Transportation Research Part D 54 (2017) 30–49

Contents lists available at ScienceDirect

Transportation Research Part D

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

The impact of coordinated policies on air pollution emissions from road transportation in China

Paul Natsuo Kishimoto [a,*], Valerie J. Karplus [b], Min Zhong [c], Eri Saikawa [c], Xu Zhang [d], Xiliang Zhang [d]

[a] *Joint Program on the Science and Policy of Global Change, Massachusetts Institute of Technology, MA, USA*
[b] *Sloan School of Management, Massachusetts Institute of Technology, MA, USA*
[c] *Department of Environmental Sciences, Emory University, Atlanta, GA, USA*
[d] *Institute of Energy, Environment & Economy, Tsinghua University, Beijing, PR China*

ARTICLE INFO

Article history:
Available online 12 May 2017

ABSTRACT

Improving air quality across mainland China is an urgent policy challenge. While much of the problem is linked to China's broader reliance on coal and other fossil fuels across the energy system, road transportation is an important and growing source of air pollution. Here we use an energy-economic model, embedded in a Regional Emissions Air Quality Climate and Health (REACH) integrated assessment framework, to analyze the impacts of implementing vehicle emissions standards (ES) together with a broader economy-wide climate policy on total air pollution in five species and 30 Chinese provinces. We find that full and immediate implementation of existing vehicle ES at China 3/III level or tighter will significantly reduce the contribution of transportation to degraded air quality by 2030. We further show that road transport ES function as an important complement to an economy-wide price on CO_2, which delivers significant co-benefits for air pollution reduction that are concentrated primarily in non-transportation sectors. Going forward, vehicle emissions standards and an economy-wide carbon price form a highly effective coordinated policy package that supports China's air quality and climate change mitigation goals.

1. Introduction

Air quality is exacting a rising toll on human health and quality of life in China. In response, a broad variety of policy measures have been announced, and some enacted – these include increasing monitoring and reporting to understand the scope and spatial/temporal nature of the problem; setting technology standards; assessing fines and pollution charges; and directly influencing the economic activities of which pollutants are a byproduct.

Transportation is the target of an important subset of these policies. Fossil fuels (gasoline and diesel) burned in road vehicles (cars, trucks, buses, taxis, etc.) result in direct emissions of pollutants, including those listed in Table 1. These direct emissions combine with emissions from other large combustion sources – especially electric power plants and industrial facilities – to affect ambient concentrations of pollutants such as fine particulate matter ($PM_{2.5}$) and ozone (O_3), which in turn impact human health.

Transportation sector policies – summarized in Section 1.1 – include standards regulating the allowable tailpipe emissions of specific pollutants from new private passenger vehicles ('light-duty' vehicles, or LDVs), and heavy-duty vehicles

* Corresponding author.
E-mail address: pnk@MIT.EDU (P.N. Kishimoto).

http://dx.doi.org/10.1016/j.trd.2017.02.012

Table 1
Primary pollutant species in this analysis, and other species included in the Regional Emissions in ASia (REAS) database, version 2.1 (Kurokawa et al., 2013), used as the basis for emissions projections in this study. 'VOCs' are volatile organic compounds.

Name	Chemical formula
This analysis:	
Black carbon	BC
Carbon monoxide	CO
Nitrogen oxides	NO_x
Organic carbon	OC
Sulfur dioxide	SO_2
Also in REAS 2.1:	
Methane	CH_4
Carbon dioxide	CO_2
Nitrous oxide	N_2O
Ammonia	NH_3
Non-methane VOCs	NMV
Particulate matter $\leqslant 10\ \mu m$	PM_{10}
Particulate matter $\leqslant 2.5\ \mu m$	$PM_{2.5}$

(HDVs) including light-, medium- and heavy trucks for freight transport, and buses for passenger transport. These standards may be set to promote installation of specific technology, such as diesel particulate filters, for compliance. Impurities in gasoline and diesel fuel are also regulated, to ensure that these emissions control technologies (ECTs) can function. Collectively, we refer to the combination of road vehicle tailpipe and fuel quality standards as "emissions standards" (ES).

At the same time, China's broader climate and energy policy agenda has important implications for air quality. The US-China Joint Announcement on Climate Change in November 2014, and China's subsequent pledged contribution to global climate mitigation efforts, target a reversal of rising CO_2 emissions by 2030 at the latest. Achieving this goal will require economy-wide policies that price CO_2 emissions through taxation or permits, which are currently being piloted in some regions and expected to be enacted nationwide within the next few years. Climate policy and vehicle emissions policies will both act on the energy and transportation systems, with important implications for future air pollution emissions and air quality outcomes.

To better understand how these policies will act together to affect future air pollution in China, we develop an integrated energy-economic modeling framework and study the interaction of road transportation emissions standards and an economy-wide CO_2 price.

Main findings. We find that ES are projected to be highly effective in reducing the total quantity of emissions from road vehicles, despite rapid growth in transportation activity to 2030 – especially when these policies are deployed and, importantly, enforced in an accelerated manner nationwide. This deployment will be important as the demand for passenger and freight vehicle travel grows, and with it the energy use associated with pollution emissions.

We further find that emission standards and economy-wide climate policy are complementary, in that they target the largest sources of primary pollutant emissions in a harmonious and non-competitive manner. Climate policy reduces CO_2 in sectors where the marginal costs of abatement are lower and delivers substantial co-benefits in the form of air pollution reduction. Since these least-cost opportunities for CO_2 abatement are mainly outside of the transportation sector, technology standards (including ES) that directly target pollution in the transportation sector deliver a significant additional contribution to reducing air pollution. Thus, a CO_2 price plus vehicle emissions standards function as effective and complementary coordinated strategies for addressing air pollution and climate change in China.

An integrated assessment framework for Chinese air quality. Integrated assessment supports the evaluation of policy by allowing comparison of policy-related changes in economic activity (i.e., costs) with the benefits of avoided economic burden of pollution. This research documents part of a Regional Emissions, Air Quality, Climate and Health (REACH) integrated assessment framework (Fig. 1), developed to quantify the human health impacts of Chinese air quality policies.

In particular, we document the first two steps in the framework. In step 1, policy affects future economic activity, including transportation, and associated energy use, within a general equilibrium framework. For this purpose, we extend the China Regional Energy Model (C-REM), as described in Section 2.1. In step 2, energy use and future, energy-denominated emissions factors derived from detailed emissions inventories and policy scenarios lead to quantified changes in emissions of primary pollutant species (Table 1) from transportation and other energy uses.

In atmospheric simulation (step 3), policy-related reductions in primary pollution, through modeled meteorology and chemistry, yield projected changes in the concentration of secondary pollutants (including fine particulate matter, $PM_{2.5}$, and tropospheric ozone,O_3) which reflect the spatial movement of species. Finally (step 4), reductions in population-weighted exposure to $PM_{2.5}$ and O_3 are translated to reductions in adverse human health effects, using future population densities and China-specific exposure-response relationships. Consequent changes in labor lost to illness and health-care expenses determine the change, due to policy, in the economic burden of pollution. These latter two steps are the subject of ongoing work, and are not discussed in the present study.

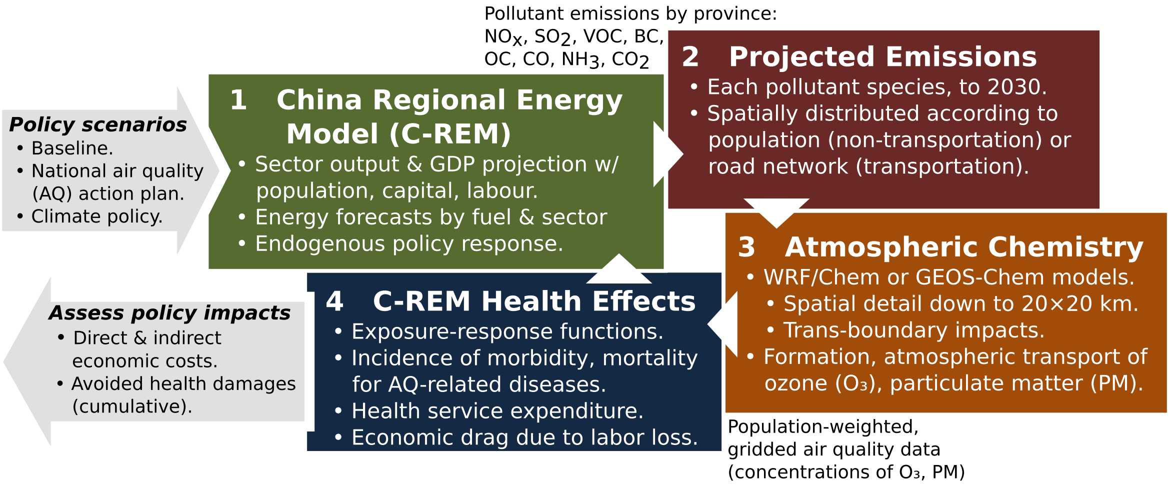


Fig. 1. Regional emissions, air quality, consumption & health (REACH) framework.

1.1. Policy background

Established emissions standards. In 2000, the Chinese Ministry of Environmental Protection issued GB 18352.1-2001, its first national standard on emissions from new road vehicles. Referred to as China 1 (for light-duty vehicles) and China I (for trucks and other heavy-duty vehicles), these specified quantities similar to the European Union's Euro 2/II (Directive 91/441/EEC and 91/542/EEC), or Euro 1/I, issued 8 years earlier.

China's national standards mandate the levels given in Table 2 for emissions from LDVs and HDVs, and in Table 3 for the presence of sulfur in fuels. Future national standards are specified, with target dates for full implementation. Some provinces have sought approval to proceed with earlier implementation of these standards. This permission is granted partly on assurances that fuel providers will be able to supply cleaner fuels that will not degrade the ECTs implied by the standards.

Finally, a variety of common and idiosyncratic local policies also aim to reduce emissions from road vehicles. These include, include other measures, prohibiting driving by some or all vehicles on certain days, accelerated retirement of older vehicles that have higher emissions or are non-compliant with recent ES, limiting the number of vehicles owned, and promoting the adoption of New Energy Vehicles (alternative fuel vehicles, such as battery-electrics).

Climate & energy policy. Climate and energy policies in the broader economy are another class of measures which can reduce emissions of the pollutants that contribute to poor air quality. Similar to transport-sector policy, these change the amount or type of energy used, or the amount of pollution emitted per unit energy. Section 2 describes in more detail how these changes contribute to reductions in total emissions.

1.2. Prior research

Integrated asssessment of policy co-benefits. The phenomenon of air pollution co-benefits of climate and energy policy has been long recognized and studied, including in Europe (van Harmelen et al., 2002; Nam et al., 2010; Rive, 2010) and more recently in China (Aunan et al., 2004; He et al., 2010; Nam et al., 2013). In particular, Nam et al. (2014) applied economy-wide, general-equilibrium models to compare the potential co-benefits in the U.S. and in China, in light of contrasting energy systems, and the stringency of existing control measures.

Air pollution from the transport sector. For assessment of pollution and health impacts within the transport sector, Yang and Ling-Yun (2016) modeled individual Chinese provinces as independent economies, using regression models and "pollution elasticities" to estimate health effects under future fuel price scenarios. Ling-Yun and Lu-Yi. (2016) took a similar approach for the country in aggregate, but studied instead the effect of mode shifts. Wu et al. (2016) used provincial-level modeling to assess emissions control policies 1998–2013, concluding that continued growth from heavy-duty (especially diesel) vehicles and enforcement of type approvals were areas of key concern. On the side of fuel quality, Yue et al. (2015) sampled fuels at about 60 sites in 2010–2011, discovering significant variation and exceedances, and suggested policy adjustments to promote compliance.

Guo et al. (2016) compared the projected effects of four transport-sector policies applied to the Beijing-Tianjin-Hebei (or Jing-Jin-Ji, JJJ) region, including accelerating the adoption of ES; and Zhang et al. (2016) similarly designed strategies for cities in the Yangtze River Delta. Lang et al. (2012) studied JJJ retroactively for the period 1999–2010, noting that increases in freight traffic were related to increases in transport NO_x and PM_{10} emissions even as other species decreased. Wang et al. (2010) developed 1995–2005 inventories of vehicle emissions for the large cities of Beijing, Shanghai and Guangzhou. Lu

Table 2
Recent Chinese tailpipe emissions standards, and selected European Union standards for comparison (ICCT, 2014 and linked documents). Note 1: two quantities are given, for gasoline and diesel passenger cars respectively. Note 2: two quantites are given, for the European Static Cycle + European Load Response test; and the European Transient Cycle respectively, versions of which are specified by the Chinese standards.

Species	CO		HC		HC + NO_x		NO_x		PM		NMHC	
Light-duty passenger vehicles (g/km)												
China 3	2.3	0.64[n.1]	0.20	–	–	0.56	0.15	0.50	–	0.05	–	
China 4	1.0	0.50	0.10	–	–	0.30	0.08	0.25	–	0.025	–	
China 5	1.0	0.50	0.10	–	–	0.23	0.06	0.18	0.0045		–	
Euro 5	0.50		–		0.23		0.18		0.0045		–	
Euro 6	0.50		–		0.17		0.08		0.0045		–	
Heavy duty vehicles (g/kW·h)												
China III	2.1	5.45[n.2]	0.66	–	–		5.0		0.10	0.16	–	0.78
China IV	1.5	4.0	0.46	–	–		3.5		0.02	0.03	–	0.55
China V	1.5	4.0	0.46	–	–		2.0		0.02	0.03	–	0.55
Euro V	1.5	–	0.46		–		2.0		0.02		–	0.55
Euro VI	1.5	4.0	0.13		–		0.4		0.01		–	0.16

Table 3
Sulfur content, in parts per million, in established and future China fuel quality standards (ICCT, 2014). Note 1: China I gasoline was required to be unleaded, but no maximum sulfur content was specified.

Level	I	II	III	IV	V
Gasoline	–[n.1]	500	150	50	10
Diesel	2000	500	350	50	10

et al. (2017) focused on school trips in particular, noting their contribution to congestion and pollution, and Deng (2006) measured the monetary costs of vehicle-related pollution in Beijing by two econometric methods.

Emissions from vehicles and uncertainty in inventories. Emissions from road vehicles have been studied by a variety of methods. Following in-use vehicles on actual roads with specialized instruments, Huang et al. (2016) measure the emissions from bi-fuel vehicles, while Zheng et al. (2015) present an instrument and drive-cycle methods focused on black carbon (BC).

For inventories of total, rather than specifically transport, emissions, studies such as Hu et al. (2015), use such direct measurements and bottom-up accounting to drive emissions inputs to atmospheric simulation models, aiming to reproduce changes in observed secondary air pollutant (*i.e.*, $PM_{2.5}$ and O_3) concentrations. Cheng et al. (2013) developed a hybrid approach incorporating ground monitoring data, focusing on Beijing only. Miyazaki and Eskes (2013) used satellite measurements and assimilation techniques to constrain the estimates of Kurokawa et al. (2013) (the REAS inventory used in the present study).

Wu et al. (2016) developed a bottom-up inventory for VOCs only at the province level, including the contribution of road vehicles. Hong et al. (2016) focus on the contribution of uncertainty in energy statistics to bottom-up methods, finding high ratios of maximum discrepancies to mean values-for instance, the total 2012 inventory of SO_2 emissions may vary up to 30%, and NO_x by 16.4%, due to energy use uncertainty alone.

Finally, Xia et al. (2016) combined satellite data with bottom-up estimates to assess the effects of industrial- and power-sector policies during the 11th (2006–2010) and 12th Five-Year Plan (2011–2015) periods, noting the growing contribution over this period of NO_x from transportation.

In terms of assessing past and future changes in transport activity – due to both growth, and policy – prior studies have focused on different geographies or aggregations, transport modes, policies, and modeling methods, yet have generally said little about the relationship with policies not focused on the transport sector. Conversely, examination of the co-benefits of climate policy has tended to focus on economy-wide impacts, or comparison with specific measures in the power- and industrial sectors, rather than the transport sector. The present work bridges this gap by providing methods for assessing both the climate policy co-benefit of air pollution reduction, and road transport emissions reductions due to ES, in a consistent, economy-wide framework.

In doing so, we note that researchers continue to work to resolve the uncertainties in the history and current state of vehicle tailpipe emissions. These are relevant to our method for deriving transport-subsector-specific emissions factors from a database (Kurokawa et al., 2013) that also covers the non-transport sectors where climate policy co-benefits also arise; a matter taken up further in Section 2.2.

1.3. Policy scenarios in this study

To investigate the contribution of road transport emissions standards to reduced pollutant emissions, their impacts can be compared with the size and distribution of impacts from current and more stringent climate policies, and also with the impacts of both types of policy implemented in concert. For this purpose, our analysis employs five model configurations, labeled A through E. Fig. 2 shows these scenarios arranged along dimensions of increasing ES stringency, and increasing stringency of CO_2 policy.

The policies implemented are as follows.

A. No Policy. Pollutant emissions from all sectors, including transportation, remain the same per unit of fossil energy consumed, as they were in 2007, the base year for the C-REM and this analysis. As energy demand grows in projections, associated pollutant emissions grow at the same rates. We also adopt the mild, autonomous reductions in energy-basis emissions factors in non-transport sectors developed by Li et al. (2014), representing the impact of learning-by-doing[1] and capital turnover.[2] The representation of these effects, which occur even in the absence of direct policy on emissions or policy co-benefits, is further described in Section 2.3.

B. Established Policies. All new road vehicles and fuels meet the China 3/III standards, so that the entire fleet converges towards this standard over time as older, dirtier vehicles are retired. In regions which have already committed to introducing vehicles cleaner than China 3/III in the near future, these lower emissions levels are used instead. In addition, a small, gradually-rising, economy-wide CO_2 price promotes energy intensity improvement and fuel switching to reduce CO_2 emissions. The price is RMB 30 (USD 3.1, in 2007 currency) in 2015, and rises 4% per year to RMB 45 (USD 5.7) in 2030. This instrument is used to model the combined effect of China's prior and established national and regional energy- and carbon-intensity targets and other direct policy measures affecting the broader economy. As a result of the energy system changes induced by the CO_2 price, there is a co-benefit of pollutant emissions reductions in these sectors, mainly through the displacement of coal. See Section 2.4 for a description of co-benefits.

C. Stringent ES. More stringent tailpipe emissions and fuel quality standards are introduced, reaching China 6/VI nationwide for all vehicles introduced from 2015. The CO_2 price is the same as in Scenario B.

D. Climate policy. A CO_2 price that is larger and rises more quickly, causing more rapid change in emissions across the entire economy. This price is consistent with the *Continued Effort* scenario of Zhang et al. (2016), in which economy-wide CO_2 emissions stabilize between 2035 and 2045, and reaches RMB 200 (USD 26) in 2030. Road transport ES are the same as in Scenario B.

E. ES and climate policy. The combination of the stringent ES from Scenario C, and the higher CO_2 price from Scenario D.

Comparison of Scenarios **A** and **B** illustrates how much established policies (in place prior to the introduction of the new nationwide China 4/IV standard) are expected to reduce pollutant emissions, compared to a future where transportation energy use has the same air pollutant emissions intensity as in 2007. Comparing Scenarios **B** and **C** illustrates the impact of accelerating road transport policies under the same CO_2 price. Comparison of Scenario **B** with Scenarios **C**, **D** and **E** illustrates the relative size and distribution of benefits from more stringent road transport policies compared to climate policy, and also the combined effect of the two.

2. Methods

2.1. China Regional Energy Model

We employ the C-REM, a multi-sector, multi-region, recursive-dynamic computable general equilibrium (CGE) model of the global economy, with provincial detail in China. The model has 30 regions within China and four international regions (see Table 4); the economy is represented in 14 sectors (see Table 5).

The C-REM projects output from each sector of each province, as well as trade and final demand (consumption), in value units, every five years to 2030. A complete description of the model is given by Zhang et al. (2013). To summarize, economic demand (*i.e.*, in units of real value, *e.g.* dollars) for energy goods is associated with physical quantities of energy consumed, and demand for transportation services is associated with passenger distance traveled or freight volume.

Kishimoto et al. (2015) discuss in detail the disaggregation of the transportation services sector, with the production structure shown in Fig. 3. Freight and passenger services are separated, each with road (denoted `FR` and `PR`, respectively) and non-road modes (`FO`, `PO`), as well as household vehicle transportation (`HVT`, representing privately-owned light-duty vehicles) following the method of Karplus et al. (2013). In total, then, there are five transport subsectors, of which three are road transport subsectors.

The base-year energy data in the C-REM supplemental accounts are derived from the China Energy Statistical Yearbook (National Bureau of Statistics of China, 2008); thus the use of refined oil, coal, natural gas and electricity by the commercial transport sector of each province reflects the flows into such sectors in the adjusted official statistics, and the household or

[1] Firms have a direct incentive to improve the efficiency of their production processes, thereby reducing costs. These improvements can have the side effect of improving energy efficiency or reducing pollution.

[2] Industrial equipment has a finite lifetime and must be periodically replaced. New, replacement equipment is often more efficient, requiring less energy or producing less emissions for the same value of production.

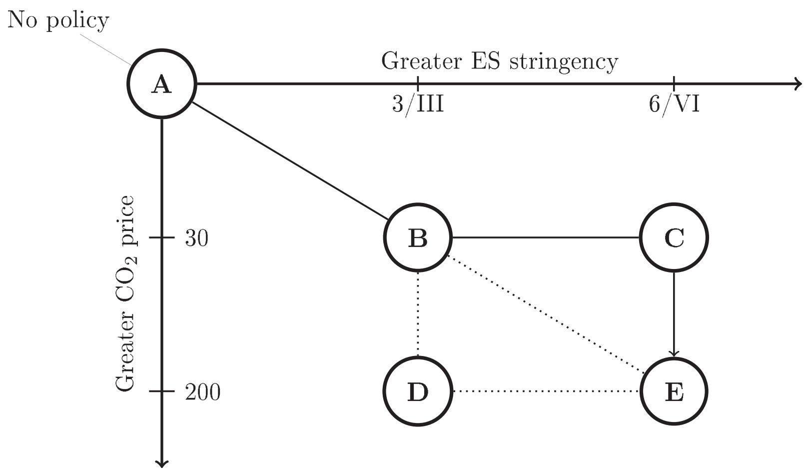
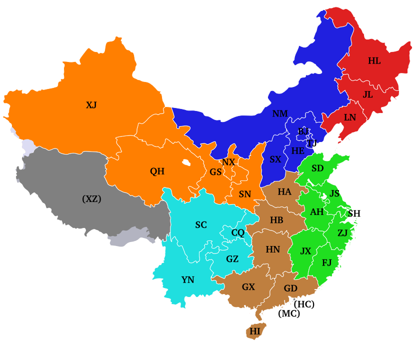


Fig. 2. Policy scenarios in this study, with the stringency of road transport emissions standards and the initial (2015) CO_2 price level in RMB per ton. Figs. 7 and 8 present comparisons along the solid lines.

Table 4
C-REM regions/Chinese provinces. Hong Kong (HK), Macau (MC) and Xizhang (Tibet, XZ) are not included in the C-REM; aggregate international regions not shown.



Code	Name	Code	Name
AH	安徽 Anhui	JS	江苏 Jiangsu
BJ	北京 Beijing	JX	江西 Jiangxi
CQ	重庆 Chongqing	LN	辽宁 Liaoning
FJ	福建 Fujian	NM	内蒙古 Inner Mongolia
GD	广东 Guangdong	NX	宁夏 Ningxia
GS	甘肃 Gansu	QH	青海 Qinghai
GX	广西 Guangxi	SC	四川 Sichuan
GZ	贵州 Guizhou	SD	山东 Shandong
HA	河南 Henan	SH	上海 Shanghai
HB	湖北 Hubei	SN	陕西 Shaanxi
HE	河北 Hebei	SX	山西 Shanxi
HI	海南 Hainan	TJ	天津 Tianjin
HL	黑龙江 Heilongjiang	XJ	新疆 Xinjiang
HN	湖南 Hunan	YN	云南 Yunnan
JL	吉林 Jilin	ZJ	浙江 Zhejiang

Table 5
List of C-REM sectors, omitting the transportation subsectors shown in Fig. 3.

Code	Sector	Code	Sector
AGR	Agriculture	MAN	Other manufacturing industries
COL	Coal mining & processing	OIL	Petroleum refining, coking and fuels
CON	Construction	OMN	Metal, minerals, other mining
CRU	Crude petroleum products	SER	Services
EIS	Energy-intensive industries	TRN	Transportation & post
ELE	Electricity & heat	WTR	Water
GAS	Natural gas products	c, g, i	Final demands

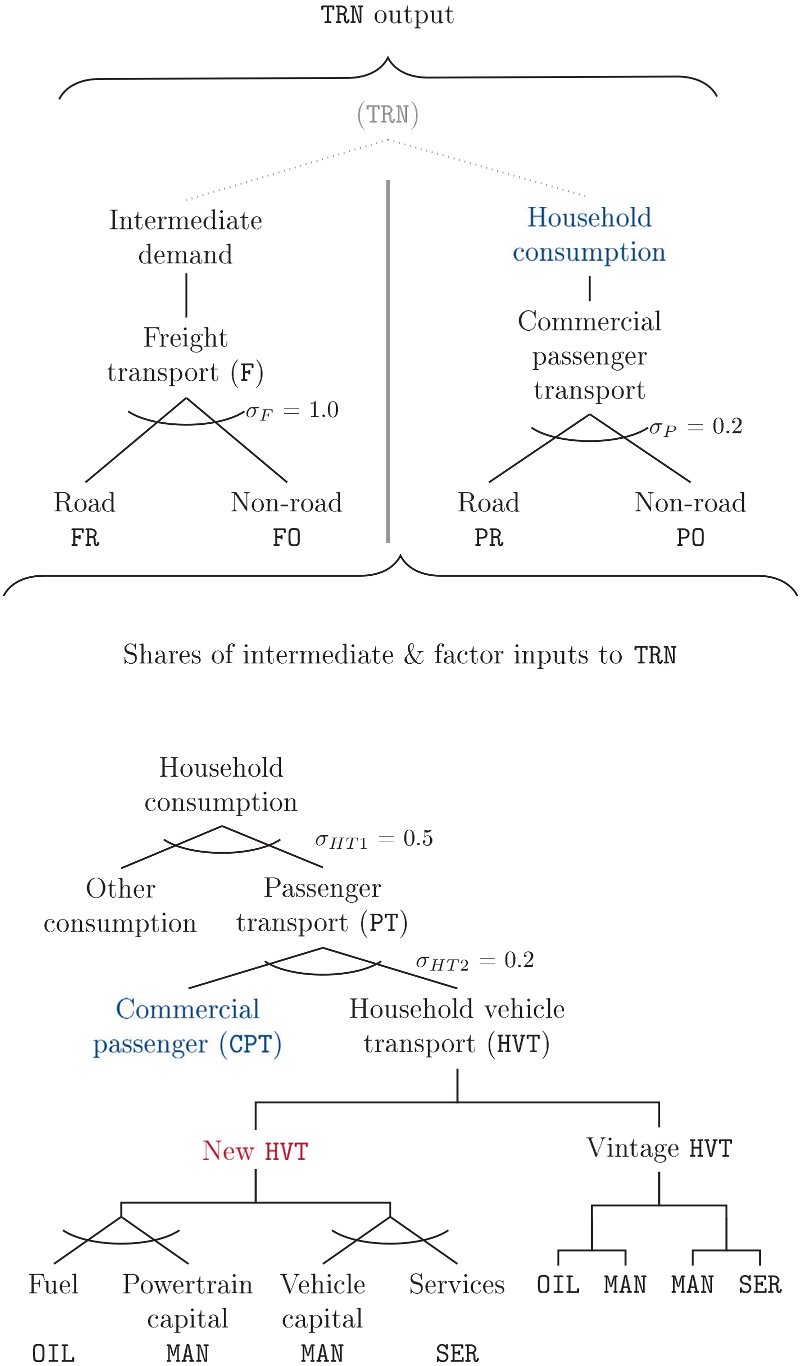


Fig. 3. Disaggregate transportation representation in the C-REM. Top: the transportation services sector is separated into two modes each of *freight* transport – supplying intermediate demand by other industrial sectors – and *passenger* transport – supplying households' consumption of commercial travel. Bottom: households' consumption contains passenger transport, which involves a bundle of commercial passenger transport and own-supplied, household private vehicle transport. The latter is produced by households themselves, using fuel and vehicles; vehicle purchases consist of inputs from the manufacturing and service sectors (Kishimoto et al., 2015).

final consumption of gasoline and diesel fuels is the energy supply for HVT. In the general equilibrium setting, households (firms) adjust the share of their total consumption (intermediate inputs) devoted to purchase of passenger (freight) transport services, given changing prices relative to other goods. Households additionally substitute PR for HVT at a calibrated price elasticity, and within the latter (Fig. 3, bottom) substitute more expensive and efficient powertrain capital for fuel consumption. This substitution represents the option to improve fuel economy. The C-REM currently represents only refined oil-fueled private vehicles, since the 2007 share of electricity in household vehicle fuel purchases was small.

To represent the effects of the emissions policies described in Section 1.1, we expand the physical accounts of the model to include primary pollutant species from the Regional Emissions in ASia (REAS) database, version 2.1 (Kurokawa et al., 2013): black carbon (BC), carbon monoxide (CO), nitrogen oxides (NO_x), organic carbon (OC), and sulfur dioxide (SO_2). Primary pollution is modeled as a byproduct of either combustion of fuels to produce energy, or of industrial or technical processes. We aggregate emissions from the REAS combustion and non-combustion sectors to C-REM sectors and REAS fuels to C-REM fuels, in order to associate emissions of each species with individual C-REM sectors, provinces, and energy sources. This connection is made by calibrating energy-basis emissions factors (EFs) using the base-year (2007) energy statistics contained in the supplemental accounts of the C-REM, as shown in (1).

$$\text{Emissions factor}_{p,f,i,r,t} = \frac{\text{Emissions of } p_{f,i,r,t}}{\text{Consumption of } f_{i,r,t}} \quad \text{for every} \begin{cases} \text{Pollutant} & p \\ \text{Fossil fuel} & f \\ \text{End-use sector} & i \\ \text{Province} & r \\ \text{Model period} & t \end{cases} \tag{1}$$

In the model base year ($t = 2007$), the EFs are a result of the calibration to REAS; in projected periods ($t = 2010 \ldots 2030$), the product of an emissions factor and the C-REM projected demand for energy gives the quantity of emissions for each p, f, i (including the transport subsectors) and r.

2.2. Emissions factors: transport subsectors

To determine future emissions from the C-REM road transport sectors (freight road or `FR`, passenger road or `PR`, and household vehicle transport or `HVT`), we determine energy-basis EFs, in mass of pollutant emitted per unit fuel energy consumed, and apply these to the C-REM projection of fuel energy consumption in these sectors. As a result of the calibration described above, the 2007 EFs exactly reproduce the 2007 REAS v2.1 emissions inventory.

In future periods, we make use of the detailed, bottom-up engineering model of Akerlind (2013). This model tracks the number of Chinese vehicles in detailed categories by year of manufacture, representing the scrappage (conversely: survival) rate of older vehicles, improving fuel economy, and annual driving distance differences between newer vehicles. The fleet model also accounts for fuel demand using these highly disaggregate categories; newer vehicles' driving activity is associated with a greater fuel economy.

We use the engineering model to determine the portion of fuel energy demand attributable to vehicles which are "new" since the prior C-REM period. For instance, in the C-REM forecast year 2010, this is the sum of fuel energy demand, in 2010, by vehicles from model year 2010, 2009, or 2008. The remainder of C-REM fuel energy demand in 2010 is associated with vehicles sold in 2007 or earlier.

Fuel demand from new vehicles is associated with the EFs required by the specific levels of Chinese ES in each scenario. The remaining fuel demand, associated with pre-existing vehicles, retains the EF of the previous period-in 2010, this is the 2007 REAS/C-REM calibrated EF, or in 2015 or later, the energy-weighted average across new and used vehicles in the previous period. Thus, policy which reduces emissions in new vehicles relative to Scenario A also reduces the emissions associated with the fuel demand of vintage (used) vehicles in subsequent C-REM periods.

China's emissions standards, like the Euro standards on which they are based (e.g., Directives 91/441/EEC and 91/542/EEC, for Euro I/1 respectively) specify distance-basis, rather than energy-basis, EFs for new vehicles. To determine energy-basis EFs, we use the on-road measurements of He et al. (2010). For future Chinese standards (China 5/V and 6/VI), we assume the on-road emissions levels will be in the same proportion to China 4/IV as the regulated levels are to the China 4/IV regulated levels. For `PR` and `HVT`, we use the figures for light-duty gasoline (passenger) vehicles, and to represent the average `FR` vehicle, we use the figures for medium-duty diesel trucks.

Fig. 4 shows these levels, and the resulting energy-weighted average EF across the combined fleet of new and old vehicles, for one example combination of fuel, primary pollutant species and transport subsector: NO_x from refined oil combustion in the `HVT` fleet. In the no-policy Scenario A, the REAS 2007 EF is used unchanged throughout the projection. Table 6 gives values across provinces for 2010, 2015 and 2030 and all species.

Because there is a base-year calibrated EF for each species and province, the relative improvement in EF due to the introduction of lower-emission vehicles differs province-to-province, and species-to-species. Absent any difference in standards or enforcement across provinces, EFs would eventually converge to the same level in all provinces, as today's heterogeneous provincial vehicle stocks are scrapped and replaced by vehicles with identical emissions characteristics. In our projection, this occurs near the very end of the C-REM forecast period – see Fig. 5.

2.3. Emissions factors: non-transport sectors

Energy-basis emissions factors for all sources are applied to the non-transportation sectors to obtain a complete picture of economy-wide emissions, as described in Li et al. (2014). Model base year (2007) EFs are calibrated, as in transportation, to

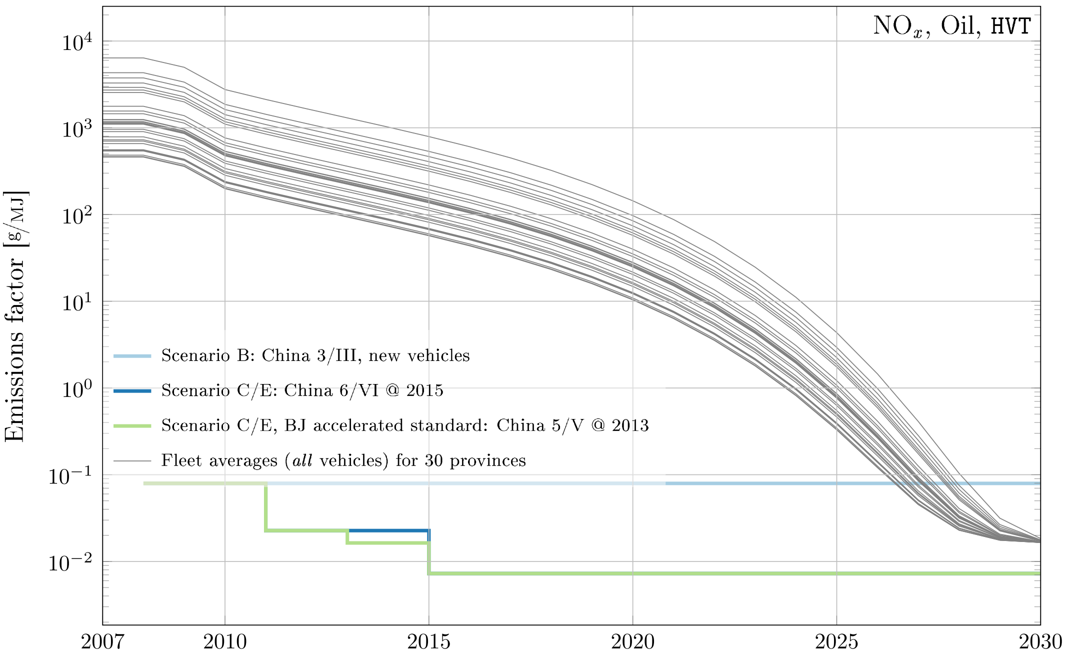


Fig. 4. Emissions factors for NO_x from refined oil (gasoline or diesel) combustion in road freight vehicles. Bold lines: mandated levels in policy scenarios. Thin lines: emissions factors for 30 provinces under Scenario C. In Scenario A, every province continues at its 2007 level. In Scenario B, new vehicles meet the China III (shown here) or 3 standard from 2008 onwards. In Scenarios C and E, new vehicles meet China 3/III from 2007–2010, China 4/IV from 2011–2015, and China 6/VI from 2016 onwards, excepting Beijing, which meets China 5/V from 2013 and China 6/VI from 2016.

Table 6
2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion ($\frac{g}{MJ}$), by road transport subsector, province and species.

Province	Road freight			Private passenger vehicles		
	2010	2015	2030	2010	2015	2030
			BC			
AH	11.50772	3.340007	0.01784278	25.85923	7.42597	0.00583426
BJ	3.20869	0.9592916	0.01783687	7.134878	2.053926	0.005802144
CQ	11.74151	3.407076	0.01784294	23.16656	6.653533	0.005832343
FJ	8.283364	2.415047	0.01784048	15.42472	4.432656	0.005826835
GD	6.752261	1.975825	0.01783939	14.67321	4.217073	0.0058263
GS	12.9262	3.746923	0.01784379	102.7668	29.4882	0.005888985
GX	6.697895	1.960229	0.01783935	17.48474	5.023608	0.0058283
GZ	12.62993	3.661933	0.01784357	27.33963	7.85065	0.005835313
HA	8.409849	2.451332	0.01784057	211.4454	60.66448	0.005966319
HB	7.555687	2.206301	0.01783996	82.15733	23.57603	0.00587432
HE	13.5947	3.938694	0.01784426	99.24966	28.47925	0.005886483
HI	11.51071	3.340865	0.01784278	13.29909	3.822884	0.005825322
HL	12.35615	3.583394	0.01784338	8.524258	2.453142	0.005821924
HN	7.936364	2.315504	0.01784023	28.34571	8.13926	0.005836029
JL	9.7755	2.843092	0.01784154	19.12953	5.495442	0.005829471
JS	10.84459	3.149779	0.0178423	45.83949	13.15764	0.005848477
JX	7.337348	2.143667	0.01783981	22.42283	6.440182	0.005831814
LN	9.37424	2.727983	0.01784126	24.14684	6.934744	0.005833041
NM	7.486177	2.186361	0.01783991	40.08636	11.50726	0.005844383
NX	8.061533	2.351411	0.01784032	91.67955	26.30764	0.005881096
QH	14.92848	4.32131	0.01784521	78.47135	22.51864	0.005871697
SC	8.406775	2.45045	0.01784057	48.74438	13.99096	0.005850544
SD	3.763694	1.118504	0.01783727	34.28837	9.844012	0.005840258
SH	14.69164	4.253368	0.01784504	5.695065	1.641541	0.005819911
SN	7.069741	2.066899	0.01783962	13.5962	3.908114	0.005825533
SX	13.56512	3.930207	0.01784424	92.82517	26.63628	0.005881911
TJ	7.686513	2.243831	0.01784006	15.16888	4.359263	0.005826652
XJ	6.504321	1.904699	0.01783922	26.27382	7.544904	0.005834555
YN	10.95964	3.182781	0.01784239	24.57936	7.05882	0.005833349
ZJ	8.904854	2.593332	0.01784092	13.75033	3.952329	0.005825643

Table 6 (*continued*)

Province	Road freight			Private passenger vehicles		
	2010	2015	2030	2010	2015	2030
			CO			
AH	1131.568	324.7649	0.2118226	7121.96	2043.204	0.1676885
BJ	522.2041	149.9586	0.211389	6548.931	1878.821	0.1672807
CQ	1188.26	341.0279	0.2118629	7938.902	2277.558	0.1682698
FJ	1000.978	287.3029	0.2117296	4345.805	1246.818	0.165713
GD	824.7293	236.743	0.2116042	3896.261	1117.858	0.1653931
GS	1264.998	363.0416	0.2119175	23564.53	6760.033	0.1793886
GX	671.3801	192.7522	0.2114951	3779.78	1084.444	0.1653102
GZ	1326.993	380.8259	0.2119616	9669.539	2774.02	0.1695013
HA	828.3376	237.7781	0.2116068	43415.31	12454.56	0.193514
HB	764.6412	219.5057	0.2115615	15403.61	4418.935	0.1735815
HE	1383.945	397.1634	0.2120021	28077.98	8054.791	0.1826003
HI	1222.033	350.7162	0.2118869	3215.819	922.6618	0.1649089
HL	1170.523	335.9397	0.2118503	2996.409	859.7206	0.1647528
HN	780.1915	223.9666	0.2115725	7592.626	2178.223	0.1680234
JL	974.3014	279.6503	0.2117107	5249.896	1506.171	0.1663563
JS	1054.397	302.6271	0.2117676	13121.54	3764.284	0.1719576
JX	709.8309	203.7825	0.2115225	5445.774	1562.362	0.1664957
LN	995.4975	285.7308	0.2117257	7754.072	2224.536	0.1681383
NM	670.7022	192.5578	0.2114946	12998.02	3728.852	0.1718697
NX	823.1702	236.2958	0.2116031	19530.24	5602.729	0.1765179
QH	1427.745	409.7282	0.2120333	17146.75	4918.983	0.1748219
SC	850.0086	243.9948	0.2116222	23111.12	6629.964	0.179066
SD	421.0803	120.9495	0.211317	10180.15	2920.496	0.1698646
SH	2332.681	669.3244	0.2126772	2887.251	828.4067	0.1646751
SN	676.0049	194.0789	0.2114984	3956.329	1135.09	0.1654359
SX	1324.652	380.1542	0.21196	36810.95	10559.99	0.1888145
TJ	908.7813	260.8548	0.211664	7202.707	2066.368	0.1677459
XJ	654.974	188.0459	0.2114834	7664.024	2198.704	0.1680742
YN	1141.027	327.4783	0.2118293	8407.991	2412.124	0.1686036
ZJ	1108.833	318.2429	0.2118064	6070.059	1741.449	0.16694
			NO_x			
AH	329.4949	94.92429	0.08210871	502.1078	144.0553	0.007770662
BJ	103.3923	29.9848	0.0796863	339.4146	97.38229	0.007604657
CQ	337.9948	97.36263	0.08211476	533.0447	152.9301	0.007792676
FJ	243.7779	70.33493	0.08204772	237.8253	68.24144	0.007582604
GD	203.8908	58.89263	0.08201933	240.9566	69.13971	0.007584832
GS	309.0178	89.0501	0.08209414	1622.673	465.5083	0.008568031
GX	205.0311	59.21977	0.08202014	200.636	57.57307	0.007556141
GZ	364.3998	104.9373	0.08213355	673.2566	193.1522	0.007892448
HA	234.4897	67.67046	0.08204111	2759.234	791.5501	0.009376784
HB	214.368	61.89822	0.08202679	313.965	90.08338	0.007636783
HE	345.6516	99.55912	0.08212021	1417.496	406.6498	0.008422032
HI	366.5796	105.5627	0.0821351	208.6495	59.87187	0.007561843
HL	280.4135	80.84446	0.08207378	198.3734	56.924	0.007554531
HN	230.5824	66.54958	0.08203833	481.8982	138.2579	0.007756281
JL	230.0566	66.39874	0.08203795	302.6407	86.83481	0.007628725
JS	308.7615	88.97658	0.08209396	626.675	179.7895	0.007859301
JX	216.4015	62.48155	0.08202824	283.6327	81.38206	0.0076152
LN	233.856	67.48868	0.08204066	537.3854	154.1753	0.007795765
NM	175.103	50.63437	0.08199885	763.9642	219.1732	0.007956993
NX	200.763	57.9954	0.08201711	1261.332	361.8515	0.008310909
QH	320.6038	92.37374	0.08210238	1101.302	315.9443	0.008197035
SC	212.3758	61.32672	0.08202537	1175.662	337.2756	0.008249948
SD	101.1797	29.42824	0.08194625	474.4597	136.124	0.007750988
SH	733.1359	210.7155	0.08239593	233.0096	66.85996	0.007579177
SN	186.2933	53.8445	0.08200681	237.4614	68.13704	0.007582345
SX	346.8681	99.9081	0.08212107	1861.856	534.1221	0.008738229
TJ	200.0259	57.78393	0.08201658	414.4664	118.9139	0.007708298
XJ	155.3776	44.97582	0.08198481	488.6262	140.1879	0.007761069
YN	321.1555	92.53201	0.08210278	507.8591	145.7052	0.007774754
ZJ	252.6122	72.8692	0.082054	389.7189	111.8147	0.007690688
			OC			
AH	4.401972	1.301605	0.01783772	18.02086	5.177401	0.005828682
BJ	1.309447	0.4144619	0.01783552	3.681778	1.063346	0.005799687
CQ	4.517558	1.334762	0.0178378	14.59869	4.195696	0.005826247

(*continued on next page*)

Table 6 (*continued*)

Province	Road freight			Private passenger vehicles		
	2010	2015	2030	2010	2015	2030
FJ	3.452279	1.029169	0.01783704	12.70255	3.651756	0.005824898
GD	2.864714	0.8606164	0.01783663	12.06638	3.46926	0.005824445
GS	4.80542	1.41734	0.01783801	43.22314	12.4071	0.005846615
GX	2.619625	0.7903084	0.01783645	12.67409	3.643592	0.005824877
GZ	4.91395	1.448474	0.01783808	14.69259	4.222631	0.005826314
HA	3.211388	0.9600655	0.01783687	102.9944	29.55348	0.005889147
HB	2.920968	0.8767538	0.01783667	83.36573	23.92268	0.00587518
HE	5.156374	1.518017	0.01783826	56.24579	16.14287	0.005855882
HI	4.660281	1.375705	0.0178379	9.657988	2.778372	0.005822731
HL	4.535099	1.339794	0.01783781	3.926093	1.134081	0.005818652
HN	3.050399	0.9138833	0.01783676	18.84	5.412388	0.005829265
JL	3.6575	1.08804	0.01783719	9.501366	2.733442	0.00582262
JS	4.12667	1.22263	0.01783752	37.62682	10.8017	0.005842633
JX	2.815987	0.8466382	0.01783659	18.25655	5.245013	0.00582885
LN	3.603658	1.072595	0.01783715	12.63828	3.633318	0.005824852
NM	2.733955	0.8231059	0.01783653	15.38447	4.42111	0.005826806
NX	3.06073	0.9168467	0.01783677	39.54005	11.35054	0.005843994
QH	5.451454	1.602666	0.01783847	30.70418	8.815826	0.005837707
SC	3.184431	0.9523325	0.01783685	32.5509	9.345589	0.005839021
SD	1.513963	0.4731305	0.01783566	27.67358	7.946447	0.005835551
SH	5.813656	1.70657	0.01783872	3.512485	1.015431	0.005818358
SN	2.655746	0.8006702	0.01783648	7.939637	2.285434	0.005821508
SX	5.078584	1.495702	0.0178382	57.8078	16.59096	0.005856993
TJ	3.095665	0.9268685	0.01783679	9.355496	2.691597	0.005822516
XJ	2.45575	0.7432981	0.01783633	12.76023	3.668301	0.005824939
YN	4.249182	1.257774	0.01783761	13.36572	3.841997	0.005825369
ZJ	3.727518	1.108126	0.01783724	7.911613	2.277395	0.005821488
			SO_2			
AH	18.47227	5.300417	0.0007822129	18.51308	5.328029	0.007426546
BJ	12.33097	3.538682	0.0007778429	12.37178	3.564558	0.007371941
CQ	33.54023	9.622915	0.000792935	33.58104	9.650528	0.007437268
FJ	20.07056	5.758914	0.0007833503	20.11137	5.786526	0.007427684
GD	16.81085	4.823811	0.0007810307	16.85166	4.851423	0.007425364
GS	26.38486	7.570278	0.0007878434	26.42567	7.59789	0.007432177
GX	25.79433	7.400873	0.0007874232	25.83514	7.428485	0.007431757
GZ	31.33104	8.989171	0.000791363	31.37185	9.016783	0.007435696
HA	19.79993	5.681279	0.0007831577	19.84074	5.708891	0.007427491
HB	26.31058	7.548968	0.0007877905	26.35139	7.576581	0.007432124
HE	10.20805	2.929687	0.0007763323	10.24886	2.957299	0.007420666
HI	14.2119	4.078257	0.0007791814	14.25271	4.105869	0.007423515
HL	10.85326	3.114777	0.0007767914	10.89407	3.142389	0.007421125
HN	26.98608	7.742747	0.0007882712	27.02689	7.770359	0.007432605
JL	10.67374	3.063278	0.0007766637	10.71455	3.09089	0.007420997
JS	20.22491	5.803191	0.0007834601	20.26572	5.830804	0.007427793
JX	23.02265	6.605771	0.0007854509	23.06346	6.633383	0.007429784
LN	16.39385	4.704187	0.000780734	16.43466	4.731799	0.007425067
NM	10.97365	3.149311	0.0007768771	11.01446	3.176923	0.00742121
NX	25.68913	7.370696	0.0007873483	25.72994	7.398308	0.007431682
QH	24.54442	7.042316	0.0007865338	24.58523	7.069928	0.007430867
SC	17.37201	4.98479	0.00078143	17.41282	5.012403	0.007425763
SD	12.43902	3.569676	0.0007779198	12.47982	3.597289	0.007422253
SH	18.5102	5.311298	0.0007822399	18.55101	5.33891	0.007426573
SN	21.80453	6.256331	0.0007845841	21.84534	6.283943	0.007428917
SX	11.54243	3.312475	0.0007772818	11.58324	3.340087	0.007421615
TJ	12.50433	3.588412	0.0007779663	12.54514	3.616025	0.0074223
XJ	21.64293	6.209974	0.0007844691	21.68374	6.237587	0.007428802
YN	27.8219	7.982517	0.0007888659	27.86271	8.010129	0.007433199
ZJ	17.51757	5.026546	0.0007815336	17.55838	5.054158	0.007425867

reflect the total energy demand in the C-REM supplemental accounts and quantities in the REAS database. EFs undergo an exogenous, exponential decline, calibrated to reflect observations in 2010 and 2013, using the method of Webster et al. (2008).

The exogenous decline in these EFs represents the continuing effect of non-market policies and actions by firms which-for instance-will retire older equipment and replace it with new equipment which produce less emissions in operation (capital turnover); or implement efficiency improvements in production processes that also reduce pollution intensity (learning-by-doing). These trends are assumed to be independent of any CO_2 price applied to fossil fuel use in these non-transport sectors;

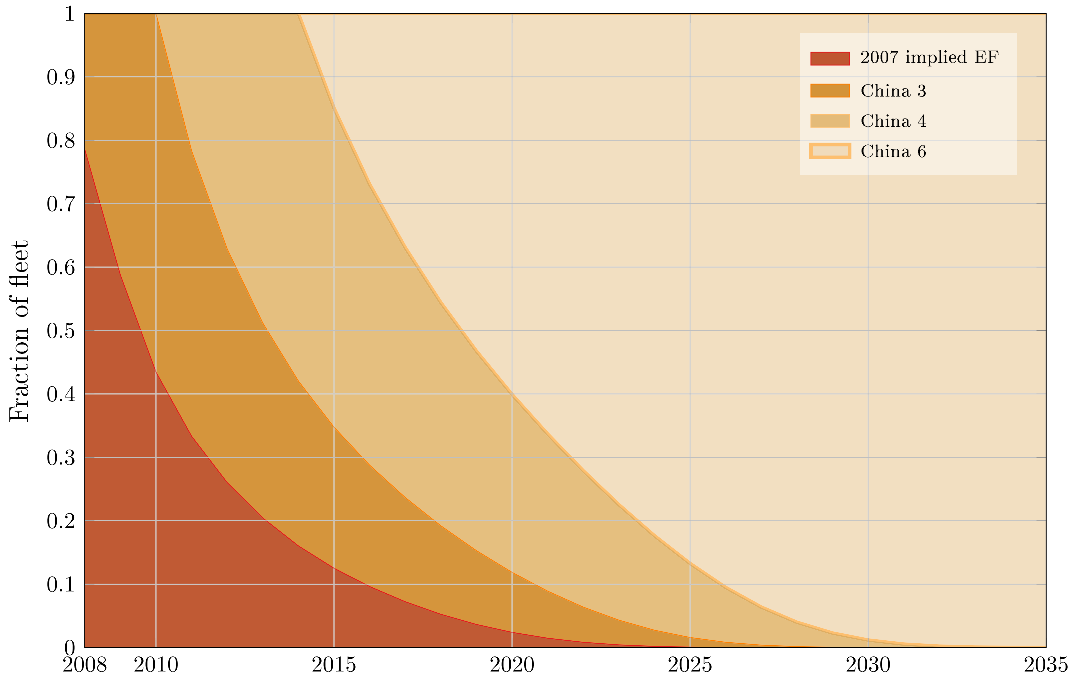


Fig. 5. Projected fraction of private vehicle stock complying with various ES, Scenario C/D.

the CO_2 price reduces emissions not by altering EFs, but by incentivizing low-carbon activities that also have lower emissions intensity, as described in the next section.

2.4. Policy mechanisms & effects

Economy-wide climate & energy policy. Climate policy, in a CGE model such as the C-REM, signals sectors and households via changes in energy prices in proportion to CO_2 content, prompting these actors to respond with energy intensity improvements and input substitution to reduce CO_2 emissions. This economic response can include reductions in energy demand and switching to low carbon fuels, which may also reduce air pollution in addition to CO_2, in proportion to differences in EFs between the energy goods that are substituted or reduced. Individual sectors in individual provinces – including electricity generation, industry, and the commercial transport subsectors – may perform these adjustments with more or less ease; as welfare is maximized subject to the CO_2 price, the sectors with lower-cost abatement opportunities will contribute larger reductions in CO_2.

Climate policy also has indirect impacts on the road transport subsectors, in two ways. First, freight transport demand arises because other sectors need to move their raw materials or finished goods to and from markets. Because a climate policy may cause each sector to increase or decrease production, freight transport demands will also change, affecting the overall level of freight transport activity, energy consumption, and pollution. Second, households use their income to purchase passenger transportation services, private vehicles, and fuel. Changes in household income mean more (due to economic growth), or less (due to stringent policy) income is available for these purchases. This, in turn, affects passenger transport demand, energy use, and pollution. Both of these indirect effects are captured by our model.

Road transport emissions standards. We model policy measures specifically aimed at reducing EFs more rapidly than they would decrease in the absence of regulation – in particular, road transport fuel quality standards and tailpipe emissions standards. The calibrated base-year EFs displayed in Table 6 vary widely – by an order of magnitude for BC, CO and NO_x from household vehicles – reflecting province-to-province variation in the emissions attributed to road vehicles (in the REAS v2.1 inventory), and the amount of energy used in household and commercial road transportation (as reflected in the official energy statistics underlying C-REM). As a result, the relative improvement in EF and in total emissions of each species due to the introduction of lower-emission vehicles differs province-to-province, and species-to-species.

3. Results

In Scenarios B and C, the small emissions price causes total economy-wide CO_2 emissions to decrease by about 5%, from 14.7 Gt to 14.2 Gt in 2030; in Scenarios D and E, emissions are 11.9 Gt CO_2 in 2030.

3.1. Road transport emissions standards

Fig. 6 gives an overview of the effects of established emission standards in the model projection, comparing Scenarios A and B in national totals and four selected provinces with distinct mixes of household vehicle, commercial passenger and freight road transport. Despite compound annual growth of 7.5% in transportation energy demand (rates vary between 4.5–9.9% across provinces) between 2010 and 2030, established polices reduce total national road transport pollution emissions to between 2% (in the case of OC) and 0.04% (CO) of their 2007 levels (Scenario B vs. Scenario A); a decrease of 2–4 orders of magnitude. In Scenario B, this decrease occurs by 2025 when (cf. Fig. 5) high emitting vehicles of 2007 and earlier model years have been entirely retired from the fleet. After this point, emissions in some regions and species exhibit a slow growth, in line with increases in transport demand and energy use.

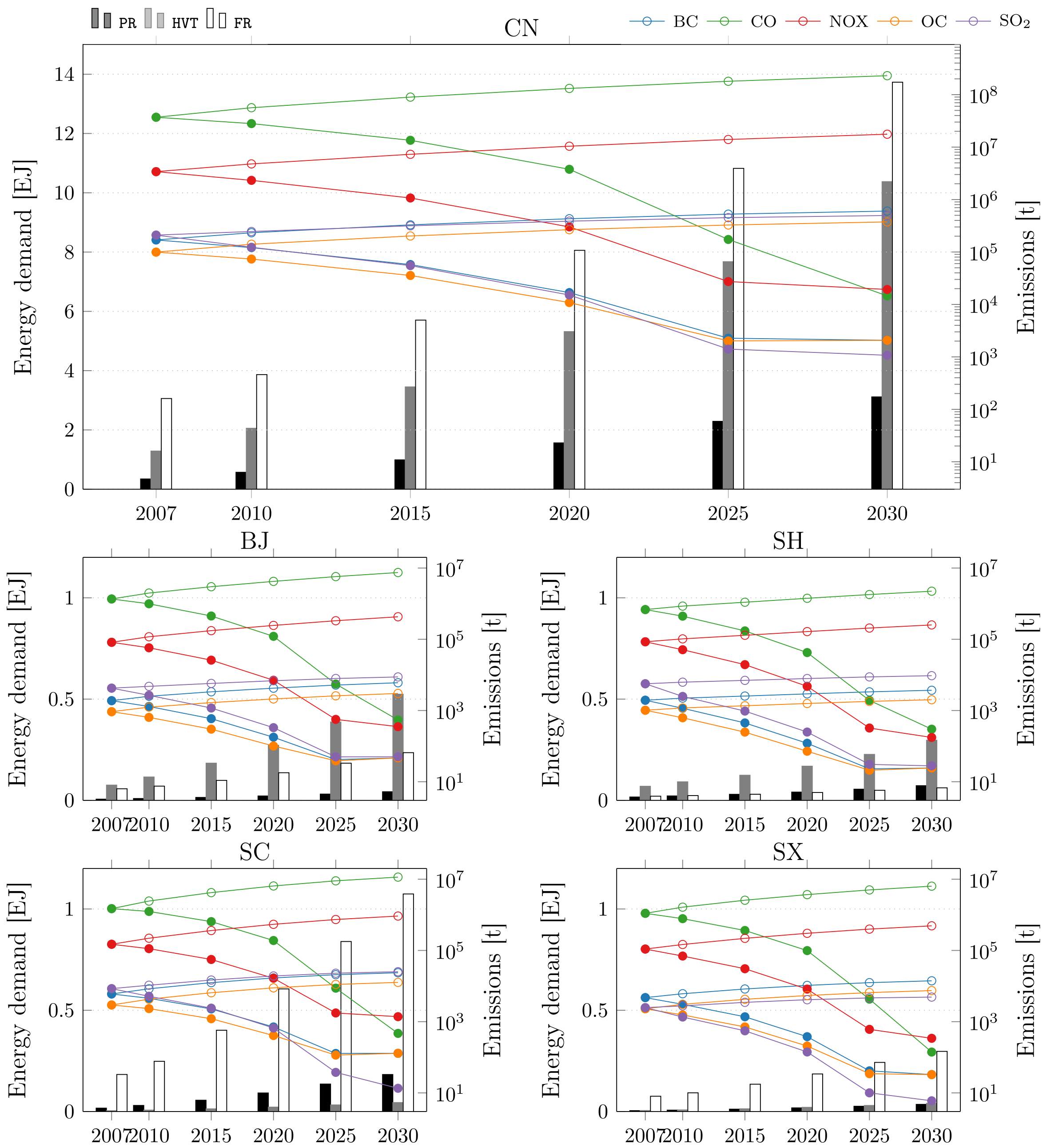


Fig. 6. Bars and left ordinate: energy demand for three road transport sectors: commercial passenger (PR), household vehicles (HVT) and freight (FR). Lines and right ordinate: road transport emissions for five species in Scenario A (open marks) and Scenario B (filled marks). Top: China (CN) total. Bottom: same quantities for Beijing (BJ), Shanghai (SH), Sichuan (SC) and Shanxi (SX).

Fig. 7 illustrates the same baseline growth (bars labeled "2007→2030"), and reductions (labeled "2030 A→B") in a linear, rather than logarithmic scale. Further reductions occur in Scenarios C and E. A contrast is visible between *percentage* changes (due to growth, or policy) and the *absolute* change in total road transport emissions measured in annual tonnes.

If fully implemented, established policies will do most of the work; the continued sale of China 3/III vehicles alone will significantly reduce emissions in 2030, compared to a mix of vehicles like those in use in the model base year. For instance, despite a 277% growth from 2007–2030 in OC emissions from road transport, the full deployment of China 3/III vehicles results in a 99.45% decrease from this no-policy counterfactual to about 0.4 kt in 2030, much lower than the REAS 2007 reference value of 99.4 kt. Future standards (China 6/VI) further reduce EFs by roughly an order of magnitude in 2030 (labeled "B→C" in Fig. 7; cf. Fig. 4). However, these translate to a smaller reduction in *absolute* road transport emissions, because they act on a base already rendered small by the adoption of China 3/III. The percentage reductions due to moving from China 3/III to China 6/VI (Scenario B→C) are between 58% (CO) and 91% (OC).

Finally, Fig. 8 illustrates that in 2020 in the significant impact of China 3/III is already visible; however, because of the remaining share of non-China 6/VI vehicles present at 2020 in Scenario C (compare Fig. 5), the reductions compared to Scenario B are smaller.

3.2. Provincial variation in reduction sources

ES impacts differ across provinces due to underlying differences in their transport systems. Across China, road freight transportation is a larger consumer of energy (3.06 EJ in 2007) compared to the combination of private and commercial passenger transport (1.63 EJ together in 2007) – yet, depending on the species, its contribution to total pollution in the REAS inventory is larger (0.6% compared to 0.03% of SO_2) or smaller (3.3% compared to 15.6% of CO) than those of other modes. Fig. 9 shows that additional 2030 emissions reductions due to increasing stringency of ES from China 3/III to China 6/VI (Scenario B→C) are contributed by different transport subsectors in different provinces. For instance, the additional reductions in OC emissions due to the China 6/VI standards come mostly from passenger road transport in Hainan (HI); from private LDVs in Shanghai (SH); and from road freight vehicles in other provinces.

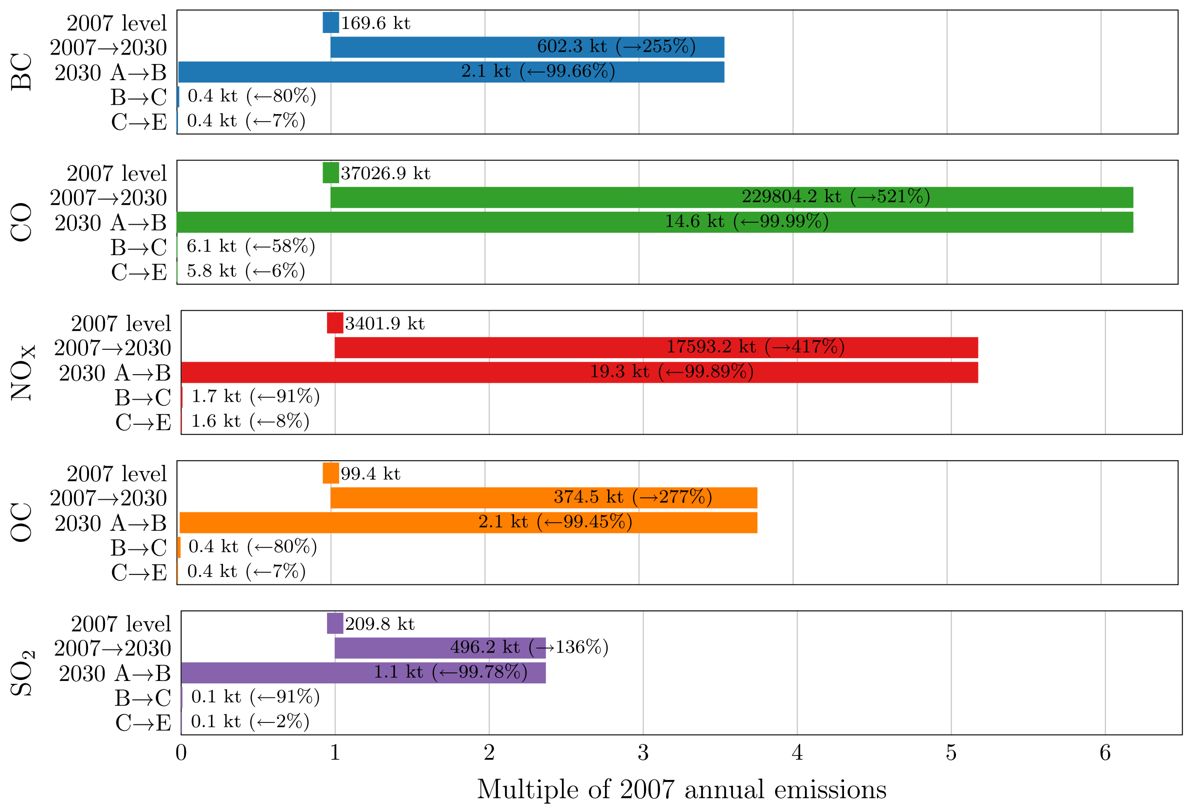


Fig. 7. Top line for each of five pollutant species: the 2007 *level* of emissions. Subsequent lines: *changes* in road transport emissions-increase from 2007→2030 in Scenario A (no policy); and decreases in 2030 from Scenario A→B (introducing China 3/III and a mild CO_2 price), Scenario B→C (increasing ES stringency to China 6/VI), and Scenario C→E (increasing CO_2 price). Annotations give nationwide road transport emissions in each year/scenario and percent change compared to the bar above.

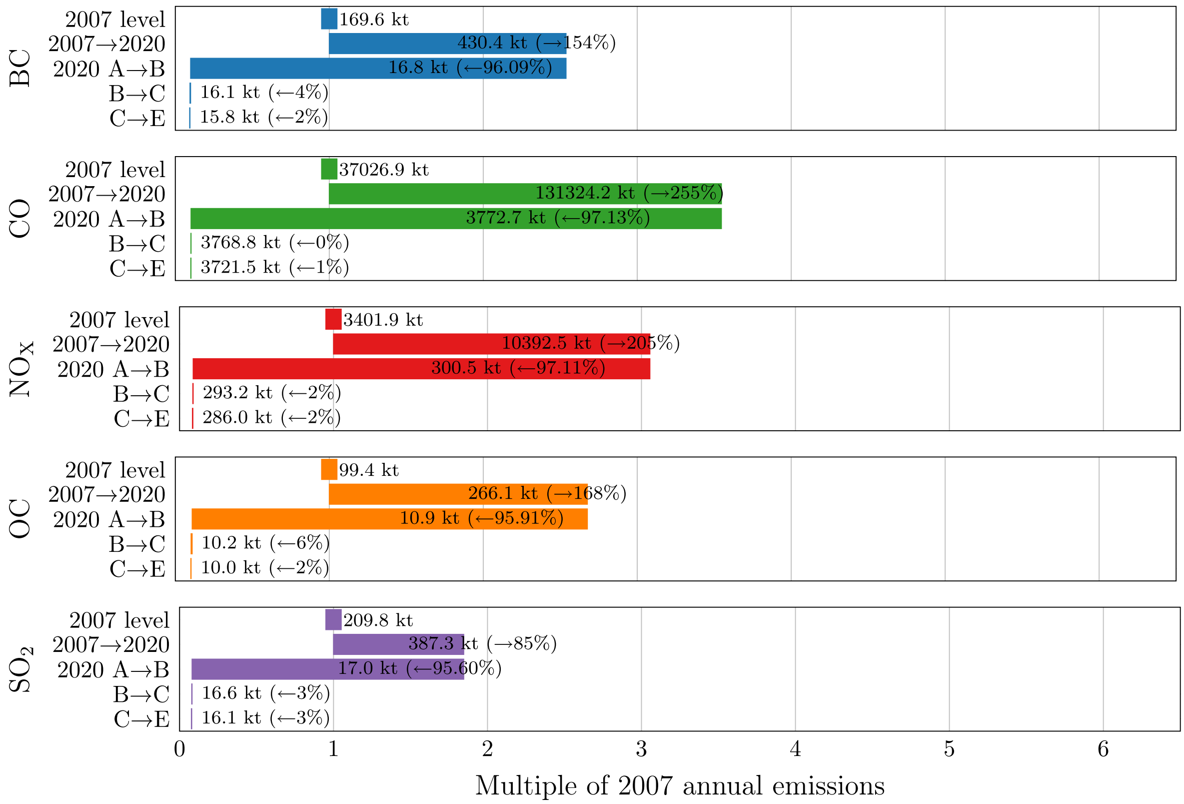


Fig. 8. Top line for each of five pollutant species: the 2007 *level* of emissions. Subsequent lines: *changes* in road transport emissions of five species: increase from 2007→2020 in Scenario A; and decreases in 2020 from Scenario A→B, B→C, and C→E. Annotations give the nationwide road transport emissions in each year/scenario and percent change compared to the bar above.

The panels for Shanghai (SH) and Shanxi (SX) in Fig. 6 also illustrate that the trajectory of reductions in pollutant species differ by province. In Shanghai, 2030 emissions of SO_2 from road vehicles roughly equal emissions of OC, while in 2007 SO_2 emissions were higher. Conversely, in Shanxi absolute emissions of these two species were roughly equal in 2007, but OC emissions are reduced less than SO_2 in 2030 by China 3/III emissions standards. Our provincially-resolved framework preserves these differences, which can affect the atmospheric chemical processes that form secondary pollutants.

3.3. Emissions standards & climate policy

Fig. 10 illustrates that stringent road transport ES cause very modest additional reductions in total emissions of pollutants, even though they are very effective in reducing emissions *within* road transport sectors. China 6/VI emissions standards reduce road transport CO emissions by 30–70% compared to China 3/III (Fig. 10, right). However, the reduction in total emissions of this species is generally smaller than 0.15% (Fig. 10, left, vertical ordinate), in part because less than 24% of 2007 CO emissions are attributed to transport (Kurokawa et al., 2013); and less to road transport.

The mild CO_2 price of Scenario B causes 9.6–48% (across species) reductions in economy-wide emissions of pollution, versus no policy (Scenario A). Tightening the CO_2 price only (Scenario D) results in 8.9–27% reduction versus established policy (Scenario B). The co-benefits of climate policy for air pollution reduction are substantial, even for the modest CO_2 price. In contrast (Fig. 10, right), increasing the CO_2 price causes a reduction in road transport emissions of less than 7% in all but three provinces – that is, smaller than the co-benefit of climate policy across the entire economy.

4. Discussion

Previous research emphasizes that the marginal cost of CO_2 emissions abatement in transportation tends to be higher than in other sectors, such as electricity and industry (Kishimoto et al., 2015). This means that responses to a given CO_2 price – efficiency improvements and fuel-switching – are smaller in transport, and the sectoral pollution co-benefit of CO_2 policy is also small. Indeed, reductions in air pollutant emissions due to CO_2 pricing in our scenarios mostly occur outside the transport sector: although increasing the CO_2 price (Scenario B→D) results in 8.9–27% additional reductions in national total emissions, road transport emissions decrease by only 2.0–7.1% across species.

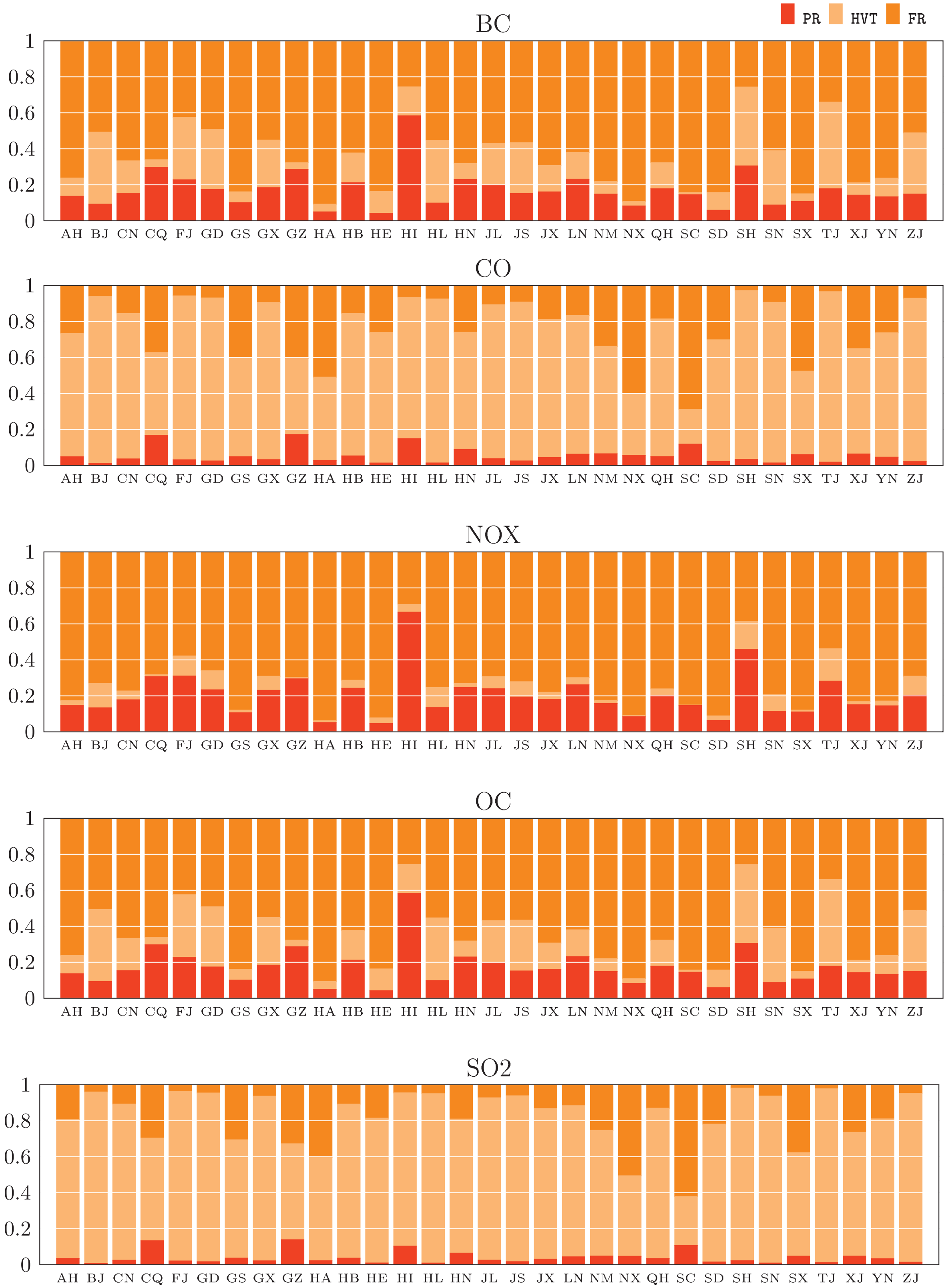


Fig. 9. Contribution of each road transport mode to the total reduction of road transport emissions in 2030 due to stringent ES (Scenario B→C), by province.

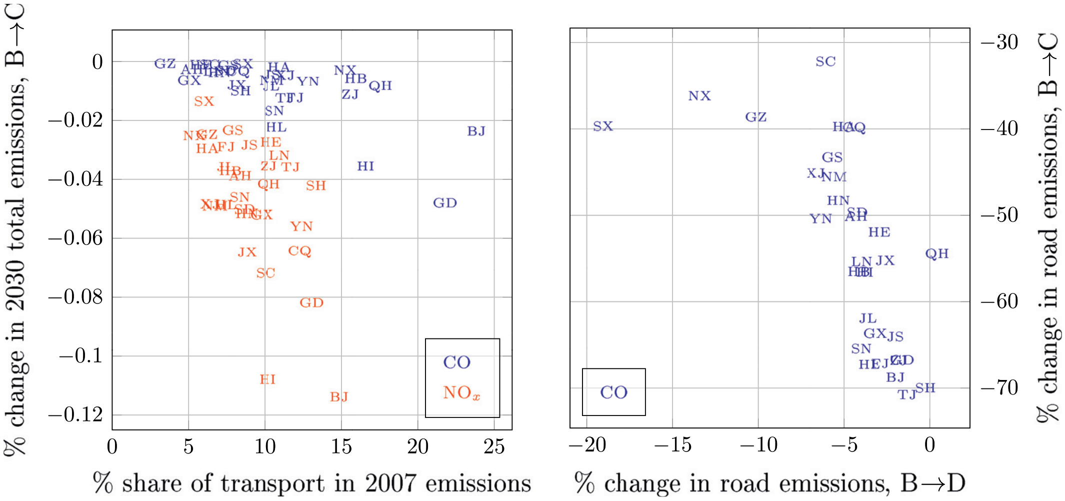


Fig. 10. Left: change in total CO and NO_x emissions due to moving from established to stringent ES, versus the share of transport in total emissions in each province. Right: change in road transport CO emissions compared to current policies (Scenario B) due to increasing stringency of climate policy (Scenario D, horizontal axis), versus change due to increasing stringency of ES (Scenario C, vertical axis).

In contrast, the within-sector reductions due ES are measured in orders of magnitude, when comparing established policies (China 3/III) to a counterfactual future where the road vehicle fleet retained its 2007 emissions characteristics. Additional reductions due to tightening ES are similarly large as a percentage of remaining road transport emissions, yet small in absolute terms when compared to the co-benefit of economy-wide CO_2 pricing.

We conclude that transport-sector ES are an important complement to economy-wide climate policy, since they can achieve deep reductions via technology and cleaner fuel, which together greatly reduce EFs. To achieve the same transport-sector reductions purely through co-benefits of climate policy would require CO_2 prices much higher than the those modeled.

4.1. Policy implications

Taken together, this work clearly illustrates the emissions reduction benefits of completely implementing emissions standards at the China 3/III level or higher. To quantify the consequences for secondary pollutant concentrations and the burden of human health impacts on the economy requires further, ongoing, research; but our results indicate the scale of the benefit available due to the two policy levers considered.

While moving to China 6/VI standards is clearly desirable, if tightening standards on the books comes at the expense of a sustained implementation effort that brings the reality on the road in line with policy aspirations, a focus on full implementation of the existing standards is advised. Our results show that the marginal benefits of accelerating the policy timeline are modest. On the other hand, a small number of non-compliant vehicles, running on non-compliant fuels, could more than offset the modest benefits of moving a large number of sales to a more stringent standard; and indeed the work of Yue et al. (2015); Wu et al., 2016 and others indicates ongoing issues with on-the-ground enforcement of existing standards.

Lessons from Europe and elsewhere that suggest significant benefits from accelerated standard implementation do not yet apply in China. Changes that result from incremental standard tightening are large relative to total emissions in the European context, but small relative to total emissions in the Chinese context. As noted in Section 3.1 and Fig. 6, we project that energy demand for road transport will continue to grow through 2030, both in absolute terms and as a share of total energy demand, as increases in demand for transport more than offset improvements in energy efficiency (*i.e.*, fuel economy). Consequently, road transport's share of total CO_2 emissions will also grow. In contrast, the large reduction in air pollutant emissions factors from implementing established (China 3/III) emissions standards means that the share of road transport in total air pollutant emissions will decrease markedly; and reductions from tightening ES can only further narrow this already-small share.

4.1.1. Policy recommendations

Therefore, we underscore that climate policies now being discussed and piloted, specifically a price on carbon such as the one we model, can serve as an important and effective complement to the full implementation of emissions standards in the road transportation sector. The work suggests two main policy recommendations.

First, strengthen mechanisms for enforcing the newly-enacted China 4/IV emissions standards. Authorities at the highest levels should clearly direct the Standardization Administration of China and Ministry of Environmental Protection to strengthen and standardize the monitoring and enforcement system for fuel quality and vehicle tailpipe emissions. Define well in advance the timing of increasing the stringency of standards to China 6/VI levels, so that the system can adjust and prepare.

Second, continue to diligently work toward establishing a national CO_2 price with broad sectoral coverage. Although it seems likely that transportation will not be included in a national CO_2 emissions trading system, reductions in fossil energy use in other parts of the economy will deliver significant and meaningful co-benefits that will contribute to improved human health in the near to medium term.

4.2. Limitations

A key difference between the carbon and ES policies implemented in our scenarios is that the price impacts of CO_2 emissions abatement is captured in the general equilibrium framework of C-REM. In contrast, the ES policies are implemented through exogenous calculation of future road transportation EFs. In practice, implementing tighter ES will require manufacturers to install more advanced ECT on passenger vehicles and road freight vehicles sold in China. Fuel economy improvements can also reduce the amount of energy used per kilometer, and thus contribute to reducing the amount of emissions per unit of fuel consumed. Both of these compliance options impose costs on manufacturers and consumers if the resulting vehicles are more expensive than those that would be sold without the standard. Zhenying and David (2015) estimate the costs of ECTs for different levels of Euro standards, with some adjustment for the Chinese fleet, and note that the absolute costs for private, light-duty vehicles are nominal. Herein we assumed that these increases in purchase price due to more advanced ECT, when considered as an increment on the cost of transport per passenger-kilometer or ton-kilometer, are not large enough to affect vehicle purchases or vehicle use intensity.

Our scenarios have assumed that ES are fully implemented – that is, 100% of new vehicles comply with the active standard as of the sale date. If a fraction of new road vehicles (either passenger or freight) are non-compliant, their much higher EFs will increase the fleet average so long as they remain in use. Similarly, timely retirement of older vehicles is important to our results. Fig. 5 shows that ~40% of vehicles in 2020 have China 4/IV, 3/III, or pre-2007 EFs; as a consequence fleet-wide emissions that year under Scenarios B, C, or E (Fig. 8) are much higher than in 2030 (Fig. 7), even though the 2030 fleets and total transport energy use are larger. If recent vehicles that do not meet future ES are preserved in the fleet longer than modeled by Akerlind (2013), then their contribution to emissions will continue to be felt until they are scrapped.

We emphasize the importance of the need for more recent emissions inventories which capture provincial and sectoral variation in the Chinese economy. Monitoring of concentrations of secondary pollutants has increased with the level of research and policy attention on China's air quality, and the literature contains much recent information on the characteristics of transport emissions. Yet comprehensive and consistent inventories of the sources of primary pollutants have not fully incorporated these advances; and such economy-wide inventories are important to studies, like the present one, that contrast policies affecting different sectoral sources in different ways.

Finally, we do not explicitly model policies aimed at reducing CO_2 emissions or fuel consumption for light-duty vehicles (`HVT` in C-REM). As noted by Paltsev et al. (2016), such policies require energy demand reduction and/or CO_2 emissions abatement efforts within the transport sector; meeting the targets affects the price of transport services and thus affects demand. In the present context, changing efficiency of road vehicles would affect the energy basis to which EFs were applied, and thus the total size of the road transport sector emissions growth or reduction. This would affect the relative scale of transport versus non-transport emissions reductions (Section 3.3), but have a smaller influence on the relative benefit of increasing stringency of ES.

4.3. Conclusions

Initial Chinese ES were based on previously established European Union regulations. Ever since, China's ES have been converging towards parity with the world's most stringent (ICCT, 2014). Because transportation is one of several major polluting sectors, and because of the varied geographic distribution of transportation activity – and of air pollution and its health impacts – it is important to understand the contribution of transport-sector policies to improving air quality in China on a detailed, regional basis. It is also important to understand how coordination of multiple policies can lead to improved air quality in China, and whether this coordination is different from that required in other contexts.

To this end, we herein developed key components of an integrated assessment framework that projects the economic activities – including energy use in non-transport and transport sectors – giving rise to air pollution. Within this framework, we implemented two types of policies affecting emissions: road transport emissions standards, and economy-wide CO_2 pricing that gives a pollution co-benefit. Examining scenarios of no policy, established policies, and more stringent policies, we characterized the relative scale of their impacts on pollutant emissions within road transportation, and across the economy. Our results indicate that increased CO_2 pricing, and full enforcement of more stringent road transport emissions standards, play complementary roles in reducing total emissions.

Acknowledgements

We thank the consortium of founding sponsors of the MIT-Tsinghua China Energy and Climate Project: Eni S.p.A., the French Development Agency (AFD), ICF International, and Shell International Limited for funding this work at MIT. We are grateful to the National Science Foundation of China (Grant No. 71573152), the Ministry of Science and Technology of China, the National Development and Reform Commission of China, the National Energy Administration of China, Rio Tinto, and Total for supporting this research at Tsinghua University. We further thank the Energy Information Administration of the US Department of Energy for support to MIT through a cooperative agreement. At MIT the China Energy and Climate Project is part of the MIT Joint Program on the Science and Policy of Global Change, which is funded through a consortium of industrial, foundation, and government sponsors. For a complete list of sponsors and U.S. government funding sources, see http://globalchange.mit.edu/sponsors/.

We also thank Prof. Zhang Qiang at Tsinghua University, and Chiao-Ting Li, Audrey Resutek and Andrew J. Cockerill at MIT for their inputs at various stages in the work.

References

Akerlind, Ingrid, 2013. Driving change: evaluating strategies to control automotive energy demand growth in China. S.M. thesis, Massachusetts Institute of Technology. HDL: 1721.1/81114.

Aunan, Kristin, Jinghua, Fang, Haakon, Vennemo, Kenneth, Oye, Seip, Hans M., 2004. Co-benefits of climate policy-lessons learned from a study in Shanxi, China. Energy Policy, 0301-4215 32 (4), 567–581. An Economic Analysis of Climate Policy: Essays in Honour of Andries Nentjes. doi: http://dx.doi.org/10.1016/S0301-4215(03)00156-3..

Cheng, Shuiyuan, Lang, Jianlei, Zhou, Ying, Han, Lihui, Wang, Gang, Chen, Dongsheng, 2013. A new monitoring-simulation-source apportionment approach for investigating the vehicular emission contribution to the PM2.5 pollution in Beijing, China. Atmos. Environ., 79, pp. 308–316. ISSN: 1352-2310. doi: http://dx.doi.org/10.1016/j.atmosenv.2013.06.043.

Deng, Xin, 2006. Economic costs of motor vehicle emissions in China: a case study. Transport. Res. Part D: Transport Environ., 1361-9209 11 (3), 216–226. http://dx.doi.org/10.1016/j.trd.2006.02.004.

Guo, Xiurui, Fu, Liwei, Ji, Muse, Lang, Jianlei, Chen, Dongsheng, Cheng, Shuiyuan, 2016. Scenario analysis to vehicular emission reduction in Beijing-Tianjin-Hebei (BTH) region, China. Environ. Pollut., 216, pp. 470–479. ISSN: 0269-7491. doi:http://dx.doi.org/10.1016/j.envpol.2016.05.082.

van Harmelen, Toon, Joost, Bakker, de Vries, Bert, van Vuuren, Detlef, den Elzen, Michel, Petra, Mayerhofer, 2002. Long-term reductions in costs of controlling regional air pollution in Europe due to climate policy. Environ. Sci. Policy, 1462-9011 5 (4), 349–365. http://dx.doi.org/10.1016/S1462-9011(02)00043. Linkages Between Regional Air Pollution and Climate Change in Europe.

He, Kebin, Lei, Y., Pan, X., Zhang, Y., Zhang, Q., Chen, D., 2010. Co-benefits from energy policies in China. Energy, 0360-5442 35 (11), 4265–4272. http://dx.doi.org/10.1016/j.energy.2008.07.02. Energy and Its Sustainable Development for China.

He, Kebin, Yao, Zhiliang, Zhang, Yingzhi, 2010. Characteristics of vehicle emissions in China based on portable emission measurement system. In: 19th Annual International Emission Inventory Conference. U.S. Environmental Protection Agency, San Antonio, TX, US http://www.epa.gov/ttnchiel/conference/eil9/session6/he.pdf.

Ling-Yun, He, Lu-Yi, Qiu, 2016. Transport demand, harmful emissions, environment and health co-benefits in China. Energy Policy, 97, pp. 267–275. ISSN: 0301-4215. doi:http://dx.doi.org/10.1016/j.enpol.2016.07.037.

Hong, Chaopeng, Zhang, Q., He, K., Guan, D., Li, M., Liu, F., Zheng, B., 2016. Variations of China's emission estimates response to uncertainties in energy statistics. In: Atmospheric Chemistry and Physics Discussions 2016, pp. 1–23. doi:http://dx.doi.org/10.5194/acp-2016-45.

Hu, Jianlin, Wu, Li, Zheng, Bo, Zhang, Qiang, He, Kebin, Chang, Qing, Li, Xinghua, Yang, Fumo, Ying, Qi, Zhang, Hongliang, 2015. Source contributions and regional transport of primary particulate matter in China. Environ. Pollut. 207, 31–42. ISSN: 0269-7491. doi:http://dx.doi.org/10.1016/j.envpol.2015.08.037.

Huang, Xiaoyan, Wang, Yang, Xing, Zhenyu, Du, Ke, 2016. Emission factors of air pollutants from CNG-gasoline bi-fuel vehicles: Part II. CO, HC and NOx. Sci. Total Environ. 565, 698–705. ISSN: 0048-9697. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.05.069.

ICCT, 2014. China: Light-duty: Emissions. <http://transportpolicy.net/index.php?title=China:_Light-duty:.Emissions>.

Karplus, Valerie J., Paltsev, Sergey, Babiker, Mustafa H., Reilly, John M., 2013. Applying engineering and fleet detail to represent passenger vehicle transport in a computable general equilibrium model. Econ. Modell. 30, 295–305. doi:http://dx.doi.org/10.1016/j.trd.2017.02.012 & http://dx.doi.org/10.1016/j.econmod.2012.08.019.

Kishimoto, Paul Natsuo, Zhang, Da, Zhang, Xiliang, Karplus, Valerie J., 2015. Modeling regional transportation demand in China and impacts of a national carbon policy. Transport. Res. Rec.: J. Transport. Res. Board 2454, 1–11. ISSN: 0361-1981. doi:http://dx.doi.org/10.3141/2454-01.

Kurokawa, J., Ohara, T., Morikawa, T., Hanayama, S., Janssens-Maenhout, G., Fukui, T., Kawashima, K., Akimoto, H., 2013. Emissions of air pollutants and greenhouse gases over Asian regions during 2000–2008: Regional Emission inventory in ASia (REAS) version 2. Atmos. Chem. Phys. 13, 11019–11058. doi:http://dx.doi.org/10.5194/acp-13-11019-2013.

Lang, Jianlei, Cheng, Shuiyuan, Wei, Wei, Zhou, Ying, Wei, Xiao, Chen, Dongsheng, 2012. A study on the trends of vehicular emissions in the Beijing-Tianjin-Hebei (BTH) region, China. Atmos. Environ. 62, 605–614. ISSN: 1352-2310. doi:http://dx.doi.org/10.1016/j.atmosenv.2012.09.006.

Li, Mingwei, Selin, Noelle, Karplus, Valerie J., Li, Chiao-Ting, Zhang, Da, Luo, Xiaohu, Zhang, Xiliang, 2014. Effects of China's Energy Policy on Future Air Quality in China and the U.S. In: 2014 AGU Fall Meeting. Session A24A-08. American Geophysical Union, San Francisco, CA, US.

Lu, Ming, Sun, Cong, Zheng, Siqi, 2017. Congestion and pollution consequences of driving-to-school trips: a case study in Beijing. Transport. Res. Part D: Transport Environ. 50, 280–291. ISSN: 1361-9209. doi:http://dx.doi.org/10.1016/j.trd.2016.10.023.

Miyazaki, Kazuyuki, Eskes, Henk, 2013. Constraints on surface NOx emissions by as-similating satellite observations of multiple species. Geophys. Res. Lett. 40 (17), 4745–4750. ISSN: 1944-8007. doi:http://dx.doi.org/10.1002/grl.50894.

Nam, Kyung-Min, Selin, Noelle E., Reilly, John M., Sergey, Paltsev, 2010. Measuring welfare loss caused by air pollution in Europe: a CGE analysis. Energy Policy 38 (9), 5059–5071. Special Section on Carbon Emissions and Carbon Management in Cities with Regular Papers. ISSN: 0301-4215. doi:http://dx.doi.org/10.1016/j.enpol.2010.04.034.

Nam, Kyung-Min, Waugh, Caleb J., Paltsev, Sergey, Reilly, John M., Karplus, Valerie J., 2013. Carbon co-benefits of tighter SO2 and NOx regulations in China. Global Environ. Change 23 (6), 1648–1661. doi:http://dx.doi.org/10.1016/j.gloenvcha.2013.09.00.

Nam, Kyung-Min, Waugh, Caleb J., Paltsev, Sergey, Reilly, John M., Karplus, Valerie J., 2014. Synergy between pollution and carbon emissions control: comparing China and the United States. Energy Econ. 46, 186–201. ISSN: 0140-9883. doi:http://dx.doi.org/10.1016/j.eneco.2014.08.013.

National Bureau of Statistics of China, 2008. China Energy Statistical Yearbook. China Statistics Press, Beijing, CN.

Paltsev, Sergey, Henry Chen, Y.-H., Karplus, Valerie J., Kishimoto, Paul Natsuo, Reilly, John M., Loeschel, Andreas, von Graevenitz, Kathrine, Koesler, Simon, 2016. Reducing CO2 from cars in the European Union. Transportation. ISSN: 1572-9435. doi:http://dx.doi.org/10.1007/sllll6-016-9741-3.

Rive, Nathan, 2010. Climate policy in Western Europe and avoided costs of air pollution control. Econ. Modell. 27 (1), 103–115. ISSN: 0264-9993. doi:http://dx.doi.org/10.1016/j.econmod.2009.07.025.

Zhenying, Shao, David, Vance Wagner, 2015. Costs and Benefits of Motor Vehicle Emission Control Programs in China. Tech. Rep., Washington, DC, US. <http://www.theicct.org/costs-and-benefits-motor-vehicle-emission-control-programs-china>.

Wang, Haikun, Fu, Lixin, Zhou, Yu, Du, Xuan, Ge, Weihua, 2010. Trends in vehicular emissions in China's mega cities from 1995 to 2005. In: Environmental Pollution 158.2, pp. 394–400. ISSN: 0269-7491. doi:10.1016/j.envpol.2009.09.002.

Webster, Mort D., Sergey, Paltsev, John, Parsons, Reilly, John M., Jacoby, Henry D, 2008. Uncertainty in Greenhouse Gas Emissions and Costs of Atmospheric Stabilization. Report #165. Cambridge, MA, US: MIT Joint Program on the Science & Policy of Global Change. HDL: 1721.1/44623.

Wu, Rongrong, Bo, Yu, Li, Jing, Li, Lingyu, Li, Yaqi, Xie, Shaodong, 2016. Method to establish the emission inventory of anthropogenic volatile organic compounds in China and its application in the period 2008–2012. Atmos. Environ. 127, 244–254. ISSN: 1352-2310. doi:http://dx.doi.org/10.1016/j.atmosenv.2015.12.015.

Wu, Xiaomeng, Wu, Ye, Zhang, Shaojun, Liu, Huan, Fu, Lixin, Hao, Jiming, 2016. Assessment of vehicle emission programs in China during 1998–2013: achievement, challenges and implications. Environ. Pollut. 214, 556–567. ISSN: 0269-7491. doi:http://dx.doi.org/10.1016/j.envpol.2016.04.042.

Xia, Yinmin, Zhao, Yu, Nielsen, Chris P., 2016. Benefits of China's efforts in gaseous pollutant control indicated by the bottom-up emissions and satellite observations 2000–2014. Atmos. Environ. 136, 43–53. ISSN: 1352-2310. doi:http://dx.doi.org/10.1016/j.atmosenv.2016.04.013.

Yang, Sheng, Ling-Yun, He, 2016. Fuel demand, road transport pollution emissions and residents' health losses in the transitional China. Transport. Res. Part D: Transport Environ. 42, 45–59. ISSN: 1361-9209. doi:http://dx.doi.org/10.1016/j.trd.2015.10.019.

Yue, Xin, Wu, Ye, Hao, Jiming, Pang, Yuan, Ma, Yao, Li, Yi, Li, Boshi, Bao, Xiaofeng, 2015. Fuel quality management versus vehicle emission control in China, status quo and future perspectives. Energy Policy 79, 87–98. ISSN: 0301-4215. doi:http://dx.doi.org/10.1016/j.enpol.2015.01.009.

Zhang, Da, Rausch, Sebastian, Karplus, Valerie J., Zhang, Xiliang, 2013. Quantifying regional economic impacts of CO2 intensity targets in China. Energy Econ. 40, 687–701. doi:http://dx.doi.org/10.1016/j.eneco.2013.08.018.

Zhang, Shaojun, Wu, Ye, Zhao, Bin, Wu, Xiaomeng, Shu, Jiawei, Hao, Jiming, 2016. City-specific vehicle emission control strategies to achieve stringent emission reduction targets in China's Yangtze River Delta region. J. Environ. Sci. ISSN: 1001-0742. doi:http://dx.doi.org/10.1016/j.jes.2016.06.038.

Zhang, Xiliang, Karplus, Valerie J., Qi, Tianyu, Zhang, Da, He, Jiankun, 2016. Carbon emissions in China: how far can new efforts bend the curve. Energy Econ. 54, 388–395. ISSN: 0140-9883. doi:http://dx.doi.org/10.1016/j.eneco.2015.12.002.

Zheng, Xuan, Wu, Ye, Jiang, Jingkun, Zhang, Shaojun, Liu, Huan, Song, Shaojie, Li, Zhenhua, Fan, Xiaoxiao, Fu, Lixin, Hao, Jiming, 2015. Characteristics of on-road diesel vehicles: black carbon emissions in Chinese cities based on portable emissions measurement. Environ. Sci. Technol. 49 (22), 13492–13500. PMID: 26462141. doi:http://dx.doi.org/10.1021/acs.est.5b04129.