} emission factors of vehicles recommended in the guideline (g/km)

			China 1	China 2	China 3	China 4	China 5
Passenger Vehicle	Light-Duty	Gasoline	0.029	0.012	0.008	0.003	0.003
		Diesel	0.070	0.058	0.036	0.034	0.034
	Medium-Duty	Gasoline	0.067	0.020	0.012	0.007	0.007
		Diesel	0.516	0.174	0.164	0.118	0.059
	Heavy-Duty	Gasoline	0.177	0.080	0.049	0.049	0.049
		Diesel	1.092	0.980	0.439	0.280	0.140
Truck	Light-Duty	Gasoline	0.067	0.020	0.012	0.007	0.007
		Diesel	0.299	0.290	0.114	0.064	0.013
	Medium-Duty	Gasoline	0.177	0.080	0.049	0.049	0.049
		Diesel	1.006	0.303	0.190	0.110	0.022
	Heavy-Duty	Gasoline	0.177	0.080	0.049	0.049	0.049
		Diesel	0.692	0.558	0.270	0.153	0.030

Table 3 PM_{10} emission factors of machinery recommended in the guideline (g/kW·h)

Power Range	Fuel Type	Before China 1	China 1	China 2	China 3
Power＜37 kW	Diesel	1.2	1	0.95	0.55
Power＞37, ＜56 kW	Diesel	1.0	0.85	0.40	0.35
Power＞75, ＜130 kW	Diesel	0.8	0.7	0.3	0.25
Power＞130 kW	Diesel	0.7	0.54	0.20	0.18

The BC/OC emission factors obtained in the project are listed in Table 5 (on-road vehicles) and Table 6 (off-road machinery) with Table 1 multiplied the data in Table 2 and Table 3.

Table 4 BC factors of inland ships from the literature

Literature	EF/(g/kg fuel)	Method
Lack D.A, Light absorbing carbon emissions from commercial shipping, Geophys. 25 Res. Lett., 35, 2008	0.36-1	Optical
Agrawal, H., Emission measurements from a crude oil tanker at sea, Environmental Science & Technology, 42 (19), 2008	0.1	Thermal
Corbett, J.J., Updated emissions from ocean shipping. Journal of Geophysical Research: Atmospheres, 108, 4650, 2003	0.37	/
Petzold, A., Recommendations for reporting ”black carbon” measurements. Atmospheric Chemistry and Physics, 13, 2013	0.06(85% load) 0.36(10% load)	Thermal
Naya Olmer, Bryan Comer, et al., Greenhouse gas emissions from global shipping, 2013-2015 Detailed methodology, ICCT, 2017	stroke, 50%laod HFO: 0.49(g/kg fuel) Distillate: 0.26(g/kg fuel) Average of HFO and Distillate: 0.375	/

Table 5 BC/OC emission factors of on-road vehicles (g/kW·h)

			China 1		China 2		China 3		China 4		China 5	
			BC	OC	BC	OC	BC	OC	BC	OC	BC	OC
Passenger Vehicle	Light-Duty	Gasoline	0.00783	0.01682	0.00324	0.00696	0.00216	0.00464	0.00081	0.00174	0.00081	0.00174
		Diesel	0.0357	0.0224	0.02958	0.01856	0.02016	0.01152	0.02244	0.00714	0.02448	0.00714
	Medium-Duty	Gasoline	0.01809	0.03886	0.0054	0.0116	0.00324	0.00696	0.00189	0.00406	0.00189	0.00406
		Diesel	0.26316	0.16512	0.08874	0.05568	0.09184	0.05248	0.07788	0.02478	0.04284	0.01239
	Heavy-Duty	Gasoline	0.04779	0.10266	0.0216	0.0464	0.01323	0.02842	0.01323	0.02842	0.01323	0.02842
		Diesel	0.55692	0.34944	0.4998	0.3136	0.24584	0.14048	0.1848	0.0588	0.1008	0.0294
Truck	Light-Duty	Gasoline	0.01809	0.03886	0.0054	0.0116	0.00324	0.00696	0.00189	0.00406	0.00189	0.00406
		Diesel	0.15249	0.09568	0.1479	0.0928	0.06384	0.03648	0.04224	0.01344	0.00936	0.00273
	Medium-Duty	Gasoline	0.04779	0.10266	0.0216	0.0464	0.01323	0.02842	0.01323	0.02842	0.01323	0.02842
		Diesel	0.51306	0.32192	0.15453	0.09696	0.1064	0.0608	0.0726	0.0231	0.01584	0.00462
	Heavy-Duty	Gasoline	0.04779	0.10266	0.0216	0.0464	0.01323	0.02842	0.01323	0.02842	0.01323	0.02842
		Diesel	0.35292	0.22144	0.28458	0.17856	0.1512	0.0864	0.10098	0.03213	0.0216	0.0063

Table 6 BC/OC emission factors of off-road machineries (g/kW·h)

Machinery Type		Before China 1		China 1		China 2		China 3	
		BC	OC	BC	OC	BC	OC	BC	OC
Construction Machinery	Excavators (100 kW)	0.248	0.352	0.217	0.308	0.093	0.132	0.1125	0.0825
	Bulldozers (120 kW)	0.248	0.352	0.217	0.308	0.093	0.132	0.1125	0.0825
	Loaders (135 kW)	0.217	0.308	0.1674	0.2376	0.062	0.088	0.081	0.0594
	Forklifts (40 kW)	0.31	0.44	0.2635	0.374	0.124	0.176	0.1575	0.1155
	Rollers (110 kW)	0.248	0.352	0.217	0.308	0.093	0.132	0.1125	0.0825
	Paving machinery (80 kW)	0.248	0.352	0.217	0.308	0.093	0.132	0.1125	0.0825
	Graders (110 kW)	0.248	0.352	0.217	0.308	0.093	0.132	0.1125	0.0825
	Others (Diesel)(30 kW)	0.372	0.528	0.31	0.44	0.2945	0.418	0.2475	0.1815
Agriculture Machinery	Larger and Medium Tractors (29.2 kW)	0.372	0.528	0.31	0.44	0.2945	0.418	0.2475	0.1815
	Small Tractors (9.6 kW)	0.372	0.528	0.31	0.44	0.2945	0.418	0.2475	0.1815
	Combine Harvesters (42.5 kW)	0.31	0.44	0.2635	0.374	0.124	0.176	0.1575	0.1155
	Irrigation Machinerys (14.9 kW)	0.372	0.528	0.31	0.44	0.2945	0.418	0.2475	0.1815
	Others (Diesel)(3.0 kW)	0.372	0.528	0.31	0.44	0.2945	0.418	0.2475	0.1815

Annex Ⅱ: List of Air Pollutant Emission Standards from 2013 to 2020

NO.	Name of standard	Code of standard
	Stationary sources	
1	Emission standard for air pollutants for electronic glass industry	GB 29495—2013
2	Emission standard for air pollutants for brick and tile industry	GB 29620—2013
3	Emission standard for air pollutants for cement industry	GB 4915—2013
4	Emission standard for pollutants for battery industry	GB 30484—2013
5	Standard for pollution control on co-processing of solid wastes in cement kilns	GB 30485—2013
6	Standard for pollution control on the municipal solid waste incineration	GB 18485—2014
7	Emission standard for air pollutants for boilers	GB 13271—2014
8	Emission standards for pollutants for stannum, antimony, and mercury industries	GB 30770—2014
9	Emission standard for pollutants for petroleum refining industry	GB 31570—2015
10	Emission standard for pollutants for petroleum chemistry industry	GB 31571—2015
11	Emission standard for pollutants for synthetic resin industry	GB 31572—2015
12	Emission standard for pollutants for inorganic chemical industry	GB 31573—2015
13	Emission standard for pollutants for secondary copper, aluminum, lead, and zinc industry	GB 31574—2015
14	Emission standard for air pollutants for crematories	GB 13801—2015
15	Emission standard for industrial pollutants for caustic alkali and polyvinyl chloride industry	GB 15581—2016
16	Standard for fugitive emission of volatile organic compounds	GB 37822—2019
17	Emission standard for air pollutants for paint, ink, and adhesive industry	GB 37824—2019
18	Emission standard for air pollutants for pharmaceutical industry	GB 37823—2019
19	Emission standard for air pollutants for foundry industry	GB 39726—2020
20	Emission standard for air pollutants for pesticide industry	GB 39727—2020
21	Emission standard for air pollutants for onshore oil and gas exploitation and production industry	GB 39728—2020
22	Emission standard for air pollutants for bulk petroleum terminals	GB 20950—2020
23	Emission standard for air pollutants for gasoline filling stations	GB 20952—2020
24	Technical specification for pollution control of fly-ash from municipal solid waste incineration	HJ 1134—2020

NO.	Name of standard	Code of standard
	Transportation	
1	Limits and measurement methods for exhaust pollutants from diesel engines of non-road mobile machinery(CHINA Ⅲ，Ⅳ)	GB 20891—2014
2	Technical requirements and measurement methods for emissions from light-duty hybrid electric vehicles	GB 19755—2016
3	Limits and measurement methods for emissions from motorcycles(CHINA Ⅳ)	GB 14622－2016
4	Limits and measurement methods for exhaust pollutants from marine engines (CHINA Ⅰ, Ⅱ)	GB 15097—2016
5	Limits and measurement methods for emissions of pollutants from mopeds(CHINA Ⅳ)	GB 18176－2016
6	Limits and measurement methods for emissions from light-duty vehicles(CHINA Ⅵ)	GB 18352.6—2016
7	Measurement method and specifications for exhaust pollutants from in-use diesel vehicles by remote sensing method	HJ 845—2017
8	Measurement method and technical specifications for PEMS test of exhaust pollutants from heavy-duty diesel and gas fuelled vehicles	HJ 857—2017
9	Limits and measurement methods for emissions from diesel fuelled heavy-duty vehicles (CHINA Ⅵ)	GB 17691—2018
10	Limits and measurement methods for emissions from diesel vehicles under free acceleration and lugdown cycle	GB 3847—2018
11	Limits and measurement methods for exhaust smoke from non-road mobile machinery equipped with diesel engine	GB 36886—2018
12	Limits and measurement methods for emissions from gasoline vehicles under two-speed idle conditions and short driving mode conditions	GB 18285—2018
13	Measurement methods for non-regulated emissions from methanol fuelled vehicles	HJ 1137—2020
14	Emission standard for air pollutants for petroleum transport	GB 20951—2020
15	Emissions control technical requirements for non-road diesel mobile machinery	HJ 1014—2020

Annex Ⅲ: EU Policies and Regulations Relevant for Black Carbon Mitigation

The EU has issued numerous ambitious climate and clean air regulations and policies. Presented here are the most relevant overarching plans and commitments for BC and OC reductions not described elsewhere in this report.

The European Green Deal provides an action plan to boost the efficient use of resources by moving to a clean, circular economy to restore biodiversity and cut pollution. The plan outlines investments needed and financing tools available. It explains how to ensure a just and inclusive transition.

The EU aims to be climate neutral in 2050. A European Climate Law has been adopted to turn this political commitment into a legal obligation. Reaching this target will require action by all sectors of the economy, including investing in environmentally friendly technologies, supporting industry to innovate cleaner, cheaper, and healthier forms of private and public transport, decarbonising the energy sector, ensuring buildings are more energy efficient, and working with international partners to improve global environmental standards.

The EU will also provide financial support and technical assistance to help those most affected by the move towards the green economy. This is called the Just Transition Mechanism. It will help mobilise at least €100 billion over the period 2021-2027 in the most affected regions.

In its Member States, the EU has adopted and implemented numerous regulations that also reduce emissions of BC/OC. Although not necessarily specified for reducing BC, but other pollutants, these implementations also reduce emissions of BC and/or OC.

Norway is also currently considering implementation of the EU's National Ceilings Directive (NEC Directive), and will eventually also have pursuant reporting obligations for NO_x, NMVOCs, SO_2, ammonia and $PM_{2.5}$.

The most relevant EU regulations on emissions of BC are covered in sections 3.3 and 3.4 of this report.

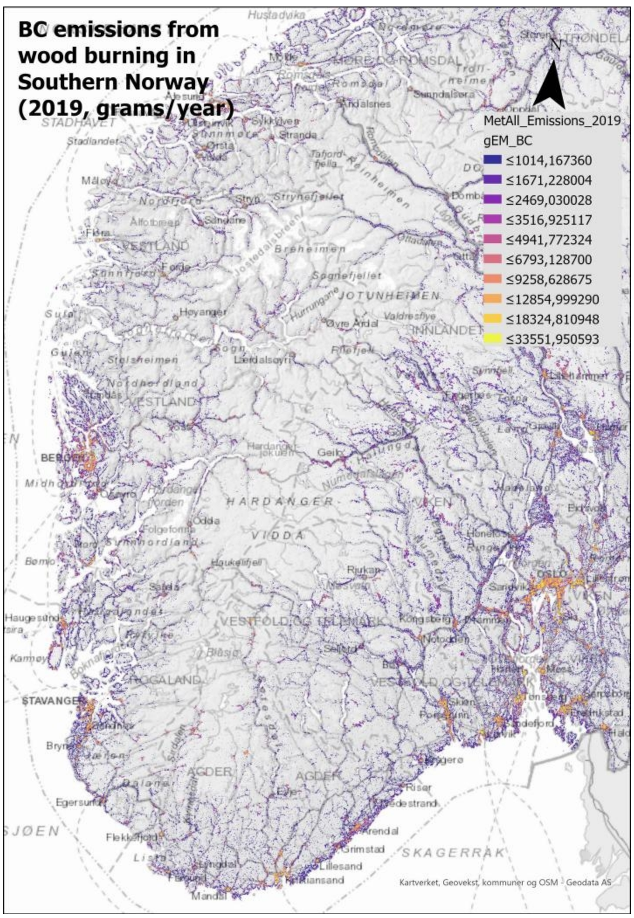
The EU-funded Action on Black Carbon in the Arctic is an initiative sponsored by the European Union to contribute to the development of collective responses to reduce BC emissions in the

Arctic and to reinforce international cooperation to protect the Arctic environment. It provides and communicates knowledge about sources and emissions of BC and supports relevant international policy processes. The EU-funded Action on Black Carbon in the Arctic is implemented through the EU Partnership Instrument providing 1.5 million EUR of funding during 2018-2020.

The initiative supports processes aimed at setting clear commitments and/or targets for reducing BC emissions from major BC sources (gas flaring, domestic heating, maritime shipping), and further enhances international cooperation on BC policy in the Arctic region—with a special focus on supporting the work of the Arctic Council and Convention on Long-range Transboundary Air Pollution and other national, regional, and international initiatives, and building strong collaboration with EU strategic partners. Thus far the project has published two reviews: Review of Observation Capacities and Data Availability for Black Carbon in the Arctic Region (EU, 2019a) and Review of Reporting Systems for National Black Carbon Emissions Inventories (EU, 2019b). Working recommendations will be explored further during the course of this EU Action on Black Carbon in the Arctic and elaborated in an upcoming Roadmap for International Cooperation on Black Carbon.

Annex Ⅳ: Fine scale emission models for BC and PM in Norway

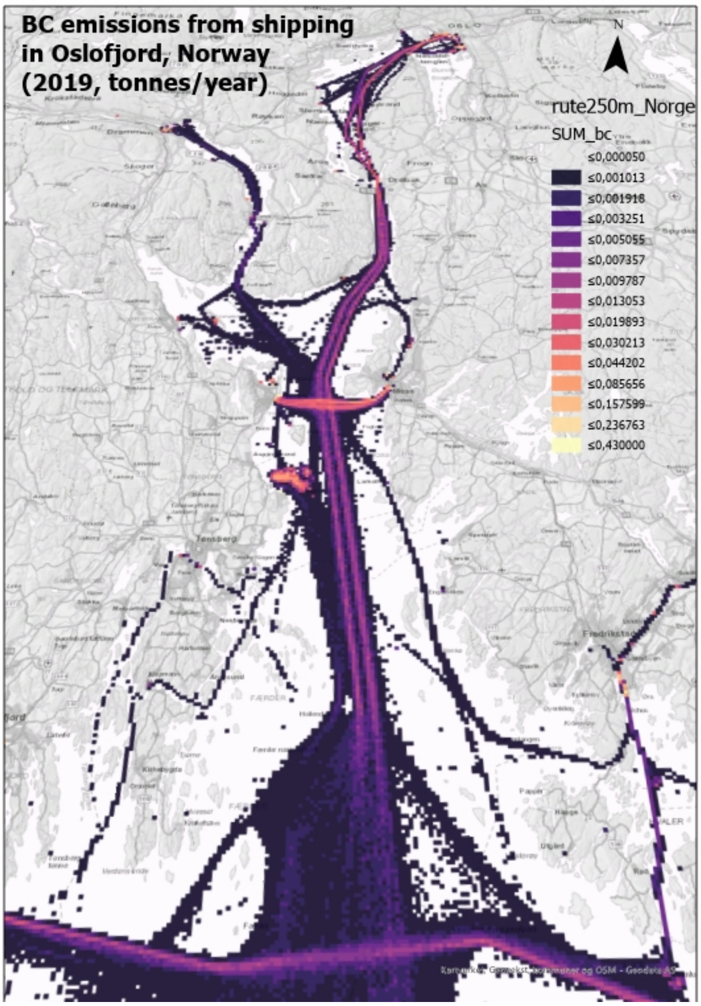
The MetVed-2 model calculates emissions (including BC) from wood burning at a high spatial and temporal resolution for the entire country (NILU, 2019) Wood burning is calculated from residential and recreational housing (Figure 1).



	Emissions 2019
MetVed-2	Metric t/a
SUM_GridConsum	890
SUM_gEM_$PM_{2.5}$	11708
SUM_gEM_PM_{10}	12741
SUM_gEM_TSP	13135
SUM_gEM_BC	740
SUM_gEM_OC	7109
SUM_gEM_CH_4	7716
SUM_gEM_CO	83107
SUM_gEM_PAH	18
SUM_gEM_PAH_O	3
SUM_gEM_ PAH_4	1
SUM_gEM_NO_x	873
SUM_gEM_PCB	52
SUM_gEM_Dioxin	0
SUM_gEM_CO_2	1426367
SUM_gEM_NMVOC	15747
SUM_gEM_N_2O	28
SUM_gEM_SO_2	36

Figure 1 Left: BC emissions from wood burning in Southern Norway for 2019 (g/a, 250 m grids).
Right: emissions total for various components from the model, 2019 (t/a).

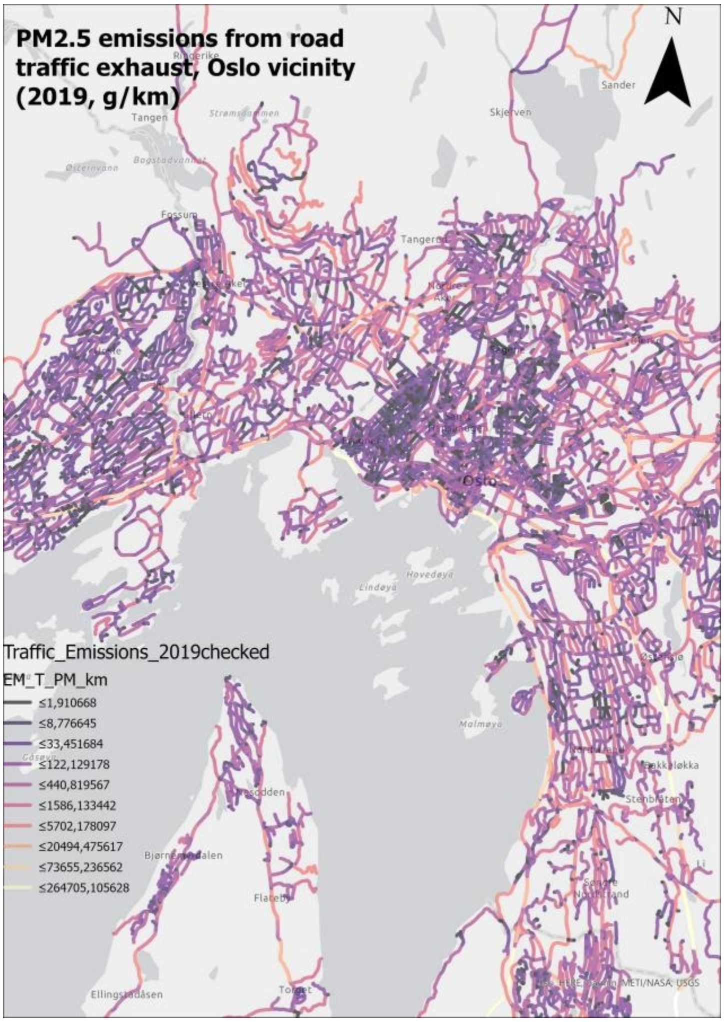
The UtAgg model distributes AIS emission data (including BC) from shipping at a high spatial and temporal resolution (Norwegian Coastal Authority, 2021). These emission data are also available for the main 15 ship categories (Figure 2).



UtAgg Shipping	Metric tonn 2019
SUM_SUM_fuelco	1547368
SUM_SUM_CO_2	4913857
SUM_SUM_CO	10583
SUM_SUM_SO_2	8085
SUM_SUM_NO_x	73950
SUM_SUM_PM	3658
SUM_SUM_nmVOC	3463
SUM_SUM_N_2O	146
SUM_SUM_CH_4	424
SUM_SUM_BC	279
SUM_SUM_OC	856

Figure 2 Left: BC emissions from shipping in the Oslofjord for 2019 (t/a, 250 m grids). Right: emissions totals for various components from the model, 2019 (t/a).

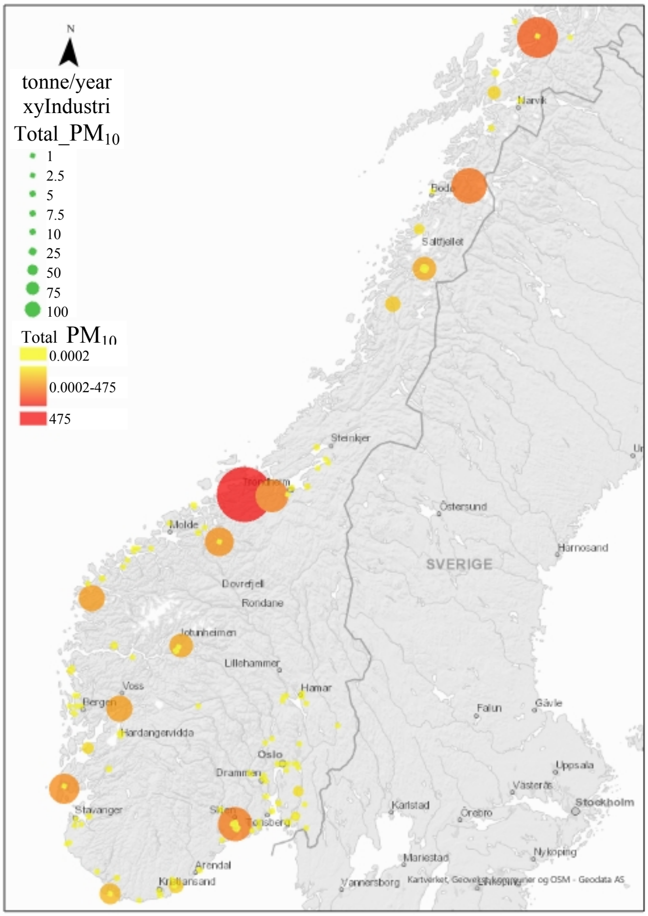
The NERVE model calculates $PM_{2.5}$ emissions from road traffic exhaust at a high spatial and temporal resolution. The emissions data are presented for selected vehicle categories (Figure 3).



Emissions exhaust	Metric tonn 2019
NO_x cars	17871
NO_x trucks	9915
NO_x busses	1526
NO_x TOTAL	**29313**
$PM_{2.5}$ cars	385
$PM_{2.5}$ trucks	1147
$PM_{2.5}$ busses	29
$PM_{2.5}$ TOTAL	**629**

Figure 3 Left: $PM_{2.5}$ emissions from road traffic exhaust in the Oslo area for 2019 (g/km). Right: emission totals for various components and vehicle types from the model, 2019 (t/a).

The Tilde database collects emission data reported for industries and includes data for soot and total PM (Figure 4).



Landbasert industri : Utslipp av Sot (SOT) (i tonn per år) *

År	Til luft	Til vann
2002	26,60	(I.R.)
2003	29,08	(I.R.)
2004	25,72	(I.R.)
2005	15,43	(I.R.)
2006	11,11	(I.R.)
2007	11,15	(I.R.)
2008	9,68	(I.R.)
2009	6,04	(I.R.)
2010	30,55	(I.R.)
2011	22,47	(I.R.)
2012	25,25	(I.R.)
2013	7,98	(I.R.)
2014	20,38	(I.R.)
2015	7,10	(I.R.)
2016	32,37	(I.R.)
2017	32,08	(I.R.)
2018	7,42	(I.R.)
2019	3,67	(I.R.)

Figure 4 Left: PM_{10} emissions from industries in Norway for 2019 (t/a).
Right: emissions totals for soot, 2002-2019.