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The effectiveness of policy on consumer choices for private road passenger transport emissions reductions in six major economies

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Abstract

The effectiveness of fiscal policy to influence vehicle purchases for emissions reductions in private passenger road transport depends on its ability to incentivise consumers to make choices oriented towards lower emissions vehicles. However, car purchase choices are known to be strongly socially determined, and this sector is highly diverse due to significant socio-economic differences between consumer groups. Here, we present a comprehensive dataset and analysis of the structure of the 2012 private passenger vehicle fleet-years in six major economies across the World (UK, USA, China, India, Japan and Brazil) in terms of price, engine size and emissions distributions. We argue that choices and aggregate elasticities of substitution can be predicted using this data, enabling us to evaluate the effectiveness of potential fiscal and technological change policies on fleet-year emissions reductions. We provide tools to do so based on the distributive structure of prices and emissions in segments of a diverse market, both for conventional as well as unconventional engine technologies. We find that markets differ significantly between nations, and that correlations between engine sizes, emissions and prices exist strongly in some markets and not strongly in others. We furthermore find that markets for unconventional engine technologies have patchy coverages of varying levels. These findings are interpreted in terms of policy strategy.

1. Introduction

Road transport generates 5.3 Gt out of 31.7 Gt of CO_2 of global fuel combustion greenhouse gas emissions contributing to climate change, and over 30% of total emissions annual growth (IEA 2013). The current trend in global emissions is leading the world towards global warming exceeding 4°C, with the most important component emitted by the electricity sector, transport coming second (IPCC 2014). Transport is not currently highly regulated for emissions, and consumer preferences with increasing income drives choices towards increasingly carbon intense engines (e.g. Gallachoir *et al* 2009, Zachariadis 2013)[1]. The lack of clear policy direction in the sector may be related to a lack of understanding of the impact of

policies aiming at incentivising consumer choice (He and Bandivadekar 2011).

The fuel efficiency of the car fleet depends directly on its composition of engine types and sizes. In other emissions intensive sectors such as industry, technologies of the same type (e.g. boilers, blast furnaces, power plants) differ modestly predominantly due to vintage[2]. The transport sector, however, features a very wide continuous array of possible fuel efficiencies (spanning a factor of 3–4, see data below), that do not depend as strongly on the state of technology and age of cars as it does on socio-economic characteristics of owners. As demonstrated by McShane *et al* 2012, vehicle choices strongly relate to social groups, and thus existing socio-economic differences are reflected in

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[1] According to our data, this is also true in the UK.

[2] By vintage we mean different variants of the same technologies of different ages, variations that stem from the gradual incorporation of particular innovations over time.

the types of vehicles that consumers purchase, consistent with socio-anthropological theory (Douglas and Isherwood 1979). We ask two questions here: do different markets respond differently to policy instruments? Is it possible to obtain quantitative insight on the effectiveness of proposed policies using market research methods?

Cars typically survive in the fleet for around 12 years[3]. The carbon intensity of the private vehicle fleet can be reduced by influencing vehicle purchase choices in two different ways[4]: by improving the average fuel economy of the new fleet-year (i.e. new models) with conventional engine technology, or by gradually replacing conventional combustion engines by hybrid and electric systems. It is clear that comprehensive emissions reduction policy must include coordinated demand-pull and supply-push policies; however, in the short term, it is policies affecting consumer choice that have the most impact, since it takes years to change production lines and supply chains of manufacturers, while sales can change overnight. Here, we focus on demand-pull policies that affect consumer choice.

Potentials for efficiency improvements in transport have been widely explored (IPCC 2014, WEC 2009, IEA 2012). In energy-environment policy modelling research, 'representative' vehicles are most of the time used, with cost-optimisation frameworks (e.g. McCollum *et al* 2014; Takeshita 2012, van der Zwaan *et al* 2013), a procedure that is, as we show here, insufficiently rich in socio-economic patterns, while the optimisation of cost driving agent choice is contradicted by empirical work (McShane *et al* 2012). It is also not clear that evaluating the average willingness to pay for higher fuel efficiency is feasible or even insightful in a highly diverse sector where prices span an order of magnitude following the income distribution (see below), and the literature is not unanimous on any value (Greene 2010, Anderson and Kellogg 2011, Gallagher and Muehlegger 2011). A more promising approach appears to be to use marketing research, where the likely response of consumers to policies is inferred from the structure of the market itself (as in He and Bandivadekar 2011), and this approach naturally introduces socio-economic distributions.

In this paper, we explore the emissions, technical and economical characteristics and potential for emissions reductions of car markets in six major representative economies of the World: the United Kingdom, the United States, China, India, Japan and Brazil. These markets possess very different characteristics that are the result of different histories of policy and regulations, but also different socio-economic

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[3] The life expectancy of cars in the UK is 12 years, as calculated by the authors using a survival analysis with data from the DVLA 2012 data, unpublished.

[4] Emissions can also be reduced by reductions of car use, for instance with fuel taxes, involving possible transport mode or lifestyle changes, not the subject of this paper.
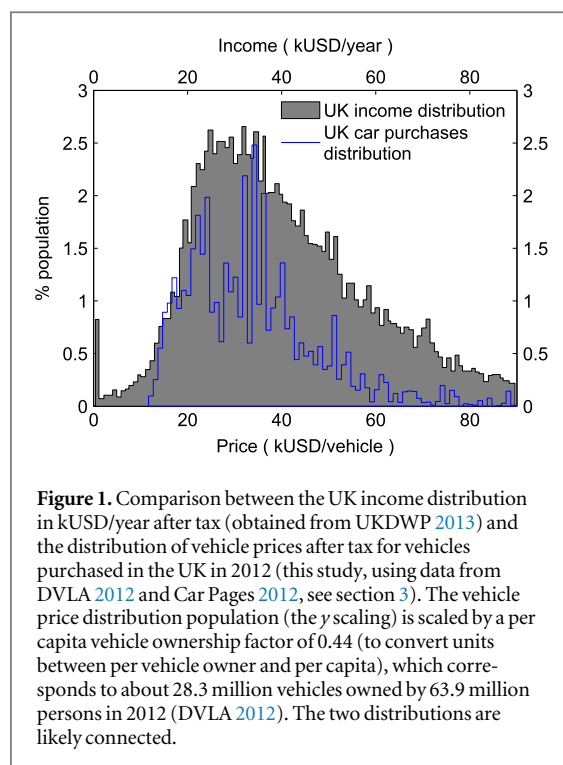

Figure 1. Comparison between the UK income distribution in kUSD/year after tax (obtained from UKDWP 2013) and the distribution of vehicle prices after tax for vehicles purchased in the UK in 2012 (this study, using data from DVLA 2012 and Car Pages 2012, see section 3). The vehicle price distribution population (the *y* scaling) is scaled by a per capita vehicle ownership factor of 0.44 (to convert units between per vehicle owner and per capita), which corresponds to about 28.3 million vehicles owned by 63.9 million persons in 2012 (DVLA 2012). The two distributions are likely connected.

characteristics of their populations as well as different cultures. We show that, even with highly globalised car manufacturers, the distribution of car purchases and their fuel efficiencies have radically different characteristics in different markets. We calculate the effectiveness of emissions reduction policies through vehicle choice in terms of emissions reductions per unit tax, applied on the price of vehicles or fuels. This is compared to the effectiveness of policy for technological change. This leads to two considerations: firstly, different car markets may require different policies for effective emissions reductions. Secondly, differences in market characteristics between countries show comparatively what may be realistically achievable elsewhere with policy. Details of calculations are given in the supplementary information (SI) available at stacks.iop.org/erl/10/064008/mmedia.

2. Materials and methods

2.1. Theory for car purchase choice
McShane *et al* 2012 makes a remarkable demonstration, using correlations spatially resolved using postcode data, that vehicle purchases in the USA are visually influenced by previous purchases of similar vehicles locally, but not by purchases that happen at distances where visual influence is weak. This takes place within *geographical areas*, *social identity groups*, and within *vehicle types* and *price tiers*. This work provides strong evidence of how vehicle choices, determined within social groups, take place within

restricted subsets the vehicle market in specific price segments.

This can be directly observed using socio-economic distribution data. Figure 1 shows the 2012 distribution of income for the whole UK population (from UKDWP 2013), along with the 2012 UK distribution of car purchase prices (this work, see below) multiplied by average car ownership. Both datasets are well described by lognormal distributions and a close relationship is observed between their scaling parameters[5], suggesting a likely proportional relationship. This indicates that particular socio-economic groups, on average, purchase vehicles of similar prices[6].

Based on this evidence, here we infer that car buyers seek to display their social identity, and thus social group characteristics, when purchasing a vehicle, an assertion supported by socio-anthropological theory (the anthropology of consumption, seminal work by Douglas and Isherwood 1979), principles generally accepted in marketing research. Following the reasoning of Douglas and Isherwood 1979, while all passenger vehicles are built for the purpose of transporting persons, they are marketed for particular social groups at particular prices that match their willingness to pay, and these prices also restrict access to other social groups. The price one is able to pay for a vehicle may be a particularly important channel for the communication of one's social identity, and, as emerges from figure 1, appears to be a good indicator of one's disposable income.

The diversity of social groups within a nation therefore strongly influences the diffusion of innovations in the private transport fleet. Formally, the *breadth of consumer diversity determines the elasticities of technology substitution*, an assertion that is best understood using discrete choice theory (for instance inBen-Akiva and Lerman 1985). This assertion is consistent with classical diffusion theory (Rogers 2010), in which the diversity of consumers determine the scaling of the typical *S*-shaped profile of diffusion. It also follows from evolutionary economics, where technology diversity continually increases to ever better match evolving consumer tastes.

2.2. Theory for evaluating the effectiveness of policy

Emissions reduction policies targeting private vehicle choices have an effectiveness that depends on the structure of local vehicle markets[7]. Intuitively, one expects that fuel efficiencies decrease with increasing levels of luxury. As we find here, however, prices do not always scale with emissions, and we describe below the implications to the response of consumers to policy-making.

Vehicles with large engine displacements have in general higher torque and power (for a review of data see WEC 2009). For this enhanced power, more fuel (energy) is used and therefore, for any carbon-based fuel, higher power means higher emissions per kilometre travelled[8]. We explore this relationship below; even though significant amounts of energy may be used in vehicles for other purposes than movement (e.g. air conditioning, electronics), we find that a relationship between emissions and engine size is always measurable, and it is linear.

Vehicles with larger engine displacements are generally thought more luxurious and expensive than vehicles with small economical engines. This, however, is not always true and, as we find below, depends on particular vehicle markets. As we show below, when it exists, it is log-linear[9]. Therefore, whether emissions, or the fuel economy, are functionally related in any particular way with vehicle price depends on these two relationships: between emissions and engine size, and between engine size and price. When it exists, it is also of the log-linear type.

The effectiveness of taxes or subsidies applied on vehicle properties such as engine size or emissions corresponds to their ability to generate substitutions between available vehicle models in order to achieve reductions in the emissions of new vehicles. We show here that at the aggregate level, the effectiveness depends on the structure of the market and on whether a relationship between emissions and price exists. If a relationship exists, the effectiveness is closely related to its correlation parameters. If no relationship exists, the aggregate outcome of fiscal policies may or may not turn out to match what was expected in their design, and high levels of uncertainty remains. Given the log-linear structure of the market, an emissions reduction tax on the purchase price that produces a non-negligible incentive across the spectrum of vehicles requires a fee that increases at least proportionally to the emissions rating. When proportional, it is consistent with the 'polluter pays' principle. Note, however, that with fiscal policies based on the polluter pays principle, according to our data, the incentive comparatively decreases with increasing car price[10]. The

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[5] The ratio between their mean and their standard deviation or median is nearly the same.

[6] Note that the strength of this correlation and its scaling is found to vary across the world, where in some countries, the importance of vehicle purchases relative to other expenditures differ. In particular, in Japan, the car is losing its significance as an identity symbol and expenditure, and the relationship between income and vehicle price is comparatively weakened (see the data in JAMA 2013).

[7] By this we mean markets for new vehicles, since the sale of second hand vehicles does not change fleet emissions, unless they are scrapped before the end of their statistical lifetime, e.g. with a dedicated scrapping policy.

[8] Two aspects of driving behaviour predominantly underlie a demand for higher power. For the same travel itinerary, (1) higher constant speeds (wind resistance increases faster than linearly with speed, which thus requires more energy per kilometre the faster the vehicle travels). (2) Higher acceleration (requires more energy since energy losses are proportionally higher at higher engine revolutions required for higher acceleration).

[9] The log of the price is a linear function of engine size.

[10] Car prices increase exponentially for corresponding emissions that increase linearly. At high car prices, emissions taxes become comparatively small.
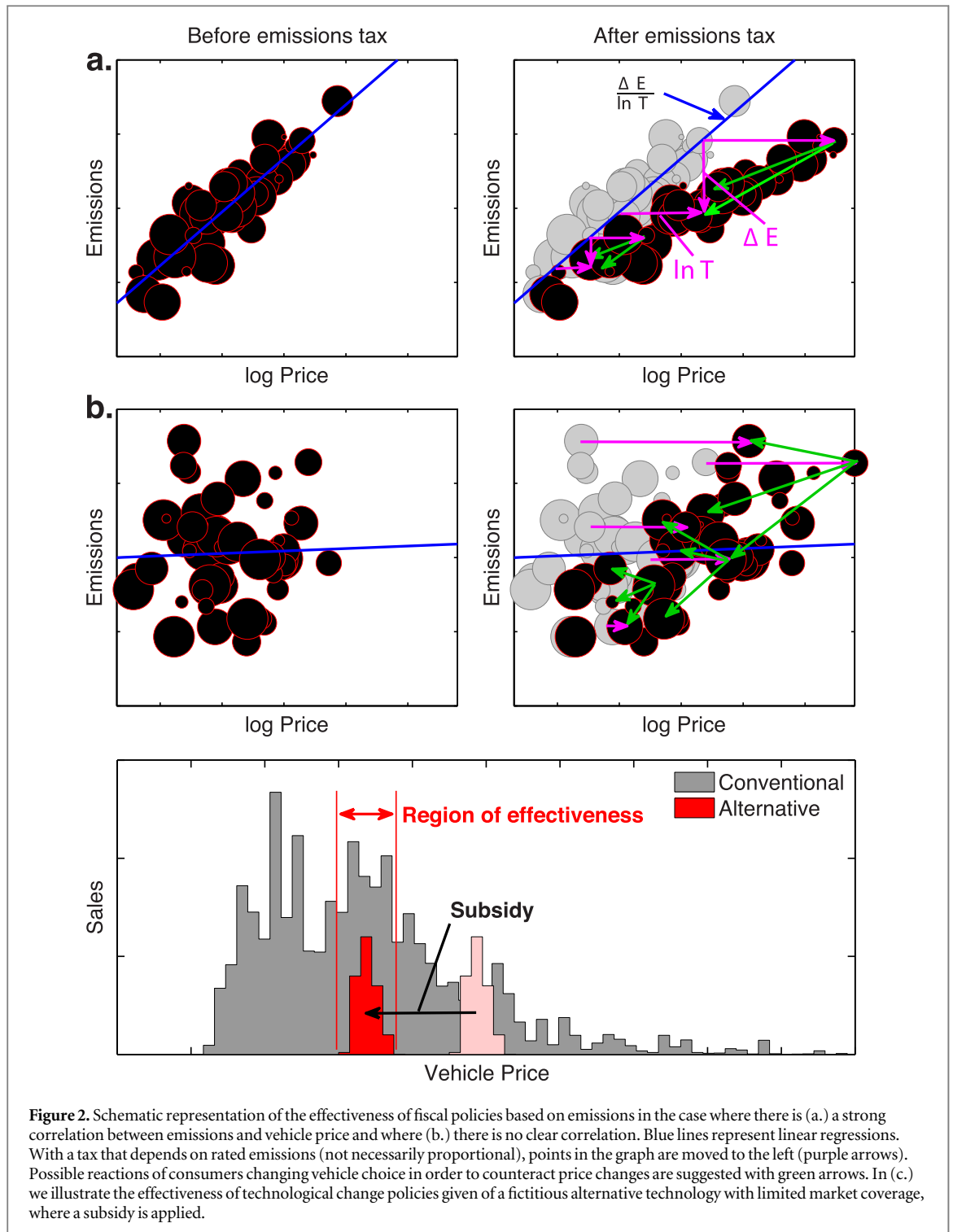
Figure 2. Schematic representation of the effectiveness of fiscal policies based on emissions in the case where there is (a.) a strong correlation between emissions and vehicle price and where (b.) there is no clear correlation. Blue lines represent linear regressions. With a tax that depends on rated emissions (not necessarily proportional), points in the graph are moved to the left (purple arrows). Possible reactions of consumers changing vehicle choice in order to counteract price changes are suggested with green arrows. In (c.) we illustrate the effectiveness of technological change policies given of a fictitious alternative technology with limited market coverage, where a subsidy is applied.

effect of taxes on the price of fuels to the choice of vehicles is technically very similar to that of taxes on rated emissions, since the consumer, at purchase time, can only at best estimate future fuel cost using manufacturer rated emissions. Thus we treat them in the same way here. Estimating their effectiveness however requires extra knowledge, or assumptions, on the extent to which consumers evaluate and take consideration of fuel costs over the vehicle's lifetime in their purchasing decisions.

The effectiveness of policy in a diverse market is explained schematically in figures 2(a) and (b), in which two fictitious vehicle markets are illustrated using one circle per vehicle model, of which the area scales with its number of sales. In *a.*, we have a market where exponentially more expensive vehicles have on average proportionally higher fuel consumption. Applying a fiscal policy based on engine sizes or rated emissions is likely to lead to some emissions reductions if consumers, when replacing a vehicle, attempt

to remain within a particular price bracket, as the work of McShane *et al* 2012 suggests they would. By seeking a price reduction to compensate for the tax, consumers are forced by available choice to pick in almost every case lower emissions vehicles. In this case the policy effectiveness is well defined, and is equal to the slope of the relationship. Note that this effect also works in reverse: if their income increases, consumers may seek higher price vehicles, which in turn would have higher emissions. Note also that the effectiveness value is independent of the shape of the chosen relation between the tax value and rated emissions (i.e. proportional or not)[11].

In *b.*, we have a situation where no correlation exists between emissions and price. In this case, the aggregate impact of a fiscal policy on emissions is ambiguous and could lead to uncertain changes in emissions. This is because there is a wide range of possible fuel economy values that consumers can access while attempting to choose lower price vehicles to compensate for the tax. In this case the policy effectiveness is itself ambiguous. Insight on the effectiveness of policy can thus be obtained from a combination of the strength of the correlation between prices and emissions (the level of confidence that a response to the policy would arise), and the slope of the relationship itself (the strength of the response).

Meanwhile, emissions can also be influenced using policies supporting changes in engine technologies, such hybrid or electric. Technological change could effectively reduce emissions if alternative engine vehicle models have very low emissions and if they are accessible to most consumers. Currently, however, (see below) most markets do not offer a very wide range of models, which may not have the ability to capture the whole breadth of existing consumer diversity, restricting their diffusion. The effectiveness of subsidies will be determined by whether they help better match new technologies to consumer tastes in market segments where they can be made attractive. This is depicted schematically in panel *c.*, where hypothetical price distributed sales for an alternative technology are shown (in red) along with a typical sales distribution of conventional vehicles. The range of the market that can be expected to be affected by a policy, according to McShane *et al* 2012, is a restricted segment of the whole market. A subsidy policy changes the market segment in which the technology is being marketed. This restricted effectiveness can be altered in the future if manufacturers succeed in broadening market choice in order to cover more social identity groups.

Finally, we may ask, is the analysis of the impact of a tax on the price of fuels conceptually any different to that for a tax on the price of vehicles proportional to

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[11] Pigouvian and fuel taxes are proportional to emissions, but other relationships, e.g. exponential, could also be used.

emissions? There is considerable controversy on how buyers of new vehicles take consideration of fuel costs when taking a decision (e.g. Busse *et al* 2013, OECD 2010). In the consumer's perspective, a fuel tax applies later in time than a tax on vehicle prices. There is no clear evidence to support a claim that high emissions vehicles are scrapped more quickly in situations of a high tax on fuels; it mostly changes their trading value in second hand markets, but they likely remain in the fleet (Busse *et al* 2013). There is evidence that fuel taxes lead to reductions of car use and associated emissions (Busse *et al* 2013), which originates from changes in lifestyles, transport modes, load factors and economic impacts, but this subject is outside of the scope of the present paper, which focuses on consumer choices for new vehicles. Thus for our purposes, for a similar tax value, a fuel tax influences the choice of new vehicles in a similar way as a registration tax proportional to emissions.

The extent to which consumers consider fuel costs when choosing a vehicle can be expressed by how much they discount future fuel costs, and the discount rate expresses their time preference. The literature reports values from around 5–10% (Busse *et al* 2013) to 20, even 40% (OECD 2010), depending how the measurement is done. We show in the SI that fleet averaged lifetime fuel costs are most of the time much lower than fleet averaged car prices. The effectiveness of a tax on fuels, per percent of tax applied, is proportionally lower than a tax on car prices proportional to emissions, by the proportion that fuel costs make in the total cost, which partly depends on the discount rate chosen. Calculations using various rates are given in the SI.

Vehicle markets, policy and consumer choices co-evolve with time (e.g. Geels 2006). It is thus understood that the analysis presented is a present day picture which will evolve as manufacturers, consumers and policies co-evolve. While the market structure will likely change, the short term analysis approach proposed here should remain valid.

2.3. Calculating the effectiveness of a fiscal policy

Given comprehensive market data, the likely effectiveness of a fiscal policy given above can be calculated using a regression. However, more detailed insight may be obtained with a model of distributed choice dynamics. We assume, as in the above, that when imposing a new tax on the purchase of vehicles based on emissions, the response of consumers will be to compensate the tax with a lower vehicle cost. We define a symmetric probability distribution (in log scale) $f(\ln P_i - \ln P_j + \ln T_j, \sigma)$, that determines the approximate region in the $(E, \ln P)$ plane where they search the market. P_i is the price of vehicles i considered before the tax comes into force, P_j is the price of vehicles j that consumers decide to purchase

instead once the tax comes into force, however without the tax included, and $r_j = T_j - 1$ is the vehicle dependent tax rate applied on models j based on their rated emissions E_j. σ is the tolerance of consumers to price differences, as a fraction of price, that we assume within 5–20%. Following McShane *et al* 2012, consumers who would, before the tax, have purchased vehicle model i, their probability of choosing model j instead will be proportional to the relative popularity of model j, (it's number of sales N_j):

$$\mathcal{P}_{i \to j} = \frac{N_j f\left(\ln P_i - \ln P_j + \ln T_j\right)}{\sum_k N_k f\left(\ln P_i - \ln P_k + \ln T_k\right)}. \quad (1)$$

Before the tax was applied, N_i consumers per unit time purchased model i. After the tax is applied, these consumers will most likely purchase other models instead, while other consumers from another price tier will purchase model i. Since there were N_i consumers initially considering model i, the number of consumers changing their choice from i to j is thus

$$\Delta N_{i \to j} = N_i N_j \frac{f\left(\ln P_i - \ln P_k + \ln T_k\right)}{\sum_k N_k f\left(\ln P_i - \ln P_k + \ln T_k\right)} \Delta t. \quad (2)$$

Consumers who would have purchased model j also have a non-zero probability of purchasing model i[12], $\Delta N_{j \to i}$. The number of changes of choices between model i and j is

$$\Delta N_{ij} = \Delta N_{i \to j} - \Delta N_{j \to i},$$
$$= N_i N_j \left[g_{ij}(\ln T_i) - g_{ji}(\ln T_i) \right] \Delta t, \quad (3)$$

$$g_{ij}(\ln T_i) = \frac{f\left(\ln P_i - \ln P_k + \ln T_k\right)}{\sum_k N_k f\left(\ln P_i - \ln P_k + \ln T_k\right)}. \quad (4)$$

A new set of vehicle sales when the tax applies can be calculated using $\sum_j \Delta N_{ij} + N_i$. Here, we are interested in calculating the average emissions change related to these change of choices. We therefore add up each choice change's impact on total fleet-year emissions:

$$\overline{\Delta E} = \frac{1}{N_{\text{tot}}} \sum_{ij} E_j \Delta N_{ij}, \quad N_{\text{tot}} = \sum_i N_i, \quad (5)$$

The cumulative amount of registration tax applied and paid associated to new choices is calculated differently since the tax value is zero before the tax is applied:

$$\overline{\ln T} = \frac{1}{N_{\text{tot}}} \sum_{ij} \left(\Delta N_{ij} + N_i \right) \ln T_j. \quad (6)$$

$\overline{\ln T}$ is approximately equal to an average tax rate $\bar{r}$ between zero and one. The value obtained from the ratio $\overline{\Delta E}/\overline{\ln T}$ gives the weighted average emissions

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[12] Especially if the tax was negative, i.e. a subsidy; this equation must be symmetric under changes of sign of $\ln T_i$.

reductions per unit of average tax paid that result from consumer choice[13].

The breadth of the variations between individual choices and the average $\overline{\Delta E}/\overline{\ln T}$ is obtained using $\delta\overline{\Delta E} = \sqrt{\langle \Delta E^2 \rangle - \langle \Delta E \rangle^2}$, where $\langle \rangle$ indicates a population weighted average[14]. This may be interpreted as an uncertainty range; however note that it actually corresponds to the standard deviation of a known distribution of individual behaviour, while $\overline{\Delta E}$ is its mean, with significant amounts of variations cancelling in the aggregate. The same calculation is done for engine sizes $\overline{\Delta S}$.

3. Data and results

Sales for new private passenger vehicles were obtained from Marklines 2014 for all six countries except the UK, for which a more detailed dataset was used for new vehicle registrations from the registration agency DVLA 2012. Entries were matched, model by model, to various data sources, all commercial websites, for vehicle price, engine size and rated emissions: (Car Pages 2012) (UK), AutoUSA.com 2013 (USA), SohuAuto 2014 and CarWale 2013 (India), Zigwheels 2013 and AutoHome 2014 (China), and individual car maker websites. For almost all models matched, we thus obtained their price, rated emissions, engine size and the number sold from the combination of only two data sources by model, enabling to look for correlations. We matched in this way over 4200 models across the six nations[15].

Prices were converted from local currencies to US dollars using www.XE.com (June 2014). Rated emissions and engine sizes are those given by the manufacturers. Marklines 2014 numbers were checked for reliability against total sales given by a number of official data sources, and proved to be reliable. We stress that Marklines data are *comprehensive* in these countries, not samples. Variations of prices for particular models related to optional features were found to remain most of the time within about 5–20% of the basic model prices, providing a rough basis for the value of σ.

Figure 3 shows distributions of private passenger vehicle prices on linear price axes for 2012 registrations in our six major economies. Sales of alternative

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[13] E.g. if $\overline{\Delta E}/\overline{\ln T} = 40$ gCO$_2$ km^{-1}, then for a 10% average tax one obtains mean emissions reductions of 4 gCO$_2$ km^{-1}.

[14] We considered that variations on $\overline{\ln T}$ are the same as those on $\overline{\Delta E}$, since the relationship between emissions and tax is exact, and thus this uncertainty is not double-counted.

[15] 2212 in the UK, 470 in the USA, 630 in China, 188 in India, 455 in Japan, 335 in Brazil. All data sets cover all types of private passenger vehicles with 4 wheels (i.e. we excluded buses and motorcycles). Where numbers of models are higher, such as in the UK, more variants of similar models were included. In the UK, the DVLA 2012 database for new registrations has over 29 000 entries, featuring large numbers of entry variants of similar or identical models. In the UK the matching was restricted to entries with sales of more than 100 units, however for other countries, all Marklines data was used.
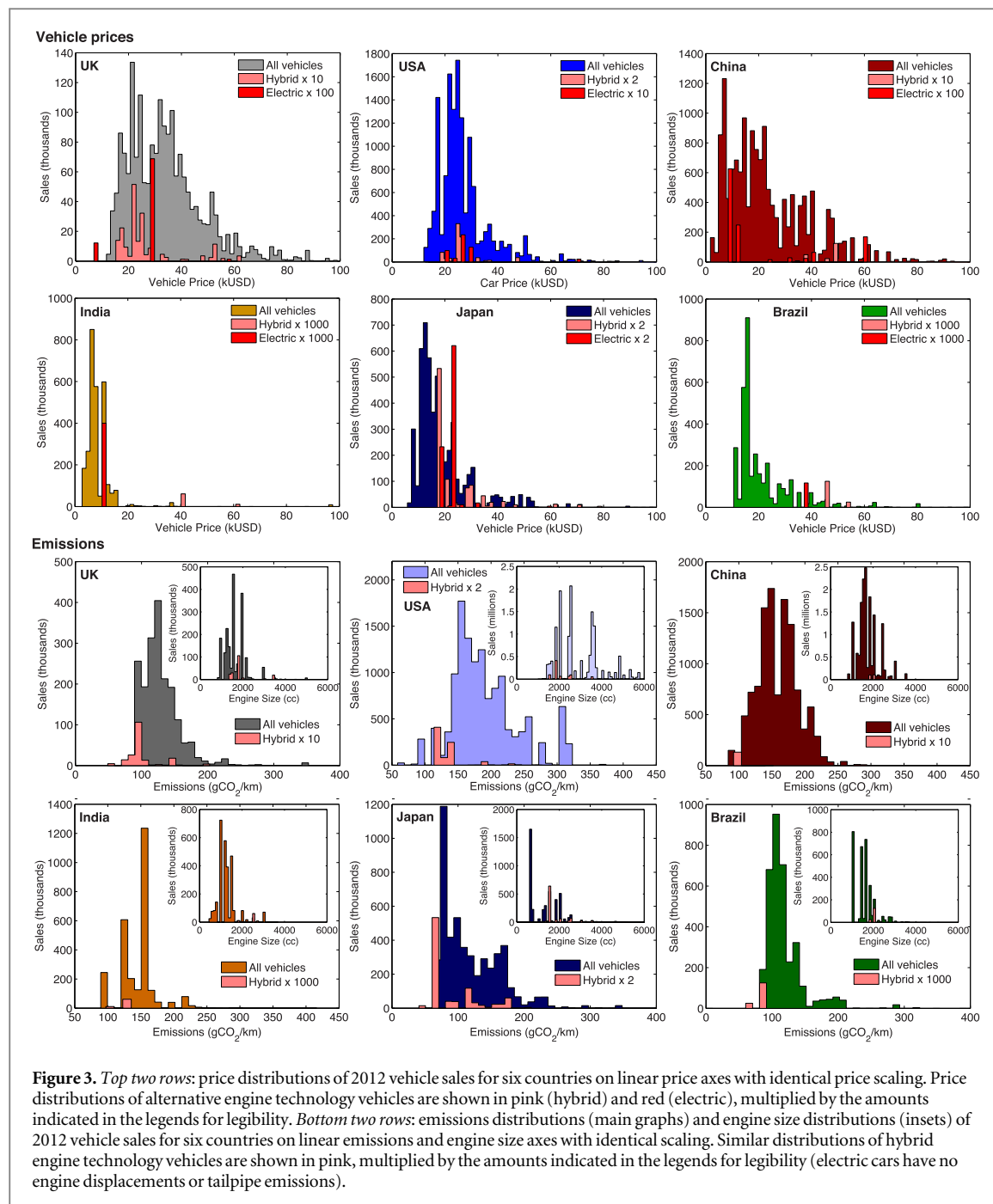
Figure 3. *Top two rows*: price distributions of 2012 vehicle sales for six countries on linear price axes with identical price scaling. Price distributions of alternative engine technology vehicles are shown in pink (hybrid) and red (electric), multiplied by the amounts indicated in the legends for legibility. *Bottom two rows*: emissions distributions (main graphs) and engine size distributions (insets) of 2012 vehicle sales for six countries on linear emissions and engine size axes with identical scaling. Similar distributions of hybrid engine technology vehicles are shown in pink, multiplied by the amounts indicated in the legends for legibility (electric cars have no engine displacements or tailpipe emissions).

technologies, hybrid and electric cars, are shown in pink and red respectively, scaled up for legibility. All graphs have identical abscissa scaling and binning for comparison. All distributions can be parameterised by log-normal distributions[16]. We provide average and median prices with their standard deviations in table 1. The distributions of emissions and engine sizes for the same data are also shown in figure 3, with engine sizes in the insets. Average and median emissions and engine sizes, with their standard deviations, are given in table 1.

Scatter plots with linear univariate correlations between engine sizes and the logarithm of vehicle prices are shown in the left hand panels of figures 4 and 5. The same is given for emissions and engine sizes, middle panels, and for emissions and the logarithm of prices, in the right hand panels. In these bubble graphs, one circle is shown per model, of which the area is scaled with the root of the number of sales[17]. The same

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[16] This was determined by both binning the distributions on a log axis and observing distributions that are roughly normal, and by fitting non-linearly log-normal distributions to the data with linear binning, not shown here.

[17] The root was used for illustrative purposes, to increase the visibility of less popular models and to limit the area of circles for popular models. This enabled the same scaling to be used in all graphs.

Table 1. Properties of vehicle fleets for six representative countries. This includes average (Avg), median (Med) and standard deviations (Std) of prices, engine sizes and emissions. Price distributions are well described by log-normal, and thus their medians are lower than the means.

| Fleet distribution properties		UK	USA	China	India	Japan	Brazil
	Avg	34285	25959	22826	8674	18317	20642
P (USD)	Med	31520	23871	18970	6947	14968	16425
	Std	18640	10570	16633	12418	11526	13770
	Avg	1576	3026	1704	1220	1286	1527
S (cc)	Med	1498	2550	1596	1170	1252	1558
	Std	544	1225	481	445	728	458
	Avg	123.3	185.4	154.1	140.4	113.4	112.2
E (g km^{-1})	Med	118.8	175.6	152.5	145.0	102.2	105.6
	Std	30.1	50.3	30.6	26.6	44.3	28.5
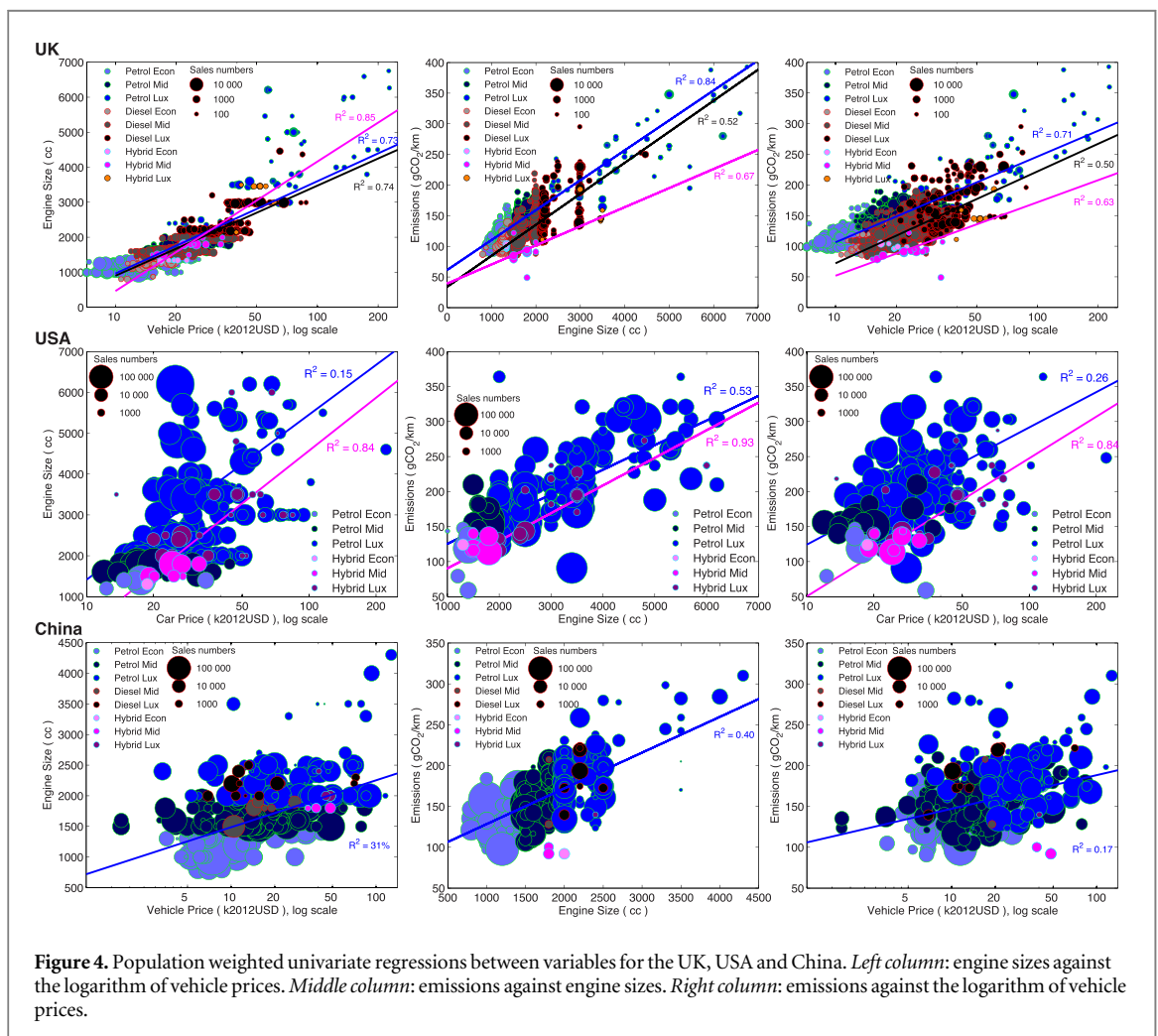
Figure 4. Population weighted univariate regressions between variables for the UK, USA and China. *Left column*: engine sizes against the logarithm of vehicle prices. *Middle column*: emissions against engine sizes. *Right column*: emissions against the logarithm of vehicle prices.

circle size scaling was used in all graphs. Lines are population weighted linear fits of two variables, with coefficients of determination R^2 indicated (all parameters given in table 2). We did not assume that emissions were correlated with engine sizes and prices simultaneously (the regressions are not multivariate); instead we considered that emissions depend on vehicle prices exclusively *through* the relationship between emissions and engine sizes. Uncertainty values on

regressions were obtained using a Monte-Carlo method (details in the SI).

Table 2 gives correlations parameters and effectiveness values ($\overline{\Delta E}/\overline{\ln T}$) calculated using equations 5 and 6, for both types of vehicle tax schemes based either on emissions or engine sizes. The effectiveness values per unit of average tax paid were calculated using three values of consumer tolerance to price changes σ, with uncertainty, over a range of 42 tax
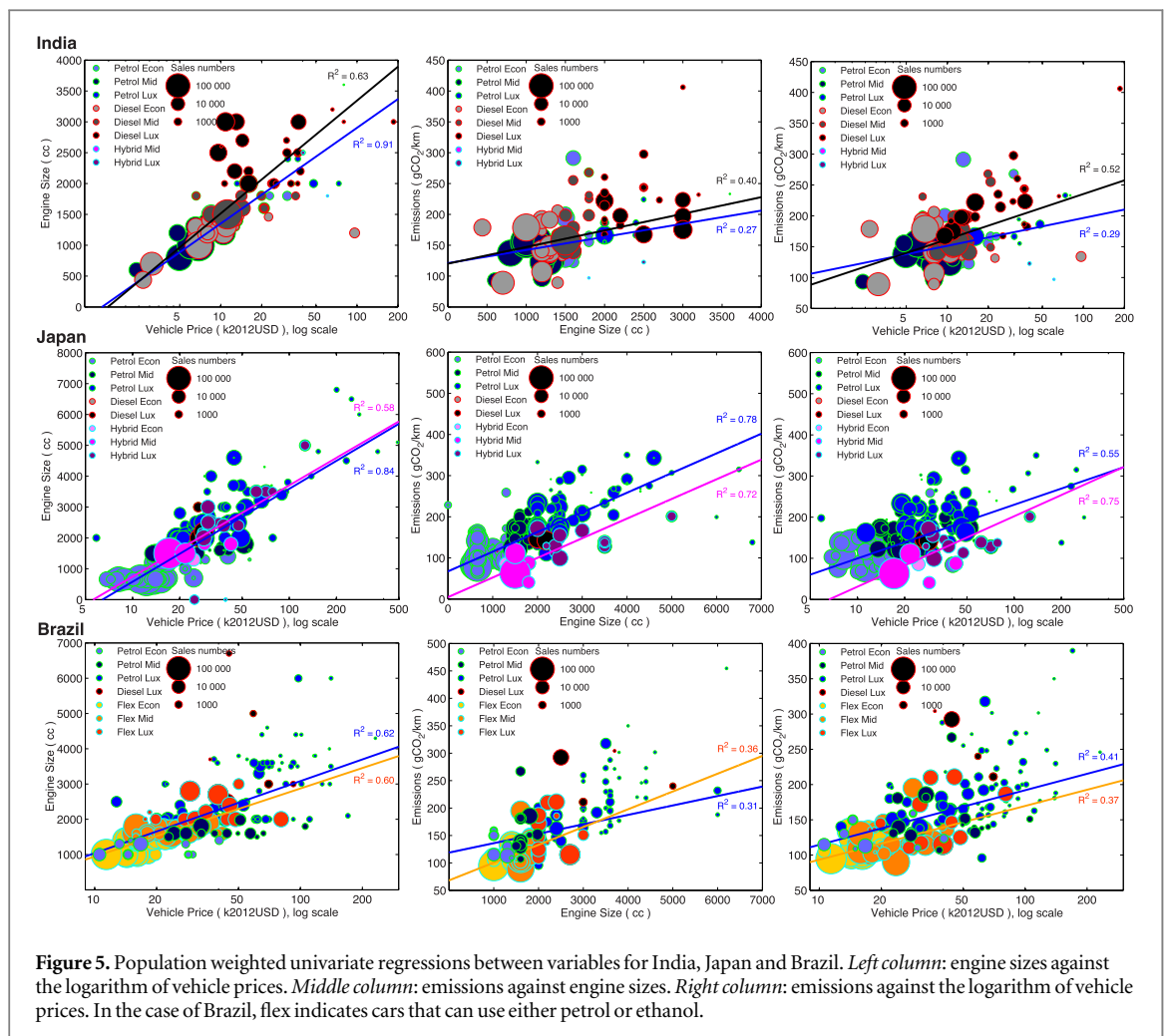
Figure 5. Population weighted univariate regressions between variables for India, Japan and Brazil. *Left column*: engine sizes against the logarithm of vehicle prices. *Middle column*: emissions against engine sizes. *Right column*: emissions against the logarithm of vehicle prices. In the case of Brazil, flex indicates cars that can use either petrol or ethanol.

schemes proportional with values of between −3kUSD up to 20kUSD at 300 gCO$_2$ km^{-1}, for taxes on emissions, and between −3 and 20kUSD cc^{-1} at 5000cc for taxes on engine sizes. The negative ranges were included to test whether negative taxes lead to higher emissions, which effectively leads to the same but negative effectiveness values (see details in the SI). This indicates that increases in relative income leads consumers to choose higher emissions vehicles by the same scaling parameter. The effectiveness of a fuel tax, since it is very similar to a registration tax proportional to emissions, is explored in the SI, however since when including fuel costs in consumer decisions, this tax is expressed in terms of a smaller component of the total operating cost, the effectiveness value obtained is proportionally smaller[18].

Finally, long term impacts of fiscal policies supporting low carbon engine technologies at low levels of diffusion is difficult to carry out using this short term analysis: it requires modelling diffusion dynamics. Here we find however that in the short term, emissions were not always reduced in the model when

subsidising the purchase of hybrid cars since the policy may actually lead consumers to replace low carbon petrol and diesel vehicles by higher emissions hybrids. This depends strongly on the specific details of the policy, which needs more careful study. Meanwhile, subsidising electric cars in all cases decreases emissions, but by amounts orders of magnitude lower than the values of table 2 in which all types of vehicles are involved.

4. Discussion

4.1. Distributions

We observe that the structures of vehicle markets are widely different across countries. Within developed nations, while the price distribution in the UK is comparable to that of the USA, the distribution in Japan is lower and narrower. For the emerging nations, in India and Brazil the distributions are narrow and similar to that of Japan, while the distribution in China is broad, comparable to that in the UK. This is likely related to income distribution, culture and social dynamics within those societies. These variations do not have any clear relationship to

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[18] I.e. this only means that a doubling of the price of fuels has a smaller effectiveness than a tax of which the fee results in a doubling of the average price of the vehicles.

Table 2. (*Top table*) Table of correlation parameters, of the form $y = ax + b$, calculated using the data of figures 4 and 5. Units are in (a^1) (cc = cubic centimetres, L = litres) cc / log P, (b^2) cc, (a^3) gCO$_2$ km^{-1} L^{-1}, (b^4) gCO$_2$ km^{-1}, (a^5) gCO$_2$ km^{-1}/log P and (b^6) gCO$_2$ km^{-1}. R^2 indicates coefficients of determination expressing the strengths of the correlations. Uncertainty values for correlation parameters were obtained using Monte-Carlo analysis, described in the SI. (*Bottom table*) Calculated values for the effectiveness of taxes on either rated emission or engine sizes are given, with their aggregate impacts on both rated emissions and engine sizes for new potential fleet-years. These are calculated using equations 5 and 6, i.e. emissions or engine size reductions expected for 100% tax applied (see calculation details with examples in the SI). Uncertainty ranges correspond to the standard deviations of variations in individual consumer choices with respect to the means. They should not be interpreted strictly as uncertainty values on the means, since significant amounts of variations in consumer choices cancel out in the aggregate, even when their variations are significant.

Correlation parameters

		Engine size versus log price			Emissions versus engine size			Emissions versus log price		
		a^1	b^2	R^2	a^3	b^4	R^2	a^5	b^6	R^2
UK	Petrol	1145	−9581	0.73	48.87	61.3	0.84	60.8	−453.2	0.71
	Diesel	1118	−9395	0.74	50.6	33.9	0.52	65.1	−527.5	0.50
	Hybrid	1605	−14314	0.85	31.2	39.2	0.67	52.0	−427.3	0.63
	All	**1089 ±76**	**−9072 ± 785**	**0.76**	**40 ± 4**	**64 ± 6**	**0.56**	**43 ± 6**	**−314 ± 57**	**0.42**
USA	Petrol	1755	−14747	0.15	35.2	90.2	0.53	72.7	−545.2	0.26
	Hybrid	1868	−16929	0.84	39.5	50.7	0.93	85.6	−738.0	0.84
	All	**1751 ± 333**	**−14742 ± 3357**	**0.13**	**36 ± 9**	**87 ± 21**	**0.54**	**73 ± 21**	**−545 ± 207**	**0.23**
China	Petrol	333	−1578	0.32	43.8	84.6	0.40	17.9	−17.6	0.17
	Diesel	259	−623	0.31	97.9	−19.2	0.31	36.2	−148.9	0.14
	Hybrid	456	−2976	0.05	23.7	51.6	0.40	−44.6	573.4	0.42
	All	**331 ± 120**	**−1560 ± 1178**	**0.32**	**43 ± 11**	**84 ± 21**	**0.40**	**18 ± 7**	**−66 ± 67**	**0.17**
India	Petrol	673	−4847	0.91	21.3	121.0	0.27	19.6	−29.2	0.29
	Diesel	794	−5796	0.63	26.9	120.2	0.40	31.9	−131.8	0.52
	All	**790 ± 273**	**−5828 ± 2455**	**0.75**	**28 ± 25**	**116 ± 39**	**0.40**	**30 ± 24**	**−114 ± 218**	**0.45**
Japan	Petrol	1308	−11466	0.84	47.8	67.2	0.78	56.7	−422.4	0.55
	Hybrid	1286	−11098	0.58	47.8	4.5	0.72	74.4	−653.8	0.75
	All	**1333 ± 210**	**−11693 ± 2051**	**0.84**	**41 ± 14**	**67 ± 14**	**0.67**	**50 ± 20**	**−349 ± /87**	**0.49**
Brazil	Petrol	887	−7130	0.62	17.2	118.8	0.31	33.6	−194.6	0.41
	Flex fuel	838	−6777	0.60	32.4	68.1	0.36	33.1	−211.6	0.37
	All	**964 ± 187**	**−7980 ± 1875**	**0.84**	**35 ± 18**	**67 ± 24**	**0.73**	**42 ± 16**	**−328 ± 157**	**0.76**

Tax effectiveness $\dfrac{\Delta E}{\ln T}$ and $\dfrac{\Delta S}{\ln T}$

Tax on emissions

	Emissions reductions (in gCO$_2$ km^{-1})			Engine size reductions (in cc)		
	$\sigma = .05$	$\sigma = 0.1$	$\sigma = 0.2$	$\sigma = 0.05$	$\sigma = 0.1$	$\sigma = 0.2$
UK	32 ± 15	32 ± 11	31 ± 8	781 ± 157	780 ± 118	775 ± 92
USA	93 ± 52	82 ± 42	64 ± 30	2128 ± 673	1941 ± 552	1465 ± 406
China	25 ± 22	25 ± 19	21 ± 14	327 ± 231	294 ± 214	263 ± 167
India	38 ± 57	37 ± 48	32 ± 36	717 ± 389	637 ± 331	549 ± 261
Japan	26 ± 39	34 ± 28	35 ± 18	859 ± 384	874 ± 278	884 ± 222
Brazil	33 ± 86	32 ± 69	26 ± 32	820 ± 902	748 ± 746	584 ± 381

Tax on engine size

UK	33 ± 12	33 ± 9	33 ± 8	858 ± 139	861 ± 112	870 ± 103
USA	83 ± 56	78 ± 44	64 ± 32	2361 ± 745	2156 ± 613	1682 ± 460
China	25 ± 25	23 ± 19	21 ± 12	265 ± 263	276 ± 211	302 ± 148
India	26 ± 93	22 ± 67	17 ± 45	781 ± 664	731 ± 475	623 ± 347
Japan	26 ± 42	36 ± 26	38 ± 23	1155 ± 470	1208 ± 318	1206 ± 302
Brazil	34 ± 77	35 ± 60	29 ± 32	965 ± 850	834 ± 656	658 ± 374

GINI coefficients for these countries or other measures of inequality[19].

[19] While a relationship with income distribution apparently exists (as suggested by figure 1), the relationship with inequality is likely more complex.

We also find disparities between distributions of engine sizes across nations, seemingly unrelated to any particular physical or geographical features or constraints. These are unrelated to differences in price distributions. For example, despite similar price distributions in the UK and the USA, the distribution

of engine sizes in the USA covers 1000–6000cc, while that of the UK is concentrated between 1000 and 3000cc.

We find similar disparities across countries in the distributions of rated emissions. Cars in the USA have the broadest emissions distribution, while Brazil and India have the narrowest, followed by the UK and Japan. The USA has the highest average and median emissions, while Brazil and Japan have the lowest. The upper end of the emissions distribution in the USA has a value of almost twice that in Brazil or India, and 50% higher than the UK and Japan. These market structure differences are identified as tied to cultural and behavioural characteristics of consumers, but also tied to their regulatory and tax history (including fuel taxes, which are markedly lower in the USA).

Price distributions of alternative technologies are also very different across countries. The UK is the nation with the widest choice for hybrid cars while it is the USA for electric cars. Japan sees the highest market penetration in absolute numbers for both, however these cover only about the upper half of the price range. The availability and penetration of alternative technologies in emerging nations is very low, and in India and Brazil the price of hybrid vehicles is prohibitively higher than what people spend on vehicles. This means that in the USA and the UK, most consumers can access an alternative vehicle technology in price ranges near what they are willing to pay for a vehicle. Meanwhile this is not the case in emerging countries where the market coverage of these technologies is very patchy. Japan is in an intermediate situation. This limits highly the effectiveness of alternative technology support policies such as subsidies.

The emissions distributions of hybrid vehicles have lower averages than those of ICE vehicles. However their emissions are not always lower than the fleet-year averages, where for example in the USA, many hybrid vehicles have higher emissions than most petrol cars. A similar observation can be made about Japan and the UK. As the availability of technologies in these countries is comparatively high, this feature correlates with the fact that emissions of hybrid vehicles sold to consumers purchasing in large engine size brackets are higher than emissions of any vehicles in lower engine power brackets; however emissions of high power hybrids are still lower than those of the vehicles they most likely replace in the same price or engine size brackets. It suggests that subsidising hybrid vehicles can lead to increased emissions.

4.2. Correlations and policy effectiveness

Correlations between emissions and engine sizes exist in all countries, and this is due to this relationship being mostly an engineering one (figures 4 and 5, middle panels, and table 2, middle columns), where larger engines are more powerful, use more energy (fuel) and therefore produce higher emissions per distance driven. The parameters of these correlations are similar but differ across countries, and this may be related to either or both: (1) additional features coming with higher power vehicles that also use energy, of which the purchase differ between countries, or (2) different levels of technology sophistication in vehicles leading to varying levels of fuel efficiency at similar engine sizes.

However, correlations do not always exist between prices and engine sizes (or power, left panels). Their strength (the R^2 parameters), when they exist, vary considerably between countries. In particular, while a very clear log-linear relationship exists in the UK, India, Japan and Brazil, the relationships are very weak in the USA and China. This indicates a clear hierarchy between engine size and prices in vehicle markets in the UK, India, Japan and Brazil, but none in the USA or China. It thus emerges that in China and the USA, the size of the engine is not a major determinant of the price in manufacturer marketing decisions, while it is elsewhere.

Depending whether relationships exist between prices and engine sizes, the existing relationships between emissions and engine sizes bridges to possible relationships between emissions and prices, although with weaker correlations than with engine sizes. Thus, wherever relationships exist between engine sizes and prices, they exist between emissions and prices, and conversely where they don't exist. Thus very weak relationships exist between emissions and prices in the USA and China, while they are stronger in other countries.

The scaling parameters of these relationships (table 2) indicate the likely consumer response to fiscal policy in these countries, with a confidence measure given by R^2. However, we consider that more reliable values, along with measures of variations in consumer behaviour, are given by using the average effectiveness of policy per average unit tax paid determined using equations 5 and 6. According to these, one expects a high response to policy of 0.6–0.9 gCO_2 km^{-1} per percent of average tax in the USA (60–90 gCO_2 km^{-1} for 100% tax), indicating a very high abatement potential with moderate tax values. Meanwhile in all other countries, the fleet-year-average effectiveness lies in the area of 0.3–0.4 gCO_2 km^{-1} per percent of average tax. These values agree with those obtained with regressions. It has been suggested that indirect emissions taxes based on engine sizes are less efficient at reducing emissions than direct taxes (He and Bandivadekar 2011). From the consumer choice perspective, this is not supported by our data, where little difference is observed in aggregate between the two tax schemes. Therefore, this assertion is likely to be only true from the supplier perspective, where manufacturers are likely to attempt to circumvent engine size-based taxes by designing vehicles with smaller engines for the same amount of power (e.g. with turbo-chargers).

Negative tax values also lead to increases in emissions by the same factors, suggesting that relative

consumer income increases lead to increasing emissions. This can explain current trends of increase in engine sizes in the UK[20], Ireland (Gallachoir *et al* 2009) and Germany (Zachariadis 2013). Finally, the effectiveness of taxes are orders of magnitude higher in the short term than those obtained when policy only supports the diffusion of alternative engine technologies with subsidies.

5. Conclusion and policy implications

The data and analysis presented in this paper has clear policy implications. We have provided qualitative and quantitative tools that can help understand and determine the likely outcome of chosen policies targeting consumer vehicle choice for emissions reductions. We find that it is possible to calculate average values for the effectiveness of fiscal policies using market data, and the results differ by country analysed. Our data show a high diversity of consumer choices, tied to socio-economic differences, different in every country, with repercussions on fleet-year emissions and prospects for reductions.

The effectiveness of fiscal policies for incentivising consumer choices towards lower emissions vehicles of any type, according to this work, is of around 0.3–0.4 $gCO_2 \, km^{-1}$ per percent of a proportional emissions tax applied to vehicle sale prices, in all countries except in the USA where it is of 0.6–0.9 $gCO \, km^{-1}$ per percent of tax, 2 to 3 times higher, with higher uncertainty. Taxes based on either emissions or engine sizes have similar impacts. However, using on their own policies to support the diffusion of new engine technologies, given our model of consumer choice, has an effectiveness orders of magnitude lower than tax policies that apply to all types of vehicles simultaneously. Fuel taxes, while reducing transport use, have a similar effect on vehicle choices; however since they only apply to a minor component of total costs, they result in a lower impact. These values, however, apply only in the short term. In the long term, the choice of consumers will influence manufacturer marketing strategy, which will alter the range of vehicles offered to consumers, which will in turn again alter consumer choices, and policy will need to be re-assessed over time. The complex co-evolutionary process between consumer choices, manufacturer strategy and policy-making is a subject that requires further research.

We conclude that both the stringency and the structure of policies are important. Fiscal policies based on emissions are essential to reduce fleet-year emissions across the range of consumer groups; policies supporting changes in engine types are not likely highly effective in the short term, and this is due to a patchy market coverage by hybrid and electric vehicles. Meanwhile, we show that fiscal policies following

the 'polluter pays' principle generate an incentive that decreases with consumer income. If, in order to completely decarbonise private transport (e.g. the UK's 2050 target), policy strategy is to transform the market towards a dominance of alternative technologies, comprehensive technology demand pull and supply push policies must be used in close collaboration with manufacturers, which may yield positive results in the long term. However, both emissions-based fiscal policies and technology policies must be used simultaneously in order to reduce emissions in both the short and the long term.

Acknowledgments

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