Climate, health, agricultural and economic impacts of tighter on-road vehicle-emission standards

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Emissions Scenarios

Baseline emissions are based on currently proposed or adopted sub-national, national and international regulations. No additional changes in policies are assumed in the baseline future. Tighter standards are based on expert judgment of the local financial, technical and institutional capacity for adopting stricter regulations. Note that tight standards in a region does not necessarily mean in every country. For example, standards are already projected to be fairly stringent in Southern Africa, so the baseline and tight standards emissions are identical for South Africa, Namibia, Mozambique and Zimbabwe though they differ for countries further north. Emissions examples for several key compounds are shown in Figure S1. Values in each region were distributed onto a regular grid using the spatial distribution of current vehicle emissions. Fuel standards that accompany the emissions standards were also included, which has a substantial effect on sulfur dioxide emissions.

The tight standards clearly reduce emissions after 2015 relative to the baseline (Figure S1). Reductions in emissions from the developed world take place in either scenario, but developing nations’ emissions are greatly reduced when the tighter standards are imposed. The increase after 2030 in the tight standards scenario is largely driven by growth in emissions from Asian countries other than China, India, Korea and Japan (“other Asia”) for BC, and in other Asia, Africa and the Middle East for NOx.

A more detailed example of the emission trends by fuel type is shown for China in Figure S2. Under the tight-standards scenario, particle filters are installed on diesel road vehicles during 2015-2030. However, prior to installation of such filters, fuel desulfurization is required. Thus the overall trends for diesel show drastic decreases in SO2 emissions from 2015 onwards, followed by filter application leading to dramatic decreases in BC and NOx (both relative to the baseline and over time). In contrast, the tight standards scenario for gasoline vehicles shows little change in NOx with time (due to offsetting influence of fleet growth and improving emissions controls), but substantial decreases in SO2, CO and BC emissions (the latter are primarily from motorcycles). Note that even a steady rate of emissions, as for NOx from gasoline vehicles in China in the tight standards scenario from 2000-2030, does not imply that the technologies are ineffective as vehicle usage is projected to increase substantially in the future.

Some scenarios proposed for future emissions from on-road transport assume that nearly all non-CO2 air pollution emissions will be eliminated over the next few decades. Our judgment is that it is not guaranteed that emissions will follow such an optimistic assumption, especially in Asian countries other than Japan, Korea, China and India (our ‘other Asia’) where emissions increase substantially from 2030 to 2050 even in our tight standards scenario (Figure S1). For a country such as Pakistan, for example, it seems...
quite plausible that issues other than air pollution will continue to be the priority of the government. Hence our baseline scenario, with no additional tightening of standards beyond those currently proposed or adopted, is in some sense near the upper limit on future emissions. Even then, however, actual emissions could be higher as regulations can be delayed or not fully implemented. Our two scenarios thus provide a comparison between no additional tightening of standards and the most aggressive near-term tightening we judged to be practical. Note the implementation of the tight standards is quite aggressive, with China and India, for example, going from Euro 4 currently in place in many cities straight to Euro 6. However, vehicle manufacturers in those countries can already produce vehicles that can meet the tighter standards. Our scenarios primarily provide useful insight into emissions changes during the next couple of decades, and the climate response over the next few decades given the system’s slow response time, while further in the future emissions projections of course become less and less reliable.

Figure S1. NO\textsubscript{x} (top row, Tg N), black carbon (center row, Tg C), and SO\textsubscript{2} (bottom row, Tg SO\textsubscript{2}) emissions from on-road vehicles by region for the baseline (left column) and tight standards scenarios (right column). OECD is Organization for Economic Cooperation and Development, EU is European Union, FSU is Former Soviet Union.
**Composition-Climate Modeling Methods**

The GISS model for Physical Understanding of Composition-Climate INteractions and Impacts (G-PUCCINI) incorporates gas-phase, sulfate, primary carbonaceous aerosols, nitrate and secondary organic aerosol chemistry within the GISS ModelE general circulation model. Evaluations of the present-day composition in the model against observations are generally quite reasonable (as documented in the references given above as well as in, e.g., 5,9,10). The aerosol optical depths and radiative forcing per unit burden change in this model have been discussed and compared with other models previously.

For these simulations, we have used the development version of the model near its “frozen” state for AR5 simulations. The model has a horizontal resolution of 2° latitude by 2.5° longitude, with increased effective resolution for tracers due to carrying higher order moments at each grid box. Tracer transport uses a non-diffusive quadratic upstream scheme. The configuration used here had 40 vertical hybrid sigma layers from the surface to 0.1 hPa. Most simulations were performed in a non-interactive mode (where composition changes do not affect climate) using observed 2000-era sea-surface temperatures, with runs extended for two years following several months initialization.

**Figure S2.** Total gasoline (left column) and diesel (right column) on-road vehicle emissions in China under the baseline (top row) and tight-standards (bottom row) scenarios (Tg CO, Tg N in NOx, Tg C in BC and Tg SO2/yr).
Additional simulations using the same atmospheric model but coupled to a mixed-layer ocean model were carried out in interactive mode for 50 years.

Direct radiative forcings for ozone and aerosols are calculated in the climate model. We use instantaneous forcings at the tropopause. Adjusted forcing, allowing stratospheric temperatures to respond, would be ~10% less for ozone and nearly identical for aerosols. We include an estimate of aerosol indirect effects (AIE) for compounds other than BC following the assumption that they are equal to the direct effect from sulfate aerosols. Calculations based on detailed modeling and observations suggest that the ratio of AIE to direct sulfate RF may be 1.5 to 2.0 \(^{15}\). We use a lower value of 1.0 because recent analyses based on satellite data suggest that at least a portion of the AIE may in fact be fairly weak \(^{16}\), while models that now include aerosol effects on mixed-phase clouds tend to show smaller indirect forcings than earlier models (Isaksen et al., 2009). We include an uncertainty of 66% on the reflective aerosols OC, nitrate and sulfate, for the latter both on the direct and indirect effects (for OC and nitrate assuming their impact is only direct effects), based on a recent assessment \(^{17}\).

For BC, the direct forcing in our model is 0.36 W/m\(^2\) from preindustrial to present. This is well within the 0.20-0.55 W/m\(^2\) range found in a summary of 18 model results \(^{12}\) (excluding a model with a very coarse horizontal resolution of 22.5 x 10 degrees), but is lower than that estimated from two studies in which model BC fields were scaled to match observations of aerosol optical absorption that found larger values of ~0.6 W/m\(^2\) \(^{18}\) and 0.4-1.2 W/m\(^2\) \(^{19}\) (though this value is the total BC forcing rather than preindustrial to present) or in some additional model studies (e.g. \(^{20}\)). The GISS model, like most others, underestimates absorbing aerosol optical depths relative to satellite or ground-based observations in at least some areas, suggesting that the higher values obtained in the semi-empirical studies may be reasonable. We use the range 0.3-0.6 W/m\(^2\) as the most probable, and scale our BC direct forcing values by 0.45/0.36 for the central value, and by 0.3/0.36 and 0.6/0.36 for the range. In other words, we calculate the fraction of anthropogenic BC forcing change in our scenarios, and multiply that by the estimated central value and range of anthropogenic BC forcing from the literature. Note that the positive scaling for even the central estimate is consistent with the underestimate of AAOD in the model in some regions noted previously. We assume the combined effect of the semi-direct and indirect effects of BC (on all phases of clouds) is from -0.4 to +0.4 W/m\(^2\) from preindustrial to present, based on the assessments of Isaksen et al., 2009 and \(^{21}\), and hence add an uncertainty of ±0.4/0.32 times the direct forcing calculated in our model (i.e. central value is assumed to be zero forcing). We include BC’s effect on snow and ice albedo \(^{22}\) as an ‘effective forcing’ equal to 0.05 to 0.25 W/m2 preindustrial to present based on the published forcings and responses \(^{23,24}\), but rather than base this on the direct forcing we use the change in BC deposition calculated in the GISS simulations relative to the preindustrial to present-day BC deposition change (i.e. we use the fractional deposition change, as for forcing). Finally, given that there are constraints on the total aerosol forcing (e.g. \(^{25-27}\)), we limit the total value of BC direct plus indirect effects (including albedo) to a maximum of 1 W/m\(^2\). This leads to an asymmetry in the forcing uncertainty.

For consistency in the treatment of different pollutants, we adopt a similar methodology for organic carbon, scaling the model forcing to match the central value and range in \(^{12}\) of -0.20±0.11 W/m\(^2\) for the anthropogenic component (vs -0.10 in the model).
and for ozone, scaling to match 0.35±0.10 W/m² as in 28 (removing the very high values 
noted there from 0.45 to 0.65 W/m² as being inconsistent with recent satellite data such 
as 9) (though there is in practice no scaling of the central value as the ozone forcing in the 
model is also 0.35, so this only provides the uncertainty range). No adjustments were 
applied to sulfate or nitrate forcings, as these were very similar to the central range from 
the IPCC AR4 at -0.29 W/m² and -0.10 W/m², respectively. We assume that aerosol 
forcings are not independent given constraints on their total, and hence sum these based 
on their absolute value to obtain upper and lower bounds. The total aerosol forcing and 
all other forcing uncertainties are assumed to be independent, so the larger and smaller 
values are summed to derive upper and lower bounds, respectively (rather than summing 
in quadrature, which assumes probability with the range is normally distributed, and 
which we believe would lead to unrealistically small uncertainties).

A recent study that examined the effects of emissions of BC and OC from diesel with 
a very sophisticated aerosol microphysical model found that the combined semi-direct 
and indirect forcing of diesel BC+OC was equal to the opposite of the direct forcing from 
BC 29, consistent with the BC-portion being well within the range given above. We point 
out that the AIE are not necessarily co-located with the aerosol direct forcing, and further 
work is clearly needed to better characterize the spatial pattern of AIE as well as their 
magnitude. For most regions, however, the sulfate forcing is a small component of the 
total, and so this contributes relatively little to the net forcing. Additionally, the 66% 
uncertainty on the sulfate AIE plus the additional range included for BC’s semi-direct 
and indirect effects is consistent with our relatively low confidence in regional AIE.

We examined the radiative forcing from changes in the long-lived species N₂O, 
methane and HFCs. The calculations included an offline estimate of the methane 
response to changes in modeled oxidants and of the slow response of ozone to the 
decadal timescale changes in methane. Radiative forceings from methane and N₂O were 
calculated using the standard IPCC TAR formulation 30, while forcing from HFCs were 
estimated based on the results of Velders et al (2009). Forcings from all these species 
were very small (<0.01 W/m²), so henceforth we neglect these molecules. Emissions of 
CO₂ may change slightly following application of diesel particle filters, but the changes 
are expected to be very small with some studies showing no statistically significant ‘fuel 
penalty’ at all attributable to the filters 31.

The simulations performed with the composition-climate model are shown in Table 
S1. The response to regional emissions changes was examined for several areas (Table 
S1) through simulations where the emissions change were applied only within the regions 
shown in Figure S3. Regional analyses used those same regions as well as the additional 
regions defined in Figure S3.

The assumption that results for 2010 to 2030 or 2030 tight standards versus 2030 
baseline can be derived by scaling the 2000 to 2030 tight standards results is justified by 
the fact that in the two regional cases modeled explicitly (diesel and gasoline), the BC 
aerosol forcing per unit emission was nearly linear (typically within a few percent). For 
ozone, an additional pair of simulation altering CO and NOₓ emissions separately 
confirms that the response per unit change of each of these precursors is also roughly 
linear for a given region, a result consistent with prior global studies of the response to 
NOₓ emissions (e.g. 32). We therefore used both the diesel and gasoline simulations to 
calculate the ozone forcing per unit CO and NOₓ emissions (having two simulations
allows us to solve for two unknowns: the regional responses per unit CO and NO\textsubscript{x} emission), and apply those responses to estimate the ozone forcing for other times/scenarios. Results were not as linear for other aerosols (using SO\textsubscript{2} for sulfate, OC for OC, and NO\textsubscript{x} for nitrate), but their forcings are fairly small and hence deviations from linearity have little effect on the net forcing and contribute less to uncertainty than the indirect impacts of BC, for example. Our regional responses per unit emission can be compared with other results by evaluating them in terms of the widely used 100-year global warming potential (GWP), the integral of the radiative forcing due to 1 kg emission over 100 years in comparison with the integrated forcing due to 1 kg of CO\textsubscript{2} emission. The results for the responses that are linear are in general reasonably consistent with other studies, though for all but BC the literature is quite limited, while for BC the range in the literature is quite large (Table S2). Our BC GWPs are generally on the high-side of the published range in other regions, which likely reflects our incorporation of the 50% added absorption due to BC coatings by other aerosols. The GWPs for Indian and North Africa/Middle East BC emissions are higher than that found in other studies, though the other values lie well within our uncertainty range.

In our evaluation of the impacts of emissions from one region on another, the rest-of-world influence is simply the difference between the response in a given region in the worldwide 2030 tight standards experiment and the sum of the response to all the localized diesel and gasoline emission changes. This again assumes linearity in the responses, but as discussed previously, this appears to be a reasonable assumption for most cases.

We estimate the surface temperature response to the calculated radiative forcings using simple analytic equations following the methodology used in calculation of global temperature potentials\textsuperscript{33}. This is further extended to regional temperatures following the method described in\textsuperscript{34}. In essence, we use prior results from simulations with the Hadley Centre and GISS climate models to obtain an approximation of global and regional responses by multiplying the calculated RF by the global or regional transient sensitivity and accounting for ocean inertia. While many simple global energy balance models exist, our calculations allow estimation of regional responses including the influence of both local and remote forcings using the results of\textsuperscript{35} and the spatial patterns of forcing calculated in the simulations performed here. For both global and regional temperature responses, the calculation includes the relatively rapid response of the land and upper ocean and the slower response of the deep ocean as reported for simulations with the Hadley Centre climate model\textsuperscript{36}, with absolute responses scaled by 0.86 to match the transient climate sensitivity of the GISS model (0.53 C per W/m\textsuperscript{2} for increasing greenhouse gases) for consistency with the other model results used here (and as the sensitivity in the simulations used to derive the responses was high even compared with standard Hadley Centre simulations). Forcings were calculated for each 5-year increment in the emissions inventory out to 2050, then linearly interpolated. We assumed that forcing remained constant after 2050. Calculations were performed for each latitude band and for the global mean. Uncertainties were derived by adding the forcing uncertainty in quadrature with the uncertainty in climate sensitivity, where for the latter we use the 2 to 4.5 C range about a central estimate of 3 C for a doubling of CO\textsubscript{2} given in\textsuperscript{37} and deemed there to represent the 67% confidence interval for climate sensitivity (corresponding to a transient climate response of 1.3 to 3 C for a doubling of CO\textsubscript{2}). It is very difficult to
assign a quantitative range to the forcing uncertainty, as some factors (e.g. ozone forcing) are comparatively well-known while others (e.g. AIE, vertical profile of BC) are not as well constrained. It is our opinion that the forcing uncertainty range used here represents roughly a standard deviation (based on the ranges used for the least well-constrained components), and hence is consistent with the climate sensitivity range.

We tested this methodology by comparing the estimated response to the forcing calculated in the diesel India case with the results of the 50-year coupled atmosphere/mixed-layer model. Note that the climate model includes direct, semi-direct, indirect and snow/ice albedo forcings from aerosols, though of course these are but a single representation of poorly understood processes. The global forcing from diesel India 2030 tight standards is 15 mW/m² relative to 2000 including our offline estimates of AIE, semi-direct/indirect and snow/ice albedo forcings. The global response calculated using the temperature potentials is 0.008±.004 C averaged over years 20-50 after this forcing is imposed (.002±.001 SHex, .011±.005 tropics, .014±.009 NHml, .010±.005 Arctic). In the full climate model simulation carried out for this case (whose total forcing was not diagnosed as analysis of AIE forcing is very computationally expensive), the average over the years 20-50 after the forcing was imposed is 0.011±.010 C globally. Regional results were not statistically significant in the climate model, where internal, unforced variability makes the signal-to-noise ratio especially low for smaller spatial scales. Hence while the significance of even the global mean climate model results is low, the method appears to at least not greatly disagree with the full climate model simulation. The results imply, however, that detection of the benefits of climate change mitigation will require substantially larger changes in radiative forcing than those that can be achieved by targeting emissions from a single sector in a single region.

Table S1. Simulations performed

<table>
<thead>
<tr>
<th>Run name and year</th>
<th>Region of emissions change</th>
<th>Fuel types for which emissions changed</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Gasoline-only</td>
</tr>
<tr>
<td>2000 Control</td>
<td>None</td>
<td></td>
</tr>
<tr>
<td>2030 Baseline</td>
<td>Worldwide</td>
<td></td>
</tr>
<tr>
<td>2030 Tight Standards</td>
<td>Worldwide</td>
<td></td>
</tr>
<tr>
<td>2030 N. America</td>
<td>N. America</td>
<td>X</td>
</tr>
<tr>
<td>2030 W. Europe</td>
<td>W. Europe</td>
<td>X</td>
</tr>
<tr>
<td>2030 Tight India</td>
<td>India</td>
<td>X</td>
</tr>
<tr>
<td>2030 Tight China</td>
<td>China</td>
<td>X</td>
</tr>
<tr>
<td>2030 Tight NA/ME</td>
<td>NA/ME</td>
<td>X</td>
</tr>
</tbody>
</table>

Regional simulations were performed for gasoline and diesel emission changes separately. Simulations for N. America and W. Europe are not identified by scenario as baseline and tight standards were identical in those areas. NA/ME is North Africa/Middle East.
### Table S2. 100-year GWPs for regional emissions

<table>
<thead>
<tr>
<th>Region</th>
<th>CO</th>
<th>NOx</th>
<th>BC total (direct only)</th>
</tr>
</thead>
<tbody>
<tr>
<td>This work</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Europe</td>
<td>1.3</td>
<td>2.3</td>
<td>1600 (630)</td>
</tr>
<tr>
<td>North America</td>
<td>1.1</td>
<td>4.4</td>
<td>1350 (580)</td>
</tr>
<tr>
<td>N. Africa/ME</td>
<td>1.3</td>
<td>3.5</td>
<td>1870 (1340)</td>
</tr>
<tr>
<td>China</td>
<td>1.4</td>
<td>8.6</td>
<td>1690 (960)</td>
</tr>
<tr>
<td>India</td>
<td>3.4</td>
<td>31.3</td>
<td>2280 (2010)</td>
</tr>
<tr>
<td>Latin America**</td>
<td>0.8</td>
<td>14.5</td>
<td>1550 (1550)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other studies</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Europe</td>
<td>-2.7 to +4.1</td>
<td>(510, 380, 430, 720)</td>
<td></td>
</tr>
<tr>
<td>North America</td>
<td></td>
<td>(550, 920, 430, 720)</td>
<td></td>
</tr>
<tr>
<td>N. Africa/ME</td>
<td></td>
<td>(340, 1300, 740, 1100)</td>
<td></td>
</tr>
<tr>
<td>China</td>
<td>25 to 29</td>
<td>(960, 420, 340, 710)</td>
<td></td>
</tr>
<tr>
<td>India</td>
<td>25 to 29</td>
<td>(920, 1400, 640, 1040)</td>
<td></td>
</tr>
<tr>
<td>Latin America</td>
<td></td>
<td>(610, 1400, 550, 620, 940)</td>
<td></td>
</tr>
</tbody>
</table>

a=5, b=38, c=39, d=40, e=41, f=42. GWPs for CO and NOx include only the ozone response (direct plus methane-induced). Total GWPs for BC include the effective forcing from BC deposition (i.e. account for the greater efficacy of this forcing). Values quoted for N. Africa/ME from Koch et al and Naik et al are actually for Africa as a whole. These are very similar in Reddy and Boucher to the values for the ME (720 vs 740). *Values listed for China and India from Berntsen et al are for SE Asia. Literature values include only direct effects for aerosols. **Results for Latin America are based on the regional emissions and forcing in the worldwide emissions change simulations since remote emissions contribute <2% of forcing there (unlike other regions (Table 2)), and do not include influence of BC deposition changes.
Health Impact Analysis Methods

Surface composition simulated in the G-PUCCINI model was used to drive the analyses of premature deaths due to outdoor air pollution. For ozone, we calculated health impacts using ozone at the GCM’s native 2 x 2.5 degree resolution, but using population data at 0.5 x 0.5 degrees. We did not apply a sub-grid correction for urban titration effects (urban ozone decrement). As BC and OC have short lifetimes and their concentrations are driven by primary emissions, their abundances are expected to be closely related to population density. Therefore these components of the PM2.5 distributions were downscaled to 0.5 x 0.5 degrees using a subgrid parameterization of urban/rural differences developed at the Joint Research Center. The parameterization is based on a population-based re-distribution of the primary PM2.5 concentration (BC and OC) on a 0.1°x0.1° degree sub-grid, within each GCM surface grid cell. Population data at the subgrid resolution from the Gridded Population of the World Version 3 (Center for International Earth Science Information Network, Columbia University, http://sedac.ciesin.columbia.edu/gpw, 2005) are used to determine the urban area fraction $f_{ua}$ and urban population fraction $f_{up}$ within each native grid cell, using a threshold population density of 600 persons per km². We then make the simple assumption that the primary rural BC concentration rescales to $(1- f_{up})/(1-f_{ua}) \times BC_{GCM}$ and the urban concentration to $f_{up}/f_{ua} \times BC_{GCM}$, which preserves the larger GCM grid box average. This assumption implies that the urban and rural emissions are mixing and diluting at the same rate. In the calculations performed here, we impose a minimum value on the rural BC concentration of one-half the GCM concentration, and then set a maximum on the urban concentration of 5 times the rural value. The 0.1 x 0.1 degree population and exposure weighted results are then aggregated to an 0.5 x 0.5 degree grid for the health analyses.

We evaluate the modeled PM2.5 against a large set of PM observations assembled in support of the Global Burden of Disease Study (GBD 2010,
The data consist of a worldwide set of annual average PM2.5 values largely drawn from official monitoring networks for 2005 (in some cases for 2004-2006). For a number of sites, PM2.5 was not measured directly but was estimated based on PM10 measurements using local PM2.5:PM10 ratios where available and a default value of 0.5 elsewhere. For North America and Europe, where more data is available, we do not use the PM2.5 values inferred from PM10. For China and India, due to a lack of measurements, we use both direct PM2.5 observations and PM2.5 inferred from PM10 (although the latter can differ substantially from the actual PM2.5 in places where both are measured). Water uptake on aerosols is accounted for using 35% relative humidity (RH) for North America and 50% for Europe following the measurement protocols (filter equilibration conditions) in place in those areas. We also assume 50% RH for China and India, though there may be less adherence to that protocol in those areas. The model values are generally in reasonable agreement with the observations, especially over North America and Europe (Figure S4). There is more scatter over China and India where the indirect PM10 data has been included. Correlations do not appear systematically biased in any of the regions other than China, where the model typically underestimates PM2.5. Note that without the urban/rural downscaling, there is a much larger low bias over China and substantial low biases in other regions as well. There may also be a low bias over India, but it is difficult to assess with such a limited dataset. Overall, our mortality results may be somewhat conservative given the low biases over the world’s most populous regions.

The health impact calculations follow established epidemiological concentration-response functions. For PM2.5, we use relative risk estimates from \(^3\), the latest reanalysis of the American Cancer Society (ACS) Study. For a 10 µg/m\(^3\) increase in annual average PM\(_{2.5}\), relative risks of death due to cardiopulmonary disease and lung cancer are 1.13 (95% confidence interval [CI]: 1.10-1.16) and 1.14 (95% CI: 1.06-1.23). For ozone, we use long-term relative risk estimates from the two-pollutant model calculated by \(^4\), also based on the ACS cohort. For a 10 ppb increase in seasonal (6 month) average 1-hr daily max ozone, relative risk of respiratory mortality is 1.04 (95% CI: 1.013-1.067). Consistent with the ACS cohort, we include only the population over 30 years of age, and for the 2030 calculations include projected changes in population (amount and distribution) following a conservative B2 scenario \(^5\) (global population increases from 6.5 billion in 2006 to 8.4 billion in 2030). The calculations also include worldwide variations in underlying baseline mortality, as described in Anenberg et al. (2010). While mortality rates are expected to change over the next several decades, projected mortality rates for future years are not available, hence current incidence rates were used to estimate 2030 incidence.

Uncertainties in mortality calculations include only the 95% CI in the concentration-response function derived from epidemiological studies. Assessing the global health impacts of air pollution is subject to a variety of other uncertainties that are not easily quantifiable. We do not explore sources of uncertainty associated with the underlying composition modeling, such as the concentration change in response to the emissions changes, and uncertainties in emissions changes due to application of emissions control technologies. These uncertainties could be important relative to the uncertainty in the concentration-response relationship, particularly for ozone \(^6\). In the absence of long-term air pollution mortality studies in the developing world, we assume the concentration-
response relationships found by Jerrett et al. (2009) and Krewski et al. (2009) in the US apply to the rest of the world, despite substantial differences in exposure levels, PM2.5 composition, and lifestyle. In particular, while the PM2.5 data used to develop the concentration-response relationships in Krewski et al. (2009) ranged as high as 30 ug/m$^3$, predicted PM2.5 concentrations in the 2030 baseline scenario ranged as high as 280 ug/m$^3$. Some evidence from indoor air pollution exposure and cigarette smoking suggests that the concentration-response relationship may flatten at high concentrations $^{47,48}$. However, since this relationship is not yet well characterized at high concentrations, we extrapolate relative risk estimates linearly from low concentrations to high concentrations. If the relationship flattens at high concentrations, these results would be overestimates. However, since the model may sometimes underestimate concentrations in urban areas in the developing world (Figure S4), these biases may offset one another to some extent. We also assume that all PM2.5 components and mixtures are equally toxic, despite wide variation in air pollution mixtures around the world. Though there is some evidence that some mixtures are more toxic than others (e.g. $^{49,50}$), we do not believe that the current available data is adequate to separate health impacts by component. While populations around the world vary in age structure, medical care, and exposure patterns, extrapolation of relative risk estimates found in the US is supported by general consistency among short-term epidemiology studies around the world for both ozone and PM2.5 $^{51,52}$. Since causes of death differ dramatically around the world, we calculate cause-specific mortality, which may be more comparable around the world than all-cause mortality.

A second set of mortality calculations, which we refer to as ‘Method 2’, was performed for the worldwide scenarios to test the robustness of the methodology. The second set of calculations only examined the effects of PM2.5 exposure, again based on $^{43}$. They differed in several respects from the first analyses. They use population from 45 to 79 years of age rather than all persons over 30 to avoid possible bias in the concentration-response function at age extremes. They also used a low concentration threshold of 5.8 ug/m$^3$ to avoid potential biases at low concentration. They attempt to capture the effect of projected changes in age structure by using age-resolved population projections by country from the US Census Bureau’s International Database (IDB; U.S. Census Bureau, Washington, D.C., http://www.census.gov/ipc/www/idb/), and baseline cardiopulmonary incidence rates for 2005 by region and age group from the World Health Organization’s Global Burden of Disease report $^{53}$. Age-resolved national population projections were merged with gridded population projections (CEISIN, 2005, http://sedac.ciesin.columbia.edu/gpw) to obtain age-resolved, gridded population for this method.

We compare the premature deaths calculated here using method 1 to the global burden of outdoor air pollution on premature mortality. In 2004, the World Health Organization estimated about 800,000 premature deaths due to urban PM2.5 $^{54}$. More recently, $^{55}$ estimated about 3.7 and 0.7 million premature deaths due to total anthropogenic PM$_{2.5}$ and O$_3$ using a similar methodology as that used here, except Anenberg et al. (2010) used 2006 population (about 6.5 billion) while the projected 2030 population (about 8.4 billion, ~30% higher) is used here. Analysis of the current modeled PM2.5 fields produces an estimate of 2.0 million premature cardiopulmonary and lung cancer deaths due to total anthropogenic PM$_{2.5}$ and O$_3$ using 2000 population, and 2.6
million using 2030 population. Hence these values are between the two prior estimates. The global avoided PM2.5 deaths calculated for the 2030 baseline scenario and tight standards scenario relative to the 2005 baseline are 2.8% and 9.2% of the global burden of urban PM$_{2.5}$ calculated here, corresponding to 6.8% and 22.6% of the burden from Cohen et al. (2004) and 1.5% and 4.9% of the burden from Anenberg et al. (2010). The global excess premature ozone deaths for the 2030 baseline scenario and avoided premature ozone deaths for the 2030 tight standards scenario are 4.5% and 1.6% of those calculated by Anenberg et al. (2010).
Figure S4. Observed and modeled PM2.5 concentrations by region. The modeled values in the top two rows include the urban/rural downscaling discussed in the text (see text for discussion of data sources and limitations).
Crop Yield Impact Analysis Methods

The impact of ozone on agricultural crop yields was calculated using established methods. The ozone metrics were calculated for the crop-specific growing season (with a standard duration of 3 months) depending on location. The methodology for the definition of the growing season makes use of multiple data sources, including temperature data, reports describing the growing season for major crops, and location of different climate zones, as described in Van Dingenen et al., 2009.

To characterize uncertainties, we use the range from calculations performed using two different exposure indicators: the seasonal mean daytime ozone concentration (indicated as M7 for the 7 h mean (09:00–15:59) or M12 for the 12 h mean (08:00–19:59)), and the accumulated daytime hourly ozone concentration above a threshold of 40 ppbv (AOT40). The M7/M12 metrics are means over the growing season, while AOT40 is a cumulative index, calculated over the same period as for the M indices. We use M7 or M12 and AOT40 as exposure–response functions are available from the literature for all four of the crops we analyze. We consider M7 and M12 as one indicator type. The only reason for considering both is that the available exposure–response functions for wheat and rice are expressed as a function of M7 whereas for maize and soybean they are a function of M12. We note that the concentration-response relationships are ‘pooled’ based on a variety of cultivars that are grown in the US and Europe. They are considered to reliably represent the average response of the commonly grown cultivar population on national or regional level in those regions. Small-scale individual studies indicate that Asian cultivars for winter wheat and rice are equally or more sensitive to ozone damage than the US cultivars, hence our results applying the US-derived exposure–response relationship are likely to be on the conservative side.

As described in more detail in Van Dingenen et al. (2009), the spatial distribution of crops and their production numbers are calculated on a 1°x1° grid resolution, based on crop suitability indices for each of the crops considered. Crop suitability grid maps are taken from. In particular, national production numbers (obtained from the FAO) are distributed over 'suitable' crop production grid cells with a weight defined by the appropriate crop suitability index. The ozone metrics are calculated for each grid cell, hence crop production loss is obtained at grid cell resolution, and afterwards aggregated to national totals. All base crop production levels are present-day rather than projections for 2030.

Uncertainty also arises from the concentration-response function and from the atmospheric processes that transform emission changes into concentration changes. For the former, the response function is relatively well known for soybean and maize, but somewhat less so for wheat and is rather poorly constrained for rice (especially at the upper end). Hence inclusion of that uncertainty would have fairly small effects on the ranges shown for maize and soybean, but would lead to substantially larger ranges for wheat and especially for rice. For rice yields in particular, there is a possibility of very large losses. While only a single model was used here, additional work by our groups have compared the ozone-related crop yield changes due to particular emissions in multiple models. They find that differences between model results are typically within 30%. The uncertainty quantified here, due to multiple concentration-response functions
(i.e. use of different metrics), has only a small impact for rice (uncertain within a factor 1.2) and wheat (within a factor 1.4). The uncertainty range for maize is between a factor 1.8 lower and a factor 1.4 higher than the central value, whereas the relative yield loss for soy may vary between a factor 2.8 lower and 1.6 higher than the central value. Hence adding the additional uncertainties due to concentration-response functions and modeling atmospheric processes would expand the ranges reported here greatly for rice, and quite substantially for wheat, but would have little effect on the ranges shown for maize and soybean. Thus we do not believe it is possible to give a single quantitative estimate of the uncertainty range associated with the values presented here. Additionally, we note that these uncertainties are all systematic, so that while the absolute change in a particular scenario (e.g. Figure 1) has substantial uncertainty the difference between two scenarios can be better constrained. In particular, the large ranges for individual emissions scenarios do not imply that the effect of the emissions control measures is not statistically significant. Finally, we note that the effects of increasing CO2 on ozone-related crop damage have not been included here, but these may lessen the impact of future ozone-related crop yield losses and merit further analyses.

The results found here are compared with an earlier analysis of the effects of modeled year 2000 ozone on crops using the same methodology in Table S3 (though the models used different emissions). Relative yield losses are generally a bit larger in this study for all crops for the EU25 and North America, but range from slightly to substantially smaller for China and India. Comparisons of the modeled ozone with observations showed that the earlier modeling study had a large low bias in AOT40 over Europe, so the larger values found here are likely more realistic (Figure S5). Both our model and the earlier model generally did a good job over North America, though our AOT40 appears to have a low bias over the Southeast US (although there are only 5 stations). Our impacts are slightly higher than the earlier study for wheat and rice, but are typically very close for maize and soybean in that region. Little evaluation data is available for China and India, unfortunately. The earlier model exhibited a positive bias compared with monthly mean observations from 4 stations in Northern India while agreeing reasonably well over Southern India. The current model is slightly closer to the observations over Northern India (Figure S5), but still overpredicts mean ozone. Biases in M7 or M12 or AOT40 are not necessarily correlated with monthly means, however. As more field data becomes available, future work could also consider the potential effects of aerosols on agriculture via altered sunlight exposure.

Valuation of crop yields is based on year 2000 producer prices per country and per crop from the Food and Agriculture Organization (FAO: faostat.fao.org). Producer price is used as a proxy for domestic market price as the latter are not readily available. For a number of minor producing countries FAO does not provide producer prices. In that case, the global median price for the commodity was used. The fraction of global production for which no price information is available is limited to 2% for wheat and maize, 6% for rice and 0.4% for soybeans.
Table S3. Present-day regionally aggregated relative yield losses

<table>
<thead>
<tr>
<th></th>
<th>World</th>
<th>EU25</th>
<th>N. America</th>
<th>China</th>
<th>India</th>
</tr>
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<tbody>
<tr>
<td><strong>Wheat</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
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<td>4.1%</td>
<td>4.1%</td>
<td>19.0%</td>
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<td>AOT40, this study</td>
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<td>11.6%</td>
<td>6.2%</td>
<td>8.4%</td>
<td>8.4%</td>
</tr>
<tr>
<td>M7, V09</td>
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<td>4.6%</td>
<td>4.4%</td>
<td>9.8%</td>
<td>13.2%</td>
</tr>
<tr>
<td>M7, this study</td>
<td>4.9%</td>
<td>5.8%</td>
<td>4.4%</td>
<td>4.5%</td>
<td>5.0%</td>
</tr>
<tr>
<td><strong>Rice</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AOT40, V09</td>
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<td>4.7%</td>
<td>3.2%</td>
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<td>8.3%</td>
</tr>
<tr>
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<td>1.9%</td>
<td>5.4%</td>
<td>6.0%</td>
<td>2.2%</td>
<td>3.0%</td>
</tr>
<tr>
<td>M7, V09</td>
<td>2.8%</td>
<td>3.5%</td>
<td>2.6%</td>
<td>3.1%</td>
<td>5.7%</td>
</tr>
<tr>
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<td>1.6%</td>
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<td>2.0%</td>
<td>2.5%</td>
</tr>
<tr>
<td><strong>Maize (Corn)</strong></td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
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<td>3.1%</td>
<td>2.2%</td>
<td>4.7%</td>
<td>2.0%</td>
</tr>
<tr>
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<td>2.0%</td>
<td>4.7%</td>
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<td>2.1%</td>
<td>0.3%</td>
</tr>
<tr>
<td>M12, V09</td>
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<td>5.1%</td>
<td>3.6%</td>
<td>7.1%</td>
<td>4.0%</td>
</tr>
<tr>
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<td>4.3%</td>
<td>1.6%</td>
</tr>
<tr>
<td><strong>Soybean</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AOT40, V09</td>
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<td>20.5%</td>
<td>7.1%</td>
<td>11.4%</td>
<td>4.7%</td>
</tr>
<tr>
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<td>8.1%</td>
<td>8.0%</td>
<td>1.5%</td>
</tr>
<tr>
<td>M12, V09</td>
<td>15.6%</td>
<td>27.3%</td>
<td>17.7%</td>
<td>20.8%</td>
<td>19.1%</td>
</tr>
<tr>
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<td>33.4%</td>
<td>21.0%</td>
<td>19.7%</td>
<td>13.0%</td>
</tr>
</tbody>
</table>

V09 refers to Van Dingenen et al. (2009).

Figure S5. Comparison of year 2000 simulations (red) with observed AOT40 (ppm.h; blue) averaged over 38 stations in Central Europe (7-17E, 48-54N; top left), over 2
stations in the Southwest US (110-125W, 30-40N; top right), 5 stations in the Southeast US (80-90W, 25-35N; bottom left), and with monthly mean ozone for 4 stations in Northern India and Nepal (70-90E, 20-30N; bottom right). Observations from Van Dingenen et al (2009). Control run output is the average of 5 years of simulation.

Health Valuation Analysis Methods

There is substantial controversy over comparison of VSL in different countries. We have therefore presented analyses using both a uniform VSL, treating all human lives as equally important, and using a country-specific income-adjusted VSL that is more logically related to local WTP.

In addition to the methodology described in the main text, we elaborate here on the income-adjusted VSL calculations. In order to compute the VSL in a particular country that lacks a credible VSL estimate (VSL₁), the VSL in a country in which there is a reported VSL estimate (VSL₂), is multiplied by the ratio of per capita incomes (Income₁/Income²) raised to the power of the selected income elasticity (0.40, in this case): VSL₁ = VSL₂ * (Income₁/Income²)₀.₄₀. The income elasticity applied in this analysis (0.40) reflects that used by USEPA in analyses that require adjustments of estimated VSLs to projected changes in personal income. This approach was applied to estimate a country-specific VSL for each of the approximately 210 countries encompassed in the analysis. Note that the approach assuming that the VSL is equal across all countries in effect sets the elasticity to zero.

We also compare the value of emissions reductions on a per ton basis to other analyses that have examined policy scenarios which impact emission rates of the similar pollutants. While we cannot determine the impacts on mortality due to specific pollutants among those abated under the tight emissions controls, we report an average impact ($/ton) in order to gauge the results presented herein against other values reported in the literature despite the inherent limitation of valuing all pollutants simply by mass.

In this analysis, the tight emission standards result in a total reduction of 119 million tons of VOC, CO, NOₓ, SO₂, and PM. The gross mortality benefit ranges between $94 billion and $1.3 trillion when using the country-specific VSLs, and between $200 billion and $1.7 trillion when using the uniform VSLs. Simply dividing through by the number of tons abated yields a range, for the country-specific VSLs, of $800 to $10,800 per ton, while for the uniform VSLs the range is between $1,700 and $13,900 per ton. Note that the ranges are entirely from the range of the underlying change in premature mortalities.

In the literature, other papers have explored emission reductions across this same set of pollutants. The imputed average value of emission reductions in the USEPA’s First Prospective Section 812 Analysis 61 was $5,000 per ton. Note that this value has been adjusted to reflect equivalent VSLs to correct for different years of analysis. Chestnut and Mills 62 explore the benefits of NOₓ and SO₂ reductions due to Title IV of the U.S. Clean Air Act. The imputed benefit ($/ton) from their analysis is $16,400 (After adjusting the VSL to $9.5 million). While this value lies just outside of the range of values from the current study, given that Chestnut and Mills focus on just NOₓ and SO₂ this difference is not surprising. Muller and Mendelsohn 63 report emission weighted average benefits per ton for NOₓ, SO₂, PM₂.₅ and VOC. The combined grand average of these pollutants is


$9,600 (again, adjusting for $9.5 million VSL). This lies within the range of values reported in this study.

We also point out that there is substantial controversy over comparison of VSL in different countries. We have therefore presented analyses using both a uniform VSL, treating all human lives as equally important, and using a country-specific income-adjusted VSL that is more logically related to local willingness to pay. We also present the results in terms of benefits per capita by regions (Table S4). This shows that regardless of the approach taken to VSL estimates, the largest per capita valuation of avoided deaths due to the tightening of emissions standards takes place in North Africa/Middle East and India, but that benefits are greater than $450 per person (over 30 years of age) in all regions where the standards are tightened. We emphasize that the regional valuation is primarily of use for comparing costs and benefits of various policies within a region rather than for comparison of the value of avoided deaths between regions.

Table S4. Valuation of air-quality related deaths due to vehicle emissions in 2030 relative to 2000 in $US per person over 30.

<table>
<thead>
<tr>
<th>Region</th>
<th>Baseline scenario</th>
<th>Tight standards scenario</th>
<th>Tight standards minus baseline</th>
</tr>
</thead>
<tbody>
<tr>
<td>North America</td>
<td>-1050 (-654, -1410)</td>
<td>-1040 (-651, -1410)</td>
<td>-1060 (-664, -1450)</td>
</tr>
<tr>
<td>China</td>
<td>378 (537, 203)</td>
<td>232 (329, 125)</td>
<td>-480 (-310, -643)</td>
</tr>
<tr>
<td>India</td>
<td>987 (1500, 401)</td>
<td>600 (910, 244)</td>
<td>-256 (-222, -289)</td>
</tr>
<tr>
<td>Western Europe</td>
<td>-3830 (-2720, -4930)</td>
<td>-3200 (-2270, -4120)</td>
<td>-3860 (-2760, -4950)</td>
</tr>
<tr>
<td>Western FSU</td>
<td>-1330 (-941, -1710)</td>
<td>-859 (-607, -1100)</td>
<td>-1470 (-1040, -1890)</td>
</tr>
<tr>
<td>South &amp; Central Africa</td>
<td>651 (947, 317)</td>
<td>394 (573, 192)</td>
<td>-97.5 (-83.1, -115)</td>
</tr>
<tr>
<td>North Africa/ Middle East</td>
<td>676 (869, 475)</td>
<td>428 (551, 301)</td>
<td>-621 (-434, -804)</td>
</tr>
<tr>
<td>Latin America</td>
<td>92.2 (121, 59.4)</td>
<td>46.6 (61.2, 31.1)</td>
<td>-741 (-302, -1150)</td>
</tr>
</tbody>
</table>

Positive values indicate increase costs and deaths, negative indicate reduced costs and deaths. Income-adjusted valuation of mortality is based on country-specific income. Ranges are based on the uncertainties in the avoided deaths only.

Radiative Forcing Response

The full geographic distributions of radiative forcing and premature deaths are shown in Figure S6 for the baseline scenario and the tight standards versus baseline.
These show the increased forcing in the baseline scenario over Africa and the Middle East seen in the regional averages (Figure 1) and the increased baseline mortality in some regions. The spatial pattern of forcing from 2000 to 2030 under the tight standards scenario is shown in Figure S7. Global mean radiative forcing values are -64±42 and -115 (+47/-102) mW/m² for the 2030 baseline and tight standards, respectively, relative to 2000. Values for the difference between the tight standards and baseline (global and regional) are shown in Table S5.

Tighter standards lead to reductions in both reflective aerosols (primarily sulfate and OC) and BC, which offset one another to some extent. Over the bright desert surfaces of North Africa and the Middle East, the effect of sulfate is minimized as reflective aerosols have a similar albedo to the surface, while the effect of absorbing BC is enhanced as its contrast with the surface is larger than elsewhere. The sensitivity of these forcing to the surface albedo can be clearly seen in the variation of the radiative forcing due to tight standards between the deserts of the Sahara and Arabia, where it is strongly negative, and the adjacent Red Sea and Persian Gulf, where it is strongly positive (Figure S6), as noted in the main text. The impact of surface albedo on forcing by BC alone can be seen by examining the forcing per unit BC (Figure S8). This illustrates how the impact of BC is enhanced over both bright desert and snow and ice covered areas. The impact of sulfate is decreased in these same areas, but is relatively larger over areas with low surface albedo such as the open oceans.

Large baseline forcings are seen over many Arctic areas where bright snow and ice surfaces are present, even those such as Northern Scandinavia and the Russian Arctic, where emissions are small. Indeed this is part of the reason why forcing over the Western FSU, which is predominantly from BC reductions, is so large in the baseline scenario (Figure 1). Similarly, baseline forcings are large at the edges of the Himalayas/Hindu-Kush, consistent with the strong localized impacts of BC in that regions (Figure S7). Hence the cooling impact of vehicle emissions reductions is substantially enhanced over bright surfaces.

There are additional regional impacts from BC deposition on snow and ice surfaces leading to darkening and melting (as noted, these are included in the area averaged forcing results, and increase the benefit of tight standards in areas other than Africa and the Middle East). While BC deposition on deserts is unlikely to lead to substantial albedo change due to mixing, our results suggest that desert regions along with snow and ice covered regions are especially sensitive to direct BC forcing (and especially insensitive to forcing by reflective aerosols), with snow and ice covered areas also influence by BC deposition. Regional variations will be greatest for realistic scenarios that include simultaneous changes in absorbing and reflecting aerosols rather than idealized cases examining the effects of a single or limited subset of aerosol types (e.g. 29).

Regional variations arise from both differences in physical conditions (e.g. snow and ice, precipitation) and differences in the mix of emissions. The impact of tighter standards on Chinese diesel vehicles, for example, is roughly neutral (Figure 2) as reduced forcing from BC and ozone is offset by increased forcing due to reductions in reflective OC and sulfate (or rather, the lack of the sizeable increase in OC emissions projected under the baseline scenario).

The larger impact of Indian diesel stems in part from the greater transport of Indian pollution over the Himalayas and desert regions to the west. Along with other factors
such as greater available sunlight at lower latitudes and differing atmospheric residence times, this leads to enhanced forcing from BC emitted in India. In our model, it is 109% more efficient than BC emitted from China in causing direct forcing, marginally greater than the ratio in other models (46-88%; Table S2). Forcing from BC emitted in NA/ME is also enhanced, and the tight standards versus baseline forcing reduction due to BC is larger there than for any other region. However, this forcing is offset by the impact of sulfate, which is extremely large due to the very high sulfur content in fuels (especially in several Middle Eastern countries). This leads to an overall increase in forcing over the next 20 years due to emissions from NA/ME under either scenario as SO\textsubscript{2} emissions are reduced.

The maximum global mean radiative forcing mitigation comes from additional tightening of standards for diesel vehicles in India (-18 \pm 12/-6 mW/m\textsuperscript{2}) and Latin America (-6\pm4 mW/m\textsuperscript{2}) and gasoline vehicles in NA/ME (-10 +9/-4 mW/m\textsuperscript{2}). The large forcing reduction for Indian diesel vehicles results from nearly equal contributions from BC and ozone, plus a contribution from nitrate and offsets from OC and sulfate (nitrate increases in part owing to reduced competition for ammonium with sulfate). Local forcings are generally much larger than the global mean. The impact of emission changes on local forcing by region and fuel type from the regional simulations is shown in Figure S9. Comparison with Figure 2 in the main text shows that local forcing can be many times greater, and longer-lived species (e.g. ozone) tend to have relatively greater impact at global scales than at local scales. For example, although the global mean forcing from tight standards applied within China is only about -10 mW/m\textsuperscript{2} for 2030 versus 2010, the forcing over China itself is roughly -130 mW/m\textsuperscript{2}. For India, the local forcing is even greater at about -400 mW/m\textsuperscript{2}, with more than half due to tight standards (the rest primarily to motorcycles under the baseline). Tight standards on Indian diesel vehicles reduces global mean forcing by 18 (+6/-12) mW/m\textsuperscript{2} relative to the baseline, but reduces local forcing by ~250 (+100/-300) mW/m\textsuperscript{2}. Due to the differing atmospheric lifetime of the various contributors to forcing, the local value can even differ in sign from the global, as for the impact of emissions controls on N. American gasoline vehicles which causes positive forcing locally (Table 2) but negative forcing globally (Figure 2). Even at local scales, however, ozone forcing can be important in addition to aerosol forcing. For example, two of the largest effects of the tight standards on local forcing are emissions controls on diesel vehicles in India and Latin America. For both of those, reductions in forcing by ozone is a substantial fraction of the added benefit of tight standards: ~1/3 the aerosol forcing for India and roughly equal to the aerosol forcing for Latin America (Figure S9). Hence not only is the balance between forcing by different aerosol types important, but so is the forcing due to changes in gaseous compounds.

For European and North American emissions changes, the global mean forcing due to ozone decreases is greater than that due to reduced BC for light-duty gasoline vehicles (again much the BC comes from motorcycles). Note that the projected decreases in forcing are substantially larger in some cases for 2000-2030 rather than 2010-2030: -6 mW/m\textsuperscript{2} for Indian motorcycles versus -4, -9 for North American light-duty gasoline vehicles versus -4, -11 for North American heavy-duty diesel trucks vs -7, and -18 for Western European heavy-duty diesel trucks versus -14. The implication is that estimated reductions between 2000 and 2010 have already achieved substantial forcing decreases for these vehicles types/regions. Note also that desulfurization is also responsible for
positive forcing over North America due to reductions in North American emissions from gasoline vehicles. This is nearly all between 2000 and 2010, however, so 2030 versus 2010 forcing is negative (see Figure 2).

The uncertainties are not necessarily proportional to the magnitude of the net forcing in each region (e.g. Figure 1). This is not due to fundamental differences in our knowledge of physical processes in different regions, but to the relative magnitudes of the individual forcings that make up the net regional response. An illustration of this is provided by comparing the forcing components for the tight standards versus baseline emissions for North Africa/ME and China. While the net forcing for tight standards relative to the baseline is less than a factor of two larger in these cases (-195 mW/m² for North Africa/ME, -107 mW/m² for China), the net forcing over N Africa/ME is primarily the difference between larger forcings from BC, sulfate and ozone (-450, 340 and -92 mW/m², respectively) than for China (-164, 106, and -61 mW/m²), leading to uncertainty about four times greater for N. Africa/ME.

Table S5. Impacts of tight standards relative to baseline.

<table>
<thead>
<tr>
<th>Region</th>
<th>Radiative Forcing (mW/m²)</th>
<th>Premature Deaths (yr⁻¹)</th>
<th>Wheat Relative Yield Loss (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>North America</td>
<td>-128</td>
<td>-300</td>
<td>-0.16</td>
</tr>
<tr>
<td></td>
<td>(-47, -210)</td>
<td>(-100, -500)</td>
<td>(-0.09, -0.22)</td>
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<td>China</td>
<td>-107</td>
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<td>-1.36</td>
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<td>(-34, -180)</td>
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<td>(-0.56, -2.16)</td>
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<td>-2.90</td>
</tr>
<tr>
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<td>(-15, -93)</td>
<td>(-38300, -89500)</td>
<td>(-1.17, -4.63)</td>
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<td>-1200</td>
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<td>(-17400, -26200)</td>
<td>(-0.35, -0.79)</td>
</tr>
<tr>
<td>North Africa/Middle East</td>
<td>-195</td>
<td>-24300</td>
<td>-3.17</td>
</tr>
<tr>
<td></td>
<td>(154, -544)</td>
<td>(-17100, -31200)</td>
<td>(-1.18, -5.15)</td>
</tr>
<tr>
<td>Latin America</td>
<td>-59</td>
<td>-10800</td>
<td>-0.22</td>
</tr>
<tr>
<td></td>
<td>(-8, -110)</td>
<td>(-4800, -16300)</td>
<td>(-0.20, -0.25)</td>
</tr>
<tr>
<td>Global</td>
<td>-51</td>
<td>-202700</td>
<td>-1.07</td>
</tr>
<tr>
<td></td>
<td>(-23, -80)</td>
<td>(-116100, -281200)</td>
<td>(-0.41, -1.72)</td>
</tr>
</tbody>
</table>

Companion to the results for each scenario shown separately in Figure 1. Ranges include estimated forcing uncertainty based on published observations and modeling results (~67% confidence interval (CI)), uncertainty due to concentration-response relationships only (health; 95% CI), and uncertainty due to concentration-response metrics (crops). Forcing is land area only for regions, and land plus ocean for global.

**Precipitation Response**

Evaluation of relatively small regional precipitation changes requires many years of simulation. We therefore repeated the control year 2000 simulations and the simulations with 2030 tight standards on diesel vehicles in China and India running for 50 years with
the atmosphere coupled to a mixed-layer ocean to provide a snapshot of potential impacts in 2030.

Precipitation responses to reductions in emissions from diesel vehicles in China averaged over the last 40 years of the 50-year mixed-layer ocean simulations are shown in Figure S10 (the spatial pattern for July-August is similar, though areas of statistical significance are smaller). We reiterate that these simulations included AIE on precipitation. Statistically significant changes are seen locally over parts of Asia. SE China shows an increase in annual mean rainfall, for example. The results shown here are indicative of the type of changes in precipitation that can be expected in response to changes in regional aerosols and ozone, even though the results from a single model must be treated with caution. They demonstrate that localized changes can be substantial and that internal variability is large. Precipitation responds to changes in both scattering and absorbing aerosols.\(^\text{64,65}\). It is not possible to isolate these in this case as both sulfate and BC emissions decreased dramatically (Figure S2). The spatial pattern we see is similar to that produced by a decrease in scattering aerosols in earlier simulations\(^\text{65}\). In contrast, another study suggested that aerosol changes over China primarily led to changes in July precipitation over South Asia with relatively little impact in China itself\(^\text{66}\), emphasizing the lack of robustness for these precipitation changes in current climate model results.

**Health Response**

Turning to premature deaths, the baseline emissions reductions clearly have great benefits over North America, Mexico and Europe, while the largest impacts of the tight standards are in West Africa, India, China and the Middle East (Figure S6). Full implementation of existing legislation covering vehicles in Western Europe and North America leads to ~54000-93000 and ~10000-18000 annual avoided premature deaths in 2030 relative to 2000, respectively. In the areas where controls are tightened, premature deaths are substantially reduced under the tight standards scenario relative to the baseline: by 33000-94000 per year in India, 37000-83000 in China, 14000-30000 in South & Central Africa, 17000-31000 in NA/ME and 5000-16000 in Latin America. Valuation of these is quite large (in billions 2006 $US): 419-1200 in India, 564-1300 in China, 160-382 in South & Central Africa, 243-447 in NA/ME, and 66-234 in Latin America (using the average of the two valuation methods’ upper bounds and lower bounds).

We also present the results in terms of costs per capita by region (Table S4). This shows that regardless of the approach taken to VSL estimates, the largest per capita valuation of avoided deaths due to the tightening of emissions standards takes place in North Africa/Middle East and India, but that benefits are greater than $450 per person (over 30 years of age) in all regions where the standards are tightened. We emphasize that the regional valuation is primarily of use for comparing costs and benefits of various policies within a region rather than for comparison of the value of avoided deaths between regions.

Note that avoided deaths over the 2010 to 2030 period would be different than the ones presented here for 2000 to 2030. In particular, sulfate is the largest component of the PM2.5 change for NA/ME and North America (87 and 38%, respectively), and trends over the 2010-2030 period are 26% larger and 96% smaller, respectively, for those regions. Thus the avoided deaths in NA/ME could be substantially greater analyzing the
2010-2030 period than those shown here, while they would likely be substantially less for North America. We also note that controls on gasoline vehicle emissions in India and China and on diesel vehicles in NA/ME can lead to increases in ozone in at least some areas rather than decreases, and hence result in more ozone-related deaths. This occurs because ozone formation chemistry depends non-linearly on the background NOx abundance, so that NOx controls can lead to ozone increases in highly polluted regions. Such a result is consistent with local analyses for Southern California. This dependence changes at lower NOx levels, so that in combination with NOx emissions reductions from other activities ozone levels could be reduced in these regions as they are elsewhere. As pointed out in the main text, ozone-related deaths in India increase under the tight standards scenario due to the influence of remote emissions increases outweighing local emissions decreases. The large increases in emissions that are projected even under the tight standards scenario for several Asian countries near India lead to strong increases in parts of South and Southeast Asia, some of which spills over into India, as illustrated by Figure S11.

Premature deaths due to PM2.5 exposure from the two methodologies utilized here are compared in Table S6. We examine the impact of the tight standards versus the baseline emissions at 2030 in the two calculations. Several main differences in methods, some of which compete in direction, explain the variations between the two sets of results. As described previously, Method 1 includes the population age 30 and older, while Method 2 uses just the population 45-79, which leads to larger estimates for Method 1. Method 1 does not use any concentration thresholds, while Method 2 uses a low-concentration threshold of 5.8 ug/m³, again leading to larger estimates using Method 1. Contrastingly, Method 2 uses age-stratification in baseline mortality rates and projected population from the IDB, which has faster growth of older populations in some rapidly aging parts of the world, such as in China. Since baseline cardiopulmonary mortality rates are higher at older ages, the effect of age stratification in the health impact assessment methodology is to give larger mortality estimates in China and India using Method 2. Mortality estimates for the developed world are not substantially different between the two methodologies since population growth is not as high.

Table S6. Annually avoided premature deaths due to PM2.5 under the tight standards scenario versus the baseline scenario at 2030 calculated by two methods.

<table>
<thead>
<tr>
<th>Region</th>
<th>Method 1</th>
<th>Method 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>N. America</td>
<td>50</td>
<td>22</td>
</tr>
<tr>
<td>China</td>
<td>42,019</td>
<td>46,978</td>
</tr>
<tr>
<td>India</td>
<td>32,350</td>
<td>42,694</td>
</tr>
<tr>
<td>W. Europe</td>
<td>683</td>
<td>769</td>
</tr>
<tr>
<td>W. FSU</td>
<td>4,734</td>
<td>3,638</td>
</tr>
<tr>
<td>S/C Africa</td>
<td>15,295</td>
<td>13,761</td>
</tr>
<tr>
<td>NA/ME</td>
<td>21,175</td>
<td>7,354</td>
</tr>
<tr>
<td>Latin America</td>
<td>3,962</td>
<td>2,279</td>
</tr>
</tbody>
</table>

**Crop Response**

Yield changes resulting from emissions controls by region, comparable to those presented for wheat in Table 2 in the main text, are presented for rice, maize and soybean...
The local versus remote impact results for ozone-related agricultural damages are qualitatively consistent with ozone-related premature deaths. Note that the magnitude of reductions in yield losses varies regionally and from crop to crop, with the largest percentage changes typically for wheat and soybeans. Crop yield impacts differ from those seen for ozone-related health impacts (in percentage relative to local emissions), and even vary from one crop to another, due to variations between the response of different ozone metrics to emissions and differences in the locations of crops and population centers.

Table S7. Regional impacts at 2030 relative to 2000 under the tight standards scenario for rice, maize and soybean.

<table>
<thead>
<tr>
<th>Impact and impact region</th>
<th>Total response</th>
<th>Percentage of response due to emission change in these regions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Local gasoline</td>
</tr>
<tr>
<td>Mean rice yield increase</td>
<td>million metric tons (%)</td>
<td></td>
</tr>
<tr>
<td>North America</td>
<td>0.23 (2.42)</td>
<td>43</td>
</tr>
<tr>
<td>Western Europe</td>
<td>0.05 (2.07)</td>
<td>30</td>
</tr>
<tr>
<td>China</td>
<td>0.30 (0.15)</td>
<td>0</td>
</tr>
<tr>
<td>India</td>
<td>-0.32 (-0.24)</td>
<td>25</td>
</tr>
<tr>
<td>North Africa/Middle East</td>
<td>0.12 (1.15)</td>
<td>3</td>
</tr>
<tr>
<td>Mean maize yield increase</td>
<td>Million metric tons (%)</td>
<td></td>
</tr>
<tr>
<td>North America</td>
<td>3.88 (1.41)</td>
<td>45</td>
</tr>
<tr>
<td>Western Europe</td>
<td>1.41 (2.75)</td>
<td>32</td>
</tr>
<tr>
<td>China</td>
<td>0.20 (0.18)</td>
<td>-6</td>
</tr>
<tr>
<td>India</td>
<td>-0.013 (-0.11)</td>
<td>27</td>
</tr>
<tr>
<td>North Africa/Middle East</td>
<td>0.10 (1.04)</td>
<td>4</td>
</tr>
<tr>
<td>Mean soybean yield increase</td>
<td>million metric tons (%)</td>
<td></td>
</tr>
<tr>
<td>North America</td>
<td>4.37 (4.28)</td>
<td>45</td>
</tr>
<tr>
<td>Western Europe</td>
<td>0.18 (8.28)</td>
<td>33</td>
</tr>
<tr>
<td>China</td>
<td>0.10 (0.50)</td>
<td>-10</td>
</tr>
<tr>
<td>India</td>
<td>-0.05 (-0.89)</td>
<td>18</td>
</tr>
<tr>
<td>Region</td>
<td>Change</td>
<td>Avoided Premature Deaths due to Ozone</td>
</tr>
<tr>
<td>------------------------</td>
<td>--------</td>
<td>---------------------------------------</td>
</tr>
<tr>
<td>North Africa/Middle East</td>
<td>0.002</td>
<td>(0.74) 26 -238 103 84 5 7 -212 113</td>
</tr>
</tbody>
</table>

Total impacts of local diesel and gasoline vehicle emissions changes on the region of emission are given in bold and are the sum of the separated impacts of local gasoline and diesel changes. Values in the right columns hence include all changes.

Figure S6. Annual average net radiative forcing (top row, mW/m²) and avoided premature deaths due to ozone (middle row, persons per 1000 km²) and due to PM2.5 (bottom row, persons per 1000 km²) for the baseline scenario at 2030 relative to 2000 (left column) and at 2030 under the tight standards scenario relative to the baseline scenario (right column). Forcing values are instantaneous at the tropopause for aerosols (direct forcing only) and ozone without adjustments to literature values.
Figure S7. Annual average total net radiative forcing (mW/m²). Values are instantaneous at the tropopause for aerosols and ozone relative to 2000 for the 2030 tight standards case (analogous to Figure S6). Aerosol forcing is direct forcing only.

Figure S8. Direct radiative forcing per g BC in the overhead column (W/m² per g), using the 2030 baseline versus 2000 model results as an example. Areas in grey had changes in BC mass that were so small (<2 µg) that the ratio is unreliable. The thick black line shows the zero BC change contour, very close to which values are also less reliable due to division by small numbers. The global mean value (excluding grey areas) is 1680 (W/m²)/g.
Figure S9. Local radiative forcing (mW/m²) over each individual region due to 2030 tight standards applied locally relative to 2010 (left column) and relative to 2030 baseline emissions (right column) for diesel and gasoline vehicles by pollutant and the net.
Figure S10. Annual average precipitation change (mm/day) following a reduction in emissions from diesel vehicles in China under the tight standards scenario for 2030 relative to the year 2000. Hatched areas show statistical significance at the 90% confidence level. Maximum changes are about 3 mm/day.

Figure S11. Surface ozone change in the worldwide tight standards scenario at 2030 relative to 2000. The quantity shown is the metric used to calculate ozone-related premature deaths, the annual average change in the 6-month maximum of the daily maximum surface ozone concentration.
References

43 Krewski, D. et al. Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality. 140 (Health Effects Institute, 2009).
52 Public Health and Air Pollution in Asia (PAPA): Coordinated Studies of Short-Term Exposure to Air Pollution and Daily Mortality in Four Cities. (Health Effects Institute, Boston, 2010).