LETTER

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Impacts and mitigation of excess diesel-related NO_x emissions in 11 major vehicle markets

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Vehicle emissions contribute to fine particulate matter ($PM_{2.5}$) and tropospheric ozone air pollution, affecting human health[1–5], crop yields[5,6] and climate[5,7] worldwide. On-road diesel vehicles produce approximately 20 per cent of global anthropogenic emissions of nitrogen oxides (NO_x), which are key $PM_{2.5}$ and ozone precursors[8,9]. Regulated NO_x emission limits in leading markets have been progressively tightened, but current diesel vehicles emit far more NO_x under real-world operating conditions than during laboratory certification testing[10–20]. Here we show that across 11 markets, representing approximately 80 per cent of global diesel vehicle sales, nearly one-third of on-road heavy-duty diesel vehicle emissions and over half of on-road light-duty diesel vehicle emissions are in excess of certification limits. These excess emissions (totalling 4.6 million tons) are associated with about 38,000 $PM_{2.5}$- and ozone-related premature deaths globally in 2015, including about 10 per cent of all ozone-related premature deaths in the 28 European Union member states. Heavy-duty vehicles are the dominant contributor to excess diesel NO_x emissions and associated health impacts in almost all regions. Adopting and enforcing next-generation standards (more stringent than Euro 6/VI) could nearly eliminate real-world diesel-related NO_x emissions in these markets, avoiding approximately 174,000 global $PM_{2.5}$- and ozone-related premature deaths in 2040. Most of these benefits can be achieved by implementing Euro VI standards where they have not yet been adopted for heavy-duty vehicles.

To reduce the health burden of ambient air pollution (estimated at 4.4 million premature deaths in 2015 globally[21]), all major vehicle markets have implemented programmes requiring new vehicle models to meet emission limits for directly emitted particulate matter, NO_x and other pollutants. The most stringent current standards—Euro VI and US EPA 2010 for heavy-duty vehicles (HDVs) and Euro 6 and Tier 2 for light-duty vehicles (LDVs)[22]—have dramatically reduced exhaust $PM_{2.5}$ and other pollutant emissions[5].

Yet reducing NO_x has proved more challenging for diesel vehicles; there is a growing gap between real-world NO_x emissions and certification limits under the tightened emission limits of Euro 4/IV and Euro 5/V[15–17]. This 'excess diesel NO_x' problem gained public prominence with the discovery that around 11 million Volkswagen LDVs in the USA, Europe and elsewhere contained a defeat device, that is, software that senses when the car is undergoing emissions testing and activates emission control equipment[10]. It is less widely understood that excess diesel NO_x stems primarily from deficiencies in LDV and HDV emission certification procedures, which legally permit higher vehicle emissions under normal driving conditions and outside a pre-defined laboratory setting[19,20]. Recent tests for Euro VI trucks and Euro 6 cars indicate that real-world NO_x emissions in line with certification limits are technically achievable[11,15].

The USA and the EU are developing more stringent policies to address excess diesel NO_x. The USA is phasing in Low Emission Vehicle III/Tier 3 standards, which are much stricter than Euro 6 and Tier 2 for model year 2017–2025 LDVs[23]. California surpassed national HDV standards with a voluntary low-NO_x standard beginning with model year 2010[24]. In 2017, the EU requires new LDV type approvals to pass a real-driving emissions (RDE) test using portable emissions measurement systems (PEMS)[25]. Over 70% of vehicles sold globally are certified to EU standards and the remainder primarily to USA standards[22], so the excess diesel NO_x problem is widespread with substantial health and environmental damages likely, especially where advanced standards are not yet adopted. Previous studies have estimated the impacts of the Volkswagen scandal for LDVs in the USA[26–29] and the benefits of Euro 6/VI standards in key countries prior to revelations about real-world NO_x emissions[5], but the impacts of excess NO_x emissions from both LDVs and HDVs at the global scale are unknown.

We develop here a detailed inventory of real-world NO_x emissions in 2015 from diesel LDVs and HDVs in 11 major vehicle markets: Australia, Brazil, Canada, China, the 28 European Union member states (EU-28), India, Japan, Mexico, Russia, South Korea, and the USA. These markets cover around 80% of new diesel vehicle sales and include those (the USA, the EU and Japan) that set the precedent for new vehicle regulations elsewhere. We examine future scenarios projecting 2040 diesel NO_x emissions under presently adopted policies, expanded implementation of current Euro 6/VI standards, strengthened RDE programmes to enhance real-world effectiveness of Euro 6 standards, and more stringent next-generation standards. We isolate the influence these policies would have on NO_x emissions to examine their effectiveness in closing the gap between certification limits and real-world emissions, a challenge for NO_x that appears not to exist for particulate matter[22]. We combine global chemical transport modelling with health, crop yield and climate models to estimate the damage caused by diesel NO_x emissions and the benefits of future regulations.

The baseline on-road diesel NO_x emissions inventory, a key innovation of this study, is built from real-world NO_x emission factors specific to each region, vehicle type and emission standard derived from an extensive review of in-use emissions testing results in the USA, the EU, China and Japan (Methods and Supplementary Methods). We estimate that real-world NO_x emissions are 3.2× and 5.7× the emission limits for Euro 4 and Euro 6 vehicles respectively (the latter is expected to decline to 4× under adopted RDE programmes[30]; Fig. 1). The emissions of US Tier 2 LDVs, including but not limited to those affected by Volkswagen defeat devices, are estimated at 5× emission limits. Real-world multipliers of HDV emission limits are highest for buses, which often operate in high-emitting low-speed, low-load conditions. The worst-performing buses are Euro IV and V buses in China (4–4.5× the limits).

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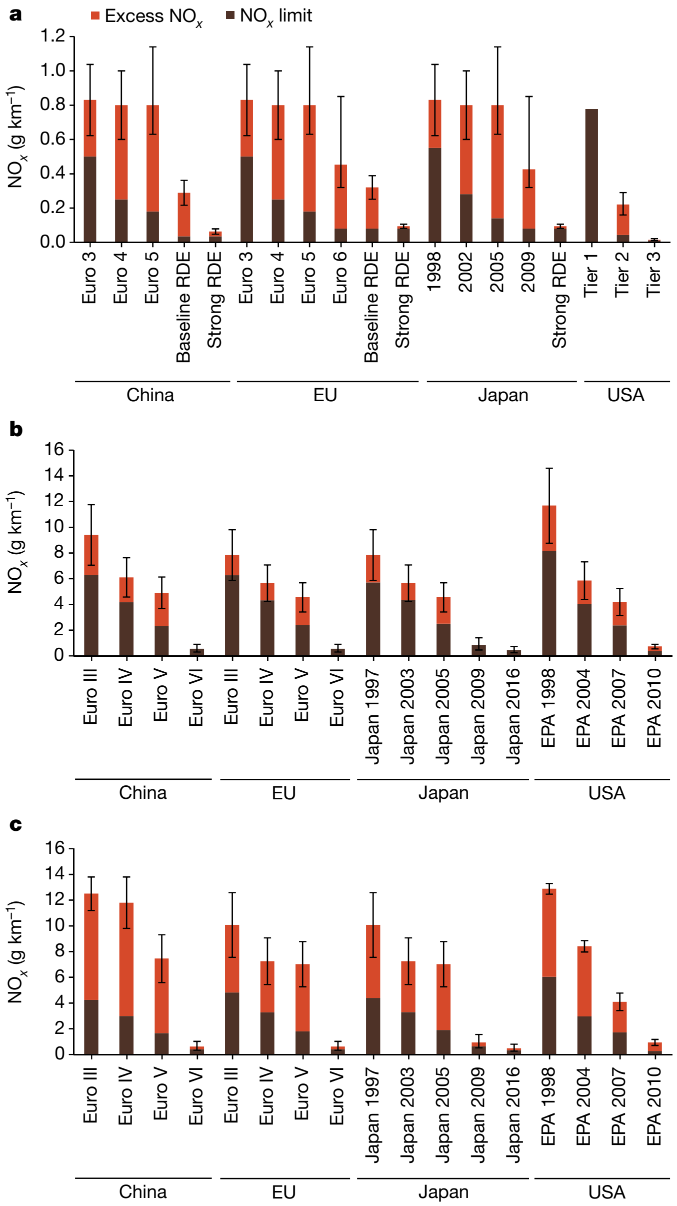

Figure 1 | Real-world NO_x emission factors by vehicle emissions standard in key regions. a, Passenger cars; **b**, heavy heavy-duty trucks; and **c**, buses. Error bars indicate high and low estimates reported in Extended Data Table 3; see Supplementary Information sections 1.2 and 1.3 for details of these methods, which are specific to each region, vehicle type and standard. Except as otherwise stated in the Supplementary Information, a 25% margin of error is assumed to account for variability in emission measurements and traffic composition.

Euro VI heavy-duty trucks and buses emit closer to certification limits (1–1.5× emission limits) than those certified to Euro IV (1.3–2.2×) and Euro V (1.9–3.9×) standards.

From these emission factors, we estimate that diesel vehicles in the 11 regions emitted about 4.6 million tonnes (Mt) of NO_x in excess of certification limits in 2015 (the Baseline minus Limits scenario; Fig. 2), constituting 31% (27%–33%, range reflecting uncertainty in emissions inventories) and 56% (47%–65%) of fleet-wide on-road HDV and LDV NO_x emissions in these regions (gridded emissions are available in Supplementary Data). This excess diesel NO_x increased $PM_{2.5}$ concentrations primarily in Europe, China and India, and increased ozone throughout the Northern Hemisphere (Fig. 3), resulting in 38,000 [95% confidence interval (CI), 23,000–47,000] premature deaths and 625,000 (95% CI, 390,000–780,000) years of life lost globally in 2015 (Fig. 4 and Extended Data Table 1; see also Supplementary Tables). In the EU-28, excess NO_x contributed 4% and 10% of total 2015 $PM_{2.5}$ and ozone mortality burdens and exacerbated ozone-related wheat production loss by 0.2%–0.3% (0.19–0.35 Mt of wheat at year 2000 production levels; Extended Data Fig. 1). Most of the global excess NO_x health impacts (80%) occurred in China, India and the EU-28, and 9% occurred outside the 11 regions due to pollution formed or carried over long distances (Extended Data Table 2). Over 80% of the excess NO_x health impacts were driven by increases in $PM_{2.5}$, which has stronger concentration-response relationships than does ozone. The net global radiative forcing impact of excess NO_x-induced changes in nitrate and other aerosols, methane and ozone is cooling (Extended Data Fig. 2), although only $-8.69\,\mathrm{mW\,m^{-2}}$ compared to the $1{,}700\,\mathrm{mW\,m^{-2}}$ of warming from preindustrial to present CO_2 levels[31].

HDVs are the dominant contributor to excess NO_x health impacts in all regions except the EU-28 (Fig. 4). Overall, fleet-wide HDV NO_x emissions in these 11 regions are 45% higher than theoretical compliance with certification limits (Extended Data Fig. 3). HDVs contribute 86% of baseline 2015 on-road diesel emissions and >75% of excess on-road diesel NO_x in 2015, about 90% of which is from China, India, the EU-28, Brazil and the USA (Fig. 2). Diesel LDV NO_x emissions are 130% higher than theoretical compliance with certification limits, with nearly 70% of excess LDV NO_x in the EU-28, followed by China (17%) and India (5%). For the USA, estimated premature deaths from LDV excess NO_x are consistent with previous studies (see Supplementary Information)[26–29] but represent only one-tenth of the impacts from HDV excess NO_x.

Implementing Euro 6/VI standards where they are not yet adopted (Australia, Brazil, China, Mexico and Russia; the Euro 6/VI scenario) would reduce HDV NO_x emissions by 80%–90% compared to the 2040 baseline and avert substantial increases in LDV NO_x emissions (Fig. 2). These emission changes lead to ozone reductions globally and $PM_{2.5}$ reductions in each implementing region (Fig. 3). Globally, these NO_x reductions could avoid 104,000 (95% CI, 56,000–129,000) $PM_{2.5}$- and ozone-related premature deaths in 2040, >80% of which occur in China (Fig. 4 and Extended Data Table 2). Regional benefits are substantial, with $PM_{2.5}$ and ozone mortality burdens reduced by 23% and 18% in Mexico, 8% and 23% in Brazil, and 4% and 10% in China. Although we isolated NO_x emissions to compare the real-world effectiveness of Euro 6/VI standards with more stringent NO_x emission policies, considering the near-elimination of black carbon emissions under Euro 6/VI would add substantially more health benefits and climate cooling[5].

The Strong RDE scenario adds provisions beyond the EU-28-adopted RDE programme that reduce LDV emission factors from 4× to 1.2× Euro 6 limits (that is, in-service vehicle testing, in-use emissions monitoring, expanded driving conditions, and independent verification)[30]. Applied in regions following EU regulations, these strengthened RDE programmes would avoid approximately 31,000 global $PM_{2.5}$- and ozone-related premature deaths in 2040 beyond the Euro 6/VI scenario (Extended Data Table 1). The greatest benefits occur in India—which has the most diesel passenger cars outside the EU-28—followed by the EU-28 and China (Extended Data Table 2). In India, a strong RDE programme makes the difference between a 4× LDV NO_x emission increase and roughly stabilized emission levels (Extended Data Fig. 3).

Progressing to more stringent next-generation standards in all 11 regions would nearly eliminate diesel NO_x emissions (the NextGen scenario; Fig. 2) and avoid approximately 38,000 additional global $PM_{2.5}$- and ozone-related premature deaths annually in 2040 beyond the Strong RDE scenario. Compared to the 2040 baseline, NO_x emission reductions from next-generation standards in all 11 regions could avoid 2% and 7% of $PM_{2.5}$- and ozone-related premature deaths globally, a total of 174,000 (95% CI, 95,000–217,000) premature deaths and 2.98 million years of life lost (95% CI, 1.62-3.68), and 1%–2% of crop production loss for Chinese wheat (1.7–4.0 Mt at year 2000

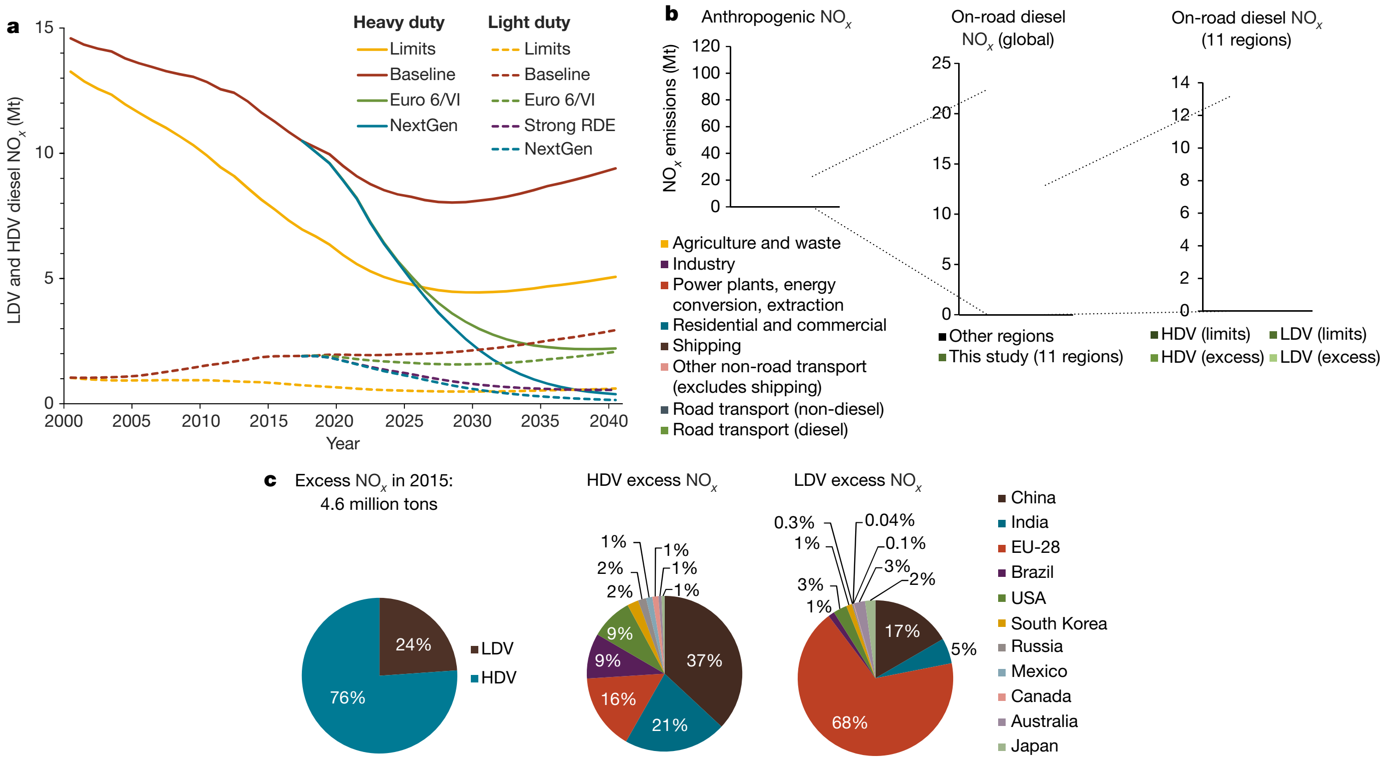

Figure 2 | NO_x emissions by scenario and region. a, From on-road diesel vehicles in 11 major vehicle markets annually. **b**, In 2015 from all emission sources globally, on-road diesel HDV and LDV globally, and on-road diesel vehicles in the 11 markets. **c**, In 2015 from on-road diesel vehicles in excess of theoretical compliance with certification limits by region.

production levels) and maize (0.6–2.0 Mt) and Brazilian soy (0.4 Mt; Extended Data Table 1 and Extended Data Fig. 1). NO_x-induced changes in aerosols, methane and ozone could cause a small net climate warming; however, this would probably be offset by Euro 6/VI black carbon emission reductions (Extended Data Fig. 2)[5].

We estimate that on-road diesel vehicles contribute 55% of global surface transportation NO_x emissions, consistent with other estimates (refs 8 and 9 and Z.K. *et al.*, manuscript in preparation). We show that in 11 major vehicle markets, about one-third of on-road diesel NO_x emissions are in excess of certification limits. Lowering emission limits

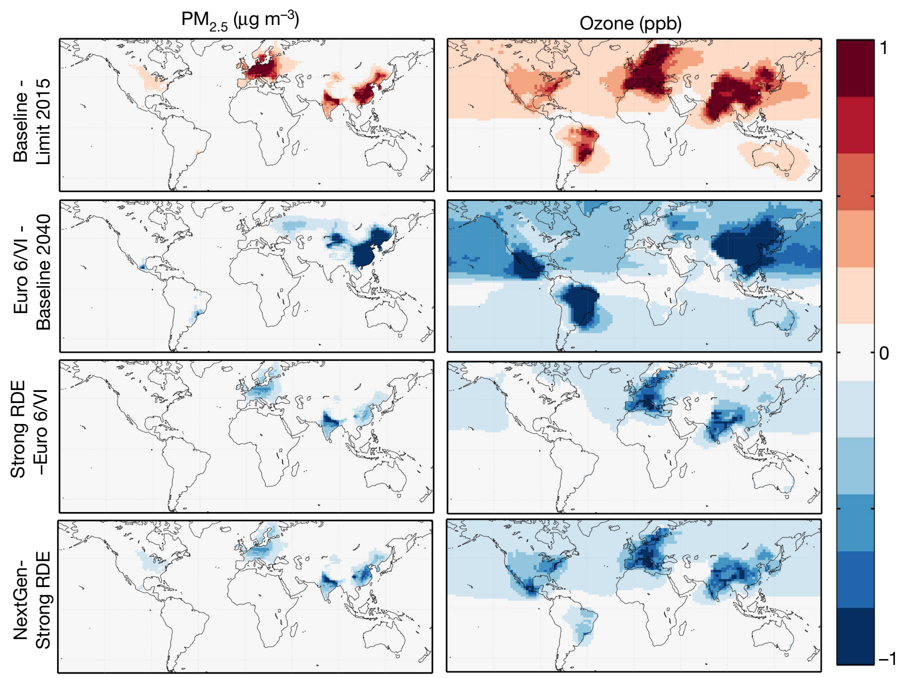

Figure 3 | Change in $PM_{2.5}$ and ozone concentration for the scenario pairs shown. Maps on the left show annual average $PM_{2.5}$ concentration and maps on the right show six-month averages of the 1-h daily maximum ozone concentration. The colour scale refers to both $PM_{2.5}$ (left panels) and ozone (right panels).

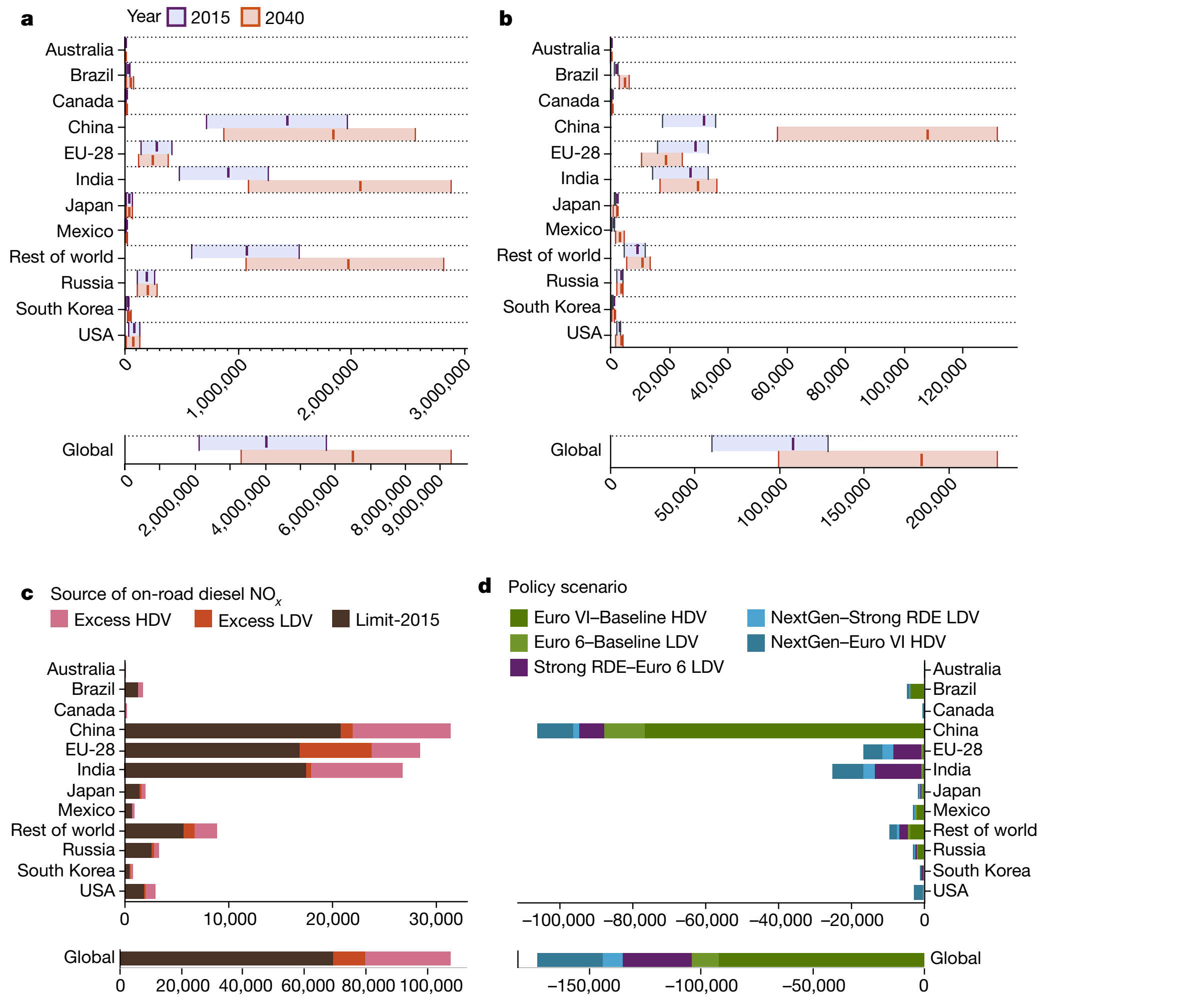

Figure 4 | Annual $PM_{2.5}$- and ozone-related premature deaths. Total $PM_{2.5}$- and ozone-related deaths associated with: **a**, all emissions sources in 2015 and 2040 (95% confidence intervals based on error in the relative risk estimates); **b**, on-road diesel NO_x emissions from the 11 regions in 2015 and 2040; **c**, on-road diesel NO_x emissions in the 11 regions in 2015 disaggregated by theoretical compliance with emission limits (Limit-2015) and excess NO_x emissions from HDV and LDV; **d**, NO_x emission reductions in 2040 under the policy scenarios applied in the 11 regions.

and strengthening compliance and enforcement practices are both essential to achieving low real-world diesel NO_x emissions. Brazil, China, Mexico and Russia can achieve most of the health benefits of stringent next-generation standards by adopting the current Euro VI standards for HDVs. For LDVs, strengthened RDE programmes that improve the real-world effectiveness of Euro 6 standards would achieve most of the health benefits of progressing to next-generation standards. Additional in-use emission testing and national assessments using more localized data can improve impact estimates and narrow uncertainties.

Online Content Methods, along with any additional Extended Data display items and Source Data, are available in the online version of the paper; references unique to these sections appear only in the online paper.

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Supplementary Information is available in the online version of the paper.

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Author Contributions S.A. and J.M. contributed equally. S.A., J.M., V.F., D.H. and R.M. planned the research, J.M., L.D., V.F. and R.M. developed the on-road diesel NO_x emissions inventories, Z.K. and C.H. developed the inventories of all other emissions, D.H. performed the model simulations, S.A., C.M., F.L. and D.H. performed the impact calculations, all authors analysed the results, and S.A. and J.M. wrote the paper with help from all authors.

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METHODS

Emission scenarios. Emission scenarios for the years 2015–2040 address LDVs (passenger cars and light commercial vehicles) and HDVs (buses and light, medium and heavy heavy-duty trucks) and are driven by assumptions of when individual countries/regions will adopt more stringent emission regulations. They exclude vehicles powered by gasoline or other non-diesel fuels and non-road diesel engines (such as locomotive, marine and off-road equipment, including diesel generators, construction and agricultural equipment). The emission scenarios (together with analysis year) for health, climate and agricultural impacts are as follows:

Emission limits 2015 and 2040. This scenario is theoretical where real-world NO_x emissions are equivalent to certification limits, reflecting what diesel NO_x emissions would be without an 'excess NO_x' problem.

Baseline 2015 and 2040. This scenario is the best estimate of how currently adopted NO_x emission standards perform in the real world. Comparison with the 'Emission limits 2015 and 2040' scenario above allows us to estimate 'excess NO_x' emissions and associated impacts.

Euro 6/VI 2040. This scenario adds to the Baseline scenario emissions standards for LDVs and HDVs equivalent to current Euro 6/VI (without modifications to existing type approval and compliance and enforcement provisions) in regions where these are not yet adopted (Australia, Brazil, China, Mexico and Russia).

Strong RDE programme for LDVs 2040. This scenario adds to the Euro 6/VI scenario strong diesel LDV RDE programmes, modelled after the EU-28's adopted RDE regulation plus the inclusion of cold-start emissions, in-use compliance testing, and expanded test procedure boundaries covering a wider range of ambient temperatures, altitudes and driving styles.

Next Generation (NextGen) 2040. This scenario adds to the Strong RDE scenario progressive implementation of next-generation emissions standards (more stringent than Euro 6/VI) based on the US Tier 3 standard for LDVs and California's voluntary NO_x rule for HDVs.

We generate emission inventories for 11 major vehicle markets by combining NO_x emission factors with dates of implemented vehicle regulations, extensive historical data on diesel vehicle activity, sales and population, and vehicle activity projections through to 2040. We adapt an established global transportation emission inventory model that since 2012 has been applied in numerous global and regional studies and validated against other leading models[32]. Most diesel vehicle activity is concentrated in the five largest markets (the EU-28, China, India, USA and Brazil), and this share is projected to grow from 2015 to 2040 (81%–88% for HDVs and 93%–96% for LDVs; Extended Data Fig. 4), driven by increasing car ownership in China and India and growing demand for road freight with increases in economic output.

Baseline emission factors for each vehicle type and region are based on a review of >30 studies of emission factor modelling and in-use emissions testing using PEMS, chassis testing, and remote sensing covering thousands of vehicles conducted mainly in the USA, Europe, China and Japan. Studies were identified by requests to experts and government contacts, supplemented by searching combinations of key words (NO_x, diesel, vehicles, road transport, PEMS, remote sensing) in academic literature databases. Increased weight was given to studies conducted within the past 5 years. EU real-world emission factors are applied to markets following EU regulations (Australia, Brazil, India, Russia and South Korea). Since Japan's LDV regulatory programme has progressed similarly to that for EU standards, the same LDV factors were applied to the EU and Japan except Euro 6, for which Japan's sales mix has led to slightly lower emissions. The same HDV factors were applied to the EU and Japan with the exception of Japan's 2009 and 2016 standards, for which EU real-world multipliers were applied to Japan-specific emission limits. Emission factors in the USA were applied to Mexico and Canada. China HDV factors were derived from local studies, whereas LDV factors were based on EU real-world multipliers.

HDV emission factors. We first convert HDV emission limits (which are based on engine work, measured in grams per kWh), to distance-based limits in grams per vehicle-kilometre (Extended Data Fig. 5) using estimates of brake-specific fuel consumption (a measure of engine efficiency over the test cycle) and in-use fuel consumption (a measure of vehicle efficiency that reflects region-specific driving conditions). We then develop real-world emission factors for each region and vehicle type using a combination of established models and results from our literature review. For most HDV emission factors, we assume a 25% margin of error to account for variability in emission measurements and traffic composition (ref. 33).

For the EU-28 and the USA, we start with established modelled estimates and update these with published in-use emissions testing results where they are substantially different. Central estimates of emission factors for Euro III, IV and V vehicles are from Emisia's Sibyl model[34], which draws its emission factors from the European Environment Agency and European Commission-supported COPERT software. These emission factors are consistent with remote sensing measurements[17,35] and other EU real-world NO_x emissions studies[16,33] showing that real-world emissions have not declined to the same extent as regulated emission limits (Extended Data Fig. 6). For Euro VI vehicles, as average chassis dynamometer test results indicate better performance than is indicated by Sibyl (80% reduction, consistent with regulated emission limits)[15], we develop new emission factors between the two estimates (see Supplementary Information section 1.3). Heavy heavy-duty truck and bus emission factors decline from 7.8 g km^{-1} to 0.54 g km^{-1} and 10 g km^{-1} to 0.61 g km^{-1} from Euro III to VI (Extended Data Table 3).

For China, we develop new HDV emission factors from five in-use emissions testing studies, which had consistent conclusions for Euro III, IV and V equivalent standards (Extended Data Fig. 6). Euro III and IV emission factors are from ref. 36 for heavy trucks and ref. 37 for buses. Emission factors for Euro V buses are from Zhang et al.[38]. Emission factors for Euro V medium and heavy trucks are estimated using the percentage reduction in real-world NO_x in the EU-28 applied to the China-specific emission factor for the previous standard. Heavy heavy-duty truck and bus emission factors decline from 9.4 g km^{-1} to 0.54 g km^{-1} and 12.5 g km^{-1} to 0.61 g km^{-1} from Euro III to VI, assuming similar performance of Euro VI HDVs in the EU-28 and China (Extended Data Table 3).

For US HDVs, central emission factor estimates are based on the United States Environmental Protection Agency (US EPA)'s MOter Vehicle Emissions Simulator (MOVES)[39] and validated against remote sensing measurements of exhaust emissions from in-use trucks in California[40], as well as PEMS testing[41]. For buses certified to US EPA 1998, 2004 and 2007 standards, average emission factors by certification level are from the Integrated Bus Information System (IBIS), which includes NO_x PEMS measurements of >3,000 buses throughout the USA[41]. We apply the same difference between IBIS and MOVES for EPA 2007 buses (a factor of 1.8) to EPA 2010 buses because they were not in the IBIS database. For heavy-duty trucks, remote sensing measurements indicate that fuel-specific NO_x emissions decreased by 83% from model years 2004 to 2012[19] while MOVES estimates an approximately 90% reduction. Limited evidence suggests that EPA 2010 HDVs[42,43] may emit more excess NO_x in urban driving conditions than equivalent Euro VI vehicles in the EU-28[44], potentially owing to USA emissions tests excluding emissions below 30% maximum engine power (EU tests are more inclusive). Since additional PEMS testing (from in-service conformity testing) is needed to establish a robust alternative estimate, we apply the MOVES estimates for EPA 2010 trucks. Lower and upper bound estimates for EPA 1998 to EPA 2007 buses are based on 95% confidence intervals estimated from the IBIS dataset. Heavy heavy-duty truck and bus emission factors decline from 11.6 g km^{-1} to 0.72 g km^{-1} and 12.8 g km^{-1} to 0.93 g km^{-1} from US EPA 1998 to US EPA 2010 (Extended Data Table 3).

LDV emission factors. Passenger cars in Europe are among the most studied with respect to real-world NO_x emissions. Emission factor estimates for Euro 1 to Euro 5 passenger cars are based on emission factor models supplemented with in-use emissions testing studies using PEMS, remote sensing, and laboratory measurements (Extended Data Fig. 6). Emission factors for Euro 6 diesel cars are estimated using the International Council on Clean Transportation's diesel PEMS database covering 32 cars over 180 h and 8,000 km of driving[11]. Light commercial vehicles (LCVs), though less studied, are shown to emit >1.5× the levels observed for passenger cars[45], generally corresponding to the difference between emission limits for heavier LCV classes versus passenger cars. (LCV emission limits depend on vehicle weight class and fall in the range 1–1.6 times the NO_x limit for cars.) Starting with Euro 4 vehicles, we therefore use average LCV emission factors of 1.5× the level estimated for passenger cars. For Euro 3 and earlier, passenger car and LCV emission factors are aligned with Sibyl, which already reflects earlier emissions testing results. Passenger car emission factors decline from 0.82 g km^{-1} to 0.45 g km^{-1} without the RDE programme and to 0.32 g km^{-1} with the Baseline RDE programme (Extended Data Table 3).

For LDVs certified to US Tier 2 standards (2.5 million vehicles from 2004 to 2015[46]), we compute a sales-weighted average of real-world emissions over the Tier 2 bin 5 emission limit (equivalent to 43 mg km^{-1}, mean adjustment factor 5) in three vehicle categories: Volkswagen vehicles with 2.0-litre (about 482,000 vehicles, mean adjustment factor 20) and 3.0-litre (about 85,000, mean adjustment factor 5) engines, and passenger cars and light trucks unaffected by the Volkswagen scandal but which may nonetheless emit NO_x over regulatory emission limits (1.9 million, mean adjustment factor 1.3). Adjustment factors for Volkswagen vehicles with 2.0- and 3.0-litre engines are generally consistent with previous studies[10,47] and those used to estimate health impact of the Volkswagen scandal in the USA[26–28]. The central estimate for unaffected vehicles is based on Vehicle C (a BMW X5) in ref. 48, with a range varying from perfect compliance (a factor of 1) to about 2× the regulated limit (accounting for the rural-uphill/downhill cycle tested, that is, 10× the limit applied to about 5%–10% of vehicle-kilometres travelled). For Tier 1 vehicles, we assume the same average emission factor as Volkswagen vehicles with 2.0-litre engines, since remote sensing measurements indicate that fuel-specific NO_x emissions of diesel passenger cars have remained statistically unchanged since the progression from Tier 1 to Tier 2, and 95% of tested Tier 2 vehicles were Volkswagen or Audi[19]. This assumption results in a central estimate of 1.1× (range 0.8×–1.4×) for the Tier 1 emission limit for 'useful life' (equivalent to 780 mg km^{-1} after 10 years or 100,000 miles).

Baseline USA LDV 2040 emissions are determined primarily by vehicles certified to Tier 3 standards phasing in 2017–2025, which are expected to match emission limits more closely, owing partly to the California Air Resources Board's new defeat device screening methods[49]. Average future Tier 3 vehicle NO_x emission factors are estimated to be within 30% of the certification limit, based on the real-world multiplier of 1.27 for a Tier 2 diesel vehicle with good performance[48]. We assume a range of 1×–2× the Tier 3 limit, similar to Tier 2 vehicles unaffected by the Volkswagen emissions scandal. The central estimate for Tier 2 vehicles (including those affected by the Volkswagen scandal) represents a 74% reduction from Tier 1 levels, reflecting that most of the USA diesel LDV fleet was unaffected by the Volkswagen emission scandal. Overall, USA LDV emission factors decline from 0.85 g km^{-1} to 0.01 g km^{-1} from Tier 1 to Tier 3 (Extended Data Table 3).

$PM_{2.5}$ and ozone concentrations. Country-level diesel vehicle NO_x emissions in the 11 regions are gridded based on population and vehicle miles travelled (see Supplementary Information). For the baseline scenario, all emissions evolve from 2015 to 2040, using our real-world on-road diesel NO_x emissions in the 11 markets combined with the ECLIPSE v5a emissions inventory[8,9] for all other emissions. For the limits and policy scenarios, all emissions are held constant at 2015 (in the case of the limits scenario) or 2040 (policy scenarios) baseline levels, except NO_x emissions in the 11 markets. Except for Euro 6/VI standards—which reduce primary $PM_{2.5}$—the policies examined are not expected to affect emissions substantially other than NO_x.

We simulate NO_x emission impacts on $PM_{2.5}$ and ozone concentrations using the GEOS-Chem chemical transport model[50] (version of forward model contained within version 35 of the model adjoint[51]), driven by GEOS-5 assimilated meteorology for 2015 from the Global Modeling and Assimilation Office at 2° × 2.5° resolution with 47 vertical layers. Simulated $PM_{2.5}$ concentrations are downscaled to 0.1° × 0.1° resolution using $PM_{2.5}$ concentrations derived from remote sensing aerosol optical depth observations[52]. For health impact calculations, simulated ozone concentrations are simply regridded to the finer resolution, as the impacts of model resolution are much less important than for $PM_{2.5}$ (ref. 53). For each scenario, we conduct four GEOS-Chem simulations: including all emissions and individually zeroing out LDV, heavy-duty bus, and heavy-duty truck NO_x emissions.

Health, climate and arable agriculture impacts. We use epidemiologically derived health impact functions to estimate premature $PM_{2.5}$- and ozone-related mortality changes between the Baseline and Limits scenarios in 2015 (using 2015 population and baseline mortality rates) and between the Baseline and policy scenarios in 2040 (using 2040 population and baseline mortality rates). Global 2015 and 2040 $PM_{2.5}$ and ozone mortality burdens are within the range of other published estimates (see Supplementary Information).

We estimate $PM_{2.5}$-related health impacts using integrated exposure response (IER) curves for five health endpoints: adult (≥25 years) ischemic heart disease (IHD), stroke, chronic obstructive pulmonary disease (COPD), lung cancer; and child (<5 years) acute lower respiratory infection (ALRI), following recent studies[21,54]. For IHD and stroke, we use the age-specific IERs for each 5-year age band. We use the IER dataset that was publicly available at the time of the analysis[55], used for the Global Burden of Disease 2010 Study[56]. The IERs take the form:

$$\mathrm{RR}_{i,h} = 1 + \alpha_h\{1 - \exp[-\gamma_h(z_i - z_{\mathrm{cf}})^{\delta_h}]\}$$

where RR is relative risk in grid cell i for health endpoint h, z is the $PM_{2.5}$ concentration in gridcell i, z_{cf} is the counterfactual $PM_{2.5}$ concentration below which we assume no additional risk, and α, γ and δ are model parameters for health endpoint h. Sensitivity results using Global Burden of Disease 2015 Study IERs[21] are in the Supplementary Information.

Ozone relative risk of chronic respiratory disease is from ref. 57. To consider ozone independently from $PM_{2.5}$—following several other studies[58–61]—we use the two-pollutant model controlling for $PM_{2.5}$, which associated a 10 parts per billion (ppb) increase in the April–September average daily 1-h maximum ozone concentration (range 33.3–104.0 ppb) with a 4% [95% CI, 1.3%–6.7%] increase in chronic respiratory disease RR. The ozone-response relationship is:

$$\mathrm{RR}_i = \exp(\beta X_i)$$

where RR is relative risk in grid cell i, β is the model parameterized slope of the log-linear relationship between concentration and mortality, and X is the maximum six-month average of the 1-hour daily maximum ozone concentration in gridcell i. We use a low-concentration threshold of 33.3 ppb (the lowest measured level in ref. 57), below which no health impacts are calculated, and examine a 41.9 ppb threshold (5th percentile) in the Supplementary Information.

We calculate the $PM_{2.5}$- and ozone-attributable disease burden within each 0.1° × 0.1° grid cell using the common population attributable fraction method:

$$M_{i,h} = P_i \times F_{c,h} \times Y_{c,h} \times [(RR_{i,h} - 1)/RR_{i,h}]$$

where M is the disease burden in grid cell i for health endpoint h, P is the population in grid cell i, F is the population fraction in country c for health endpoint h, Y is the baseline incidence rate in country c for health endpoint h. Health damages or benefits are estimated by subtracting disease burdens at the grid cell level between two scenarios. To ascertain HDV and LDV contributions to health impacts, we use the "proportional approach"[1] wherein we scale the HDV + LDV change in disease burden by the fraction of HDV + LDV concentration change affected by HDVs and LDVs individually. This method allows us to consider HDV and LDV emissions simultaneously, since removing each from the model separately would lead to lower health impact results for the quantity removed first (and thus on the flatter portion of the non-linear exposure response curve) and higher results for the quantity removed second (on the steeper portion of the non-linear exposure response curve). Uncertainty bounds for health impacts are based only on uncertainty in these concentration-response functions. Uncertainty between two scenarios is calculated by differencing gridded scenario burden estimates using the same relative risks for each (for $PM_{2.5}$, using the mean, 2.5 percentile, or 97.5 percentiles of the 1,000 RR estimates).

Present-day (2015) baseline incidence rates are from the Institute for Health Metrics and Evaluation (IHME) Global Burden of Disease 2015 Study (http://ghdx.healthdata.org/gbd-results-tool, accessed 1 November 2016). We use country- and cause-specific rates for ages ≥25 years in 5-year age groups (IHD, stroke, COPD, lung cancer for $PM_{2.5}$ mortality, and chronic respiratory disease for ozone mortality) and <5 years (for ALRI), using regional rates where country rates were unavailable. We scale chronic disease mortality rates to 2040 using International Futures model projections, following other studies[60,61] (see Supplementary Information).

Gridded 2015 population (total 6.83 billion) is from Columbia University's Center for International Earth Science Information Network and projected to 2040 using United Nations country projections (total 8.79 billion; see Supplementary Information). Age-specific population fractions for each country are calculated from the IHME data on number of cases and incidence rates.

We estimate ozone-related crop production loss for maize, wheat and soy following ref. 62 (see Supplementary Information). We calculate global radiative forcing of methane and ozone using regional radiative forcing efficiencies (mW m^{-2} per Tg of emission) from ref. 63. We calculate aerosol (nitrate, sulfate, and ammonia) radiative forcing from NO_x emission changes using GEOS-Chem with offline Mie theory calculations of aerosol optical properties and the LIDORT radiative transfer model[64–66]. Central estimates and lower and upper bounds of direct aerosol radiative forcing are scaled based on model comparison to the model ensemble radiative forcing in ref. 31. We include aerosol cloud interactions by scaling the direct radiative forcing to the net effective radiative forcing following UNEP/WMO[67].

Sensitivity analysis, limitations and uncertainties. Our scenario-modelling methods assume that diesel NO_x emissions are controlled before other air pollution controls are introduced, which might realistically be implemented concurrently. Health benefits of $PM_{2.5}$ reductions are therefore calculated at the exposure-response curve's flatter end. Here we examine health benefits of the future policy scenarios using instead the 'proportional approach', as was used to separate HDV versus LDV impacts in the core results. To implement the proportional approach, we scaled gridded baseline 2040 $PM_{2.5}$ mortality burdens by the gridded fraction of the baseline 2040 $PM_{2.5}$ concentration reduced for each policy scenario. Using this approach results in about 40% more $PM_{2.5}$-related health benefits for each policy scenario relative to the baseline.

Benefits of implementing Euro 6/VI are undercounted because the near elimination of black carbon emissions would yield additional substantial health and climate benefits[5,68]. Health impacts of all scenarios could be underestimated because we excluded direct health effects from NO_x exposure[69], morbidity impacts (such as asthma attacks and hospital visits), and health impacts for populations aged 5–24 years. Ozone-related mortality could be underestimated because recent studies indicate larger associations of ozone with respiratory and cardiovascular disease[70]. Our inclusion of only three major crops and exclusion of impacts on productive grasslands also underestimates agricultural impacts[71].

We excluded uncertainty in simulated concentrations (for $PM_{2.5}$ we attempted to address this by assimilating with satellite observations), present and future disease incidence rates, and population growth. Though we estimated both, we did not combine uncertainties in emissions and concentration-response functions.

We excluded potentially important subnational variation in baseline incidence rates and age stratification[72]. We assumed that nitrate, the main $PM_{2.5}$ component affected by NO_x, is equally as toxic as other $PM_{2.5}$ components and mixtures. For crop impacts, we excluded uncertainty about crop spatial extent and growing season and assumed that ozone concentration metrics are reasonable predictors of crop impacts. The direction in which these uncertainties and assumptions may influence results is unknown.

Data availability. Gridded real-world on-road diesel NO_x emissions datasets are available from figshare (https://figshare.com/articles/DieselNOxEmissionsInventory_zip/4748425). All other data generated during the study are included in the paper or available upon request from the corresponding authors.

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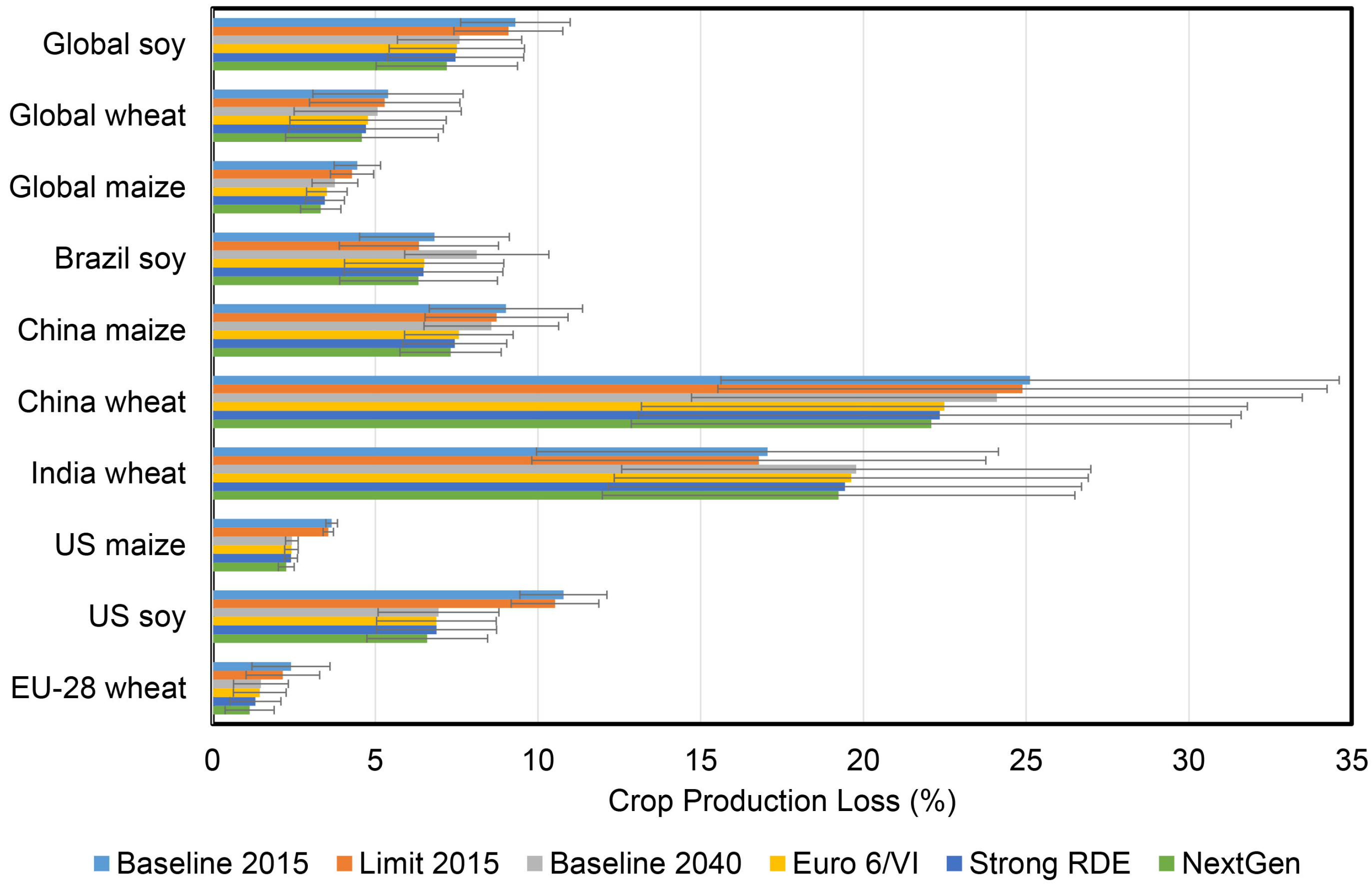
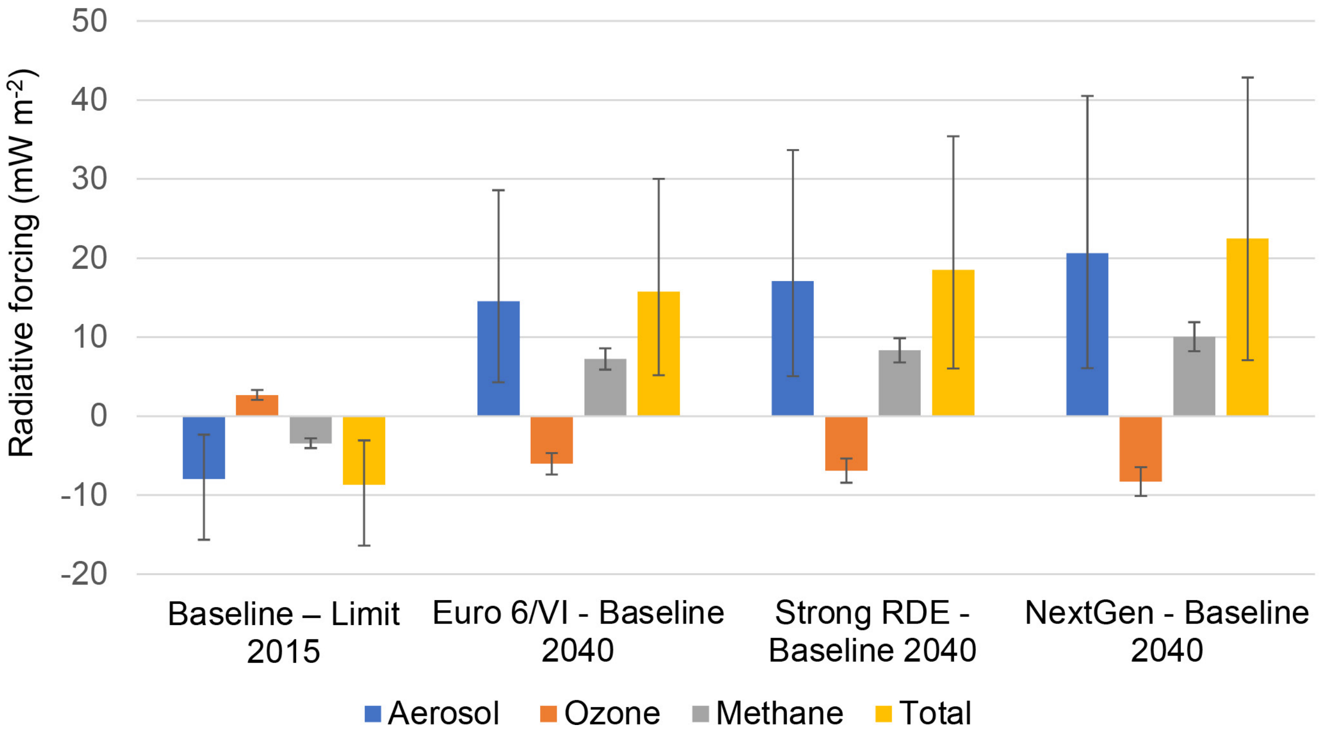
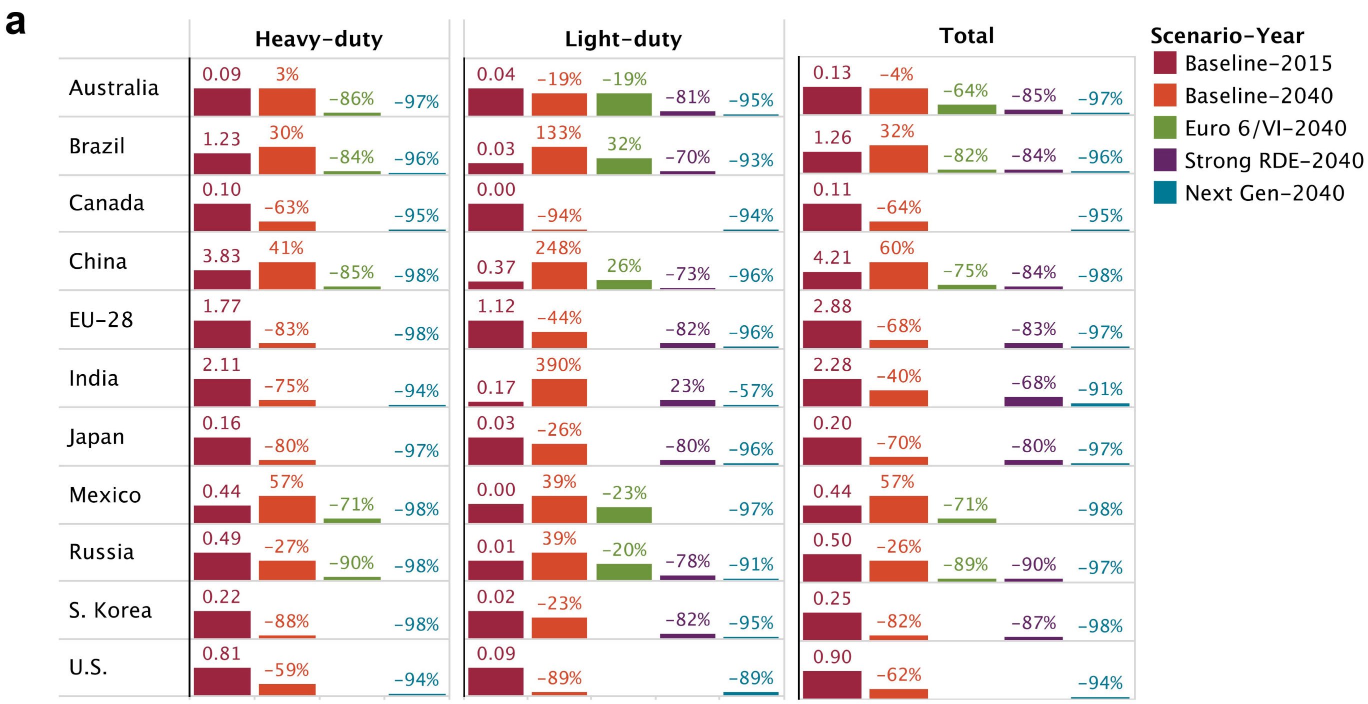


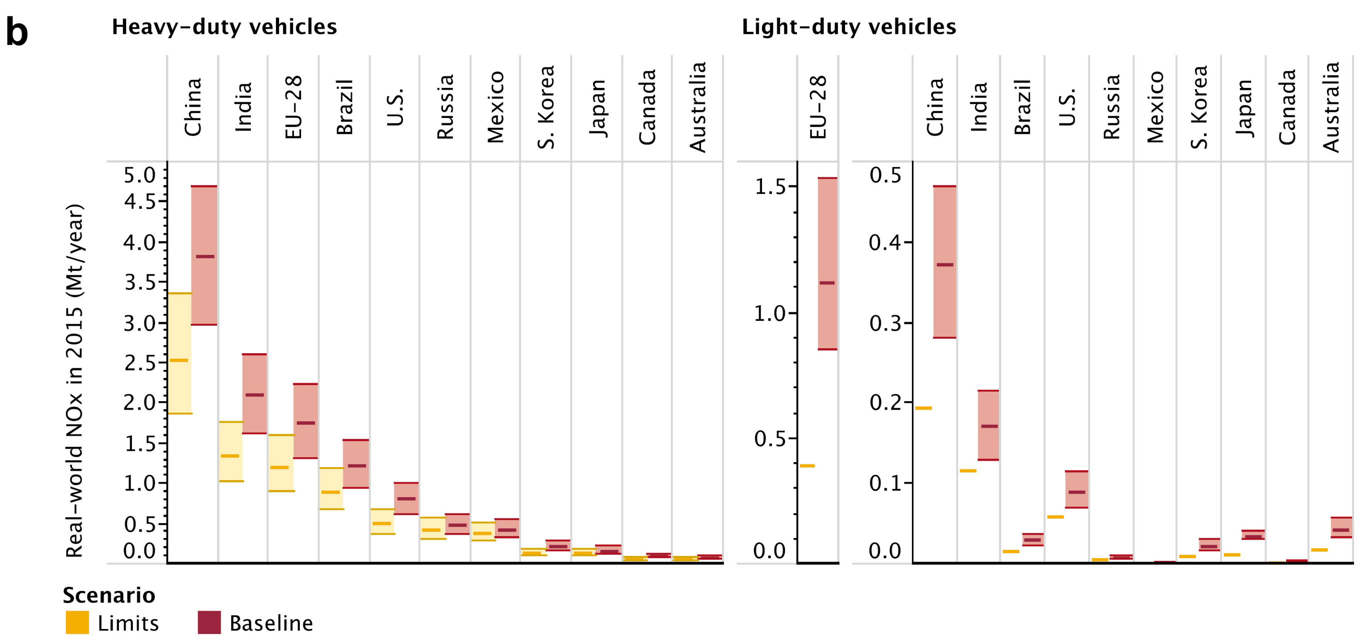
Extended Data Figure 1 | Ozone-related percentage crop production loss by region and scenario. Results are shown for maize, wheat and soy globally and in major producing regions (central estimate and uncertainty bars showing range using two exposure metrics).



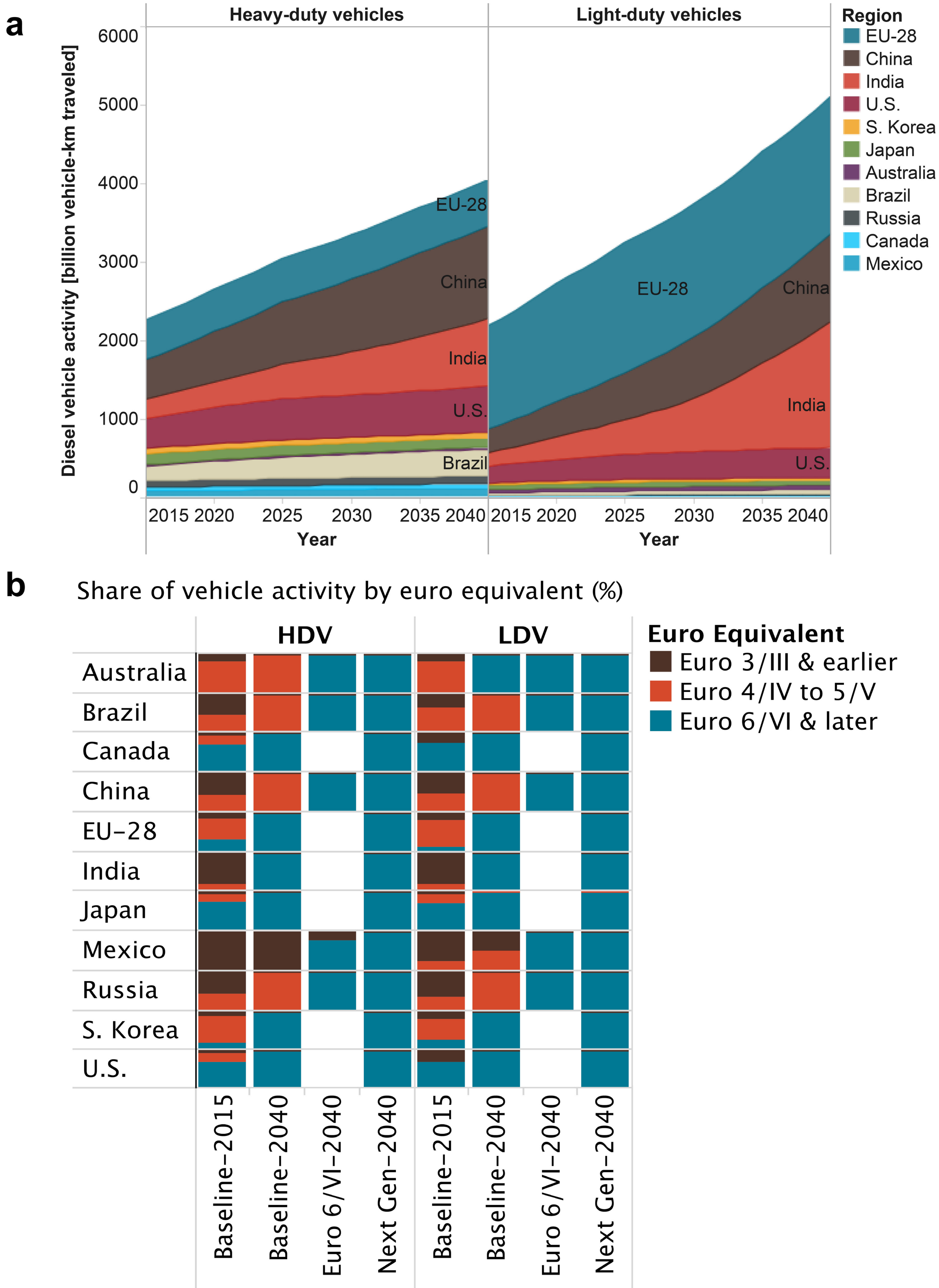
Extended Data Figure 2 | Radiative forcing from change in NO_x emissions. Results shown are central estimates and error bars show 95% confidence intervals based on error in the conversion from concentrations to climate impacts.



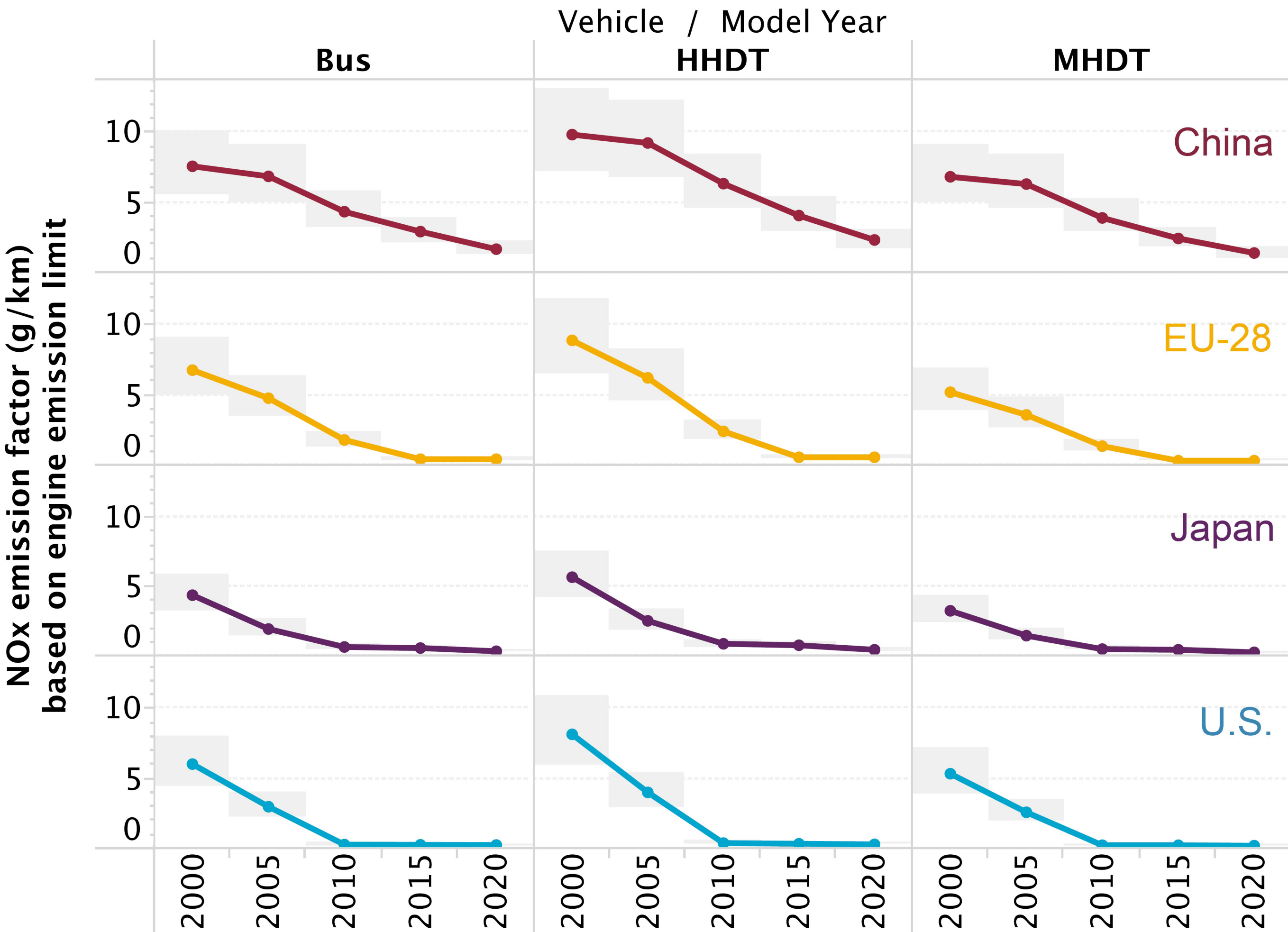
	Heavy-duty				Light-duty					Total				
11 markets	11.3	-17%	-80%	-97%	1.9	55%	9%	-71%	-92%	13.2	-6%	-67%	-79%	-96%



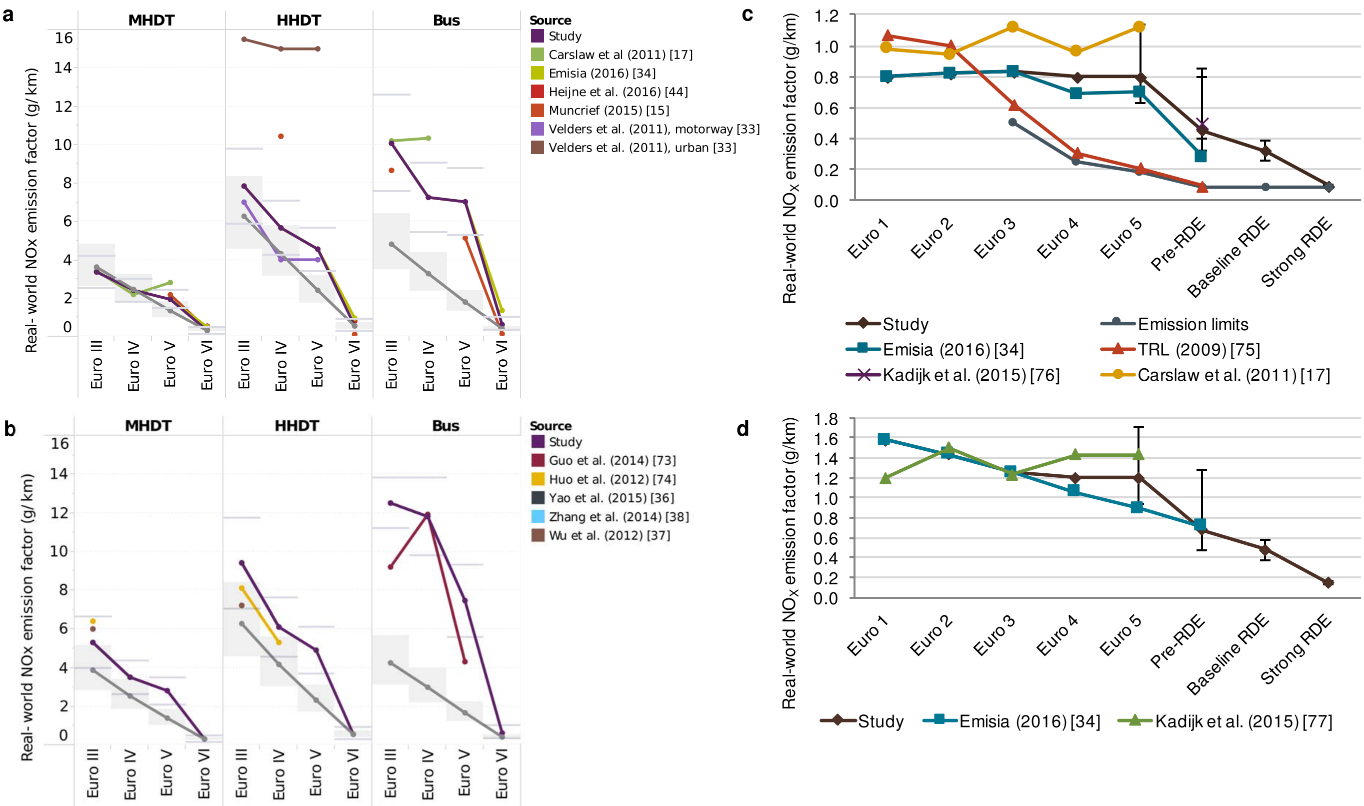
Extended Data Figure 3 | Annual on-road diesel vehicle NO_x emissions in 2015. a, Total on-road diesel vehicle NO_x emissions for the baseline in 2015 (Mt/yr) with percentage change relative to Baseline-2015 for each scenario, where Baseline-2015 labels indicate millions of tons of on-road diesel NO_x emissions. **b**, Comparison of the 2015 baseline with theoretical compliance with emission limits (Limits-2015), where bars indicate uncertainty for HDV emission limits (see Supplementary Information) and Baseline-2015, calculated as described in the Methods.



Extended Data Figure 4 | Diesel LDV and HDV activity. a, By region from 2015 to 2040. **b,** By Euro-equivalent standard and policy scenario in 2015 and 2040.



Extended Data Figure 5 | Distance-specific NO_x emission rates. Uncertainty bands are based on engine emission limits by model year. HHDT, heavy heavy-duty trucks; MHDT, medium heavy-duty trucks. Lines are guides to the eye.



Extended Data Figure 6 | Review of diesel NO_x emission factors. a, HDV in EU-28. (Data are taken from refs 17, 34, 44, 15 and 33. We note that ref. 17 indicates that remote sensing estimates are typical of urban driving conditions in the UK.) **b**, HDV in China. (Data are taken from refs 73, 74, 36, 38 and 37.) **c**, Passenger cars in EU-28. (Data are taken from refs 75, 34, 76 and 17.) Emission limits are from corresponding regulations. Data from ref. 75 are averaged over urban, rural and highway for diesel cars weighing 2.5–3.5 tons.) **d**, Light commercial vehicles in EU-28. (Data are taken from refs 75 and 34.) In **a** and **b**, horizontal lines indicate distance-based emission factors based on engine emission limits. In **a** and **b**, horizontal bars indicate upper and lower bound for emission factor estimates, calculated as described in the Methods.

Extended Data Table 1 | Global air quality and health impacts of emission scenarios in 2015 and 2040

	Baseline burden (all emission sources)		Change due to excess NOx	Change due to policies relative to 2040 baseline (scenario minus baseline)		
Impact category	Baseline 2015	Baseline 2040	Baseline – Limits 2015	Euro 6/VI	Strong RDE	NextGen
Population-weighted $PM_{2.5}$ concentration (μg/m^3)	31.63	32.39	0.38	-0.69	-0.89	-1.12
Population-weighted ozone concentration (ppb)	55.68	57.28	0.66	-0.89	-1.19	-1.57
$PM_{2.5}$-releated deaths (millions)	3.82 (2.03, 5.73)	5.99 (3.13, 8.50)	0.031 (0.020, 0.037)	-0.082 (-0.047, -0.095)	-0.107 (-0.062, -0.124)	-0.137 (-0.080, -0.160)
Ozone-related deaths (millions)	0.216 (0.080, 0.343)	0.500 (0.186, 0.791)	0.007 (0.003, 0.010)	-0.022 (-0.008, -0.034)	-0.029 (-0.011, -0.044)	-0.037 (-0.014, -0.057)
$PM_{2.5}$-related years of life lost (millions)	69.8 (37.1, 105)	116 (60.6, 165)	0.52 (0.34, 0.62)	-1.48 (-0.84, -1.71)	-1.92 (-1.11, -2.23)	-2.44 (-1.42, -2.85)
Ozone-related years of life lost (millions)	3.41 (1.26, 5.41)	8.23 (3.06, 13.0)	0.11 (0.05, 0.16)	-0.29 (-0.12, -0.49)	-0.41 (-0.16, -0.62)	-0.54 (-0.20, -0.83)

Impacts are reported for the year shown in the column headings. Values in parentheses show 95% confidence interval, reflecting uncertainty in concentration-response functions.

Extended Data Table 2 | Regional $PM_{2.5}$- and ozone-related premature deaths in 2015 and 2040

		Baseline burden (all emission sources)		Change due to excess NOx	Change due to future policies relative to baseline in 2040 (scenario minus baseline)		
Region	Pollutant	Baseline 2015	Baseline 2040	Baseline – Limits 2015	Euro 6/VI	Strong RDE	NextGen
Australia	$PM_{2.5}$	0	0	0 (0)	0 (-5)	0 (-5)	0 (-5)
	Ozone	0	0	0 (0)	0 (0)	0 (0)	0 (0)
Brazil	$PM_{2.5}$	21	37	0.4 (2)	-2.9 (-8)	-3.1 (-8)	-3.4 (-9)
	Ozone	1	4	0.1 (10)	-1 (-23)	-1.1 (-24)	-1.2 (-26)
Canada	$PM_{2.5}$	7	8	0.1 (1)	0 (-1)	0 (-1)	-0.1 (-2)
	Ozone	1	1	0 (2)	0 (-2)	0 (-3)	0 (-6)
China	$PM_{2.5}$	1,343	1,650	8.6 (1)	-69.5 (-4)	-74.8 (-5)	-83.5 (-5)
	Ozone	89	179	2 (2)	-18.2 (-10)	-19.8 (-11)	-22.6 (-13)
EU-28	$PM_{2.5}$	264	227	10.6 (4)	-0.5 (0)	-7.7 (-3)	-14.8 (-7)
	Ozone	10	11	0.9 (10)	-0.2 (-2)	-1 (-9)	-1.8 (-17)
India	$PM_{2.5}$	832	1,861	6.6 (1)	-0.3 (0)	-9.5 (-1)	-17.5 (-1)
	Ozone	69	209	2.7 (4)	-0.4 (0)	-4.1 (-2)	-7.5 (-4)
Japan	$PM_{2.5}$	33	29	0.4 (1)	-0.6 (-2)	-1 (-3)	-1.4 (-5)
	Ozone	3	5	0.1 (2)	-0.2 (-4)	-0.2 (-5)	-0.3 (-7)
Mexico	$PM_{2.5}$	6	8	0.1 (2)	-2 (-23)	-2 (-23)	-2.4 (-28)
	Ozone	1	3	0 (2)	-0.6 (-18)	-0.6 (-18)	-0.7 (-23)
Russia	$PM_{2.5}$	181	192	0.8 (0)	-2.1 (-1)	-2.4 (-1)	-2.9 (-2)
	Ozone	1	2	0 (3)	-0.1 (-4)	-0.1 (-5)	-0.1 (-7)
South Korea	$PM_{2.5}$	22	32	0.2 (1)	-0.6 (-2)	-0.7 (-2)	-1 (-3)
	Ozone	1	3	0 (3)	-0.1 (-3)	-0.1 (-4)	-0.2 (-6)
United States	$PM_{2.5}$	70	50	0.9 (1)	-0.3 (-1)	-0.4 (-1)	-2.1 (-4)
	Ozone	9	13	0.2 (2)	-0.2 (-2)	-0.3 (-2)	-0.7 (-5)
Rest of world	$PM_{2.5}$	1,037	1,893	2.7 (0)	-3.4 (0)	-5.5 (0)	-7.5 (0)
	Ozone	30	71	0.6 (2)	-1 (-1)	-1.5 (-2)	-2 (-3)

Values are central estimates of thousands of deaths using the mean relative risk estimates for $PM_{2.5}$ and central estimates from the epidemiology study for ozone. Values in parentheses indicate a percentage change in regional total number of $PM_{2.5}$- or ozone-related premature deaths. The 'Rest of world' row indicates the health benefits occurring outside the 11 implementing regions. Impacts are reported for the year indicated in the column headings.

Extended Data Table 3 | Selected diesel NO_x emission factors, emission limits and multipliers

Vehicle Category	Region	Emission level	Real-world NOx (g/km)	NOx emission limit (g/km)	Real-world multiplier
PC	China	Euro 3	0.82 (0.62, 1.03)	0.5	1.65 (1.24, 2.07)
PC	China	Euro 4	0.80 (0.60, 1)	0.25	3.20 (2.40, 4)
PC	China	Euro 5	0.80 (0.63, 1.13)	0.17	4.44 (3.5, 6.33)
PC	China	Baseline RDE	0.28 (0.21, 0.36)	0.03	8.25 (6.19, 10.3)
PC	China	Strong RDE	0.06 (0.04, 0.07)	0.03	1.79 (1.34, 2.25)
PC	EU	Euro 3	0.82 (0.62, 1.03)	0.5	1.65 (1.24, 2.07)
PC	EU	Euro 4	0.80 (0.60, 1)	0.25	3.20 (2.40, 4)
PC	EU	Euro 5	0.80 (0.63, 1.13)	0.17	4.44 (3.5, 6.33)
PC	EU	Euro 6	0.45 (0.32, 0.85)	0.08	5.66 (4, 10.6)
PC	EU	Baseline RDE	0.32 (0.25, 0.38)	0.08	4 (3.14, 4.86)
PC	EU	Strong RDE	0.09 (0.08, 0.10)	0.08	1.17 (1.02, 1.32)
PC	Japan	1998	0.82 (0.62, 1.03)	0.55	1.50 (1.13, 1.88)
PC	Japan	2002	0.80 (0.60, 1)	0.28	2.85 (2.14, 3.57)
PC	Japan	2005	0.80 (0.63, 1.13)	0.14	5.71 (4.5, 8.14)
PC	Japan	2009	0.42 (0.32, 0.85)	0.08	5.32 (4, 10.6)
PC	Japan	Strong RDE	0.09 (0.08, 0.10)	0.08	1.17 (1.02, 1.32)
LDV	US	Tier 1	0.85 (0.65, 1.08)	0.78	1.1 (0.8, 1.4)
LDV	US	Tier 2	0.22 (0.16, 0.29)	0.04	5.0 (3.8, 6.7)
LDV	US	Tier 3	0.01 (0.01, 0.02)	0.01	1.3 (1.0, 2.0)
HHDT	China	Euro III	9.40 (7.05, 11.7)	6.27 (4.56, 8.36)	1.49 (0.84, 2.57)
HHDT	China	Euro IV	6.09 (4.57, 7.62)	4.16 (3.02, 5.55)	1.46 (0.82, 2.51)
HHDT	China	Euro V	4.90 (3.67, 6.12)	2.31 (1.68, 3.09)	2.11 (1.18, 3.63)
HHDT	China	Euro VI	0.54 (0.29, 0.91)	0.54 (0.39, 0.73)	1 (0.40, 2.29)
HHDT	EU	Euro III	7.83 (5.87, 9.80)	6.26 (4.55, 8.35)	1.25 (0.70, 2.15)
HHDT	EU	Euro IV	5.66 (4.24, 7.07)	4.31 (3.13, 5.74)	1.31 (0.73, 2.25)
HHDT	EU	Euro V	4.54 (3.41, 5.68)	2.40 (1.74, 3.20)	1.89 (1.06, 3.25)
HHDT	EU	Euro VI	0.54 (0.29, 0.91)	0.54 (0.39, 0.73)	1 (0.40, 2.29)
HHDT	Japan	Japan 1997	7.83 (5.87, 9.80)	5.70 (4.14, 7.60)	1.37 (0.77, 2.36)
HHDT	Japan	Japan 2003	5.66 (4.24, 7.07)	4.33 (3.14, 5.77)	1.30 (0.73, 2.24)
HHDT	Japan	Japan 2005	4.54 (3.41, 5.68)	2.50 (1.81, 3.33)	1.81 (1.02, 3.12)
HHDT	Japan	Japan 2009	0.84 (0.46, 1.40)	0.84 (0.61, 1.12)	1 (0.40, 2.29)
HHDT	Japan	Japan 2016	0.43 (0.23, 0.72)	0.43 (0.31, 0.58)	1 (0.40, 2.29)
HHDT	US	EPA 1998	11.6 (8.75, 14.6)	8.16 (5.93, 10.8)	1.43 (0.80, 2.45)
HHDT	US	EPA 2004	5.84 (4.38, 7.31)	4.01 (2.92, 5.35)	1.45 (0.81, 2.50)
HHDT	US	EPA 2007	4.17 (3.13, 5.22)	2.36 (1.71, 3.15)	1.76 (0.99, 3.03)
HHDT	US	EPA 2010	0.72 (0.54, 0.91)	0.36 (0.26, 0.48)	2.00 (1.12, 3.44)
Bus	China	Euro III	12.5 (11.1, 13.8)	4.24 (3.08, 5.65)	2.94 (1.97, 4.47)
Bus	China	Euro IV	11.8 (9.80, 13.8)	2.98 (2.16, 3.97)	3.95 (2.46, 6.36)
Bus	China	Euro V	7.45 (5.58, 9.31)	1.66 (1.20, 2.21)	4.48 (2.52, 7.71)
Bus	China	Euro VI	0.61 (0.33, 1.01)	0.40 (0.29, 0.54)	1.5 (0.61, 3.43)
Bus	EU	Euro III	10.0 (7.55, 12.5)	4.80 (3.49, 6.41)	2.09 (1.17, 3.59)
Bus	EU	Euro IV	7.25 (5.43, 9.06)	3.27 (2.37, 4.36)	2.21 (1.24, 3.81)
Bus	EU	Euro V	7.01 (5.26, 8.77)	1.79 (1.30, 2.38)	3.92 (2.20, 6.73)
Bus	EU	Euro VI	0.61 (0.33, 1.01)	0.40 (0.29, 0.54)	1.5 (0.61, 3.43)
Bus	Japan	Japan 1997	10.0 (7.55, 12.5)	4.39 (3.19, 5.85)	2.29 (1.29, 3.94)
Bus	Japan	Japan 2003	7.25 (5.43, 9.06)	3.29 (2.39, 4.39)	2.20 (1.23, 3.78)
Bus	Japan	Japan 2005	7.01 (5.26, 8.77)	1.89 (1.37, 2.52)	3.70 (2.08, 6.36)
Bus	Japan	Japan 2009	0.93 (0.50, 1.55)	0.62 (0.45, 0.82)	1.5 (0.61, 3.43)
Bus	Japan	Japan 2016	0.48 (0.26, 0.80)	0.32 (0.23, 0.42)	1.5 (0.61, 3.43)
Bus	US	EPA 1998	12.8 (12.4, 13.3)	6.03 (4.39, 8.05)	2.13 (1.54, 3.02)
Bus	US	EPA 2004	8.41 (7.96, 8.84)	2.95 (2.14, 3.93)	2.85 (2.02, 4.12)
Bus	US	EPA 2007	4.08 (3.41, 4.78)	1.70 (1.24, 2.27)	2.39 (1.49, 3.84)
Bus	US	EPA 2010	0.93 (0.70, 1.17)	0.26 (0.19, 0.35)	3.51 (1.97, 6.04)

Values in parentheses indicate uncertainty ranges, calculated as described in the Methods. PC, passenger car.