

# Air Quality Impacts of Climate Mitigation: UK Policy and Passenger Vehicle Choice

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In 2001–2002 the UK began taxing vehicles according to CO<sub>2</sub> emission rates. Since then, there has been a significant increase in consumer choice of small cars and diesel engines. We estimate CO<sub>2</sub> reductions and air quality impacts resulting from UK consumers switching from petrol to diesel cars from 2001 to 2020. Annual reductions of 0.4 megatons (Mt) of CO<sub>2</sub> and 1 million barrels of oil are estimated from switching to diesels. However, diesels emit higher levels of particulate matter estimated to result in 90 deaths annually (range 20–300). We estimate 570, 460, and 0 additional deaths per Mt of CO<sub>2</sub> abated, for Euro III, Euro IV, and post-Euro IV emission class vehicles, respectively. CO<sub>2</sub> policies are suspected to have contributed substantially to diesel growth, but the magnitude of impact has yet to be quantified rigorously. To the extent that CO<sub>2</sub> policies contribute to diesel growth, coordinating CO<sub>2</sub> controls with tightening of emission standards would save lives. This research shows that climate policy, while reducing fuel use and CO<sub>2</sub>, does not always ensure ancillary health benefits. Lessons from the UK can help inform policies designed elsewhere which strive to balance near-term ambient air quality and health with long-term climate mitigation.

## 1. Introduction

Climate mitigation policies have been promoted on the basis that reducing fossil fuel use provides dual benefits in terms of long-term climate change attenuation and short-term air quality improvements. Models predict that climate policies result in reduced fossil fuel combustion and lower air emissions, and subsequently provide public health benefits (1–4).

In Europe, diesel cars are viewed as a promising option to reduce greenhouse gas emissions from personal transportation. Diesel fuel has a higher energy and carbon density than petrol (38.5 MJ/L gross heating value and 778 g C/L, versus 34.9 MJ/L and 659 g C/L, but diesels have 25% better fuel economy based on matched pair vehicle models and thus emit 15–20% less CO<sub>2</sub> per kilometer (5, 6). Beginning March 2001 a new vehicle excise duty (VED) was introduced in the UK whereby vehicles are taxed annually based upon

certified CO<sub>2</sub> emissions (7). In April 2002 the UK's company car benefit-in-kind tax was changed to a CO<sub>2</sub>-based system as well (8). Although a surcharge for diesel vehicles was applied to reflect their impact on air quality, the cost of owning diesel cars was and is lower (see illustrative cost comparison in Supporting Information Table S1). These CO<sub>2</sub> policies are credited with contributing to the UK's success in reducing CO<sub>2</sub> emissions (8–10). Emissions of CO<sub>2</sub> from new passenger vehicles registered in the UK between 2000 and 2005 have fallen from a fleet average of 181 to 169 g of CO<sub>2</sub> per km (10). Diesel vehicles have made a contribution to the CO<sub>2</sub> reductions as their market share has grown exponentially at an annual compounded rate of 21% from 2001 to 2005. As shown in Figure 1, diesel market share in the UK had declined from 1995 onward while diesels were gaining market share in the rest of the European Union (EU) (11). The changes in the tax regime are demonstrably the turning point for the resurgence of diesels in the UK.

On a parallel track and lagging the CO<sub>2</sub> policies, new EU emission standards for passenger vehicles are being promulgated to converge emissions from diesels and petrol engines to the best that either technology can achieve. Thus, 2001 Euro III standards and 2006 Euro IV standards both drive down the higher emissions of PM<sub>10</sub> (particulate matter less than 10 micrometers) and NO<sub>x</sub> (nitrogen oxides) from diesels, and force gasoline engines to lower their CO (carbon monoxide) and HC (hydrocarbons) emissions.

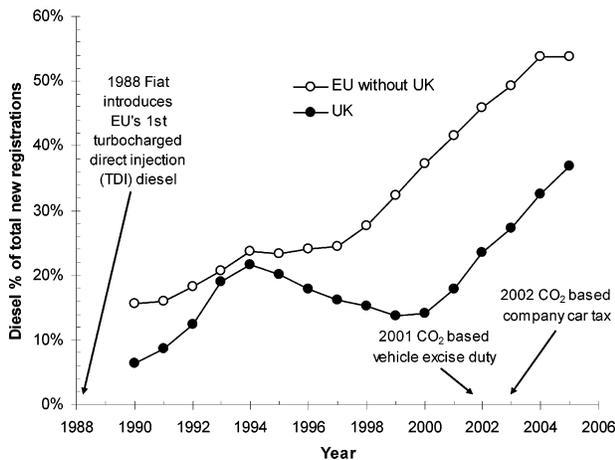
Here we examine consumer switching from petrol to diesel-fueled passenger cars in the UK (all vehicles designed to carry passengers, but excluding freight vehicles). The tradeoffs between greenhouse gas reductions and air pollution impacts are assessed over a 20 year study period from 2001 to 2020. While diesel substitution for petrol cars scenarios have been assessed elsewhere (12), this is an empirical study in the context of an actual CO<sub>2</sub> policy and targets an acknowledged gap in the climate policy literature (13, 14). These research findings have direct relevance to the design of climate policies in the transportation sector throughout the developed world.

## 2. Methods

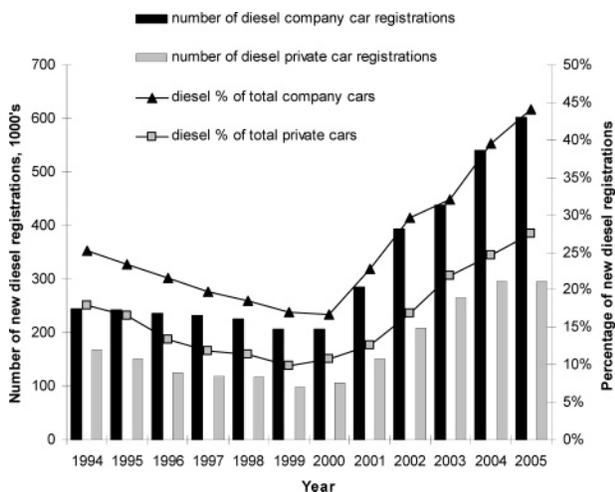
We define “additional diesels” as the number of newly registered diesel vehicles additional to an estimated “no growth” diesel market share. Growth in new diesel registrations is estimated relative to historical average market share for private and company car registrations (roughly half of all newly registered cars in the UK are for company fleets). Figure 2 shows 1994–2005 data for new diesel registrations for private and company vehicles. To quantify the overall growth in diesel vehicles, we assign 15% as the “no growth” diesel market share from 2001 to 2020. This is based on the average diesel share for 11 years from 1990 to 2000 leading up to the CO<sub>2</sub> policies, and agrees closely with government projected diesel share (15). Factors that likely have contributed to the growth in diesel market share, including CO<sub>2</sub> policy, are described in the Discussion section.

An annual time series of additional diesels for 2001–2020 is developed by adding new registrations and subtracting estimates of scrapped cars. An estimate of the yearly number of scrapped cars is made using a model developed and calibrated using UK deregistration data as described in Figure S1 of the Supporting Information (16). The annual time series of additional diesels is used to estimate changes in fuel consumption, emissions of CO<sub>2</sub> and common air contaminants, and human health effects.

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**FIGURE 1. Diesel share of new car registrations in the European Union (EU). While aggregate EU demand for diesels began increasing in 1995, UK demand continued to decline until the first CO<sub>2</sub> policy incentive came into effect in 2001 and has been increasing continuously.**



**FIGURE 2. Total number and percentage market share of new registrations of private and company diesel cars in the UK 1994–2005. During this period, the ratio of the price of petrol to diesel was remarkably stable, averaging 0.98 (range 0.95–1.00) and fuel price advantages experienced elsewhere do not provide a plausible explanation for the observed changes in diesel registrations in the UK.**

Changes in fuel consumption and CO<sub>2</sub> emissions are estimated based on the difference between petrol and diesel fleet average emission factors shown in Figure S2 of the SI. These emission factors use UK-specific, certified CO<sub>2</sub>/km (17) and are based on actual fleet average CO<sub>2</sub> emissions from 2001 to 2005 (10), and projected fleet average emissions through 2020. Changes in fuel consumption are estimated using fleet CO<sub>2</sub>/km and conversion factors of 2763 g CO<sub>2</sub>/L for diesel and 2504 g/L for petrol (18). We highlight that fleet average diesel CO<sub>2</sub>/km is only 4% lower than petrol as of model year 2005, because diesel technology's inherent advantage has been offset by consumer preference for larger cars (10).

Changes in emissions of common air contaminants are estimated based on UK-specific emission factors for PM<sub>2.5</sub> (2.5 micrometer aerodynamic diameter particulate), NO<sub>x</sub>, CO, HC, benzene, and 1,3 butadiene as provided in Table S2 of the SI. As with CO<sub>2</sub>, changes in emissions are quantified based on the difference between emission factors for petrol and diesel cars. Emissions are estimated for each year from 2001 to 2020 based on the applicable EU emission class (Euro

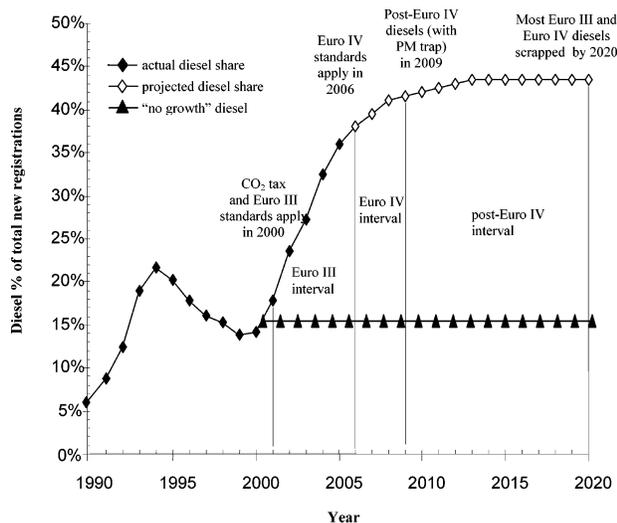
III, Euro IV, post-Euro IV), and using the UK-specific emission factors derived based on measurements under actual driving conditions (15). Automobile usage is a constant 15,800 km annually in our analysis, based on average driving distances for all UK cars (19). We assumed the same pattern of vehicle usage regardless of type (diesel or petrol) because of the high market share of diesels, and did not include a rebound effect (higher usage) due to the lower operating cost of diesels. A sensitivity estimate of the rebound effect is provided in the discussion. Further data and Discussion of the annual mileage assumptions are contained in the SI.

Human health impacts resulting from changes in common air contaminants are estimated solely on changes in particulate matter emissions employing a conventional *impact pathway* method (20, 21). The rationale for using particulate matter is that it tends to dominate human health impacts from air pollution based on current science (20, 22–24). Moreover, a recent study of transportation and air pollution in the UK concluded that reduction of PM<sub>10</sub> should be the top priority if the goal is to reduce air pollution impacts on public health in the UK (25). Another recent source apportionment study of London also concluded that controls on particulate matter emissions from vehicles are most likely to result in the greatest improvements in ambient particulate matter concentrations (26). The impact of excluding ambient ozone, NO<sub>x</sub>, CO, HC, benzene, and 1,3-butadiene in the quantification of health effects is addressed in the Discussion section and in the SI.

Published emission factors and modeling studies for the UK are based exclusively on PM<sub>10</sub>. However, in general, more than 99% of particulate emissions by mass from diesel cars are PM<sub>2.5</sub> (27). Hence, for all practical purposes, *changes in PM<sub>10</sub> and PM<sub>2.5</sub> emissions are equal*. This allows the use of the health coefficients that are based on either PM<sub>10</sub> or PM<sub>2.5</sub>. We use concentration–response coefficients based on PM<sub>10</sub> and on PM<sub>2.5</sub> where supporting evidence is available. To maintain clarity, for the remainder of this article we will only refer to particulate matter emissions or ambient concentrations as PM<sub>2.5</sub>.

We quantify changes in ambient PM<sub>2.5</sub> concentrations by employing the results of atmospheric modeling studies of vehicle emission reduction measures in the UK (28–30). The analysis scenario most applicable to our research estimated that an annual UK-wide reduction of 3.76 kt of particulate matter due to traffic emission controls would result in a UK-wide, population-weighted change in average annual ambient concentration of 0.277 μg/m<sup>3</sup>. This is a ratio of 0.0737 μg/m<sup>3</sup> change in annual mean ambient PM<sub>2.5</sub> concentration per 1 kt change in annual PM<sub>2.5</sub> emissions, which includes a component of secondary nitrate particles as described in the SI (29). This ratio was multiplied by our estimated annual change in PM<sub>2.5</sub> emissions to estimate changes in population-weighted ambient PM<sub>2.5</sub> concentrations. This PM<sub>2.5</sub> ambient concentration estimate is equivalent to an intake fraction of 20 grams per million grams emitted—in general agreement with estimates of vehicle emissions found elsewhere (31, 32).

For the association between changes in ambient PM<sub>2.5</sub> and mortality, we employ low, central, and high concentration–response coefficients. Our central mortality coefficient uses the results of the American Cancer Society cohort study from Pope and colleagues (2002) which found a 4% increase in chronic, all cause mortality per 10 μg/m<sup>3</sup> increase in PM<sub>2.5</sub> for a cohort of subjects age 30 and older in the United States (33). We chose this study because it is by far the largest cohort study published, it estimated chronic rather than acute mortality, it used the same exposure metric as we estimate in our study (annual average concentration), and is the preferred study for chronic mortality estimates according to the latest World Health Organization guidelines (34). For our low mortality coefficient, we use 0.75% change in mortality



**FIGURE 3. Integrated framework for assessing emissions from additional diesels (i.e., diesels substituted for petrol vehicles). Actual diesel share of new registrations from 1990 to 2005 are based on industry data. Projected shares from 2006 to 2007 are based on industry forecasts, and from 2008 to 2020 are based on authors' projections. The focus of this study is on the area between the actual/projection curve and the "no growth" curve which is split into three time intervals defined by the applicable emission standard: Euro III, Euro IV, and post-Euro IV.**

annually per  $10 \mu\text{g}/\text{m}^3$  change in  $\text{PM}_{2.5}$ , applied to the entire UK population. This coefficient was derived through meta-analyses by the UK Department of Health Expert Committee on the Medical Effects of Air Pollution (COMEAP) applying the results of time series epidemiological studies (35). We use this coefficient because it has been endorsed specifically for the UK, however there are arguments that time series results are inappropriate for estimating long-term death rates (36, 37). For our high mortality coefficient, we use the results of the Harvard Six Cities study which found a 13% increase in chronic mortality per  $10 \mu\text{g}/\text{m}^3$  change in  $\text{PM}_{2.5}$  for a cohort of about 8,000 subjects aged 25 and older in the United States (38). More recent research on intra-urban variation in  $\text{PM}_{2.5}$  and mortality in Los Angeles indicates that the true mortality coefficient may be somewhere between the central and high coefficients we employ in this study (39).

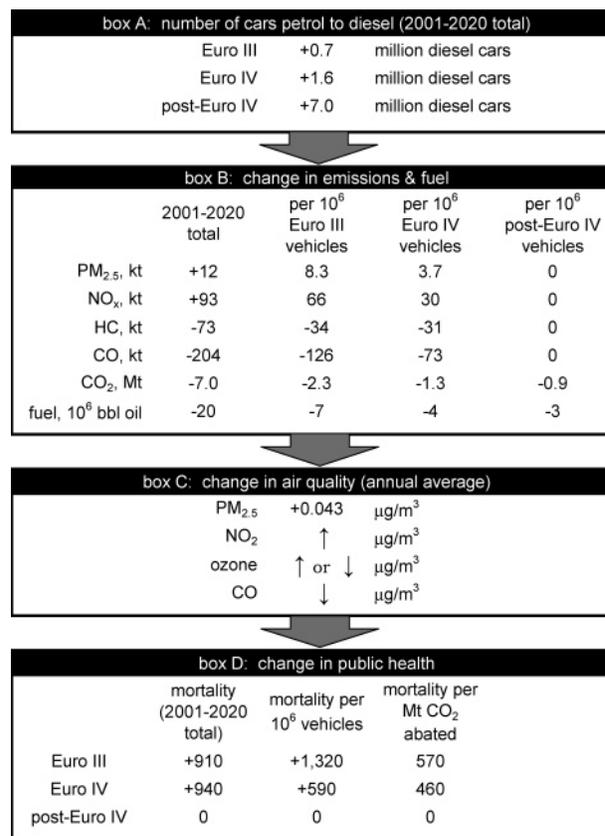
We quantify morbidity by estimating respiratory and cardiovascular hospitalizations using rate coefficients adopted by COMEAP (28). Changes in both hospitalization rates are estimated at 0.8% per  $10 \mu\text{g}/\text{m}^3$  change in  $\text{PM}_{2.5}$  applied to the entire UK population.

To tie together the linked steps in this analysis, an integrated framework is provided in Figure 3 that illustrates the timing of  $\text{CO}_2$  policy incentives, EU emission standards, and changes in diesel market share over time. To assess timing of the  $\text{CO}_2$  policies and EU vehicle emission standards, we divide the study time horizon into three intervals: a "Euro III interval" from 2001 to 2005 where Euro III emissions standards apply, a "Euro IV interval" from 2006 to 2008, and the "post-Euro IV interval" from 2009 to 2020 when progressively higher emission standards apply to all new vehicle purchases. Early adoption of Euro IV diesels is incorporated into our estimates as described in the SI.

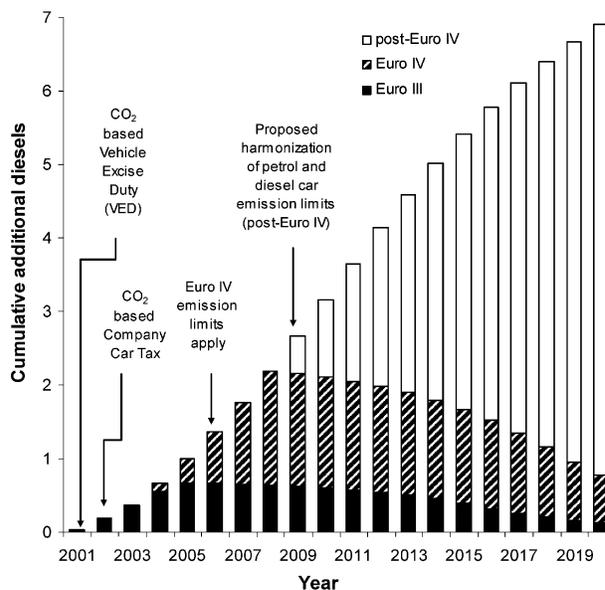
### 3. Results

Figure 4 summarizes all the main results of this study. The estimated additional diesel vehicle counts (box A), emissions and fuel consumption (box B), ambient concentrations (box C), and health effects (box D) are shown.

The estimated number of new additional diesels over the 20 year study period is 0.7 million for Euro III, 1.6 million for

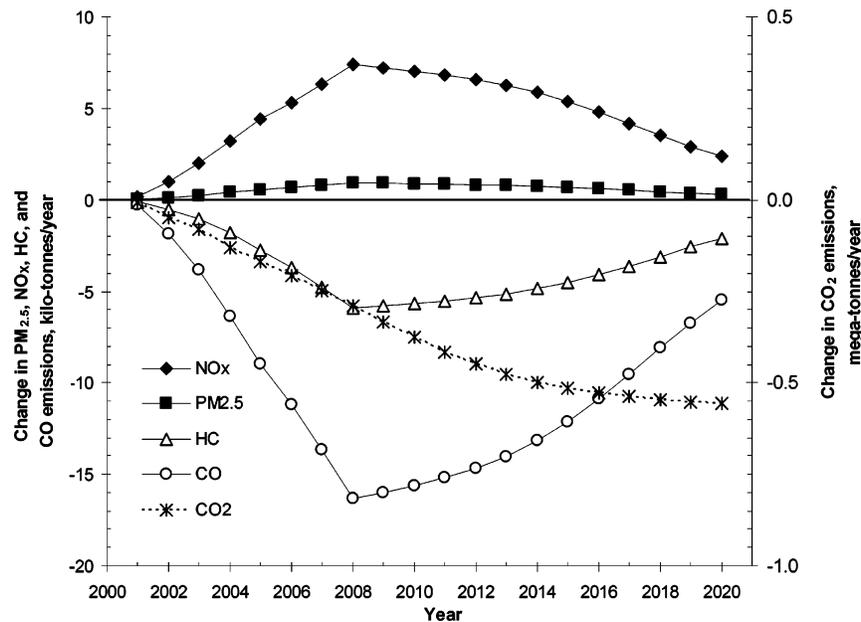


**FIGURE 4. Summary Results of the Impact of Additional Diesels in the UK from 2001 to 2020.**



**FIGURE 5. Estimates of additional diesels in the UK 2001–2020 disaggregated by Euro emission class. "Additional diesels" are defined as the number of petrol vehicles switched to diesel beyond the "no growth" estimate. Euro III and Euro IV emission standards apply in 2001 and 2006, respectively. Early adoption of some Euro IV diesels is incorporated into our estimates. Legislation to harmonize diesel and petrol particulate matter emission limits is proposed by 2009, described as "post-Euro IV" in this study.**

Euro IV, and 7 million for post-Euro IV. Figure 5 shows the cumulative time series of additional diesels disaggregated by emission class, which is quantified by adding newly registered cars less the estimated number of scrapped cars.



**FIGURE 6.** Estimated changes in emissions from 2001 to 2020 due to additional diesels. The solid lines (y-axis to left) show estimated changes in emissions of common air contaminants in kilotons (kt), while the dashed line (y-axis to right) shows CO<sub>2</sub> in megatons (Mt). Diesels emit higher rates of PM<sub>2.5</sub> and NO<sub>x</sub>, and lower rates of HC, CO, and CO<sub>2</sub>. Emissions of common air contaminants are assumed to be harmonized for diesel and petrol vehicles beginning 2009, so differences in all emissions except CO<sub>2</sub> approach zero from 2009 to 2020 as higher polluting diesels are scrapped.

Over the 20 year study period, additional diesels are estimated to increase PM<sub>2.5</sub> and NO<sub>x</sub> emissions by 12 kt and 93 kt, respectively. HC and CO emissions are estimated to decrease by 73 kt and 204 kt, respectively. CO<sub>2</sub> emissions are estimated to decrease by 7 Mt. These total changes in emissions were obtained by integrating the annual time series of emissions shown in Figure 6 over the 20 year study period. It is important to note that the lower polluting emission classes provide progressively lower CO<sub>2</sub> and fuel saving benefits.

As a result of the 12 kt increase in particulate emissions, the average annual change in ambient PM<sub>2.5</sub> concentration was estimated to be 0.043 μg/m<sup>3</sup> over the 20 year study period. Ambient concentration of NO<sub>2</sub> is determined to increase by an unquantified amount, while CO decreases. Changes in ambient ozone concentrations were not estimated because of the complexity and uncertainty in atmospheric chemistry that would result from an increase in NO<sub>x</sub> and a decrease in HC, the two principal precursors. No known UK studies have modeled such a scenario. The available studies conclude that ozone formation might be limited by NO<sub>x</sub> or HC under different conditions (40). The most recent study indicated that reductions in HC usually improved ozone while “NO<sub>x</sub> emission control gave a more complex response, which was metric and region specific. Generally NO<sub>x</sub> emission control had an adverse effect on ozone air quality.” (41) Therefore we show that ambient ozone levels may increase or decrease as indicated in Figure 4.

Over the 20 year study period, the average annual mortality is estimated at 90 deaths per year applying the central concentration–response coefficient. The low and high mortality estimates are 20 and 300 deaths per year, respectively. Annual average hospitalizations total 32 per year. All mortality and morbidity impacts are associated with the Euro III and Euro IV additional diesels, based on the assumption of harmonization of emissions for diesel and petrol in the post-Euro IV standards. Included in box D of Figure 4 are the ratios of the central mortality estimate per million additional diesels and per Mt of CO<sub>2</sub> abated, averaged over the 20 year study period. The additional mortality per million Euro III diesels and Euro IV diesels is estimated at 1320 deaths and

590 deaths, respectively. The average mortality per Mt of CO<sub>2</sub> abated is 570 for Euro III, 460 for Euro IV, and 0 for post-Euro IV.

#### 4. Discussion

There are many uncertainties and limitations associated with this analysis. The SI document contains a detailed discussion of how these uncertainties and limitations are likely to have over- or underestimated fuel savings, emissions, exposure, and health effects estimates in the following five areas:

(1) The number of registered vehicles subject to successive EU emission standards is not stated in the data. We used the range of manufacturer offerings meeting the different standards as the guide to apportion registrations to each emission class. Our assumptions with respect to emission class likely result in a low bias of the estimated air quality and health impacts.

(2) The assumption of equal annual mileage for petrol and diesel does not account for variation in driver types (company car and high-mileage drivers), nor an economic rebound effect. Our assumption of driver types is consistent with a saturation of high mileage drivers choosing diesels as described by Schipper (5). In general there is a measured rebound effect elasticity of 0.1–0.3% increase in annual mileage for every 1% decrease in fuel costs per kilometer (42). Therefore our annual mileage assumptions likely underestimate air quality impacts and overestimate CO<sub>2</sub> reductions and fuel savings.

(3) Available evidence indicates that our assumptions of spatial distribution of vehicles (urban/rural/motorway) does not substantially bias our estimates high or low.

(4) Our assumptions of PM<sub>2.5</sub> emissions and ambient concentrations, including meteorology, secondary particulate matter (sulfate, nitrates, and organic), fuel quality, brake and tire wear, and vehicle age likely bias our estimates high in some cases and low in others. Overall, the air pollutant emissions and ambient concentrations are likely underestimated.

(5) Health effects are likely underestimated, specifically the limitation of estimating health outcomes solely from

PM<sub>2.5</sub>. The likely impacts on ambient levels and subsequently health effects of ozone, SO<sub>2</sub>, HC, NO<sub>x</sub>, CO, benzene, and 1,3-butadiene are discussed further in the SI.

Given these various sources of uncertainty, we think our basic findings are robust. Overall, the study may be biased in overestimating fuel savings and CO<sub>2</sub> emissions and underestimating the health-damaging emissions and associated health impacts. Additional disbenefits may also be present that have not been considered in this study, namely black carbon emissions from diesel engines and their exacerbation of climate change (43, 44).

We estimate that consumers switching from petrol to diesel cars in the UK over the time period of 2001–2020 will reduce CO<sub>2</sub> by 7 Mt and save 20 million barrels of oil. However, ancillary air quality effects hinge upon the fuel properties and conversion technology, not just the quantity of fuel consumed, and adverse air quality is estimated to result in 90 additional deaths annually (range 20–300). The CO<sub>2</sub> reductions, fuel savings, and additional mortality are not necessarily all attributable to the CO<sub>2</sub> policies. Econometric models are a valuable tool to explain changes in consumer choice of fuel and vehicle types (45), but to our knowledge no econometric models have been developed to accurately quantify UK CO<sub>2</sub> policy influence on diesel growth. Various reports attribute no impact to the VED CO<sub>2</sub> tax (7), while estimates of the impact of the company car CO<sub>2</sub> tax on company car diesel growth range from 33% (8) to 100% (9). However, other factors such as the European manufacturer voluntary CO<sub>2</sub> program, oil prices, and technological change are potential influences as well (8, 10, 46), but not fuel price ratio or taxes (Figure 2). To the extent that CO<sub>2</sub> policies contributed to diesel growth, coordinating CO<sub>2</sub> controls with tightening of emission standards would save lives. Because of the uncertain CO<sub>2</sub> policy impact, our estimates of emissions, fuel savings, and health impacts per unit quantity of additional diesel cars (Figure 4) are emphasized. As the rest of the EU and other developed countries prepare to integrate transportation into climate mitigation programs, the lessons learned from the UK experience can help inform the design of climate policies in the transport sector strive to better balance near-term health effects with long-term climate mitigation.

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## Supporting Information Available

Additional discussion, tables, and figures. This material is available free of charge via the Internet at <http://pubs.acs.org>.

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