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The Impact of Coordinated Policies on Air Pollution Emissions from Road Transportation in China

Paul N. Kishimoto,[*†] Valerie J. Karplus,[‡] Min Zhong,[§] Eri Saikawa,[§] Xu Zhang[¶] and Xiliang Zhang[¶]

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 the broader Regional Emissions Air Quality Climate and Health (REACH) modeling framework, to analyze the impacts of implementing vehicle emissions together with a broader economy-wide climate policy on total air pollution and its spatial distribution. We find that full and immediate implementation of existing vehicle emissions standards at China 3/III level or tighter will significantly reduce the contribution of transportation to degraded air quality by 2030. We further show that transportation emissions standards 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.*

Contents

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1. INTRODUCTION

Air quality is exacting a rising toll on human health and quality of life in China. 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 which produce pollutants as 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** (p. 4). These direct emissions mix with emissions from other large combustion sources—especially electric power plants and industry—and 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 2—include standards regulating the allowable tailpipe emissions of specific pollutants from new private passenger vehicles ('light-duty' vehicles, or LDVs), and heavy-duty vehicles (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 (DPF), for compliance. Impurities in gasoline and diesel fuel are also regulated, to ensure that these emissions control technologies (ECT) 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 targets a reversal of rising CO_2 emissions at latest by 2030. Achieving this goal will require economy-wide policies, such as a CO_2 price, which is currently being piloted in some regions and is expected nationwide within the next five years. Climate policy and vehicle emissions policies will both act on the energy and transportation system, 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 simulate the interaction of road transportation emissions standards and an economy-wide CO_2 price.

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 travels grows, and associated emissions increase from a small share of the total today to a much more substantial share. We further find that an emissions standard is complimentary to economy-wide climate policy that 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 the least cost opportunities to reduce CO_2 are mainly concentrated outside of the transportation sector, an emissions standard that directly targets pollution in the transportation sector delivers a significant additional contribution to air pollution reductions. 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.

1.1 REACH Framework

The Regional Emissions, Air Quality, Climate and Health (REACH) integrated assessment framework (**Figure 1**) was developed to quantify the human health impacts of Chinese air quality policies. Analysis using the framework involves several steps:

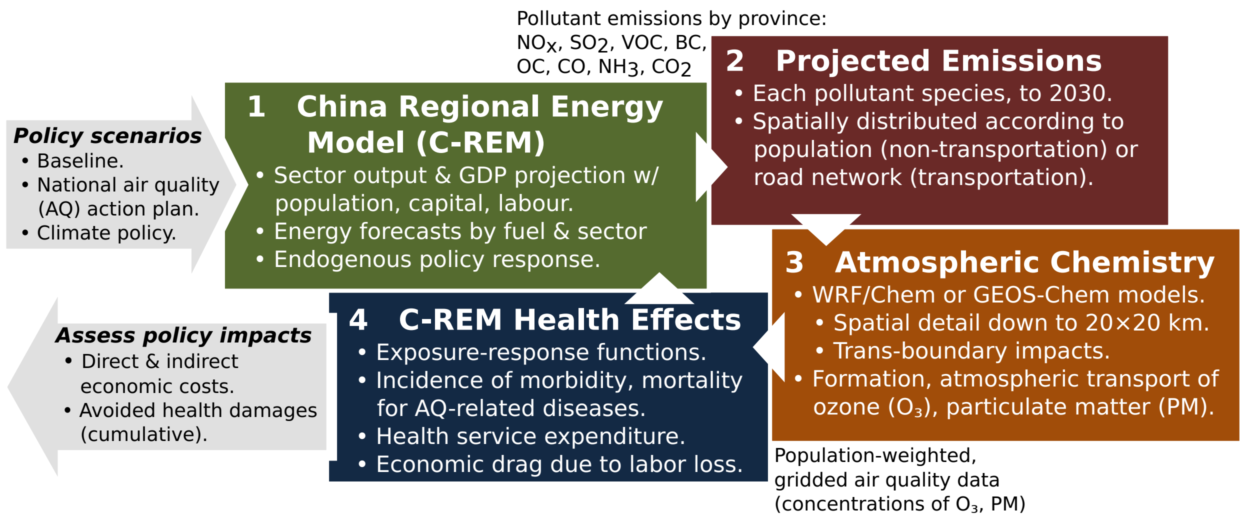

Figure 1. Regional Emissions, Air Quality, Consumption & Health (REACH) framework.

1. Projecting future economic activity, including transportation, and associated energy use, using the China Regional Energy Model (see Appendix A for technical details).

2. Determining the emissions of primary pollutant species (Table 1) due to transportation and other energy uses.

3. Modeling meteorology, atmospheric transport and chemical reactions to estimate concentrations of harmful pollutants, including particulate matter ($PM_{2.5}$) and tropospheric ozone (O_3).

4. Quantifying the population-weighted health impacts of pollution concentrations, and calculating the economic impact of lost labor and health-care expenses.

This framework supports policy impact assessment by allowing comparison of policy-related changes in economic activity (including transportation demand), energy use, and emissions associated with energy use (step 1). This leads to quantification of the reduced or avoided primary pollution (step 2). In atmospheric simulation (step 3), policy-related reductions in primary pollution affect chemistry and dynamics, leading to changes in the concentration of secondary pollutants which reflect the spatial movement of species. In step 4, reductions in population-weighted

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 $\leq$10 μm	PM_{10}
Particulate matter $\leq$2.5 μm	$PM_{2.5}$

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, in order to determine the change, due to policy, in the economic burden of pollution.

2. POLICIES AND MODEL SCENARIOS

Established emissions standards. In 2000, the 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 B1** (p. 26) for emissions from LDVs and HDVs, and in **Table B2** (p. 26) for the presence of sulfur in fuels. Future national standards are specified, with final implementation dates. Some provinces have sought approval to proceed with earlier implementation of these standards. This permission is partly predicated on the ability of fuel providers to supply cleaner fuels that will not degrade the ECTs implied by the standards.

Finally, a variety of local, ad-hoc policies also aim to reduce emissions from road vehicles. These include prohibiting driving by some or all vehicles on certain days, accelerated retirement of older vehicles, 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 3.1 describes in more detail how these changes contribute to reductions in total emissions.

Policy scenarios in this study. To investigate transport-sector emissions standards, the size and distribution of their impact can be compared with the size and distribution of impacts from current and more stringent climate policies, and also with the effects of both implemented in concert. Our analysis employs five model configurations, labeled A–E, as shown in **Figure 2**.

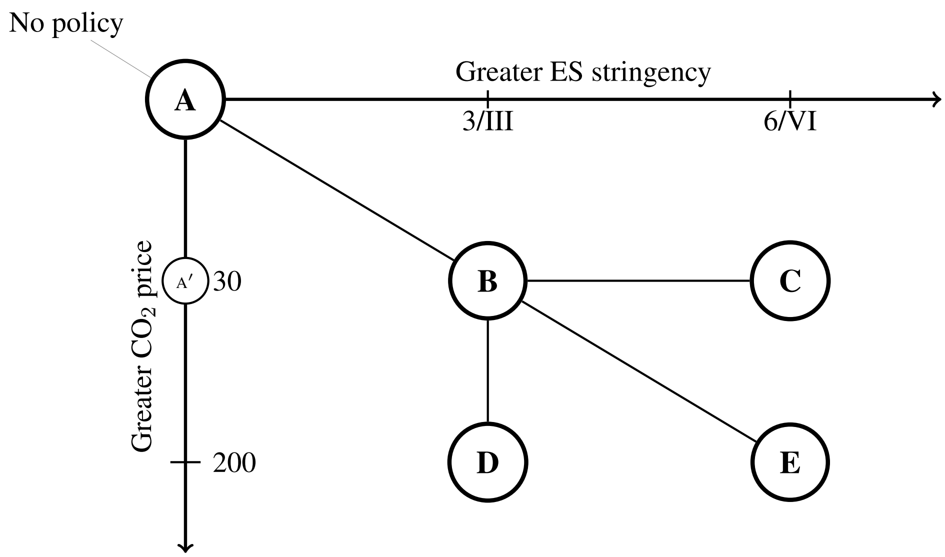

Figure 2. Policy scenarios in this study, with the stringency of road transport emissions standards and the initial (2015) CO_2 price level in RMB per ton.

The policies implemented in these scenarios 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 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] (see Appendix B).

We also consider a sensitivity scenario, termed **A′**, with no transportation ES implemented, and the small CO_2 price of Scenario B.

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

[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 production.

3/III in the near future, the 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. 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 as a result of displacing coal).

C. Stringent ES. More stringent tailpipe emissions and fuel quality standards are introduced, reaching China 6/VI nationwide starting 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. 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.

Comparing 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 today. Comparing Scenario **A′** to **B** further isolates the effect of ES, as both scenarios have the same CO_2 price. Comparing Scenarios **B** and **C** similarly illustrates the impact of accelerating road transport policies under the same CO_2 price. Comparing Scenario **B** with Scenarios **C**, **D** and **E** illustrates the relative size and distribution of benefits from road transport policies compared to climate policy, and also the combined effect of the two.

3. METHODOLOGY

3.1 China Regional Energy Model

We use the China Regional Energy Model (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 4 international regions (see **Table A1**, p. 21); the economy is represented in 14 sectors (see **Table A2**, p. 21). The C-REM projects output from each sector of each province, as well as trade and final demand (consumption), in value units, every 5 years to 2030. Further details of the model are given in Appendix A and references.

Demand 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 (see **Figure A1**, p. 22). Kishimoto *et al.*, (2015) discuss in detail the disaggregation of the transportation services sector to separate freight and passenger services, each with road (denoted `FR` and `PR`, respectively) and non-road modes, as well as household vehicle transportation (`HVT`, representing privately-owned light-duty vehicles) using the method of Karplus *et al.*, (2013).

To represent the effects of the emissions policies discussed in Section 2, the physical accounts of the model were expanded 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. Using emissions totals from the REAS database, version 2.1 (Kurokawa *et al.*, 2013), we associate emissions of each species with individual sectors, provinces, and energy sources. This connection is made by calibrating energy-basis emissions factors (EFs) using the base-year (2007) energy statistics already contained in the C-REM social accounting matrix (SAM):

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

See Appendix B for details of the calibration. In the model projection, the product of an emissions factor and the C-REM projected demand for energy gives the quantity of emissions for each p, f, i, r.

3.2 Policy Impacts: Modeled Mechanisms & Effects

Economy-wide climate & energy policy. Climate policy in a CGE model such as 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 pollution in addition to CO_2.

Climate policy also has indirect impacts on the road transport sectors in two ways. First, freight transport demand arises in the economy because other economic 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, their freight transport demands will also change, affecting the overall level of freight transport energy consumption and pollution. Second, households use their income to purchase passenger transportation services, private vehicles, and fuel. In a CGE framework, 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.

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 implied base-year EFs displayed in **Table B3** 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 inventory), and the amount of energy used in household and commercial road transportation (as reflected in the official energy data underlying C-REM). As a result, the relative improvement in EF due to the introduction of lower-emission vehicles differs province-to-province, and species-to-species.

3.3 WRT-Chem Atmospheric Model

To simulate air quality under each policy scenario, we use the fully coupled "online" regional chemical transport model WRF-Chem version 3.5 (Grell *et al.*, 2005) at a horizontal resolution of $20\,\mathrm{km} \times 20\,\mathrm{km}$ with 31 vertical levels. The initial and lateral chemical boundary conditions are taken from a present-day simulation of the NOAA Geophysical Fluid Dynamics Laboratory (GFDL) global atmospheric chemistry-climate model with atmospheric components (AM3) (Donner *et al.*, 2011; Naik *et al.*, 2013). The Regional Acid Deposition Model version 2 (RADM2) atmospheric chemical mechanism (Stockwell *et al.*, 1990) is used for gas-phase chemistry. Aerosol chemistry is represented by the Model Aerosol Dynamics for Europe with the Secondary Organic Aerosol Model (MADE/SORGAM) (Ackermann *et al.*, 1998; Schell *et al.*, 2001) with some aqueous reactions. The 2007 meteorological data are obtained from the National Center for Environmental Prediction (NCEP) Global Forecast System final gridded analysis datasets (http://rda.ucar.edu/datasets/ds083.0/).

For anthropogenic emissions in China, we use the provincial emissions created from C-REM and regrid these provincial emissions to $0.25° \times 0.25°$ spatial emissions based on REAS. For the areas outside of China, we use the Intergovernmental Panel on Climate Change Fifth Assessment Report (IPCC AR5) Representative Concentration Pathway 8.5 (RCP 8.5) emission dataset (Riahi *et al.*, 2011). For the entire model domain, we use the Fire INventory from NCAR (FINN) (Wiedinmyer *et al.*, 2011) for biomass burning emissions and the Model of Emissions of Gases and Aerosols from Nature (MEGAN 2.1) (Guenther *et al.*, 2012) for biogenic emissions. Aircraft emissions are created by the Task Force Hemispheric Transport of Air Pollution (HTAP). Dust and sea salt emissions are calculated online.

4. RESULTS

4.1 Road Transport Reductions from ES Are Large

Despite average annual growth of 7.5% in transportation energy demand (4.5–9.9% across provinces) between 2010 and 2030, established polices reduce total national road transport pollution emissions to between 2% (OC) and 0.04% (CO) of their 2007 levels (Scenario B vs. Scenario A) (**Figure 3**).

Further reductions occur in Scenarios C and E, as shown in **Figure 4**. 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 the mix of vehicles currently in use. Although future standards (China 6/VI) reduce EFs by further orders of magnitude (cf. **Figure B1**), these translate to a smaller absolute reduction in road transport emissions, because they act on a small base.

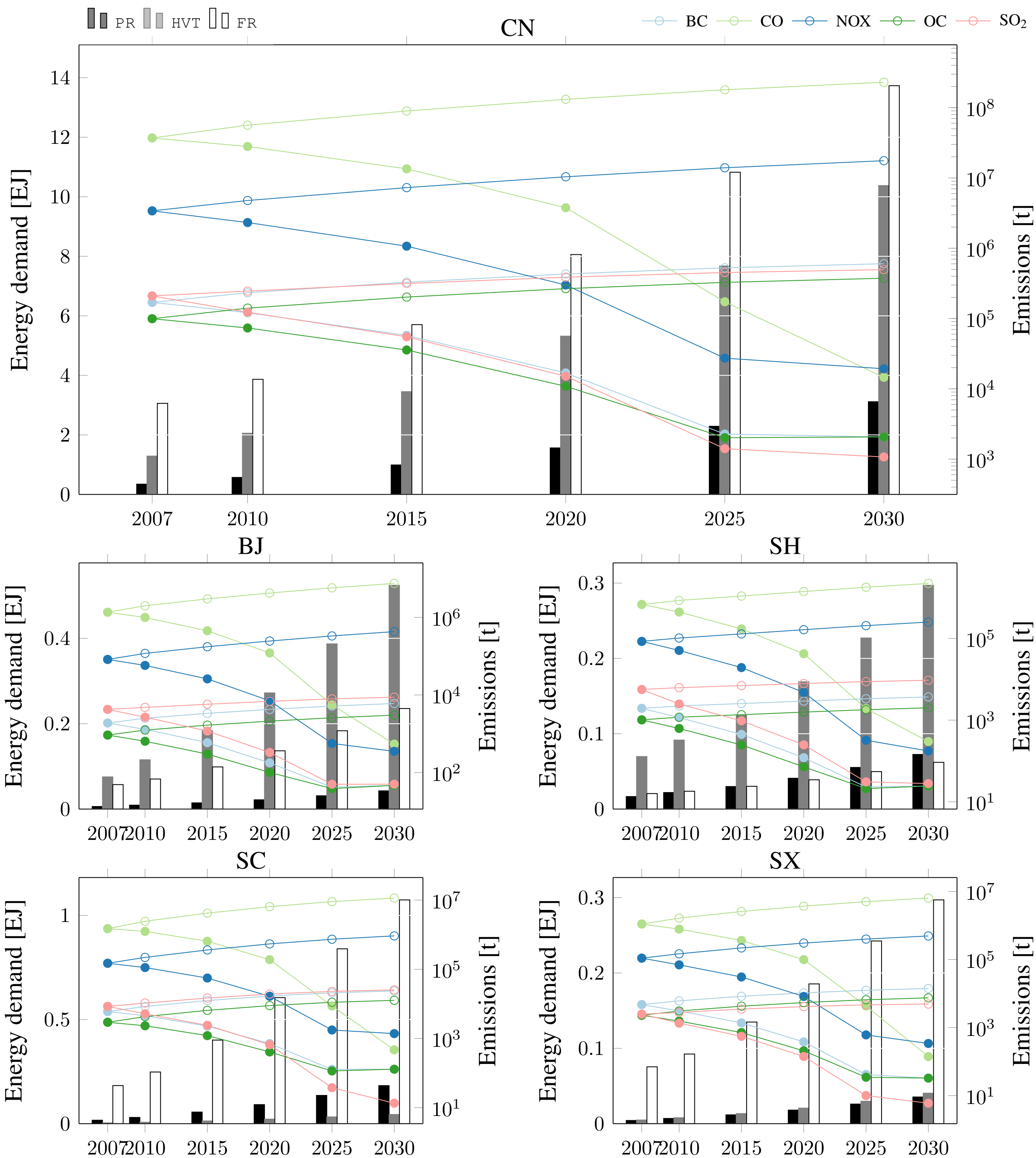

Figure 3. Bars and left ordinate: energy demand for three road transport sectors: commercial passenger (PR), household vehicles (HVT) and freight (FR). Lines and right ordinate: total emissions for five species in Scenario A (open marks) and Scenario B (filled marks). Top: China (CN) total; bottom: four selected provinces with distinct mixes of household vehicle, commercial passenger, and freight road transport—Beijing (BJ), Shanghai (SH), Sichuan (SC) and Shanxi (SX).

4.2 ES Impacts Differ Across Provincial 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 in 2007)—and it is also a larger contributor of pollution: 0.6–6.9% of the national total in species besides CO, versus 0.03–6.8% for other road modes. Consequently, most pollution reductions due to ES occur in road freight transport—see Appendix C, **Figure C1** (p. 32).

Figure 3 and Figure C1 also illustrate that the extent of emissions reductions differs between provinces where passenger road travel activity is relatively large, and those where passenger road activity is small in comparison to freight. For instance, passenger road transport accounts for three quarters of road transport black carbon (BC) in Beijing, but only 36% in Chongqing.

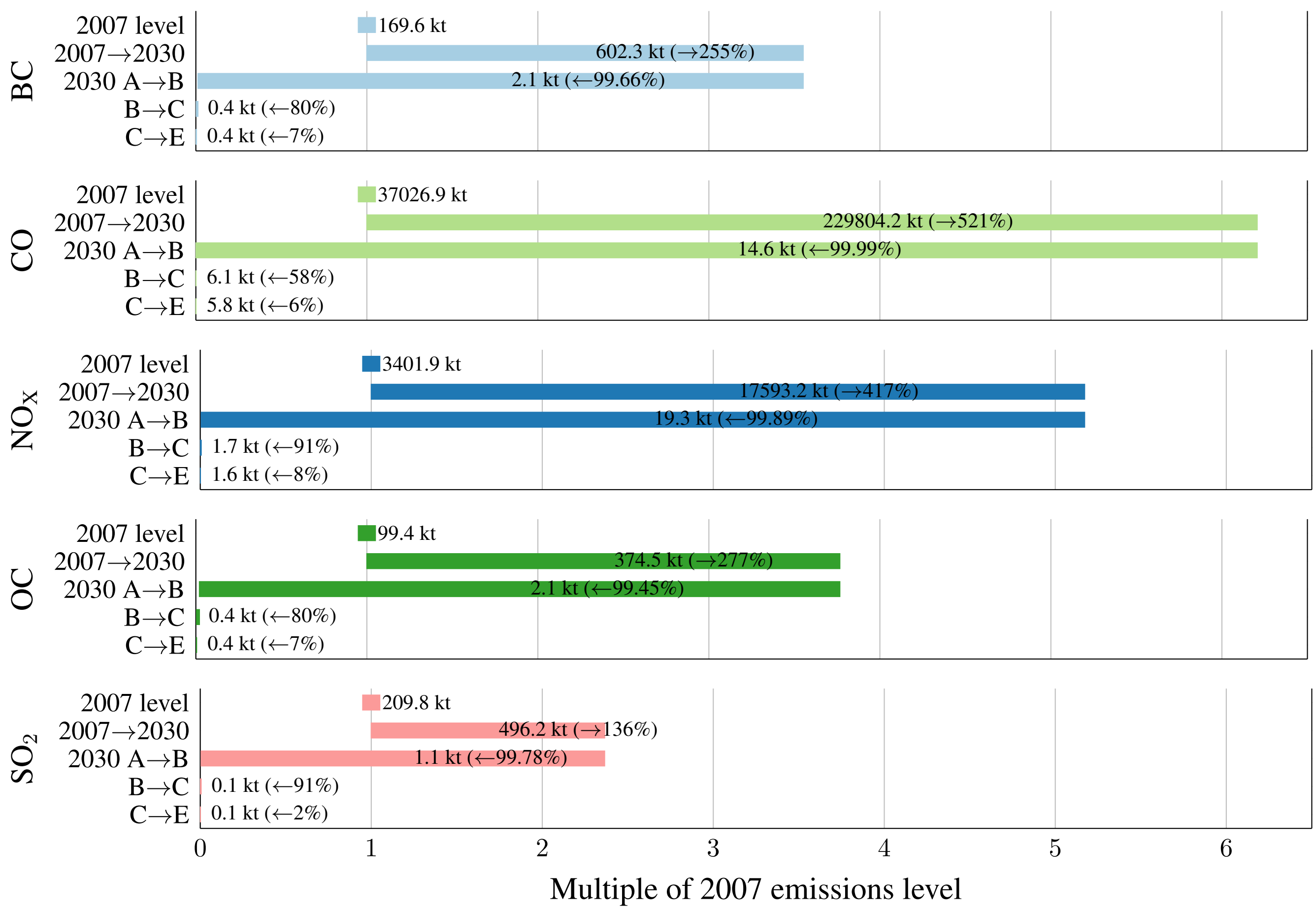

Figure 4. Changes in road transport emissions of five species: increase from 2007–2030 in Scenario A (no policy); and reductions in 2030 from Scenario A–B (introducing China 3/III and a mild CO_2 price), B–C (increasing ES stringency to China 6/VI), and C–E (increasing CO_2 price). The 2007 level is also shown, for reference. Annotations give the total emissions in each scenario and percent change in each year/scenario compared to the bar above.

4.3 ES Are Complementary to Economy-wide Climate & Energy Policy

Figure 5 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. For instance, China 6/VI emissions standards reduce road transport OC emissions by about 80% versus China 3/III, while more stringent climate policy reduces the same emissions by only 2–22% across provinces. This contrast is due to the small share of transport in overall emissions. In comparison, the mild CO_2 price of Scenario B causes 9.6–48% reductions in total emissions of pollution, versus no policy (Scenario A); similarly, tightening the CO_2 price only (Scenario D) results in 8.9–27% reduction versus established policy (Scenario B). Indeed, the co-benefits of climate policy for air pollution reduction are substantial, even for a relatively modest CO_2 price. As discussed above, co-benefits largely come from non-transport sectors, so emissions standards for road transportation are highly complementary in reducing pollution emissions.

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 CO_2 pricing—efficiency improvements and fuel-

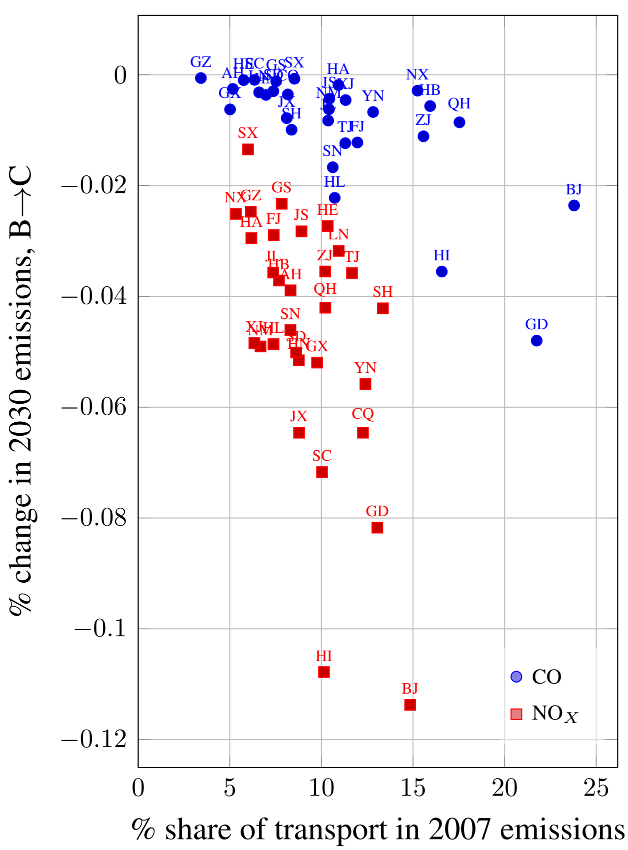

Figure 5. Fraction of transport CO and NO$_X$in *total* emissions in each province, versus the change in total CO and NO$_X$emissions due to moving from established to stringent ES. The reduction is generally smaller than 0.15%, in part because less than 24% or 15% of these species, respectively, is attributed to transport; and less to road transport.

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 total emissions, road transport emissions decrease by only 2.0–7.1% across species.

Consequently, 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.

4.4 ES Impacts Spatial Emissions and Air Quality

The implementation of China 3 (Scenario B) alone could significantly reduce pollutant emissions in eastern China in 2020 and 2030. **Figure 6** shows that the reduction of BC, OC, CO and NO_Xmainly occurs in the North China Plain, Pearl River Delta, Yangtze River Delta, and Sichuan Basin in 2020. This emissions reduction would reduce the concentrations of $PM_{2.5}$ (8–20 μg·m^{-3}), NO_X(more than 20 μg·m^{-3}), SO_2 (3–5 μg·m^{-3}) in Eastern China, and O_3 (12–16 μg·m^{-3}) in Southern China (**Figure 7**).

In the North China Plain, O_3 concentrations would increase by 4–8 μg·m^{-3} due to less $NO_X$for O_3 titration. As a result, population-weighted average exposure across China decreases from 73 μg·m^{-3} to 66.4 μg·m^{-3} in 2020, equivalent to 94,450 avoided premature mortalities each year.

When ES increases in stringency from China 3/III (Scenario B) to China 6/VI (Scenario C), further emission reduction of pollutants can be seen in urban areas of eastern China (**Figure 8**).

However, no significant changes in air pollutant concentrations are observed, as illustrated in **Figure 9** , because road transport is a relatively small emissions source compared to other sources such as domestic energy uses and industry. The character of emissions changes moving from Scenario A→B and B→C is also similar to the 2020 result when looking at 2030 emissions differences, as shown in Appendix C, **Figure C3** (p. 34) and **Figure C4** (p. 35).

5. DISCUSSION

5.1 Costs And Implementation of Tailpipe Controls

A key difference between the carbon and ES policies implemented in our scenarios is that the cost of carbon emissions abatement is captured in the general equilibrium framework of C-REM. In contrast, the ES policies are implemented through exogenous calculation of road transportation EFs. Implementing tighter ES would require vehicle manufacturers to install more advanced ECT on passenger vehicles and road freight vehicles sold in China. Fuel economy improvements would 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. Shao *et al.*, (2015) estimate of 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.

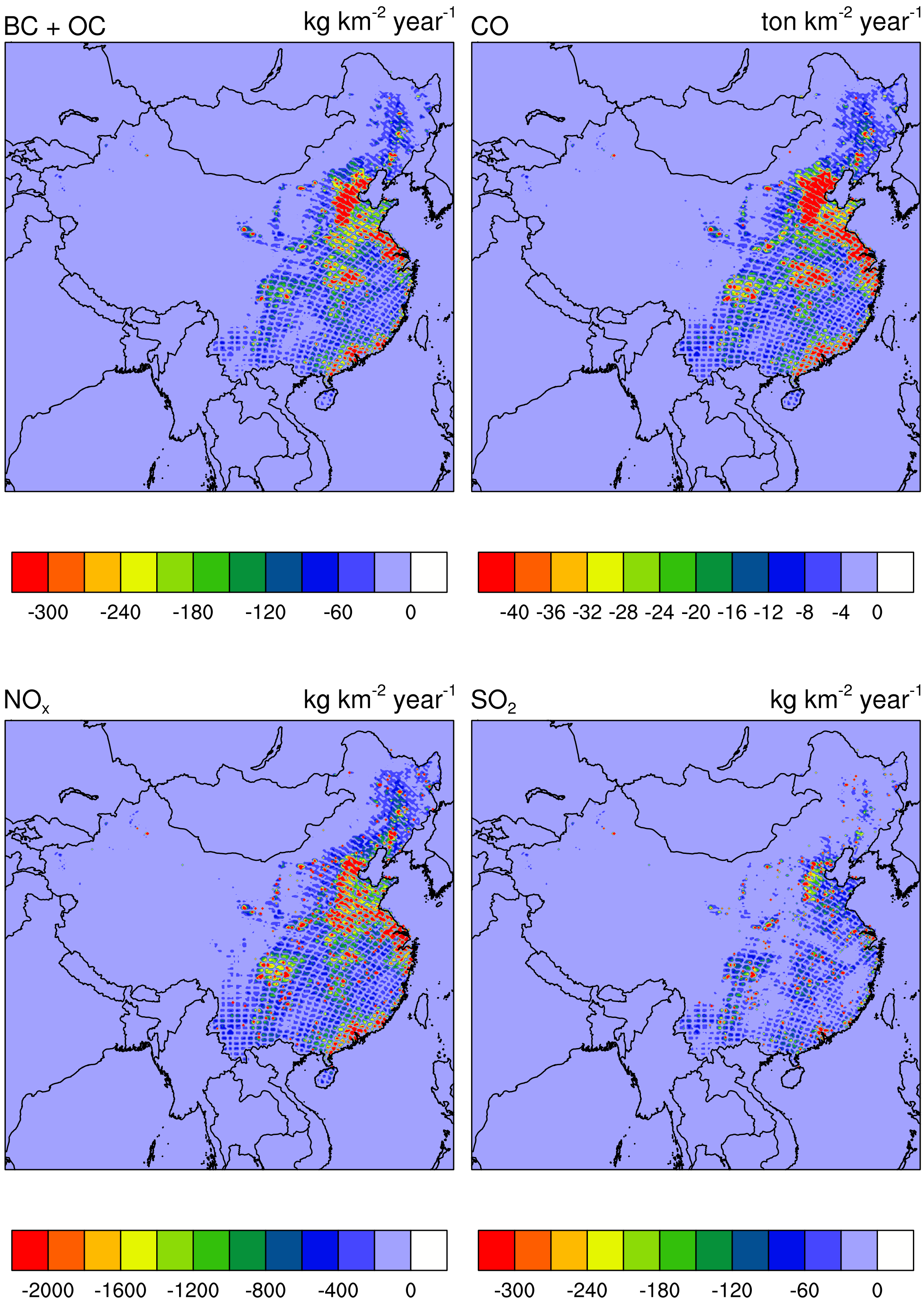

Figure 6. Spatial emissions differences of BC+OC, CO, NO_X, and SO_2 in 2020 between China 3/III (Scenario B) and no China 3/III (Scenario A′).

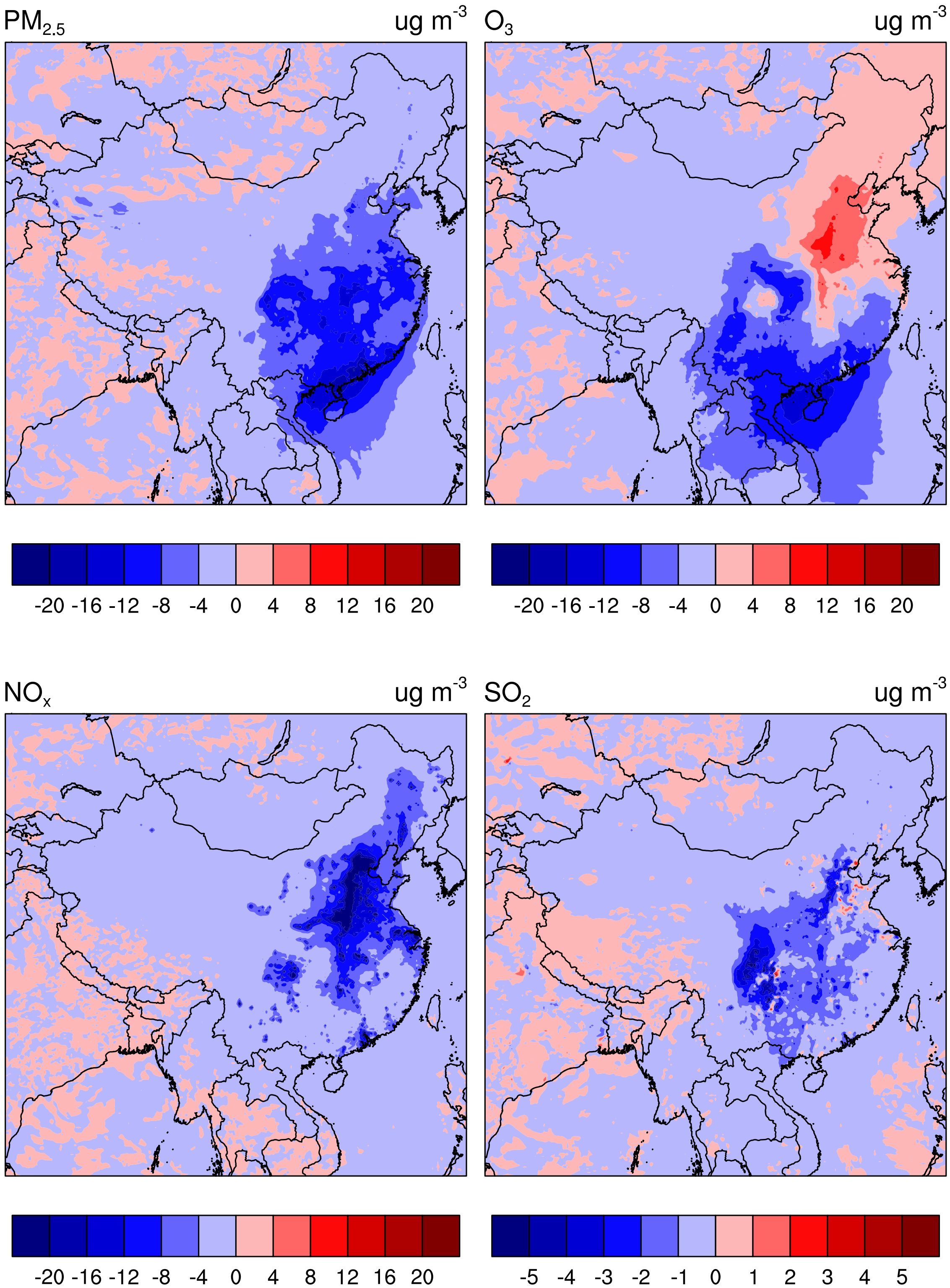

Figure 7. Concentration differences of $PM_{2.5}$, O_3, NO_X and SO_2 in 2020 between China 3/III (Scenario B) and no China 3/III (Scenario A′).

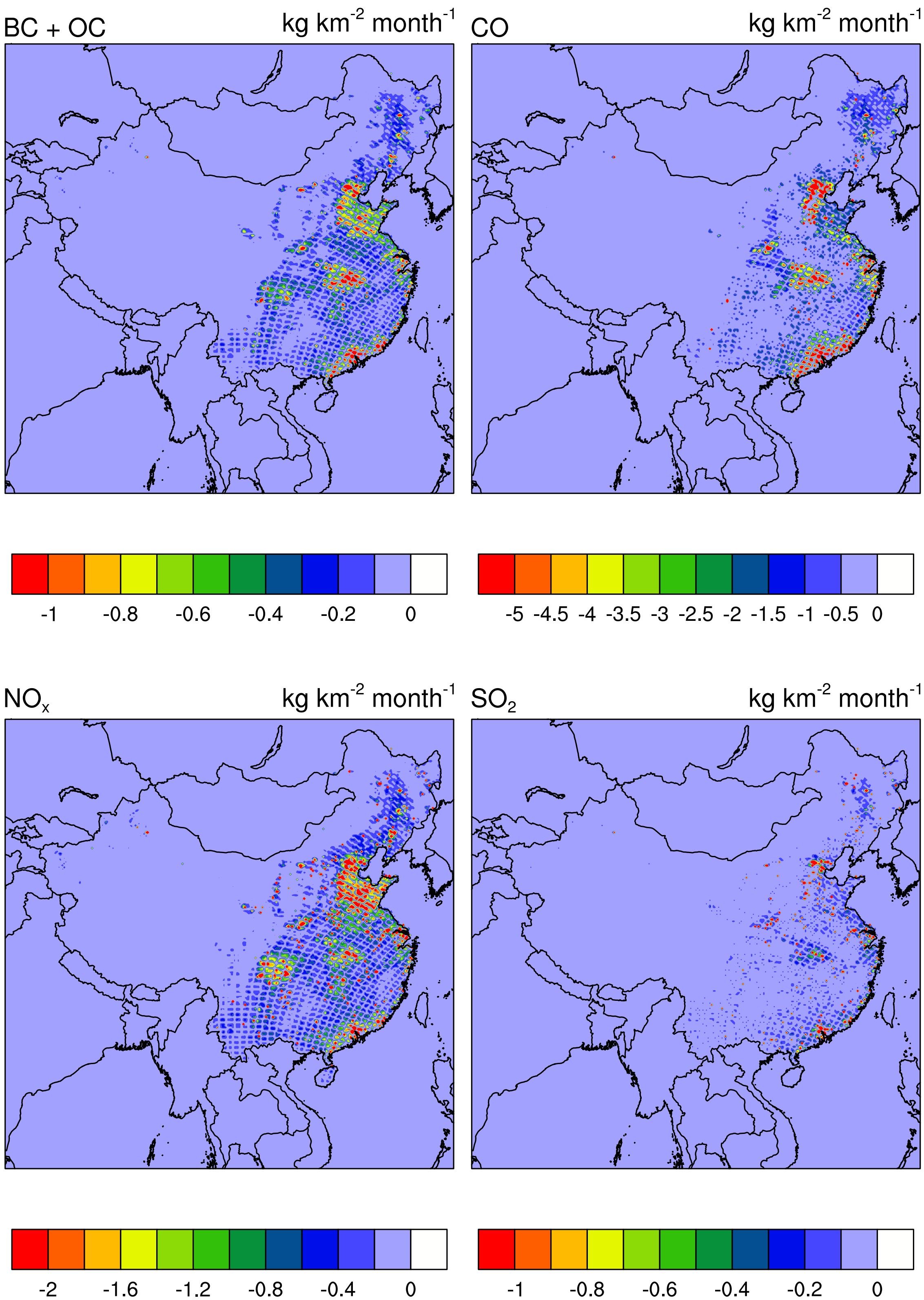

Figure 8. Spatial emissions differences of BC+OC, CO, NO_X, and SO_2 in 2020 between China 6/VI (Scenario C) and China 3/III (Scenario B).

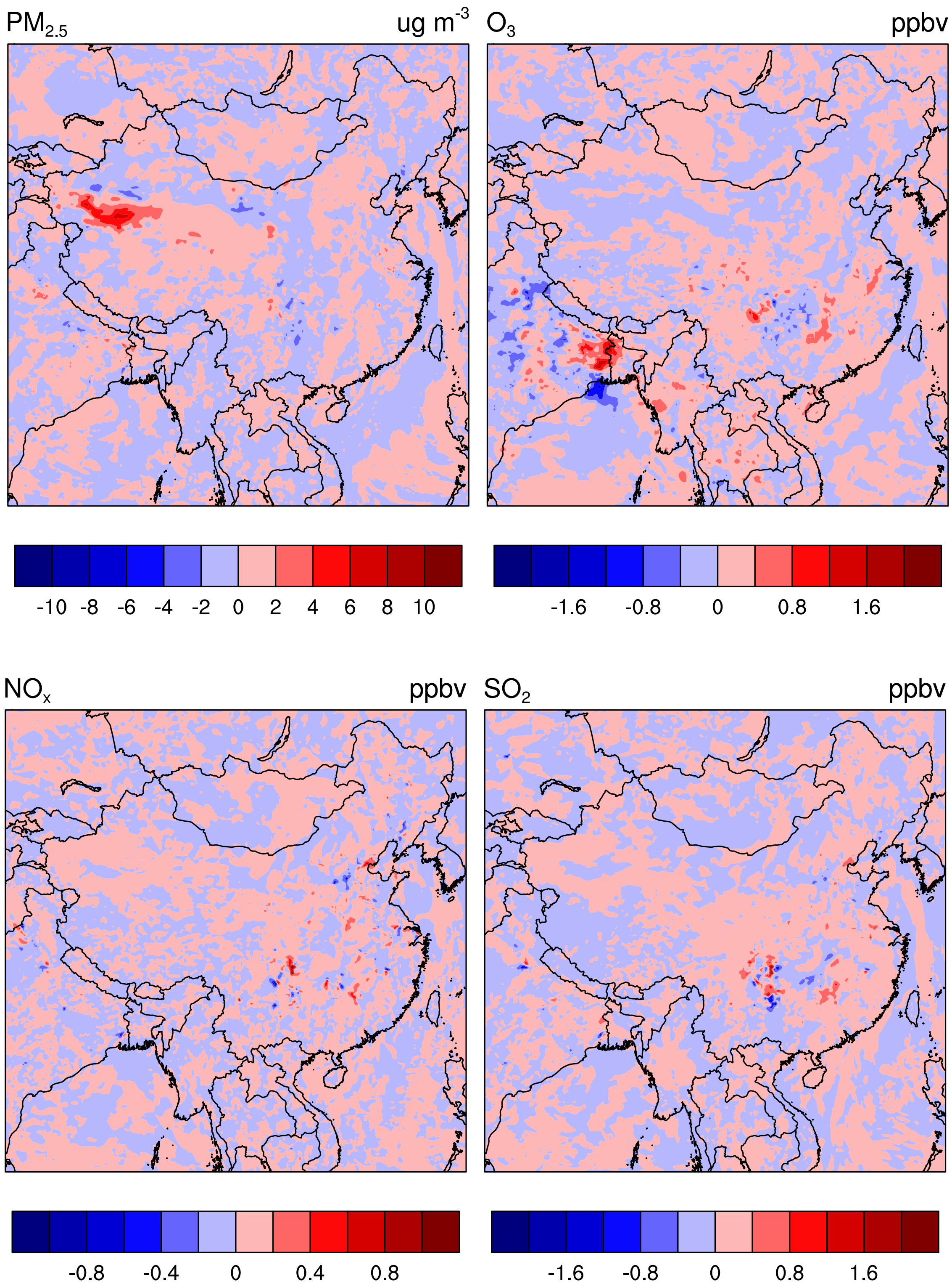

Figure 9. Concentration differences of $PM_{2.5}$, O_3, NO_X and SO_2 in 2020 between China 6/VI (Scenario C) and China 3/III (Scenario B).

This study assumes 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. More importantly, all scenarios assume that ES are *fully* implemented—that is, 100% of new vehicles comply with the active standard as of the sale date. In order to realize the air pollution emissions reductions identified here, ES implementation and compliance are critical; 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. **Figure B2** (p. 25) 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 (Figure C2, p. 33) are much higher than in 2030 (cf. Figure 4), even though the 2020 fleets and total transport energy use are larger.

5.2 Conclusion & Policy Implications

Taken together, this work clearly illustrates the emissions reduction and human health benefits of immediately and uniformly implementing emissions standards at the China III level or higher. While moving to China 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, sticking with the existing standards is advised. Our results show that the marginal benefits of accelerating the policy timeline are modest. 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 in Europe are large relative to total emissions in that context, but small relative to total emissions in the Chinese context. We also 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:

1. **Strengthen mechanisms for enforcing 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.

2. **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.

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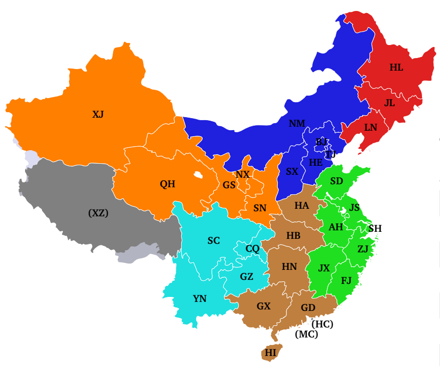
APPENDIX A: China Regional Energy Model

For complete description and discussion of the C-REM and its transportation subsector representation, see Kishimoto *et al.*, (2014, 2015) and Zhang *et al.*, (2013).

Table A1. 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 A2. List of C-REM sectors, omitting the transportation subsectors shown in Figure A1.

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

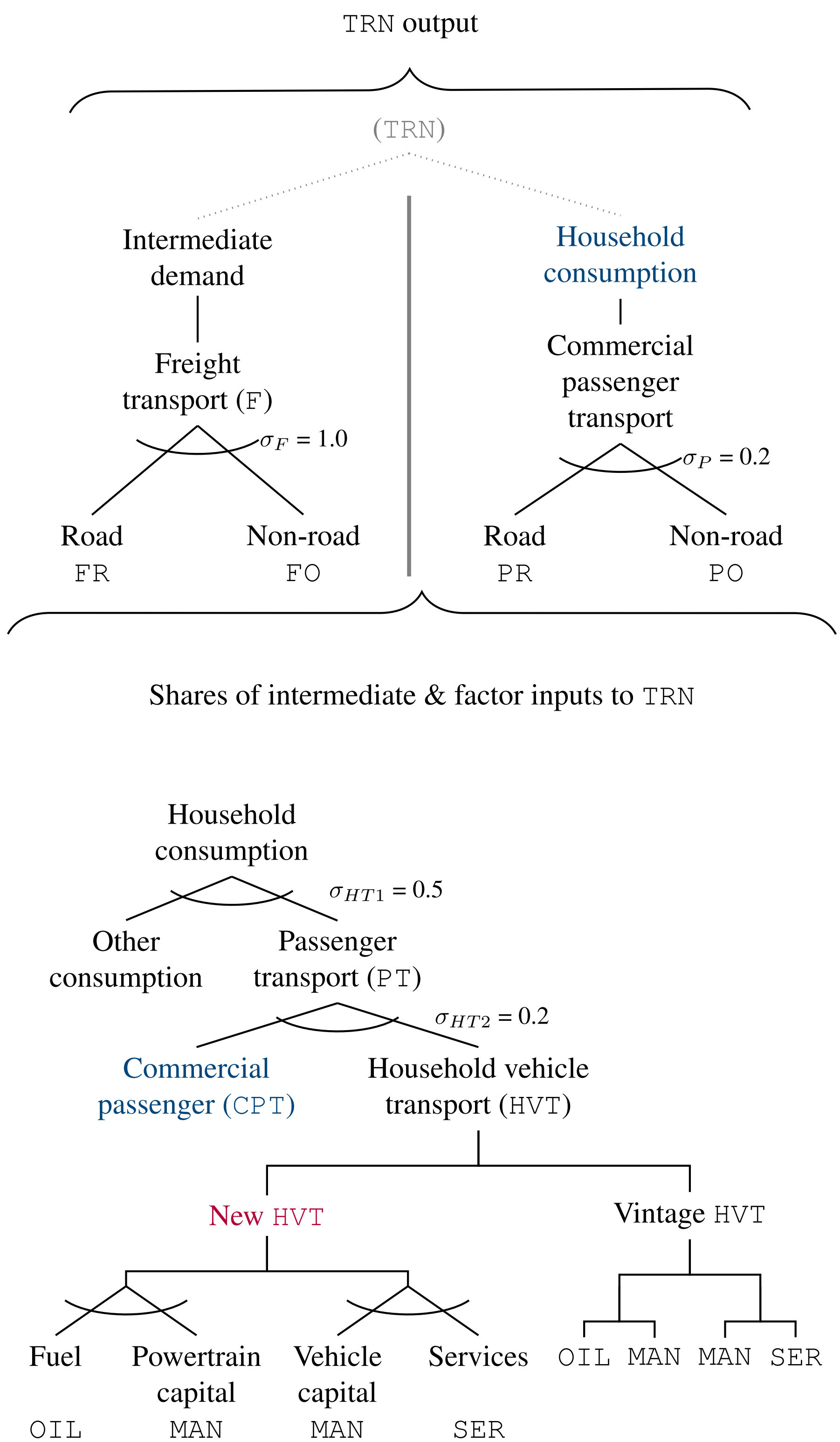


Figure A1. Disaggregate transportation representation in the C-REM. Top: a monolithic transportation services sector is broken into *freight* transport—supplying intermediate demand by other economic sectors—and *passenger* transport—supplying the demand of households for commercial travel. Bottom: households' consumption contains passenger transport, which consists of commercial passenger transport or 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.

APPENDIX B: Energy-basis Emissions Factors

B.1 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 emissions factors (EF), 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.

The base-year (2007) data in the REAS v2.1 emissions inventory and the C-REM social accounting matrix imply EFs in the following way. For each province and species, we aggregate emissions from the REAS combustion and non-combustion sectors to C-REM sectors and REAS fuels to C-REM fuels. We divide these emissions totals by the corresponding energy flow from the C-REM supplemental accounts. The resulting 2007 EFs thus exactly reproduce the 2007 REAS v2.1 emissions totals.

In the future, we make use of the detailed bottom-up engineering model of Akerlind, (2013). This model tracks total 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.

In C-REM periods beyond the base year, 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, for the C-REM forecast year 2010, this is the sum of fuel energy demand, in 2010, by vehicles sold in 2010, 2009, or 2008. The remainder of fuel energy demand in 2010 is associated with vehicles sold in 2007 or earlier.

Fuel demand from new vehicles is associated with EFs in Figure B1 (bold, horizontal lines). 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 implied 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.

In Scenario B, new vehicles meet the China 3 (passenger road and household vehicle transport) or China III (freight road) standard from 2008 onwards. In Scenarios C and E, new vehicles meet China 3/III from 2007–2010, China 4/IV (etc.) from 2011–2015, and China 6/VI from 2016 onwards, excepting Beijing, which meets China 5/V in 2013 and China 6/VI in 2016.

China's emissions standards, like the Euro standards on which they are based,[3] specify distance-basis, rather than energy-basis, EFs for new vehicles. To determine energy-basis EFs for the calculation just described, 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.

Figure B1 also shows the energy-weighted average EF across the entire fleet, for $NO_X$from refined oil combustion in the `HVT` fleet. Table B3 gives a complete list across provinces for 2010, 2015 and 2020; in Scenario A, the implied 2007 EF is used unchanged throughout the projection.

[3] e.g., Directives 91/441/EEC and 91/542/EEC, for Euro I/1 respectively.

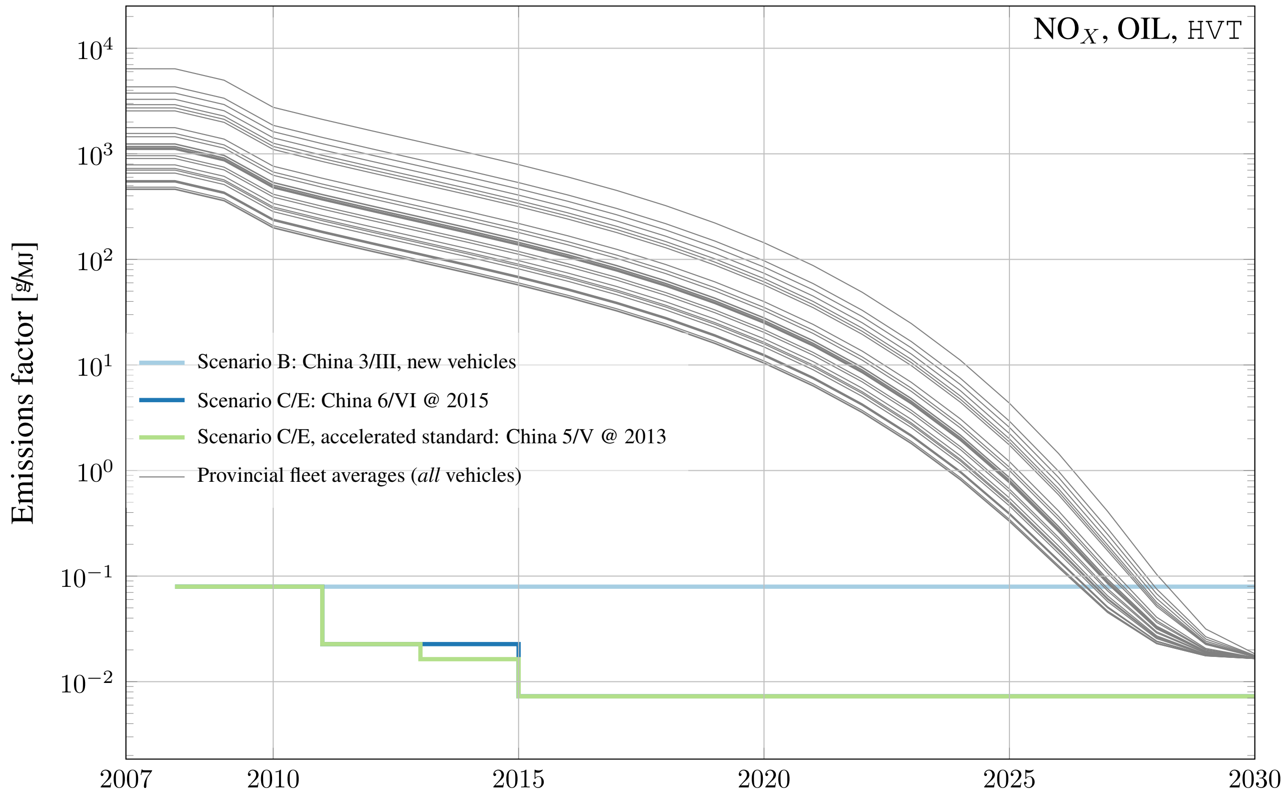


Figure B1. Bold lines: mandated emissions factors of $NO_X$from refined oil (gasoline or diesel) combustion in *new*, private, light-duty vehicles under baselines and policy scenarios. Thin lines: fleet-average OIL NO_Xemissions factors for 30 provinces under Scenario C.

Because there is a base-year implied 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 differences in policy 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; on our projection, this occurs near the very end of the C-REM forecast period—see Figure B2.

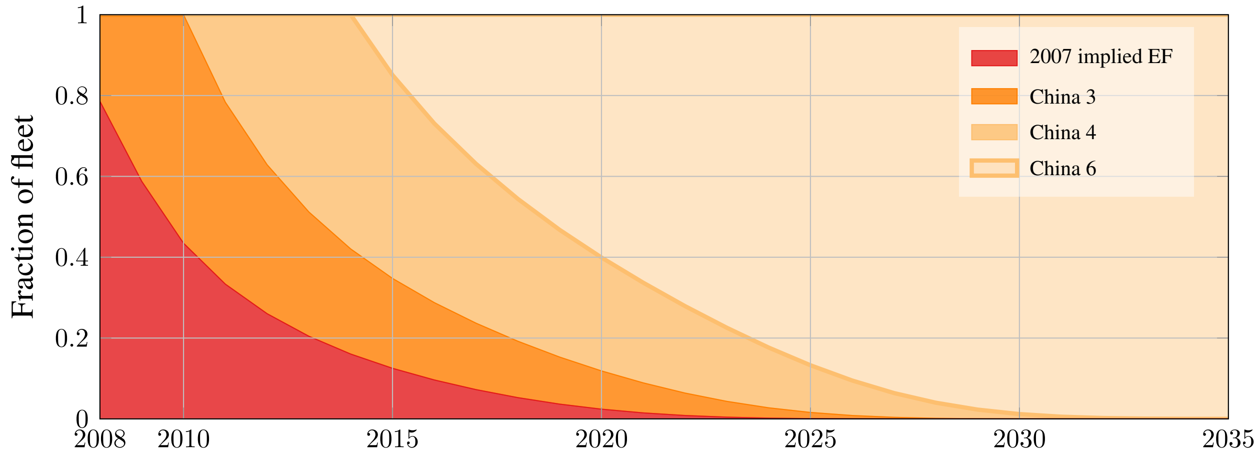


Figure B2. Projected fraction of private vehicle stock complying with various ES, Scenario C/D.

B.2 Non-transportation 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.* (Li *et al.*, 2014). Model base year (2007) EFs are calibrated, as in transportation, to reflect the total energy demand in the C-REM SAM 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.* (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; the CO_2 price reduces emissions not by altering EFs, but by incentivizing low-emissions-intensity activities.

Table B1. 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 B2. 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

Table B3. 2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion (g/MJ), by road transport subsector, province and species.

	Road freight			Private passenger vehicles		
Province	**2010**	**2015**	**2030**	**2010**	**2015**	**2030**
			BC			
AH	11.5	3.34	1.78×10^{-2}	25.9	7.43	5.83×10^{-3}
BJ	3.21	0.959	1.78×10^{-2}	7.13	2.05	5.80×10^{-3}
CQ	11.7	3.41	1.78×10^{-2}	23.2	6.65	5.83×10^{-3}
FJ	8.28	2.42	1.78×10^{-2}	15.4	4.43	5.83×10^{-3}
GD	6.75	1.98	1.78×10^{-2}	14.7	4.22	5.83×10^{-3}
GS	12.9	3.75	1.78×10^{-2}	103	29.5	5.89×10^{-3}
GX	6.70	1.96	1.78×10^{-2}	17.5	5.02	5.83×10^{-3}
GZ	12.6	3.66	1.78×10^{-2}	27.3	7.85	5.84×10^{-3}
HA	8.41	2.45	1.78×10^{-2}	211	60.7	5.97×10^{-3}
HB	7.56	2.21	1.78×10^{-2}	82.2	23.6	5.87×10^{-3}
HE	13.6	3.94	1.78×10^{-2}	99.2	28.5	5.89×10^{-3}
HI	11.5	3.34	1.78×10^{-2}	13.3	3.82	5.83×10^{-3}
HL	12.4	3.58	1.78×10^{-2}	8.52	2.45	5.82×10^{-3}
HN	7.94	2.32	1.78×10^{-2}	28.3	8.14	5.84×10^{-3}
JL	9.78	2.84	1.78×10^{-2}	19.1	5.50	5.83×10^{-3}
JS	10.8	3.15	1.78×10^{-2}	45.8	13.2	5.85×10^{-3}
JX	7.34	2.14	1.78×10^{-2}	22.4	6.44	5.83×10^{-3}
LN	9.37	2.73	1.78×10^{-2}	24.1	6.93	5.83×10^{-3}
NM	7.49	2.19	1.78×10^{-2}	40.1	11.5	5.84×10^{-3}
NX	8.06	2.35	1.78×10^{-2}	91.7	26.3	5.88×10^{-3}
QH	14.9	4.32	1.78×10^{-2}	78.5	22.5	5.87×10^{-3}
SC	8.41	2.45	1.78×10^{-2}	48.7	14.0	5.85×10^{-3}
SD	3.76	1.12	1.78×10^{-2}	34.3	9.84	5.84×10^{-3}
SH	14.7	4.25	1.78×10^{-2}	5.70	1.64	5.82×10^{-3}
SN	7.07	2.07	1.78×10^{-2}	13.6	3.91	5.83×10^{-3}
SX	13.6	3.93	1.78×10^{-2}	92.8	26.6	5.88×10^{-3}
TJ	7.69	2.24	1.78×10^{-2}	15.2	4.36	5.83×10^{-3}
XJ	6.50	1.90	1.78×10^{-2}	26.3	7.54	5.83×10^{-3}
YN	11.0	3.18	1.78×10^{-2}	24.6	7.06	5.83×10^{-3}
ZJ	8.90	2.59	1.78×10^{-2}	13.8	3.95	5.83×10^{-3}

Table B3 (continued). 2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion ($^{g}/_{MJ}$), by road transport subsector, province and species.

	Road freight			Private passenger vehicles		
Province	2010	2015	2030	2010	2015	2030
			CO			
AH	1130	325	2.12×10^{-1}	7120	2040	1.68×10^{-1}
BJ	522	150	2.11×10^{-1}	6550	1880	1.67×10^{-1}
CQ	1190	341	2.12×10^{-1}	7940	2280	1.68×10^{-1}
FJ	1000	287	2.12×10^{-1}	4350	1250	1.66×10^{-1}
GD	825	237	2.12×10^{-1}	3900	1120	1.65×10^{-1}
GS	1260	363	2.12×10^{-1}	23 600	6760	1.79×10^{-1}
GX	671	193	2.11×10^{-1}	3780	1080	1.65×10^{-1}
GZ	1330	381	2.12×10^{-1}	9670	2770	1.70×10^{-1}
HA	828	238	2.12×10^{-1}	43 400	12 500	1.94×10^{-1}
HB	765	220	2.12×10^{-1}	15 400	4420	1.74×10^{-1}
HE	1380	397	2.12×10^{-1}	28 100	8050	1.83×10^{-1}
HI	1220	351	2.12×10^{-1}	3220	923	1.65×10^{-1}
HL	1170	336	2.12×10^{-1}	3000	860	1.65×10^{-1}
HN	780	224	2.12×10^{-1}	7590	2180	1.68×10^{-1}
JL	974	280	2.12×10^{-1}	5250	1510	1.66×10^{-1}
JS	1050	303	2.12×10^{-1}	13 100	3760	1.72×10^{-1}
JX	710	204	2.12×10^{-1}	5450	1560	1.66×10^{-1}
LN	995	286	2.12×10^{-1}	7750	2220	1.68×10^{-1}
NM	671	193	2.11×10^{-1}	13 000	3730	1.72×10^{-1}
NX	823	236	2.12×10^{-1}	19 500	5600	1.77×10^{-1}
QH	1430	410	2.12×10^{-1}	17 100	4920	1.75×10^{-1}
SC	850	244	2.12×10^{-1}	23 100	6630	1.79×10^{-1}
SD	421	121	2.11×10^{-1}	10 200	2920	1.70×10^{-1}
SH	2330	669	2.13×10^{-1}	2890	828	1.65×10^{-1}
SN	676	194	2.11×10^{-1}	3960	1140	1.65×10^{-1}
SX	1320	380	2.12×10^{-1}	36 800	10 600	1.89×10^{-1}
TJ	909	261	2.12×10^{-1}	7200	2070	1.68×10^{-1}
XJ	655	188	2.11×10^{-1}	7660	2200	1.68×10^{-1}
YN	1140	327	2.12×10^{-1}	8410	2410	1.69×10^{-1}
ZJ	1110	318	2.12×10^{-1}	6070	1740	1.67×10^{-1}

Table B3 (continued). 2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion (g/MJ), by road transport subsector, province and species.

Province	Road freight 2010	Road freight 2015	Road freight 2030	Private passenger vehicles 2010	Private passenger vehicles 2015	Private passenger vehicles 2030
			NO_X			
AH	329	94.9	8.21×10^{-2}	502	144	7.77×10^{-3}
BJ	103	30.0	7.97×10^{-2}	339	97.4	7.60×10^{-3}
CQ	338	97.4	8.21×10^{-2}	533	153	7.79×10^{-3}
FJ	244	70.3	8.20×10^{-2}	238	68.2	7.58×10^{-3}
GD	204	58.9	8.20×10^{-2}	241	69.1	7.58×10^{-3}
GS	309	89.1	8.21×10^{-2}	1620	466	8.57×10^{-3}
GX	205	59.2	8.20×10^{-2}	201	57.6	7.56×10^{-3}
GZ	364	105	8.21×10^{-2}	673	193	7.89×10^{-3}
HA	234	67.7	8.20×10^{-2}	2760	792	9.38×10^{-3}
HB	214	61.9	8.20×10^{-2}	314	90.1	7.64×10^{-3}
HE	346	99.6	8.21×10^{-2}	1420	407	8.42×10^{-3}
HI	367	106	8.21×10^{-2}	209	59.9	7.56×10^{-3}
HL	280	80.8	8.21×10^{-2}	198	56.9	7.55×10^{-3}
HN	231	66.5	8.20×10^{-2}	482	138	7.76×10^{-3}
JL	230	66.4	8.20×10^{-2}	303	86.8	7.63×10^{-3}
JS	309	89.0	8.21×10^{-2}	627	180	7.86×10^{-3}
JX	216	62.5	8.20×10^{-2}	284	81.4	7.62×10^{-3}
LN	234	67.5	8.20×10^{-2}	537	154	7.80×10^{-3}
NM	175	50.6	8.20×10^{-2}	764	219	7.96×10^{-3}
NX	201	58.0	8.20×10^{-2}	1260	362	8.31×10^{-3}
QH	321	92.4	8.21×10^{-2}	1100	316	8.20×10^{-3}
SC	212	61.3	8.20×10^{-2}	1180	337	8.25×10^{-3}
SD	101	29.4	8.19×10^{-2}	474	136	7.75×10^{-3}
SH	733	211	8.24×10^{-2}	233	66.9	7.58×10^{-3}
SN	186	53.8	8.20×10^{-2}	237	68.1	7.58×10^{-3}
SX	347	99.9	8.21×10^{-2}	1860	534	8.74×10^{-3}
TJ	200	57.8	8.20×10^{-2}	414	119	7.71×10^{-3}
XJ	155	45.0	8.20×10^{-2}	489	140	7.76×10^{-3}
YN	321	92.5	8.21×10^{-2}	508	146	7.77×10^{-3}
ZJ	253	72.9	8.21×10^{-2}	390	112	7.69×10^{-3}

Table B3 (continued). 2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion (g/MJ), by road transport subsector, province and species.

Province	Road freight 2010	Road freight 2015	Road freight 2030	Private passenger vehicles 2010	Private passenger vehicles 2015	Private passenger vehicles 2030
			OC			
AH	4.40	1.30	1.78×10^{-2}	18.0	5.18	5.83×10^{-3}
BJ	1.31	0.414	1.78×10^{-2}	3.68	1.06	5.80×10^{-3}
CQ	4.52	1.33	1.78×10^{-2}	14.6	4.20	5.83×10^{-3}
FJ	3.45	1.03	1.78×10^{-2}	12.7	3.65	5.82×10^{-3}
GD	2.86	0.861	1.78×10^{-2}	12.1	3.47	5.82×10^{-3}
GS	4.81	1.42	1.78×10^{-2}	43.2	12.4	5.85×10^{-3}
GX	2.62	0.790	1.78×10^{-2}	12.7	3.64	5.82×10^{-3}
GZ	4.91	1.45	1.78×10^{-2}	14.7	4.22	5.83×10^{-3}
HA	3.21	0.960	1.78×10^{-2}	103	29.6	5.89×10^{-3}
HB	2.92	0.877	1.78×10^{-2}	83.4	23.9	5.88×10^{-3}
HE	5.16	1.52	1.78×10^{-2}	56.2	16.1	5.86×10^{-3}
HI	4.66	1.38	1.78×10^{-2}	9.66	2.78	5.82×10^{-3}
HL	4.54	1.34	1.78×10^{-2}	3.93	1.13	5.82×10^{-3}
HN	3.05	0.914	1.78×10^{-2}	18.8	5.41	5.83×10^{-3}
JL	3.66	1.09	1.78×10^{-2}	9.50	2.73	5.82×10^{-3}
JS	4.13	1.22	1.78×10^{-2}	37.6	10.8	5.84×10^{-3}
JX	2.82	0.847	1.78×10^{-2}	18.3	5.25	5.83×10^{-3}
LN	3.60	1.07	1.78×10^{-2}	12.6	3.63	5.82×10^{-3}
NM	2.73	0.823	1.78×10^{-2}	15.4	4.42	5.83×10^{-3}
NX	3.06	0.917	1.78×10^{-2}	39.5	11.4	5.84×10^{-3}
QH	5.45	1.60	1.78×10^{-2}	30.7	8.82	5.84×10^{-3}
SC	3.18	0.952	1.78×10^{-2}	32.6	9.35	5.84×10^{-3}
SD	1.51	0.473	1.78×10^{-2}	27.7	7.95	5.84×10^{-3}
SH	5.81	1.71	1.78×10^{-2}	3.51	1.02	5.82×10^{-3}
SN	2.66	0.801	1.78×10^{-2}	7.94	2.29	5.82×10^{-3}
SX	5.08	1.50	1.78×10^{-2}	57.8	16.6	5.86×10^{-3}
TJ	3.10	0.927	1.78×10^{-2}	9.36	2.69	5.82×10^{-3}
XJ	2.46	0.743	1.78×10^{-2}	12.8	3.67	5.82×10^{-3}
YN	4.25	1.26	1.78×10^{-2}	13.4	3.84	5.83×10^{-3}
ZJ	3.73	1.11	1.78×10^{-2}	7.91	2.28	5.82×10^{-3}

Table B3 (continued). 2010, 2015 and 2030 energy-basis emissions factors from refined oil combustion (g/MJ), by road transport subsector, province and species.

	Road freight			Private passenger vehicles		
Province	**2010**	**2015**	**2030**	**2010**	**2015**	**2030**
			SO_2			
AH	18.5	5.30	7.82×10^{-4}	18.5	5.33	7.43×10^{-3}
BJ	12.3	3.54	7.78×10^{-4}	12.4	3.56	7.37×10^{-3}
CQ	33.5	9.62	7.93×10^{-4}	33.6	9.65	7.44×10^{-3}
FJ	20.1	5.76	7.83×10^{-4}	20.1	5.79	7.43×10^{-3}
GD	16.8	4.82	7.81×10^{-4}	16.9	4.85	7.43×10^{-3}
GS	26.4	7.57	7.88×10^{-4}	26.4	7.60	7.43×10^{-3}
GX	25.8	7.40	7.87×10^{-4}	25.8	7.43	7.43×10^{-3}
GZ	31.3	8.99	7.91×10^{-4}	31.4	9.02	7.44×10^{-3}
HA	19.8	5.68	7.83×10^{-4}	19.8	5.71	7.43×10^{-3}
HB	26.3	7.55	7.88×10^{-4}	26.4	7.58	7.43×10^{-3}
HE	10.2	2.93	7.76×10^{-4}	10.2	2.96	7.42×10^{-3}
HI	14.2	4.08	7.79×10^{-4}	14.3	4.11	7.42×10^{-3}
HL	10.9	3.11	7.77×10^{-4}	10.9	3.14	7.42×10^{-3}
HN	27.0	7.74	7.88×10^{-4}	27.0	7.77	7.43×10^{-3}
JL	10.7	3.06	7.77×10^{-4}	10.7	3.09	7.42×10^{-3}
JS	20.2	5.80	7.83×10^{-4}	20.3	5.83	7.43×10^{-3}
JX	23.0	6.61	7.85×10^{-4}	23.1	6.63	7.43×10^{-3}
LN	16.4	4.70	7.81×10^{-4}	16.4	4.73	7.43×10^{-3}
NM	11.0	3.15	7.77×10^{-4}	11.0	3.18	7.42×10^{-3}
NX	25.7	7.37	7.87×10^{-4}	25.7	7.40	7.43×10^{-3}
QH	24.5	7.04	7.87×10^{-4}	24.6	7.07	7.43×10^{-3}
SC	17.4	4.98	7.81×10^{-4}	17.4	5.01	7.43×10^{-3}
SD	12.4	3.57	7.78×10^{-4}	12.5	3.60	7.42×10^{-3}
SH	18.5	5.31	7.82×10^{-4}	18.6	5.34	7.43×10^{-3}
SN	21.8	6.26	7.85×10^{-4}	21.8	6.28	7.43×10^{-3}
SX	11.5	3.31	7.77×10^{-4}	11.6	3.34	7.42×10^{-3}
TJ	12.5	3.59	7.78×10^{-4}	12.5	3.62	7.42×10^{-3}
XJ	21.6	6.21	7.84×10^{-4}	21.7	6.24	7.43×10^{-3}
YN	27.8	7.98	7.89×10^{-4}	27.9	8.01	7.43×10^{-3}
ZJ	17.5	5.03	7.82×10^{-4}	17.6	5.05	7.43×10^{-3}

APPENDIX C: Additional Figures

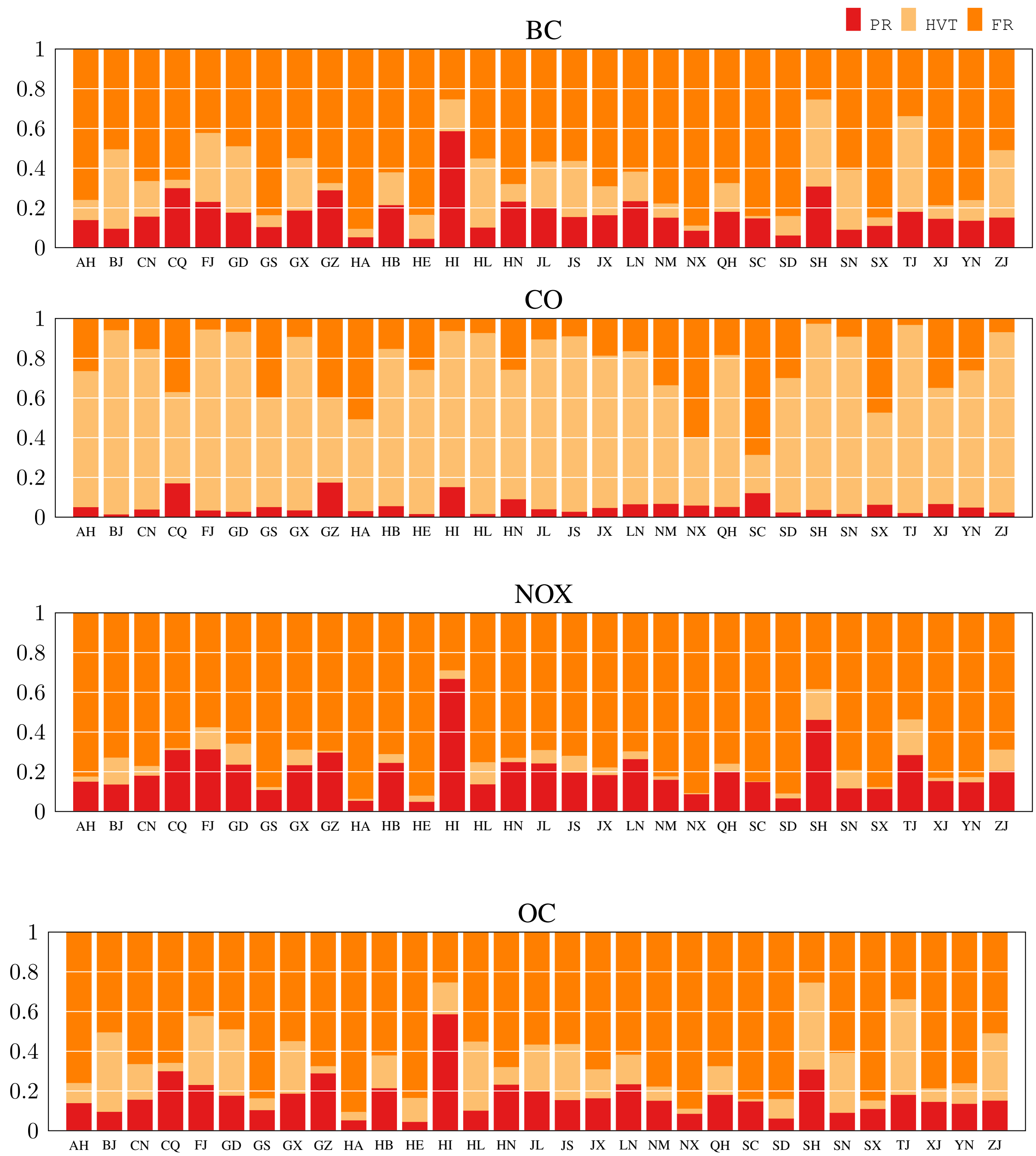


Figure C1. Contribution of each road transport mode to the total reduction of road transport emissions in 2030 due to stringent ES (Scenario B→C), by province.

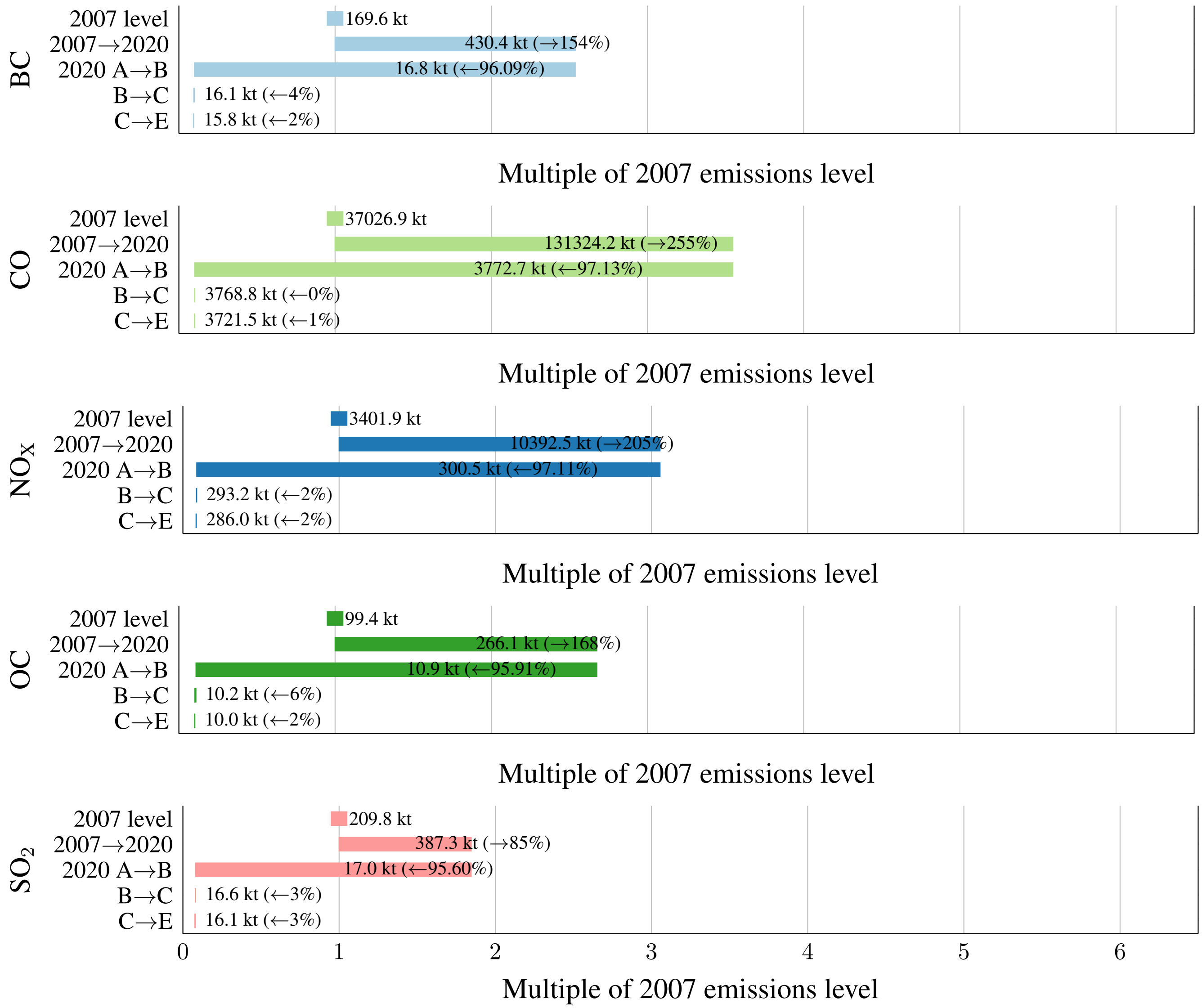


Figure C2. Changes in road transport emissions of five species: increase from 2007–2020 in Scenario A (no policy); and reductions in 2020 from Scenario A–B (introducing China 3/III and a mild CO_2 price), B–C (increasing ES stringency to China 6/VI), and C–E (increasing CO_2 price). The 2007 level is also shown, for reference. Annotations give the total emissions in each scenario and percent change in each year/scenario compared to the bar above.

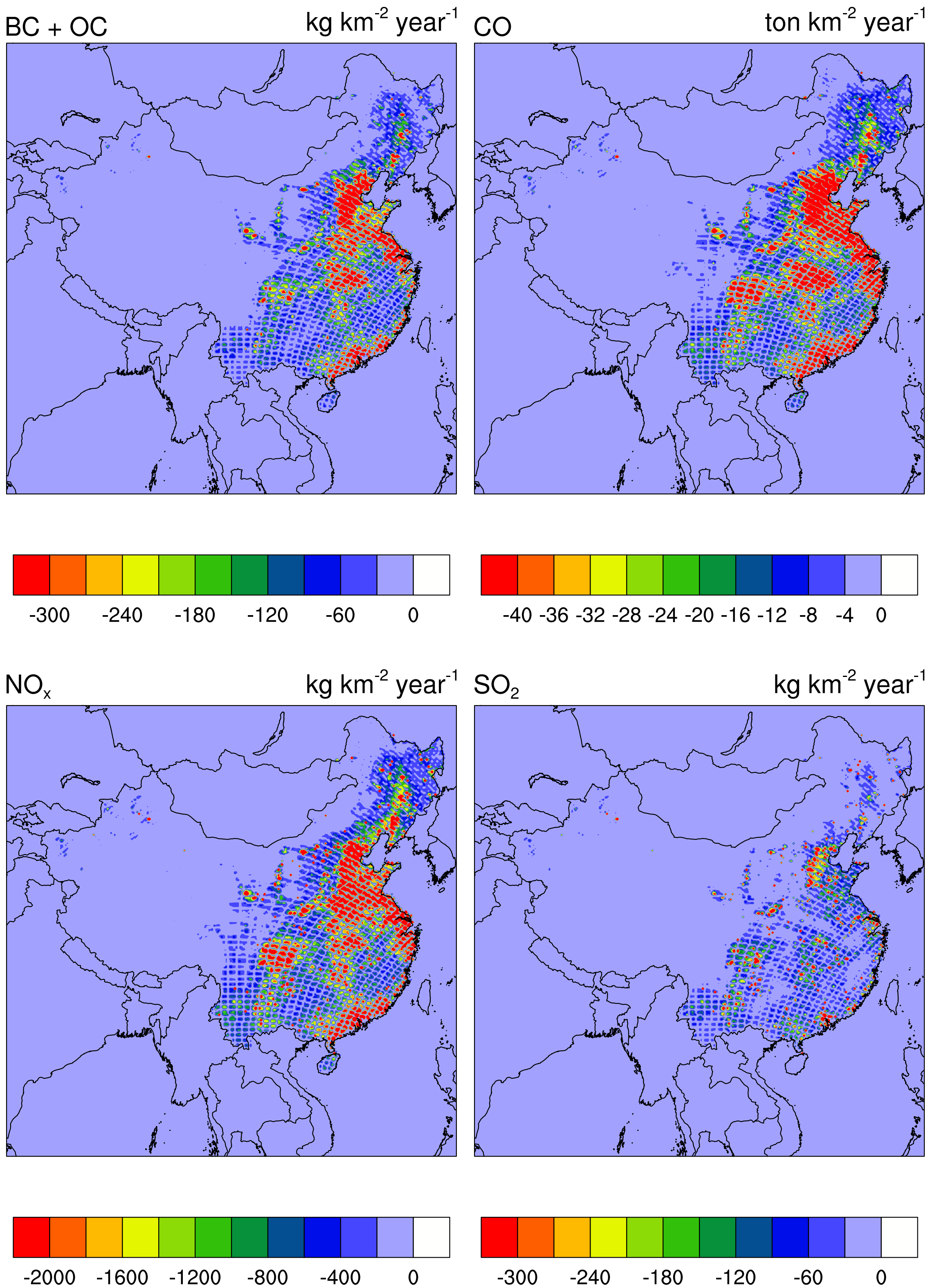


Figure C3. Spatial emissions differences of BC+OC, CO, NO_X, and SO_2 in 2030 between China 3/III (Scenario B) and no China 3/III (Scenario A′).

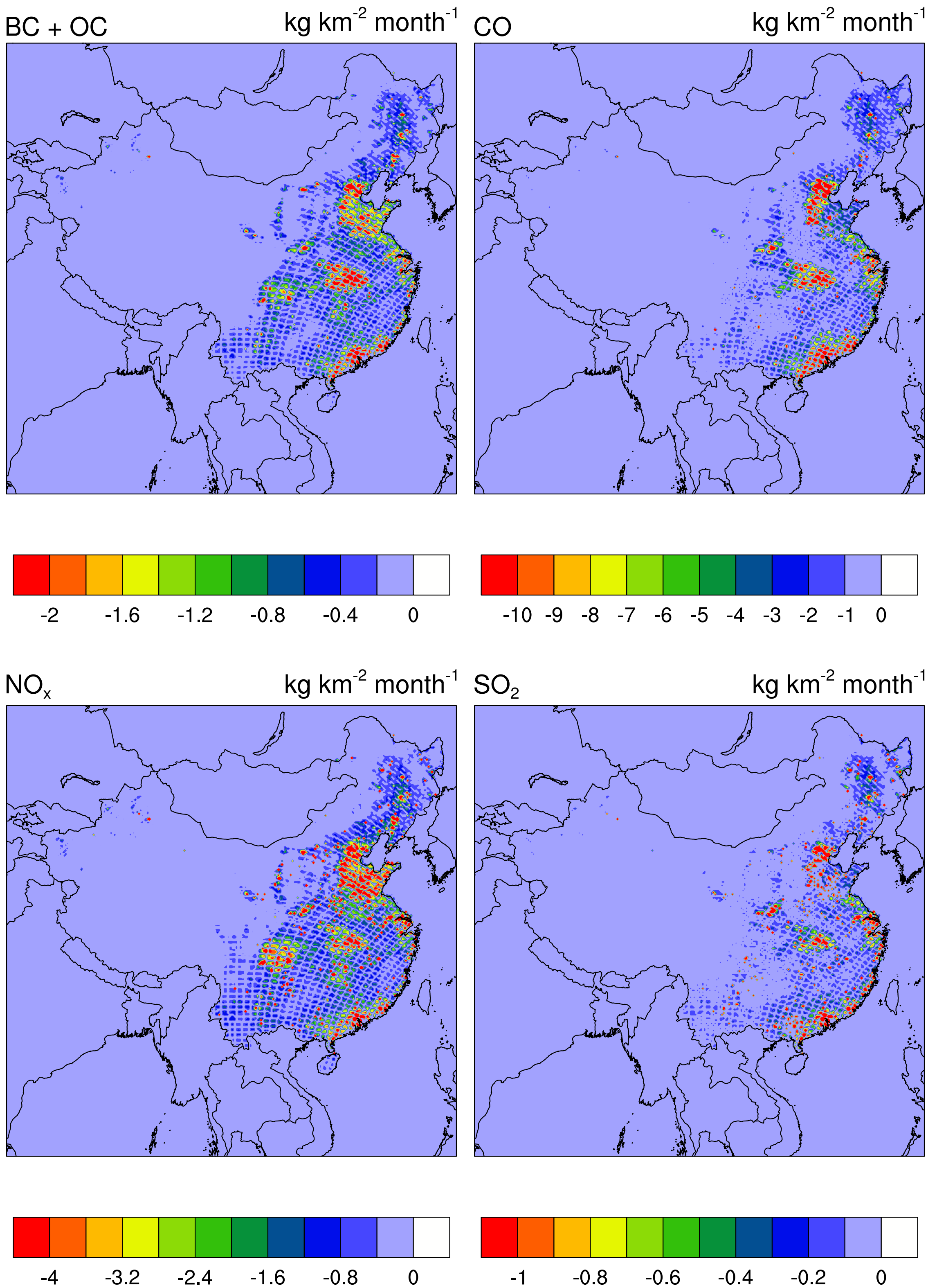


Figure C4. Spatial emissions differences of BC+OC, CO, NO_X, and SO_2 in 2030 between China 3/III (Scenario B) and China 6/VI (Scenario C).

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