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Energy Policy

Beyond 'Dieselgate': Implications of unaccounted and future air pollutant emissions and energy use for cars in the United Kingdom

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HIGHLIGHTS

- Explores the recent 'Dieselgate' affair and potential policy responses.
- Impacts on human health of 'real world' NOX emissions in the UK are significant.
- De-dieselization can have air quality benefits while showing few carbon disbenefits.
- Electrification shows largest 'co-benefits' but needs transformative approach.
- Electrification may mean faster decline in road tax revenues.

ARTICLE INFO

ABSTRACT

The 'Dieselgate' emissions scandal has highlighted long standing concerns that the performance gap between 'real world' and 'official' energy use and pollutant emissions of cars is increasing to a level that renders 'official' certification ratings virtually ineffective while misleading consumers and damaging human health of the wider population. This paper aims to explore the scale and timing of historic and future impacts on energy use and emissions of the UK car market. To achieve this aim it applies a bespoke disaggregated model of the transport-energy-environment system to explore the impacts of retrospective and future policy scenarios on the UK car market, trade-offs between greenhouse gas and air quality emissions, and fuel use and associated tax revenues. The results suggest that the impacts on human health of 'real world' excess NO$_X$ emissions in the UK are significant. Future 'low diesel' policies can have significant air quality benefits while showing few (if any) carbon disbenefits, suggesting future car pricing incentives may need to be rebalanced taking more account of effects of local air pollution. Car pricing incentives are however unlikely to transform the car market without additional market changes, industry push, infrastructure investment and policy pull aimed at cleaner, lower carbon vehicles.

1. Introduction

1.1. The challenge – and potential opportunity?

The 'Dieselgate'[1] emissions scandal has highlighted growing concerns that the performance gap between 'real world' and 'official' energy use and air pollutant emissions of road vehicles is increasing to a level that renders 'official' certification ratings virtually ineffective while misleading consumers and damaging human health of the wider population. While real world CO$_2$ emissions have been shown to be on average a third higher (CCC, 2015b; ICCT, 2014a; TandE, 2015a), NO$_X$ emissions can be up to 40 times higher than official certification values and standards operating in the EU (Hagman et al., 2015; ICCT, 2014b; Weiss et al., 2012), the US (Barrett et al., 2015) and China (Lau et al., 2015; Shen et al., 2015). For CO$_2$, the gap between test results and real-world

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[1] While there is no official definition of the term 'Dieselgate', it has become synonymous with the use by the Volkswagen Group of a 'defeat device' that detects when a diesel car is undergoing an official emissions tests and optimises engine performance to minimise air pollutant emissions to meet stringent emissions regulations. The device is only activated during the official test. Vehicles by other manufacturers have also been shown to exceed emissions in real world driving conditions; however, there has been no evidence of 'defeat devices' being used outside Volkswagen.

performance has increased from 8% in 2001–31% in 2012 and 40% in 2014 (TandE, 2015a). Up to nine out of ten diesel cars on Europe's roads are said to break European NO$_X$ pollution limits,[2] and the current generation of Euro 6 vehicles have been shown to emit, on average, seven times more NO$_X$ than certified European values (Beevers et al., 2012; Dehmer, 2015; ICCT, 2014b). While the emissions scandal began in the US, little quantified evidence exists on the effects of excess emissions in the UK, which is a larger market for diesel cars. There is also a lack of quantification of the potential trade-offs between human health and climate change mitigation effects in the UK.

Given that transport accounts for 46 per cent of all NOx emissions in the region (EEA, 2015), the performance gap partly explains why NOx emissions in many European countries continue to miss targets (Beevers et al., 2012). The other reasons are the growth in overall traffic and the increase in the diesel market share for cars. In the UK, for instance, diesel vehicles accounted for fewer than 1 per cent of cars on the road in 1984 – last year that figure had risen to more than a third, with new registrations totalling about 1.35 million (or half) of all new cars in 2014 (SMMT, 2014). This compares to the almost 1.2 million VW diesel vehicles affected in the UK, and about 11 million VW diesel vehicles worldwide (Sheffield, 2015; Yeomans, 2015).

1.2. Lack of effective policy response in the UK

The human health 'costs' of diesel related air pollution (see Supplementary material S1 for further details) are highly policy-relevant. The UK Government has been subject to legal proceedings for failing to meet European Limit Values for NO$_2$, and their plans to reduce NO$_2$ concentrations and meet these limits was submitted to the European Commission at the end of 2015 (DEFRA, 2015a). Some major urban hotspots will continue to exceed EC limits for another decade to come (DEFRA, 2015a) as effective mitigation of local air pollution is proving to be an enormous challenge in cities across the globe (Barrett et al., 2015; Carrington, 2016; Walton et al., 2015; Woodcock et al., 2009). The policy response so far has been slow and ineffective as the reliance on updating the vehicle type-approval testing procedure and associated legislation are still on-going (CCC, 2015b) and will not have significant effects for another decade or more.

The proposed policy and technological solutions include actions at national and local levels: a national diesel car scrappage scheme; a ban on (older) diesel vehicles in cities; the requirement for new taxis to be ULEV by a certain date; rebalancing of national fuel duty and road tax (Vehicle Excise Duty, or VED) consistent with reducing not just CO$_2$ emissions but also NO$_2$ and particulate matter (PM); and national monetary incentives for switching to cleaner vehicles including ultra-low emission cars, taxis, vans and buses (DEFRA, 2015b; UK EAC, 2015). The quantified effects of these measures in the UK are largely unknown.

Partly fuelled by the 'Dieselgate' affair, the electrification of transport has gained further momentum, with Germany announcing it is to provide financial incentives of around €5000 for people to buy electric cars (EurActiv, 2015a, 2015b) and the UK extending its plug-in vehicle grant (£5000 per ULEV) that has led to a 1% take up of plug-in vehicles amongst total new vehicle registrations of 2.6 million in 2015 (SMMT, 2016). Some commentators suggest that continuing reductions in battery prices will bring the total cost of ownership of plug-in vehicles below that for conventional-fuel vehicles by 2025, even with low oil prices (BNEF, 2016). However, doubts remain whether plug-in vehicles are direct

replacements of incumbent technologies or perceived as higher risk investments, thus limiting potential take-up (AEA Technology, 2009; BERR and DfT, 2008; Offer et al., 2011).

Some commentators are favouring a diesel scrappage scheme similar to that of 2009/10, with diesel car owners being offered up to £2000 to scrap their car and buy an ultra low emission vehicle (ULEV) instead (Cellan-Jones, 2015; Kollewe, 2015a; Vaughan, 2014). However, such a scheme may come at a cost to the taxpayer in the hundreds of millions (Vaughan, 2014) and may be considered unfair as it constitutes a direct subsidy to existing car owners. Moreover, as diesel vehicles emit 15–20% less CO$_2$ than a petrol equivalent, they have also made a significant contribution to climate change mitigation – an argument that was explored in this paper by investigating the trade-offs between meeting air quality and climate change objectives (Kollewe, 2015b; van der Zwaan et al., 2013; Vaughan, 2016).

1.3. Aims and objectives of this paper

This paper addressed the above challenge and potential policy and market solutions in two ways. First, it quantified the human health impacts and associated costs of 'real world' excess NO$_X$ emissions in the UK context and compare this with an alternative pathway simulating a retrospective purchase penalty for diesel cars between 2009 and 2015. Second, it quantified the NO$_X$-related human health and climate change mitigation impacts of future policy scenarios aimed at the UK diesel car market.[3] This paper thus aims to fill existing gaps in the work going on relating to the assessment of 'real world' vs 'official' emissions and potential policy responses elsewhere which: (a) ignores the potential trade-offs between human health and climate change mitigation impacts; (b) lacks detailed analysis of how policy and market signals can change the evolution of the car market; (c) ignores wider fuel and/or vehicle life cycle emissions impacts in comparing different pathways; and (d) lacks investigating the impacts on transport fuel use and associated tax revenues.

2. Approach, methods and data

2.1. Approach and choice of modelling tool

To achieve the above aims the study applied an existing and previously published transport-energy-environment modelling framework that has been applied in a number of policy modelling studies (Anable et al., 2011; Anable et al., 2012; Brand et al., 2013; Brand et al., 2012). The UK Transport Carbon Model (UKTCM) was the tool of choice for this analysis because it integrates a household car ownership model, vehicle consumer choice model, vehicle stock evolution model and vehicle and fuel life cycle emissions model in a single scenario modelling framework. The integrated transport sector tool is able to provide policy-focused conclusions which allow an assessment of the effectiveness of different policy instruments (including regulation, pricing and availability of charging infrastructure) on different vehicle *and* consumer segments. UKTCM has the ability to place the 'de-dieselization' and electrification of the car market in the context of other (low carbon) transport behaviours on the basis of their whole life cycle emissions and impacts on human health, including potential changes in the way in which cars are used, together with

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[2] European emission standards for vehicles http://ec.europa.eu/environment/air/transport/road.htm

[3] This paper focused on NO$_X$ pollution and standards as this was the issue surrounding the Dieselgate affair. It is important to note that the inclusion in the damage cost calculation of other pollutants, notably PM, may change the damage cost values. Since total multi-pollutant valuations were likely to increase the totals, the figures reported in this study can be considered on the conservative side.

the impacts on government tax revenue. It may therefore have a much broader remit and wider range of applications in scenario and policy analyses than, for instance, the top-down 'ASIF' (Schipper, 2011) decomposition framework, sectoral models that lack endogenizing consumer behaviour (Fontes and Pereira, 2014; Rogan et al., 2011), or integrated assessment models that by and large favour technology solutions and fuel shifts over travel activity and consumer behaviour modelling (Creutzig, 2015; Oxley et al., 2013).

The modelling framework was first applied in a UK case study to quantify the implications of the performance gap between 'real world' and 'official' energy use and air pollutant emissions of EURO5 and EURO6 diesel cars. In a forward looking modelling exercise it was then applied to explore the energy and emissions impacts of alternative policy scenarios of a CO_2-graded purchase/scrappage tax on diesel cars and an ambitious electrification pathways for UK car fleet. The modelling framework is briefly summarised below followed by describing the core methods, data and assumptions for the scenario portfolio used in this study.

2.2. The UK Transport Carbon Model (UKTCM): outline and key methods

The UKTCM is a highly disaggregated, bottom-up modelling framework of the transport-energy-environment system. Built around a flexible and modular database structure, it models annual projections of transport supply and demand, for all passenger and freight modes of transport, and calculates the corresponding energy use, life cycle emissions and environmental impacts year-by-year up to 2050. It takes a holistic view of the transport system, built around a set of exogenous scenarios of socio-economic, socio-technical and political developments. The model is technology rich and, in its current version, provides projections of how different vehicle technologies evolve over time for 770 vehicle technology categories,[4] including 283 car technologies such as increasingly efficient gasoline internal combustion engine (ICE) vehicles, hybrid electric vehicles (HEV), battery electric vehicles (BEV), plug-in hybrid electric vehicles (PHEV) and hydrogen fuel cell vehicles (HFCV). UKTCM played a key role in developing the *Energy2050* 'lifestyle' scenarios (Anable et al., 2011; Anable et al., 2012) for the UK Energy Research Centre (UKERC) and in exploring the effectiveness of low carbon car purchasing incentives in the UK (Brand et al., 2013). An overview of the model has been published in Brand et al. (2012). For the analysis presented in this paper, UKTCM was developed, updated and recalibrated from version 2.0 (as reported in Brand et al., 2013) to the current version 3.1.

Car technology evolution, use and impacts are modelled in four areas of modelling: (1) the car ownership model, (2) the car sales, choice and use model, (3) the direct energy use and emissions model and (4) the life cycle energy and environmental impacts model.

2.2.1. Modelling car sales, technology choice and use

New car sales are a function of total car ownership and car scrappage, with the latter modelled as a function of average life expectancy via a S-shaped (modified Weibull) scrappage probability curve (Brand, 2010; Brand et al., 2012). The new car market is first segmented into private and company/fleet markets, then

into three vehicle segments according to common definitions of car size and class (A/B – 'small', C/D – 'medium', E/F/G/H – 'large'). Using UK data to illustrate the segmentation, Fig. 1a shows the sales by ownership and size in 2013, highlighting the significance of the fleet/company market (52.5% of all new cars). This is an important distinction for modelling the car market as the fleet/company segment has historically been dominated by diesel car technology, and plug-in vehicles now taking in-roads in the same segment, but in smaller sizes. Fig. 1b shows the split by ownership and consumer segment, following the approach used by Element Energy and Aberdeen University for the Energy Technologies Institute in the UK (Element Energy, 2013).

The UKTCM's car choice model is a discrete choice model that estimates the purchase choice probability based on an assessment of overall vehicle 'attractiveness' (or 'utility') from amongst a set of vehicle choices (or 'alternatives'), each with their own financial and non-financial 'attributes'. The weighting of attributes varies across consumer segments, because consumers' opinions on the importance of different vehicle attributes vary. The model therefore reproduces the variation in utility of different vehicles across consumer segments, and the variation over time as vehicle attributes improve. The choice model was run for each vehicle segment and consumer segment, with the share of vehicle and consumer segments being kept constant in the Reference (REF) case. The modelling methods are described in more detail in the Supplementary material S2.

2.2.2. Energy and emissions from vehicle operation

In-use energy consumption (in volume and energy units) and air pollutant emissions (in tonnes of CO_2, NO_X, PM, CH_4, NMVOC, and so on) from motorised travel were computed by using disaggregate sets of emission factors, which were based on the results of large scale vehicle emissions testing programmes. For road transport, speed distributions for each vehicle type (car, motorcycle, van, HGV) and road segment type (urban, rural, motorway) were used to calculate energy consumption and emissions, based on average speed-emissions curves developed in previous research and emissions inventories such as COPERT II and III (EEA, 1998, 2000), MEET (Hickman et al., 1999), HBEFA (INFRAS, 2004) and NAEI (NETCEN, 2003). Emissions factors for road vehicles at normal operating temperatures (often called 'hot') were a polynomial function of average speed, with up to ten coefficients for each pollutant. The fitted average-speed emissions curves typically resemble U-shapes, with NO_X emissions curves showing relatively higher emissions at speeds above 100 km/h. The road transport module also takes account of 'cold start' effects, which mainly depend on ambient temperatures and trip distances. The approach allowed us to model the combined effects of different fleet compositions, different sets of emission factors (e.g. 'official' vs 'real world'), traffic characteristics, cold starts, fuel quality and driver behaviour. The UKTCM Reference Guide (Brand, 2010) describes the functional relationships in detail. 'Official' NO_X and CO_2 emissions factors and electricity consumption for cars are given in Supplementary material S3.

2.2.3. Life cycle energy use, emissions and impacts

Based on a typical environmental life cycle assessment framework (ICO, 2006), the UKTCM includes a life cycle inventory model and an environmental impacts assessment model. The former computes energy use and emissions (including primary energy and land use) from the manufacture, maintenance and disposal of vehicles; the construction, maintenance, and disposal of infrastructure; and the supply of energy (fuels). The latter then provides an assessment of the damage caused by calculating impact indicators and external costs. The life cycle inventory model uses the 'hybrid approach' of process-chain analysis and input-output

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[4] A UKTCM 'vehicle technology category' is defined as a typical representative of a combination of transport type (passenger or freight), vehicle type (e.g. motorcycle, car, HGV, train), vehicle size (e.g. small car of segment A/B, van, heavy truck, intercity rail), primary fuel type (e.g. gasoline, diesel, electricity), vintage (e.g. ICE Euro 5 2009–15, PHEV 'Euro 7' 2020–25) and hybridisation (ICE, HEV, PHEV). 'Vintaging' is used to simulate changes in performance, efficiencies, consumer preferences, costs and policy levers over time.
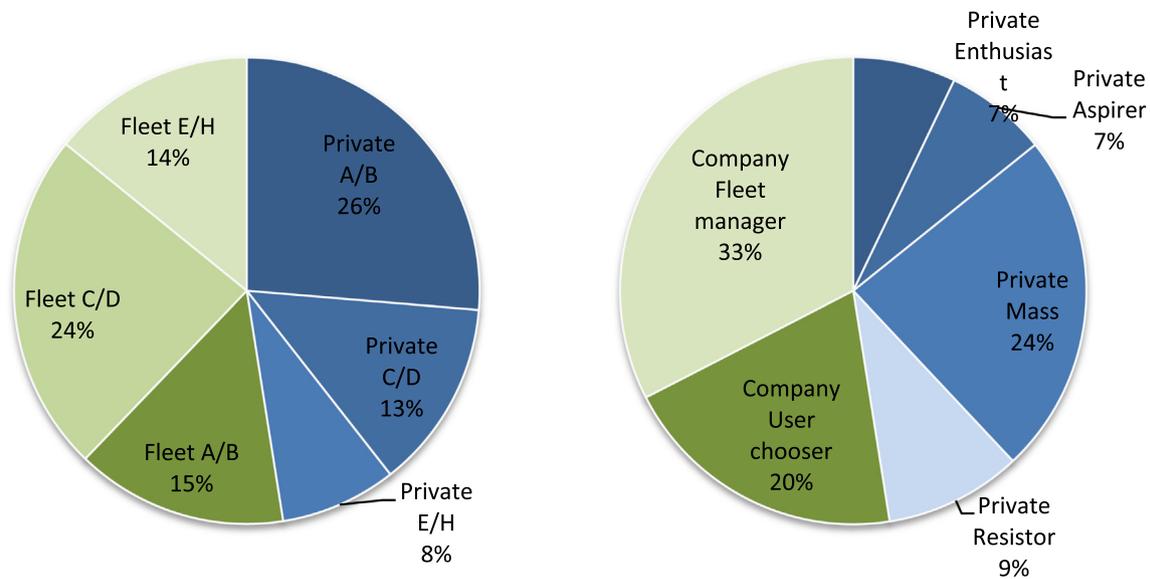
Fig. 1. a/b: Car sales (a: left) and consumer split (b: right) in the UK market (2013 data). Source: adapted from SMMT (2014), Element Energy (2013) and ETI (ETI, 2014)

analysis (Marheineke et al., 1998). Process chain analysis is used for the main supply paths, and aggregated values for complete process chains are used within the model. For additional upstream processes, considered to be second or third-order effects, input-output analysis is used. This hybrid approach is seen as appropriate as much of the evidence in the literature suggests that, in most cases, over the lifetime of a vehicle, vehicle operation produces the vast majority of energy use and emissions (Bastani et al., 2012; Lane, 2006; MacLean and Lave, 2003; Ogden et al., 2004; Sperling and Yeh, 2010). While the fuel supply and vehicle manufacture stages typically account for about 20% of total lifetime GHG emissions – being roughly equal in magnitude – vehicle maintenance and disposal account for a much smaller share (ibid.).

The methodology for determining external costs is based on an evaluation of marginal effects (marginal external costs/benefits), which were estimated using the Impact Pathway Approach developed previously in European research (Bickel et al., 2003; EC, 2005; Rabl and Holland, 2008; Rabl et al., 2014) and is commonly used in transport modelling and appraisal (DfT, 2014a; Macharis and Bernardini, 2015; Michiels et al., 2012; Mulley et al., 2013). For instance, average damage costs to human health of direct NO_X and $PM_{2.5}$ emissions by population density are shown in Table 1 below.[5] The UKTCM Reference Guide (Brand, 2010) describes the functional relationships and data used in detail.

2.2.4. Limitations of the approach used

There are some important limitations and uncertainties in the approach. The data underlying the car choice model used in the

future scenario analysis are based on stated preference and 'what-if' type assumptions on exogenous factors. More up-to-date evidence is needed on the characteristics, behaviours and attitudes of current diesel, gasoline and EV owners in the UK (towards revealed preference). In order to keep pace with the rapid development of the market and inform future policy making aimed at limiting damage to health from diesel vehicles and at the same time supporting the growth of the EV market, evidence on vehicle ownership and use should ideally be collected on a continuous or semi-regular basis (Brook Lyndhurst, 2015). This study has adopted a relatively simple analysis of linking emissions with impacts, the marginal damage costs approach based on aggregated results using the more detailed impact pathway approach. While the limited sensitivity analysis around the key factors determining damage costs on human health helps to explore the underlying uncertainty, further work may be required on impact modelling at the roadside and local levels, e.g. by linking place-based models such as UKTCM with integrated assessment models as has been done in some notable UK studies on cross-sectoral implications and climate 'co-benefits' in road transport pollution abatement (Oxley et al., 2015; Oxley et al., 2013; Oxley et al., 2012; Oxley et al., 2009).

2.3. The UK case study – modelling 'official' and 'real world' pollutant emissions

2.3.1. Reference pathway (REF) – key data and assumptions

UKTCM v3.1 was calibrated to UK national statistics for the year

Table 1
Damage costs per ton (2015 prices) for NO_X and $PM_{2.5}$ by population density.Source: DEFRA (2015a)

Population/building density	NO_X damage costs [£/ton][a]			$PM_{2.5}$ damage costs [£/ton]		
	Low[b]	Central	High[a]	Low[a]	Central	High[a]
High ('transport inner conurbation')	20,455	51,137	81,820	110,590	141,248	160,507
Medium ('transport average')	8417	21,044	33,670	45,510	58,125	66,052
Low ('rural')	2610	6524	10,438	14,108	18,020	20,476

Notes:

[a] The central sensitivities reflect uncertainties around the lag between exposure and the health impact. The sensitivity for NO_X also reflects the uncertainty around the link between NO_2 exposure and mortality.

[b] NO_X damage costs if $PM_{2.5}$ is also valued (per ton, 2015 prices).

2013 (DfT, 2014b). The base case or 'Reference scenario' (REF) broadly depicts a projection of transport demand, supply, energy use and emissions as if there were no changes to transport and energy policy beyond October 2015. It was modelled using UKTCM based on exogenous assumptions and projections of socio-demographic, economic, technological and (firm and committed) policy developments, including the complex CO_2-graded road tax[6] and company car tax regimes. Economic growth data up to 2014 were based on UK government figures. Future GDP/capita growth were assumed to average 1.7% p.a. up to 2050 – in line with the historic 50-year average for the UK. Transport demand projections were modelled based on average demand elasticities (of GDP/capita, population and generalised cost) for the 1995–2014 period. Fuel price and retail electricity price projections were based on 2014 UK Government forecasts (DECC, 2014). Vehicle Excise Duty (VED, i.e. annual road tax) and other fuel duties were assumed to remain constant at 2015 levels. Following an approach commonly used in technology futures and modelling studies (European Commission, 2005; Fulton et al., 2009; Strachan and Kannan, 2008; Strachan et al., 2008; UK Energy Research Centre, 2009; WEC, 2007), pre-tax vehicle purchase costs were kept constant over time for established technologies and gradually decreased for advanced and future technologies, thus exogenously simulating improvements in production costs, economies of scale and market push by manufacturers.[7] For example, average purchase prices for BEV cars were assumed to decrease by 2.8% pa from 2015 to 2020, by 1.6% pa until 2030 and 0.6% pa until 2050. The Reference scenario further assumed gradual improvements in specific fuel consumption and tailpipe CO_2 emissions per distance travelled. The rates of improvement were based on technological innovation driven entirely by market competition, not on policy or regulatory push.[8] Fuel consumption and CO_2 improvement rates for future car vintages were assumed to be 1.5% pa (a somewhat lower and more conservative rate than the average rate of 4% pa observed for all new cars between 2008 and 2013). Indirect emissions from fuel supply and vehicle manufacture, maintenance and scrappage have been updated with data from a recent UK based review (Kay et al., 2013). Finally, the default electricity generation mix follows central government projections (mainly natural gas, wind and nuclear – with some CCC coal and gas by 2030), implying the carbon content of retail electricity is gradually decreasing from about 390 gCO₂/kWh in 2015 to about 160 gCO₂/kWh in 2030 (then staying constant to 2050).

2.3.2. Retrospective pathways exploring unaccounted for emissions

Two retrospective pathways coined Real World (RW) and Real World plus Diesel Purchase Penalty (RW_DPP) were developed to assess unaccounted-for emissions from diesel cars and the wider effects on energy use and GHG emissions in the UK. First, scenario RW simply assessed the UK-wide implications of the gap between 'official' and 'real world' NOₓ emissions by assuming that all EURO5 and EURO6 diesel cars (not just VWs) bought and used in

the UK since 2009 perform 4 times worse than official ratings suggest. Thus, EURO5 diesel cars emitted on average 0.72 gNOₓ/km instead of 0.18 gNOₓ/km; and EURO6 cars emitted on average 0.32 gNOₓ/km instead of 0.08 gNOₓ/km. This factor 4 is in line with the real world test data for EURO5 cars reported in ICCT, (2014b). For EURO6 cars it is, however, on the conservative side as the average gap between official and real word was reported to be higher at a factor of 7 (ibid). In terms of market shares, sales data for the UK show that VW had a 20.5% market share in the UK in 2013 (SMMT, 2014), so the factor 4 assumption implies that VW cars of that generation performed about 20 times worse than expected – which falls into the reported range of between 5 and 35 times. Any future EURO standards and performance gaps beyond EURO6 (so from about 2020 onwards) are currently uncertain, so it was assumed that any future vintages (labelled 'EURO7′ from 2020 to 2024, 'EURO8′ from 2025 to 2029, and so on) would meet emissions standards of at least EURO6 in 'real world' conditions. This was considered a reasonable assumption given the increased efforts to improve real world performance by industry and government and the likely tightening of standards beyond EURO6.

Second, to explore how consumers may have responded if they had known about the gap in emissions performance, an alternative 'what if' scenario (RW_DPP) was developed that modelled a 'purchase penalty' of GBP 2500 for all new EURO5 diesel-fuelled cars (ICE, HEV and PHEV) from 2009 to 2015. The penalty fee of GBP2,500 was estimated from summing up annual car mileages over the lifetime of a vehicle (∼180k kilometres over 10 years), then multiplying this by the difference in NOₓ emissions between 'real world' and 'official' rates (0.72–0.18 = 0.54 gNOₓ/km) and the average damage costs of NOₓ on human health of GBP 21,044 per ton (Table 1) (DEFRA, 2015a). By placing a penalty at the time of purchasing a diesel car the study effectively explored a recent past with lower diesel car market shares and different energy use and emissions when compared to the 'real world' case (RW). The core scenarios are summarized in Table 2.

2.3.3. Forward looking pathways exploring the effects of potential policy and market responses

Recent pronouncements by market analysts, city authorities and business leaders (e.g. BBC News, 2015; TfL, 2015) have supported policy measures such as diesel scrappage fees and promotion of ultra-low emissions vehicles (ULEV) and a switch to electro mobility to reduce the damage to human health by (underperforming) diesel cars (EurActiv, 2015b). Three further scenarios were developed to explore future policy and market responses. First, scenario REF' depicts a 'revised baseline' for comparison by assuming that all existing and future diesel cars underperform by a factor of 4 when compared to legislated standards (from EURO6 in 2015 onwards).[9] This is at the lower end of the reported range of 4–20 times higher NOₓ for EURO6 type approved private cars with diesel engines in city traffic and during cold weather (Hagman et al., 2015). In essence this depicts a future where NOₓ emissions control will not be achieved as planned. Second, DPT explores the effects of a variable scrappage/purchase tax on new diesel cars, with the tax graded by the vehicle's purchase price and specific fuel consumption (Tax∼[purchase price]* [specific fuel consumption, SFC]). In 2015, the graded tax ranged from approx. GBP800 for small (A/B class) diesel cars to approx. GBP2,500 for large (E-H class) ICE diesel cars, with the tax gradually decreasing due to improved SFC. Purchase taxes in this range have shown to be fairly effective in accelerating change in vehicle uptake in the low carbon context (Brand et al., 2013). Third,

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[5] The full Impact Pathway (IPA) modelling is relatively resource and time intensive. To support decision making, damage costs per ton of emissions were developed by DEFRA for use by analysts. These damage costs were derived from the full IPA analysis, providing a proportionate approach to valuing of the impacts of changes in air quality.

[6] In the UK this is called 'vehicle excise duty', or VED.

[7] The assumption that alternative technologies improve (cost, energy and environmental performance, consumer preferences) at a faster rate over time applies equally to all scenarios modelled here, not just the reference scenario.

[8] This implies that the EU mandatory agreement on new car CO_2 emissions would not be met. However, separating innovation by competition and innovation by regulation/policy push is slightly arbitrary here as the effects are never easy to untangle. It was merely assumed that half of the recent improvement came from market competition and the other half from policy (mainly fiscal) and regulation.

[9] This compares to the REF case assuming that only EURO5 and EURO6 diesel cars underperformed in real world driving conditions.

Table 2
Overview of core scenarios.

		Scenarios		
		Baseline for comparison	Alternative 1	Alternative 2
Type of analysis	'Retrospective' (from 2009)[a]	REF: reference scenario of transport demand, supply, energy use and emissions as if there were no changes to transport and energy policy	RW: EURO5 and EURO6 diesel cars emit 4 times the regulated amount (both 'hot' and 'cold start' emissions)	RW_DPP: as RW but with a purchase penalty of GBP2,456 per EURO5 diesel car, from 2009 to 2015 only
	Forward looking (from 2015)[b]	REF': adjusted baseline assuming all future diesel cars emit 4 times the regulated (EURO6+) amount	DPT: as REF' but with a scrappage/purchase tax on diesel cars, graded by purchase price and 'official' fuel economy	DPT_EV: combined DPT with 'high electrification' pathway (supply side measures + pricing)

Notes:
[a] Both RW and RW_DPP assumed that only EURO5 and EURO6 diesel cars emit 4 times the regulated amount through their lifetimes, from first take-up in 2009 until they are eventually scrapped (i.e. well into the late 2020s).
[b] In contrast, all forward looking scenarios assumed that all future diesel cars emit 4 times the regulated amount.

in order to compare the above with a high electrification, low dieselization pathway, scenario DPT_EV combined the above scrappage/purchase tax (DPT) with a transformative pathway developed for the UK's Committee on Climate Change (CCC) and focusing on supply measures for plug-in vehicles as an alternative to diesel cars. The analysis by the CCC (CCC, 2013, 2015a) suggested plug-in vehicle deployment targets for 2020 and 2030 at 9% and 60% respectively. A small number of scenarios were run using UKTCM in an iterative process that led to the high electrification scenario underlying DPT_EV. This implied transformational change including: ULEV being available in all vehicle segments and by all major brands by 2030; nationwide consumer awareness and acceptance of ULEV by the 2030s; significant investment and repositioning towards ULEV by the main vehicle manufactures; significant investment in recharging infrastructure; reduced (perceived) recharging times; and continued and improved equivalent value support for ULEVs for both private and company/fleet buyers.

2.3.4. Sensitivity analysis

In order to assess uncertainty in the economic valuation estimates a limited sensitivity analysis was conducted, including testing a range of low and high values (Table 1) for air quality damage costs of NO_X emissions in line with UK government guidelines used to assess national policies, programmes and projects (DEFRA, 2015a).

3. Results

This section presents the results structured by the main findings on the impacts on human health (Section 3.1), the future car market evolution (Section 3.2), the trade-offs between GHG and air quality emissions (Section 3.3), and the impacts of future policy on fuel use and associated tax revenues (Section 3.4).

3.1. The impacts on human health of unaccounted-for NO_X emissions could be significant

The retrospective analysis suggests significant impacts on human health that have not been accounted for using 'official' NO_X emissions ratings. When comparing RW with REF, 'real-world' NO_X emissions from Britain's car fleet were 12 Kilo-tonnes of NO_X ($KtNO_X$) (+6%) higher in 2009 (the first year of the EURO5 period) when compared to 'official' rating values, rising steadily over the following ten years to 72 $KtNO_X$ (+137%) in 2019 when the diesel car stock of the EURO5 and EURO6 generations would peak (see green lines/bars in Figs. 2 and 3).
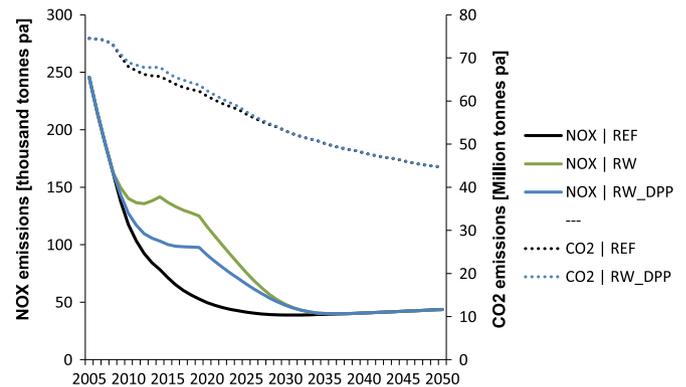
Fig. 2. : Retrospective gap analysis, total NO_X and CO_2 emissions from cars, 2005–2050 Notes: For NO_X emissions read left y axis; for CO_2 emissions read right y axis.
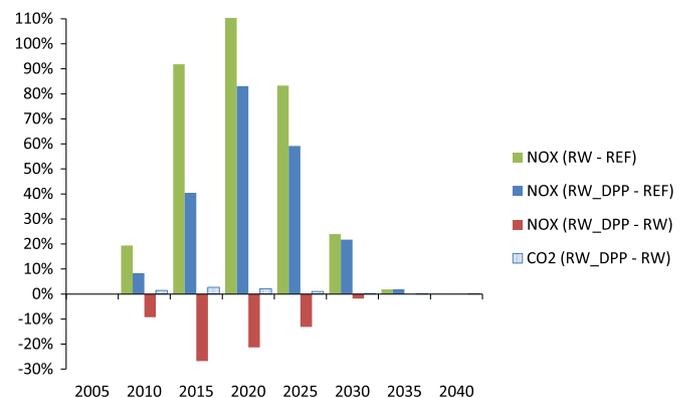
Fig. 3. : Retrospective gap analysis, differences between scenarios in tailpipe NO_X and CO_2 emissions from cars, 2005–2040 Notes: REF=Reference scenario; RW=Real World scenario; RW_DPP=Real World and Diesel Purchase Penalty scenario.

The associated marginal damage cost to human health in 2009 was £356 million p.a. (low £142 million, high £569 million) rising to £2.13 billion p.a. (low £852 million, high £3.41 billion) in 2019. Between 2009 and 2039 the additional damage costs of 'real world' NO_X totalled £29.1 billion (low £11.6 billion, high £46.5 billion). While these are big numbers, they are considerably lower than the total economic cost from the impacts of air pollution in the UK of around £20 billion every year (RCP, 2016), which is also comparable to the wider economic cost of obesity of up to £27 billion p.a. (Morgan and Dent, 2010). The damage costs are also smaller than annual tax revenues from diesel cars that have were estimated in this study at around £5.6 billion in 2015 (see Fig. 9).

An emissions related purchase price premium on diesel cars (as in RW_DPP) could have avoided about 43% (a total of 419 KtNO$_X$) of the 'real world' gap in NO$_X$ emissions if consumers had perceived diesel cars as being more damaging to health and chosen higher shares of alternative drivetrains to diesel ICEs (NB: RW_DPP was compared with RW here). However, a fleet with fewer diesels would have emitted somewhat higher tailpipe CO$_2$ emissions, with the difference to baseline peaking at 2.2 Million tons of CO$_2$ (MtCO$_2$) per year in 2014 (or 3.3% of direct CO$_2$ emissions from cars) and summing up to 23.1 MtCO$_2$ between 2009 and 2030. By comparing RW_DPP to RW (i.e. fewer diesel cars in the fleet between about 2010 and 2030), the modelling suggests that the avoided damage cost to human health of £12.2 billion (low £4.6 billion, high £19.5 billion) clearly outweighed the carbon costs[10] of £607 million (low £243 million, high £971 million).

3.2. A diesel purchase tax alone will not transform the car market without supply side measures aimed at clean vehicles

The retrospective 'purchase penalty' showed a significant drop in the market share for diesel cars. While the baseline (REF) market share of new diesels rapidly increased from 27% in the early 2000s to about 45% in 2015, it was only about 19% during the six years the 'purchase penalty' was in place (RW_DPP). Most of the fuel shift in car purchasing was to gasoline ICV, with some shifting to gasoline HEV, resulting in the NO$_X$ and CO$_2$ emissions changes shown above.

When switching focus to the forward-looking scenarios, the main effect of the graded purchase/scrappage tax on diesel cars (DPT) was a moderate shift in preferences from diesel (ICV, HEV) to conventional gasoline (mainly ICV but also some HEV) and plug-in vehicles (BEV, gasoline PHEV), particularly for the more price sensitive fleet/company car market. While diesel market shares in the baseline scenario (REF') stayed at just below 50% between 2015 and 2050, they were somewhat lower at 35% in DPT (or 26% lower than the REF' baseline), with gasoline ICV and BEV increasing their shares by about 25% each (Fig. 4).

In contrast to DPT, the DPT_EV pathway showed a marked transition from conventional ICV to ULEV from the mid 2020s onwards (Fig. 4). While new BEV and PHEV only made up 4% of the market share in 2020, this had risen to nearly half of all new cars being plug-in (39% PHEV, 9% BEV) by 2030. By 2050, the share had risen further to 60% (24% PHEV, 36% BEV). In contrast, new diesel ICV and diesel HEV sales dropped from nearly half in 2015–31% in 2020, 20% in 2030 and 17% in 2050.

The evolution of the *total* car fleet is shown in Fig. 5, suggesting that the 2020 stock will look pretty much the same as it is today. By 2030, the car fleet would include significantly fewer diesel ICV in the DPT case, and plug-in vehicles (mainly PHEV) would have taken significant shares away from ICVs and HEVs in the DPT_EV case. By 2050, the majority of the fleet would be plugged-in only if the UK adopted appropriate ULEV measures (as in DPT_EV but *not* in DPT). However, the modelling suggests that even higher ULEV uptake of the fleet of more than 60% is unlikely to materialize without further policy incentives (e.g. free parking, free electricity, new business models of EV ownership and use), supply shift to EV (e.g. decreasing model/brand supply of gasoline ICV cars) and regulation (e.g. eventually banning gasoline and diesel cars in
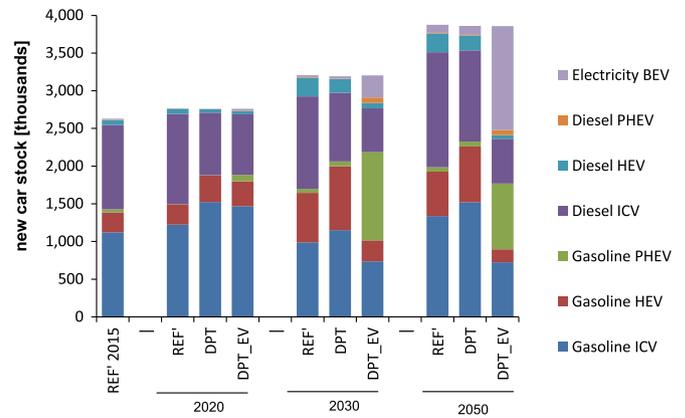
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[10] Carbon valuation was based on the 'social cost of carbon' (SCC) that measures the full cost of an incremental unit of carbon (or greenhouse gas equivalent) emitted now, calculating the full cost of the damage it imposes over the whole of its time in the atmosphere. The SCC was preferred over market prices (e.g. EU trading scheme) as the road transport sector is a non-traded sector. Values were taken from UK guidance, ranging from GBP 13.5/tCO$_2$e (Low) to 22.5 (central) and GBP 41.5/tCO$_2$e (High).

Fig. 4. : Scenario comparison of new car sales by propulsion system Notes: REF' 2015 refers to base year sales. REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario. The analysis suggested no significant take up of hydrogen FCV, dedicated biofuel ICV and gas powered ICV, largely due to the assumed lack of any model/brand supply. Therefore, these propulsion systems were omitted from the chart.
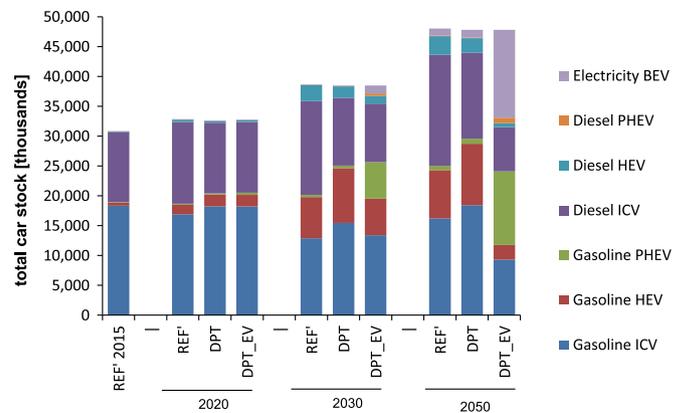

Fig. 5. : Scenario comparison of total car stock by propulsion system in 2020, 2030 and 2050 Note: REF' 2015 refers to base year sales. REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario. The total stock of dedicated biofuel ICV, (LPG/CNG) gas powered ICV and hydrogen FCV was negligible (<10 thousand), largely due to the assumed lack of any model/brand supply. Therefore, these propulsion systems were omitted from the chart.
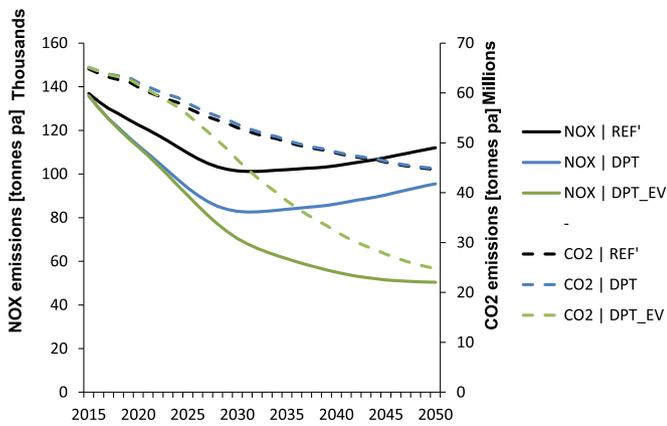
urban areas).

3.3. Future 'low diesel' policies and supply measures can have significant air quality benefits while showing few (if any) carbon disbenefits
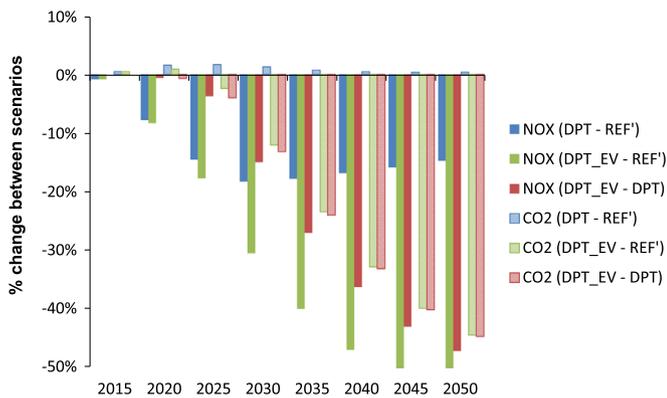
The results of the forward looking analysis suggests that policies designed to 'discourage' diesel sales and/or promote ULEV as cleaner alternatives can have significant impacts on air quality and carbon emissions (Figs. 6 and 7). When compared to the 'real world' baseline (REF'), the variable purchase/scrappage tax on new diesel cars (DPT) resulted in moderate NO$_X$ emissions reductions, rising steadily from 1.1 KtNO$_X$ p.a. (0.8%) in 2015 – the first year of the policy – to around 16 KtNO$_X$ p.a. (15%) in 2025. The total avoided marginal damage costs to human health between 2015 and 2025 were £2.91 billion (low £1.04 billion, high £4.66 billion). Beyond 2025 the emissions savings levelled off then stayed roughly constant with average reductions of 18 KtNO$_X$ p.a. out to 2050, indicating a saturation of the policy's main effect of fuel switching away from diesel. In that period the annual avoided damage cost to human health due to reduced NO$_X$ averaged £516 million p.a. (low £193 million p.a., high £826 million p.a.).

To achieve higher and *increasing* emissions savings and avoided

Fig. 6. : Future policy analysis, total NO_X and CO_2 emissions from cars, 2015–2050 Note: REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario.

Fig. 7. : Future policy analysis, differences between scenarios in car NO_X and CO_2 emissions, 2015–2050 Note: REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario.

damage costs beyond 2025 additional market changes, industry push, infrastructure investment and policy pull would be required as explored in DPT_EV. When compared to baseline (REF'), NO_X emissions in DPT_EV were lower at 31 $KtNO_X$ p.a. (-30%) in 2030 and 62 $KtNO_X$ p.a. (-55%) in 2050, by which time about three quarters of the reduction was due to the additional supply side measures (Figs. 6 and 7). Between 2025 and 2050 the total avoided damage cost of reduced NO_X emissions came to a substantial £35 billion (low £15 billion, high £56 billion).

In terms of carbon emissions the results showed that already in the forward-looking baseline case (REF') direct emissions of CO_2 from cars fell substantially, from the 2015 level of 65 $MtCO_2$ to 61 $MtCO_2$ in 2020, 53 $MtCO_2$ in 2030 and 45 $MtCO_2$ in 2050.[11] While the post-2008 economic downturn and rising fuel costs were major factors underlying the short term fall before 2015 (Fig. 2), the longer-term decrease of about 18% between 2015 and 2030 is largely the result of improvements in fuel efficiency and emissions performance of new cars penetrating the fleet and some fuel switching to HEVs and PHEVs, thus offsetting the overall growth in the demand for car travel. The diesel car purchase/scrappage tax (DPT) showed *higher* carbon emissions when compared to baseline

(REF'), with the difference first increasing to 1.0 $MtCO_2$ by 2025 then gradually decreasing to 0.2 $MtCO_2$ by the 2040s (Fig. 7). As with the retrospective analysis, this was due to the lower share of (lower carbon) diesel cars in the fleet. As expected the combination of the diesel purchase/ scrappage tax with the higher uptake of ULEV in scenario DPT_EV yielded significant emissions savings from 2025 onwards, gradually reducing emissions from $+0.6$ $MtCO_2$ in 2018 to -6.4 $MtCO_2$ in 2030 and -19.9 $MtCO_2$ p.a. in 2050. DPT_EV thus achieved significant 'co-benefits' in the longer term by incentivizing and promoting cleaner and lower carbon alternatives to diesel cars.

As shown in the Supplementary material (S4), intra-scenario differences in *total life cycle* emissions impacts were not as significant as with direct emissions, as direct GHG emissions savings were being offset by gradual increases in indirect GHG emissions from the increase of emissions from upstream electricity generation. As expected, the biggest changes came from the radical (perhaps necessary from a climate perspective) changes in scenario DPT_EV.

3.4. Wider impact on energy use, fuel demand and fuel tax revenues

In the short term all scenarios showed only a small increase in overall energy use and a switch from diesel to gasoline (ICV, HEV), which can be explained by gasoline ICV being less energy efficient than diesel ICV (Fig. 8). In the medium to longer term the modelling showed modest (2030) to large (2050) fuel switching and decreases in energy consumption due to the uptake of more energy efficient propulsion systems in the form of plug-in vehicles (gasoline PHEV, BEV). The diesel purchase tax + high electrification scenario (DPT_EV) showed total fuel demand decreasing by up to 52% by 2050 when compared to 2015. This contrasts to a decrease of 31% in baseline scenario (REF') by 2050. By 2050, diesel demand drops from about half in the reference case (REF') to 35% (DPT) and 26% (DPT_EV). By comparison, electricity demand grows steeply in the DPT_EV scenario, particularly in the second half of the period, accounting for 21% of total energy consumption by 2050. However, in all scenarios, conventional fuels (gasoline + diesel) still dominate energy use, never falling below 79% of total demand.

In 2014/15, about £16.2 billion were raised from cars through road fuel tax, which was almost entirely from the duty on gasoline and diesel of £0.61/litre (DfT, 2015). As shown in Fig. 9, the results suggest that the road tax revenue stream would not change much in the short term. However, in the long term the modelling suggests that road tax revenues would fall sharply to about £11 billion

Fig. 8. : Pathways comparison of car transport fuel demand (in PJ) by transport fuel, 2020–2050. Note: The demand for LPG, CNG and hydrogen was negligible, largely due to the lack of any take up in the car fleet. Therefore, these fuels were omitted from the chart. REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario.

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[11] Changes in carbon emissions are the result of a number of interrelated factors, including the penetration of lower emission cars into the vehicle fleet, changes in demand for cars and other modes, changes in car total ownership (e.g. a decrease in total ownership means lower indirect carbon emissions from manufacture, maintenance and scrappage) and changes in upstream fuel emissions. For further details on how this is done in UKTCM see Brand (2010a/b) and Brand et al. (2012).
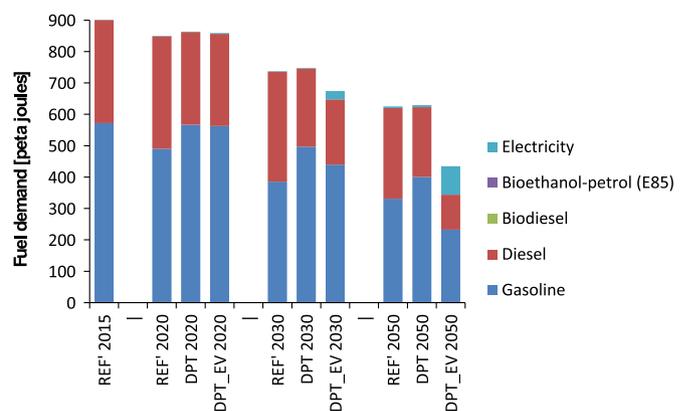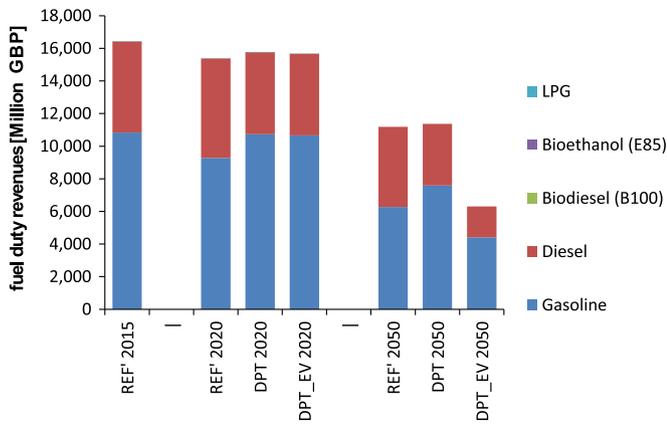
Fig. 9. : Pathways comparison of fuel tax revenues from cars (in £Million) by transport fuel – 2015, 2020 and 2050 Note: The demand for CNG and H_2 was negligible, largely due to the lack of any take up in the car fleet. Therefore, these fuels were omitted from the chart. REF'=Reference scenario (revised); DPT=Diesel Purchase Tax scenario; DPT_EV=Diesel Purchase Tax and EV package scenario.

p.a. (REF', DPT), and even lower to £6 billion p.a. in the high electrification case (DPT_EV), which while necessary in climate terms may limit the government's ability to pay for the health damage costs.

4. Conclusion and policy implications

4.1. Key results: 'real world' excess emissions

This study has quantified the human health impacts and associated costs of excess NO_X emissions in the UK context and found that the size and timespan of unaccounted-for NO_X emissions was significant, with up to more than double the amount of NO_X in the atmosphere than official ratings suggest. This is in line with recent studies that investigated air quality impacts of excess diesel emissions in the UK (Beevers et al., 2012; Dunmore et al., 2015; Walton et al., 2015). It is significantly higher than the most recent estimates of excess emissions in the US (Barrett et al., 2015), reflecting the different vehicle fleet compositions and emissions standards operating in the two countries. The additional damage costs of 'real world' NO_X in the UK were significantly higher than the US estimates reported in Barrett et al. (2015), reflecting differences in methodologies (valuation of 'indirect' $PM_{2.5}$ and ozone only in the US study), 'doses' and population densities. The impact valuation results of this study compare to the total economic cost from the impacts of air pollution in the UK of around £20 billion every year (RCP, 2016), which is also comparable to the wider economic cost of obesity of up to £27 billion p.a. (Morgan and Dent, 2010). More widely, the damage costs reported here were broadly consistent with estimates reported in a recent WHO study, which estimated the cost of disease and the premature deaths associated with a wider set of air pollutants (incl. PM) in the UK at around $83bn (£54bn) (Vidal, 2016; WHO, 2016). Note this includes all economic sectors and modes of transport, so is expected to be higher.

The results of the retrospective diesel 'purchase penalty' suggest a trade-off between a large *decrease* in local air pollutants against a modest *increase* in climate change pollutants. This can be explained by the significant fuel switching away from diesels in the UK car fleet during the 2009–2015 modelling period. The relative size of the effects was in line with other modelling exercises looking at CO_2 and air quality effects of policy (e.g. Leinert et al., 2013).

4.2. Key results: future policy

The finding that a diesel purchase tax is unlikely to transform the car market without considerable supply side and tax incentive measures promoting ULEVs (scenario DPT_EV) contributes to the debate on what policy options and industry investments are required to meet air quality and climate mitigation goals. While scrappage schemes can be effective in reducing emissions (CEMT, 1999; Kagawa et al., 2013), they have issues around reliance on increasingly scarce public funds, are potentially regressive (benefitting the rich more than the poor), and have potential rebound effects (Brand et al., 2013; Vaughan, 2014). A dynamic and revenue neutral 'feebate' system could be the better option in the medium term, as has been shown in a number of studies (BenDor and Ford, 2006; Brand et al., 2013). Furthermore, a purchase/scrappage tax should not counteract any CO_2-graded road tax regime that typically favours diesel cars due to lower CO_2 ratings (Leinert et al., 2013). One solution would be to disaggregate CO_2-graded taxation levels further by fuel type, as is currently the case for new company cars in the UK through differential BIK rates (HM Treasury, 2015).

To achieve higher and increasing emissions savings (up to 55% less NO_X and by 2050), 'co-benefits' (CO_2 *and* NO_X emissions reductions) and avoided damage costs beyond 2025 additional market changes, industry push, infrastructure investment and policy pull would be required. The marked transition to plug-in vehicles from the mid 2020s onwards explored in DPT_EV can be explained by the underlying transformational change in a number of areas beyond purchase price policy. First, DPT_EV implied that EV availability would increase following existing trends, meaning they will be widely available in all vehicle segments and by all major brands by 2030 (in the REF' baseline and DPT policy scenario, vehicle supply stays constant at 2015 levels, implying perceived supply penalties). Significant investment and repositioning by the major manufactures would be required, potentially driven by increasingly stringent new car CO_2 regulation after 2020 that eventually can only be met by ULEVs (Berggren and Magnusson, 2012). Second, consumer awareness and acceptance were assumed to increase significantly, with a steep increase in the 2020s (simulated by an S-curve) leading to 95% of potential buyers being aware of ULEVs and their incentives by 2030, and 100% by 2040. To achieve the critical mass for acceptance and awareness would involve require promotional campaigns, large field trials, dedicated car clubs and the 'neighbour' effect to diffuse widely and even convince the 'Resistors'. Third, the scenario further assumed investment in the next 15 years in high levels of overnight (mainly off-street) charging complemented by a national network of about 2000 rapid charging points for day charging to increase the market base for plug-in vehicles (in particular for the fleet segment) and provide national coverage by 2030. This effectively meant that by 2030 74% of private buyers (compared to 70% in REF'/DPT) and 80% of fleet buyers (compared to 40% in REF'/DPT) would have 'certainty of access' to charging. The investment needed would be in the tens of millions of GBP. Fourth, with a growing fast charging network happening over time the *perceived* EV charging times were decreasing with increasing BEV power rates (assumed to increase rapidly from 3 kW in 2015–7 kW in 2020 and then to 50 kW; for PHEV, this maxed out in 2020 at 7 kW). Last, in order to mitigate the purchase price premium of ULEVs the scenario assumed continued and improved equivalent value support for ULEVs for both private and company/fleet buyers, through capital incentives and continuation of the CO_2-graded VED that incentivizes ULEV uptake. The plug-in car-grant was recently extended to 2019 (instead of stopped after 2017) at the current rate of £5000, then reduced by half to 2024 (no grant from 2025 onwards). In addition, the company car tax regime was revised so

that cars emitting 50gCO$_2$/km or less (effectively BEV and PHEV) see the 9% Benefit-in-Kind (BIK) rate (as opposed to 13–16% as currently planned) (HM Treasury, 2015).

The results contribute to the growing body of evidence that while the health and environmental benefits related to fuel switching can be significant, the pace and scale of achieving those benefits is somewhat uncertain (Åström et al., 2013; van der Zwaan et al., 2013), particularly in the UK context where the projected deep decarbonisation of the electricity system over the longer term may prove difficult to achieve (Buekers et al., 2014). However, lower carbon content of future road electricity is the key component that drives the carbon reductions in DPT_EV. While the UK Government does not expect London and other AQ 'hot-spots' to meet legal pollution levels until at least 2025 (DEFRA, 2015a), ambitious taxation policy and further investment in electrified mobility will play important roles in meeting those targets in the medium term. However, this result cannot easily be translated to other countries which rely more on higher carbon (coal fired) power stations that can offset the life cycle carbon and health benefits of replacing diesel ICV with plug-in cars (Baptista et al., 2012).

Finally, fuel tax remains an important policy instrument (Montag, 2015) and source of government revenue (HM Treasury, 2015). While the UK already taxes diesel and gasoline at the same rate per litre, diesel is taxed 10% less per unit of energy (TandE, 2015a). Electricity is only taxed through VAT (currently 20% for road transport). This suggests there may be a case for revising the fuel taxation regime taking into account energy, CO$_2$ and air quality impacts – not just those associated with NO$_X$ but also PM. In the longer run, the reductions in fuel duty revenues in *all* future scenarios can be explained by the take-up and use of more fuel efficient cars and, in the DPT_EV case, the zero duty on electricity as a road transport fuel. The latter issue has been recognised by the UK Government and other countries around the world – and it is expected that once plug-in vehicles make up significant market shares, electricity as a transport fuel will have to be taxed accordingly, with expected rebound effects on take up rates. However, as shown in a recent study for the UK (Brand et al., 2013), a fuel duty on electricity of 5 pence per kWh (roughly the gasoline-equivalent to the current duty rate for gasoline) would show relatively small reductions in plug-in vehicle uptake – reflecting the comparative energy efficiency advantage of electric drivetrains.

4.3. Outlook and future work

The approach used for this study contributes to the growing consensus that regulation and emissions budgeting based on tailpipe emissions is increasingly no longer fit for purpose and should be changed to be based on well-to-wheel, and ultimately life cycle, emissions (IEA, 2013). Currently the average fuel life cycle greenhouse gas (GHG) saving for a BEV over its full life has been estimated at about 50% under UK conditions – that is, with the current mix of grid electricity generation (Kay et al., 2013). This could increase to 75% in 2020 and to 83% by 2030 with the anticipated decarbonisation of grid electricity. Also, vehicle life cycle emissions (from manufacture, maintenance and scrappage) add significantly to emissions from vehicle use (IEA, 2013; Lane, 2006; MacLean and Lave, 2003) and can be significantly higher for BEV and PHEV than for ICV (Baptista et al., 2012; Kay et al., 2013).

Further work is required in exploring sensitivities around 'real-world' vehicle emissions factors of other pollutants affecting human health, most notably PM and hydrocarbons. While this paper focused on Dieselgate and related NO$_X$ pollution and standards it is important to note that the inclusion in the damage cost calculation of other pollutants, notably PM, may change the damage cost values. Since total multi-pollutant valuations were likely to

increase the totals, the figures reported in this study could be considered on the conservative side. Further work could also investigate the acceptance of various vehicle propulsion systems by a wider set of the heterogeneous fleet/company market actors. This could be achieved by employing Monte Carlo analysis, which can help analyse the propagation of multiple uncertainties in an integrated transport-energy-environment modelling system such as UKTCM (Int Panis et al., 2004).

4.4. Final thoughts

The policy and industry response in the aftermath of the 'Dieselgate' affair is in full swing. However, there are concerns in Europe whether more realistic 'real world' emissions test cycles will be approved and implemented anytime soon, and that the European Commission's car emissions testing may not have 'the muscle like US watchdog' (Stupp, 2016). Some of the major diesel car manufacturers have agreed to cooperate on real-world emissions testing and reductions, including Peugeot Citroen (TandE, 2015b) and Renault (AFP, 2016). The regulatory response should go hand in hand with further development of technological solutions to meet NO$_X$ standards. These have been available for some time, including cooled exhaust gas recirculation, lean NO$_X$ traps or selective catalytic reduction with ammonia (Bertelsen, 2001; Faiz et al., 1996; Sanchez et al., 2012). Manufacturers are usually chosing the NOx aftertreatment technology based on a combination of cost, reliability, fuel economy, and consumer acceptance.

By assessing the potential impact of different policy approaches and consumer responses to the 'de-dieselization' (Leinert et al., 2013) of cars, this study contributes to the growing consensus (Barrett et al., 2015; Carrington, 2016; Walton et al., 2015; Woodcock et al., 2009) that future policy may have to go the extra mile (pun not intended) by promoting additional market changes, industry push, infrastructure investment and policy pull in order to achieve the emissions savings, 'co-benefits' and avoided damage costs of a range of pollutants required to meet climate, air quality and health damage goals. Given the UK's strategic commitments to meeting its stringent climate objectives and realisation that this is likely to be achieved by a pathway similar to DPT_EV (CCC, 2015a), NO$_X$ and other air quality pollutant emissions may be significantly reduced providing significant 'co-benefits'.

Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.enpol.2016.06.036.

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