Equity Implications of Vehicle Emissions Taxes

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Abstract

This paper considers the equity implications of vehicle emissions taxes by examining the incidence of a tax on local pollutants. It uses emissions data from the California Air Resources Board and household vehicle and income data from the US Consumer Expenditure Survey. It incorporates household price responsiveness that differs across income groups into a consumer surplus measure of tax burden. Since poor vehicle owners spend more on miles as a proportion of their income and drive vehicles that pollute more per mile than those owned by the wealthy, the incidence of tax on vehicle emissions falls relatively heavily on them. This burden, however, is mitigated to some extent by low vehicle ownership rates and high price responsiveness in the lower half of the income distribution. A uniform tax on miles that does not distinguish between dirty and clean vehicles is less regressive than the emissions tax.

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1.0 Introduction

Between 1980 and 2000, vehicle miles travelled (VMT) in the United States increased by over 76 per cent (FHWA, 2003). This dramatic increase in vehicle use frustrates attempts to reduce local air pollution. As of January 2004, for example, fourteen metropolitan areas in the United States are classified as extreme or severe ozone nonattainment areas.[1] Faced with these challenges, policy makers seek cost-effective means to attain abatement goals. While command-and-control standards still dominate vehicle pollution policy in the United States, many studies have shown that taxes and other incentives can attain the same amount of pollution reduction at lower cost (see for example Bohm and Russell (1985) and Harrington *et al.* (1994)). Pollution taxes also generate revenue that can be used to fund government projects or to reduce taxes on labour or investment.

One argument against taxes on vehicle pollution and gasoline is that they disproportionately burden poor households. Several studies confirm fears that a tax on gasoline is regressive. Only Walls and Hanson (1999) and Sevigny (1998), however, explicitly consider the incidence of a vehicle emissions tax.[2] Both those studies consider local pollutants: Walls and Hanson examines a tax on hydrocarbons (HC) and Sevigny examines a tax on HC, carbon monoxide (CO), and oxides of nitrogen (NO_x).

Five main contributions distinguish this paper. First, it considers a tax on two particularly damaging pollutants, particulate matter (PM) and sulphates (SO_4) along with CO, HC, and NO_x. Second, it uses data from the California Air Resources Board to estimate vehicle emissions per mile as a function of vehicle vintage, engine size, import status, and vehicle type. Walls and Hanson (1999) and Sevigny (1998) assign emissions per mile to vehicles in households according to vehicle vintage. Other papers estimate vehicle emissions functions as complicated as those estimated here but do not conduct incidence analysis (see for example Harrington (1997) and Kahn (1996a)).

Third, my analysis examines the effect of including households that do not own vehicles. Walls and Hanson (1999) includes households without

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[1] See the US Environmental Protection Agency's Green Book Nonattainment listings at http://www. epa.gov/oar/oaqps/greenbk.

[2] Kahn (1996b) determines the effect of households' neighbourhoods' median incomes on the likelihood of passing vehicle emissions tests. Dill *et al.* (1999) consider the incidence of existing California vehicle licence fees that are not based on emissions. Other papers analyse the incidence of other pollution taxes. Brännlund and Nordström (2004) and Cornwell and Creedy (1997), for example, examine the incidence of carbon taxes. Poterba (1991) and West and Williams (2004) consider the incidence of a gasoline tax, West (2004) examines taxes on miles and subsidies to new vehicles, and Metcalf (1999) calculates incidence for a variety of environmental taxes but not a tax on vehicle emissions.

vehicles but does not compare results with simulations that omit these households. Sevigny (1998) considers only vehicle owners. Measures that ignore households that do not own vehicles will overstate the incidence on income groups with fewer vehicle owners, and understate the incidence on groups with more vehicle owners.

Fourth, I incorporate households' responses to an emissions tax and a tax on vehicle miles travelled into incidence calculations. I use elasticities of demand for VMT that differ across income groups, obtained from estimation using household data from the 1997 US Consumer Expenditure Survey. Other papers calculate incidence with no price responsiveness or with no income-varying price responsiveness.[3] They assume away the possibilities that, for example, poorer households are more price responsive because their spending on miles occupies a larger fraction of their budget, or that wealthier households are more price-responsive because they have more transport options. To the extent that demand elasticities vary with income, measures that ignore this will overstate the incidence on income groups with relatively elastic demand, and understate the incidence on groups with relatively inelastic demand.

Fifth, incorporation of price responsiveness allows me to use the change in consumer surplus to calculate the reduction in welfare due to the tax.[4] This measure adds the Harberger triangle (Harberger (1962)), to the rectangle of tax paid.[5] Ignoring this triangle biases incidence estimates. For example, if poor households are more price responsive than wealthy households, they will escape more of the tax than wealthy households, mitigating regressivity, but their Harberger triangles will be larger, exacerbating regressivity.

The incidence of vehicle emissions taxes depends on the relationships between emissions per mile, VMT, and income. Thus the income measure used is a critical determinant of incidence. For the purposes of incidence analysis, an ideal income measure places a money value on material well being with which the value of welfare loss from the policy change can be scaled. Consumption is a better indicator of material well being than

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[3]Metcalf (1999), Poterba (1991), and Walls and Hanson (1999) calculate incidence assuming no price responsiveness. Sevigny (1998) allows price responsiveness to differ according to how many vehicles a household owns but not according to income.

[4]West (2004) uses consumer surplus when examining incidence of a miles tax. Brännlund and Nord-ström (2004) and Cornwell and Creedy (1997) use equivalent and compensating variation to measure the welfare effects of carbon taxes.

[5]The Harberger triangle is usually associated with deadweight loss. A distorting tax, such as a tax on labour or capital, creates a deadweight loss by raising price above marginal cost. A tax on emissions, on the other hand, is corrective rather than distorting — it raises price *to* marginal cost. Rather than creating a deadweight loss, emission taxes eliminate one.

annual income.[6] University students and retired people, for example, may have very low incomes but high levels of consumption and thus high material well being and working households can maintain levels of well being in the face of temporary reductions in income by taking money from savings or by borrowing. In good economic times, households smooth consumption by saving.[7] For these reasons, and since the Consumer Expenditure Survey contains detailed and accurate data on expenditures, I use total consumption expenditures as my measure of income.

This paper focuses on incidence of the costs of vehicle emissions taxes, ignoring the distribution of the external benefits of reduced vehicle pollution and miles driven. Incorporating such benefits would reduce tax burdens for all income groups and might have important distributional effects if the benefits are unevenly distributed across income groups.[8] In addition, I do not consider the distributional effects of rebating tax revenue or using it to reduce other taxes.[9] As pointed out in Small (1983), the concept of regressivity is not appropriate for a corrective tax where there are net welfare increases. For the purposes of this paper, therefore, I use the terms "regressive" and "regressivity" when comparing one scenario to another, not to characterise the incidence of one particular policy.

Among vehicle owners, emissions per mile of local pollutants decrease as income increases; poor vehicle owning households drive vehicles that pollute more than those owned by wealthy households. But a significant proportion of the population owns no vehicles and therefore emits no vehicle pollution.[10] This population is concentrated in the lower third of the income distribution. In addition, poor households are more price responsive than wealthy ones; elasticities of *VMT* demand in the poorer deciles of the income distribution are about twice as large as those in the wealthier deciles. While the incidence of tax on local vehicle pollutants falls relatively heavily on poor households, this burden is mitigated to

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[6] Sevigny (1998) uses annual income. Walls and Hanson (1999) use a measure of lifetime income, which usually produces incidence estimates more similar to those that use consumption. Poterba (1991), West (2004), and West and Williams (2004) use consumption to analyse environmental policy incidence but do not consider a vehicle emissions tax.

[7] See Slesnick (2001) for further discussion of differences among income measures. Consumption may be a better approximation to welfare than income, but it is still an approximation. It is problematic, for example, for households that cannot smooth consumption by saving or borrowing.

[8] See Baumol and Oates (1988) and Brooks and Sethi (1997) for general discussion of the distribution of benefits.

[9] I assume that the government discards all tax revenue. For incidence analysis of a gasoline tax whose revenues are used to reduce a labour tax, see West and Williams (2004). For analysis of a carbon tax whose revenues are used to reduce a variety of taxes, see Brännlund and Nordström (2004).

[10] Of course, households that do not own vehicles may pollute indirectly by riding diesel-fuelled buses or trains fuelled by electricity generated by coal-burning power plants.

some extent by low vehicle ownership rates and high price responsiveness in the lower half of the income distribution.

Since poor households drive dirtier vehicles than wealthy households, a uniform tax on miles that does not distinguish among vehicles is less regressive than the emissions tax. In the case of vehicle emissions of local pollutants, the pollution control policy that is easier to implement is also less regressive.

Section 2 provides details on the incidence measures employed in the paper. Section 3 discusses the data, emissions per mile estimation, and elasticity estimation. Section 4 presents the incidence results and Section 5 concludes.

2.0 Measuring the Incidence of Emissions Taxes

A household's total emissions equals the sum of emissions per mile of each pollutant i (EPM_i^h) times vehicle miles travelled (VMT^h). An efficient tax on emissions equals the money value of marginal external damages (MED) per unit of emissions.[11] A tax per gram of emissions could be assessed by measuring the emissions per mile and multiplying by miles travelled, where miles travelled is obtained by comparing current odometer readings with previous readings. The total tax paid by a household (T^h) on local pollutants would equal:

$$T^h = \sum_i MED_i EPM_i^h VMT^h. \tag{1}$$

The tax on emissions is thus equivalent to a vehicle-specific tax on miles.

An ideal measure of the incidence of a tax on vehicle emissions would calculate the general-equilibrium changes in prices that would occur throughout the economy in response to the tax, and then calculate the effects of those price changes on households' welfare. A tax on vehicle emissions, even if it was assessed only on households, would presumably affect prices of fuel and automobiles and thereby affect many sectors of the economy. Calculating such effects, however, requires a great deal of information, most notably the demand and supply elasticities for all affected industries, and the distribution of ownership of firms in those industries. Thus, for simplicity, I focus on the short-run partial equilibrium incidence of the tax. I assume that in the short run, households respond to the emissions tax by reducing the number of miles they drive, but not by

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[11]This is the familiar Pigouvian tax, introduced in Pigou (1932).

switching to less polluting vehicles or other modes of transport.[12] I also assume that the supply of consumer goods is perfectly elastic. This implies that the imposition of the emissions tax does not affect the producer prices of related goods such as fuel and vehicles, and thus the entire burden of the tax falls on consumers.[13]

Each household faces its own initial price per mile, determined by fuel costs and other variable costs per mile. Each household also faces its own with-tax price per mile, which is the initial price per mile plus $\sum_i MED_i EPM_i^h$. While I assume that MED are the same for all households, emissions per mile are vehicle specific.[14] In addition, I allow VMT demand elasticities to vary across income deciles.

I use the change in household consumer surplus to measure the change in household welfare due to the tax. Because I consider short-run incidence, where households respond to the emissions tax only by reducing miles, consumer surplus is defined as the change in the area under the household's VMT demand curve over the quantity purchased, which reflects the effect of changing prices on utility. Assuming a linear VMT demand curve, the change in consumer surplus for household h can be expressed:

$$\Delta CS^h = \sum_i MED_i EPM_i^h VMT_1$$

$$+ \tfrac{1}{2} \left(\sum_i MED_i EPM_i^h \right) (VMT_1^h - VMT_0^h), \qquad (2)$$

where the first term is T^h, the rectangle of tax paid defined in equation (1), after the household has adjusted miles in response to the tax. The second

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[12]Studies confirm that in the long run, households respond to higher gasoline prices by switching to more fuel efficient vehicles (see Dahl and Sterner (1991)), but none examine the differences in this adjustment across income groups. We can therefore only speculate that since wealthier households can more easily afford to replace a dirty car with a less polluting one and thus avoid more of the tax, the emissions tax may be more regressive in the long run.

[13]While no study examines the degree to which a vehicle emissions tax would be borne by gas and vehicle producers versus consumers, we might expect the gas tax to have a similar incidence. CBO (2003) finds that gasoline consumers would bear nearly 85 per cent of the total long run costs of an increase in the gas tax. Vehicle manufacturers would bear part of the burden by lowering vehicle prices; Goldberg (1998) finds that carmakers would respond to gas tax increases by lowering the relative prices of less fuel efficient luxury and standard sized cars. These studies, however, do not address the possible differences in the producer incidence of a gas tax across different income groups. To the extent that gasoline and vehicle suppliers bear part of the tax burden, my estimates will overstate the incidence on households that consume gasoline and vehicles, and will understate the incidence on households that own firms that supply gasoline and vehicles. If firm owners are concentrated in the top deciles, this would mean that my estimates would overstate the tax's regressivity.

[14]A referee points out that to the extent that poorer people live and drive in relatively densely populated areas, the true MED for them will be higher because MED is nearly proportional to population density.

term is the Harberger triangle, which, with a linear demand for VMT, is half of the tax per VMT times the change in VMT that occurs in response to the tax.[15]

For ease of comparison with previous studies, I also calculate the change in household welfare as the tax paid by that household, using equation (2) but omitting the second term. I divide both welfare loss measures by total expenditures to obtain tax burden as a percentage of income. Comparison of these ratios across deciles allows one to determine which income groups bear more of the burden of a particular tax relative to their incomes.

To compare the overall incidence of two different tax policies, however, one needs to measure the distribution of the tax burden across all deciles for each policy. Suits (1977) derives just such a measure. The Suits index is traditionally calculated using tax paid as the measure of welfare change. For a given tax policy T_x, the Suits index calculated over ten income deciles is:

$$S_x = 1 - \left[\sum_{i=1}^{10} \tfrac{1}{2}[T_x(y_i) + T_x(y_{i-1})](y_i - y_{i-1}) \right] / K, \qquad (3)$$

where y_i is the accumulated percentage of income through decile i, $T_x(y_i)$ is the accumulated percentage of total tax paid through decile i, and K is a constant. Scaling the term in brackets by K and subtracting it from one gives an index similar to the Gini coefficient. The Suits index ranges from -1 for a perfectly regressive tax, to zero for a proportional tax, and to $+1$ for a perfectly progressive tax.

For ease of comparison with other studies, I present the traditional Suits index using tax paid. I also calculate a Suits index using consumer surplus as the measure of welfare loss. The Suits index equivalent for consumer surplus (taxes paid plus the Harberger triangle) is obtained by defining $T_x(y_i)$ as the accumulated percentage of consumer surplus lost through decile i.

3.0 Data, Emissions per Mile Estimation, and Elasticity Estimation

Determining the incidence of a tax on vehicle emissions requires data on household income, emissions per mile, vehicle miles travelled, and a measure of households' responsiveness to the tax. This section describes

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[15]For detailed discussion of this and related measures of welfare loss, see Willig (1976).

the three main sources of data used here: the US Consumer Expenditure Survey (CEX), the California Air Resources Board Light Duty Surveillance Program (CARB), and the American Chambers of Commerce Researchers Association (ACCRA) cost of living index. It also explains estimation of emissions per mile and estimation of elasticities of VMT with respect to operating costs per mile.

3.1 Household data, operating costs, and VMT derivation

The household data consists of 7,073 households from the 1997 US Consumer Expenditure Survey that own zero, one, or two vehicles.[16] Households that own no cars make up 24 per cent of the sample, 45 per cent own one vehicle, and 31 per cent own two vehicles. The CEX includes total expenditures, the amount spent on gasoline, and detailed information on each household's vehicles. Variables in the vehicle file include year, make, model, number of cylinders, odometer reading, the amount paid for the vehicle, and other characteristics.

I define the operating cost per mile for each vehicle as the price of gasoline divided by fuel efficiency, plus maintenance and tyre costs per mile. The ACCRA cost of living index lists for each quarter the average prices of regular, unleaded, national-brand gasoline for over 300 US cities. Since the CEX reports state of residence of each household, but not city, I average the city prices within each state to obtain a state gasoline price for each calendar quarter. Then I assign a gas price to each CEX household based on state of residence and CEX quarter.

Unfortunately, the CEX does not record vehicles' fuel efficiencies. I therefore use data from the California Air Resources Board (CARB) to estimate a regression of miles per gallon (MPG) on engine size and vehicle vintage.[17] Smaller or newer vehicles are more fuel efficient. For one-vehicle households, fuel efficiency is calculated directly from the regression results. For two-vehicle households, I calculate the fuel efficiency of each two car pair by averaging the two cars' estimated efficiencies.

I then calculate the fuel cost per mile as the price of gasoline divided by fuel efficiency. The ORNL (1998) provides maintenance and tyre costs per mile, by vehicle vintage. I add these per mile costs to the fuel cost per mile to obtain operating costs per mile.

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[16] Eighty-two per cent of 1997 CEX households own zero, one, or two vehicles (for comparison, the 1995 US Nationwide Personal Transportation Survey lists this number as 81 per cent (ORNL (2000)). The CEX data for households with more than two vehicles is very spotty; 70 per cent of these households have missing data for engine size or vintage of at least one vehicle. Households with three or more vehicles have higher average expenditures than the households included here; by ignoring them this study focuses on a less wealthy portion of the income distribution.

[17] For more detail on these data and results from the MPG regression, see West (2004).

The CEX reports one number per household for gasoline expenditures. To obtain VMT driven per household, I divide the household's gas expenditure by its gas price to get gallons of gas consumed. Then, I multiply gallons by fuel efficiency to obtain VMT for the household.[18,19]

3.2 Emissions per mile estimation

The CEX does not include information on vehicle emissions. The CARB contains complete information on 671 vehicles' emissions of carbon monoxide (CO), hydrocarbons (HC), and oxides of nitrogen (NO_x) in grams per mile.[20] Several studies find that important determinants of vehicle emissions include vintage, cumulative miles (obtained from an odometer reading), engine size, vehicle class (light-duty truck versus car), and import status (that is, manufactured by a US company versus a non-US company) (see for example Bin (2003), Harrington (1997), and Kahn (1996a)). All of these variables are included in both the CEX and the CARB data. Odometer readings, however, are missing for more than 10 per cent of the CEX vehicles in my sample and, since CEX interviewers do not check the odometers of the vehicles but instead rely on the household to report accurately their vehicles' mileage, readings are subject

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[18]Using total VMT versus the VMT in each vehicle ignores the possibility that households respond to changes in the operating cost per mile by driving more in one vehicle and less in another. Green and Hu (1985) find that substitution among vehicles within a household in response to changes to the price per mile is negligible, and so the bias is not likely to be large.

[19]Miles data by vehicle is available in two data sets that were conducted close in time to the 1997 CEX: the 1994 Residential Transportation Energy Consumption Survey (RTECS) and the 1995 Nationwide Personal Transportation Survey (NPTS). Since both surveys, however, report income only as categorical indicators for annual income ranges, I do not use them in this paper. I did conduct a simple comparison of the distribution of VMT across deciles from these surveys. I assigned each RTECS and NPTS household that owns fewer than three vehicles a random income from a uniform distribution within the household's income range category and divided the households into deciles. Both the RTECS and NPTS show more miles driven in lower deciles and fewer in upper deciles than shown in the CEX. This is due in part to the fact that both surveys contain fewer households with no vehicles than the CEX. Using those surveys' miles and income data (but CEX-derived price elasticities and assuming that NPTS and RTECS vehicle characteristics distributions are similar to those in the CEX) would result in incidence estimates that are more regressive than those found here.

[20]The CARB's Light-Duty Vehicle Surveillance Program, Series 13 and 14, was part of ongoing efforts by the CARB to accumulate vehicle emissions data, to investigate vehicle maintenance practices, and to determine the frequency and effect of tampering with pollution control equipment. The programme performed dynamometer tests used in California's Smog Check II Program in Enhanced Areas. Such tests do not capture all increased emissions that occur in real-world driving conditions. They miss those emitted, for example, during heavy acceleration or high-load operating conditions. To the extent that driving behaviour and loads differ across income groups, my estimates will understate incidence on households that are aggressive drivers or carry heavy loads and overstate incidence on households that drive less aggressively or with light loads. For more details on the benefits and disadvantages of different vehicle emissions testing methods, see BEST (2001).

to extreme rounding and other measurement error, and so I do not include odometer readings in emissions per mile estimation.[21]

I estimate one regression each for CO, HC, and NO_x as functions of indicators for the number of cylinders in the engine (four, six, or eight), vintage (older than 1980, 1980 through 1989, and 1990 and newer), interactions between number of cylinder and vintage indicators, an indicator equal to one if the vehicle is a light truck (pick-up truck, van or minivan, or sport-utility vehicle), and an indicator equal to one if the vehicle is an import (made by a non-US manufacturer).[22]

Since the CARB data contain only vehicles from California, the sample is not representative of the US vehicle fleet.[23] I therefore use population weights from the CEX to weight the vehicles in the CARB data so that the data represent the distribution of all combinations of vintage, engine size, vehicle type, and import status in the US vehicle fleet.[24] Examination of the relationship between emissions per mile of the three pollutants revealed that a semilog specification best fits the data.[25] I estimate the regressions using weighted ordinary least squares.

Table 1 contains results from this estimation. In the CO regression, 6-cylinder terms and import are not statistically significant. For HC, the 6-cylinder interaction terms and import are not statistically significant. Other coefficients in those regressions are significant at the 5 per cent level or better. In the NO_x regression, the 6-cylinder-1980s-vintage interaction term, truck, and import are not statistically significant. The 6-cylinder indicator is significant at the 10 per cent level and all other coefficients are significant at the 5 per cent level or better.

Older and larger vehicles pollute with more CO, HC, and NO_x per mile than newer and smaller vehicles. The average 6-cylinder vehicle emits 17 per

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[21] Inspection of households' reported odometer readings across quarters confirms these measurement error problems; if reported odometer readings were accurate, the data would imply that many households are driving a negative number of miles per quarter.

[22] An alternative specification might replace the vintage ranges with indicator variables for each year. Some years, however, are very sparsely represented in the CARB.

[23] The CARB chose a random sample of all vehicles in California, and then sent requests to owners of such vehicles within a 25-mile radius of the CARB office in El Monte, California. The final sample includes only those who responded.

[24] Even using these weights, because California (CA) vehicles certify to different emission standards than do vehicles in the rest of the US, a relationship between emissions and vehicle characteristics based on CA vehicles may not be accurate for all vehicles nationally. Since CA standards tend to be more stringent for earlier vintages than national standards, we might expect my CA-based estimates to underestimate emissions per mile for older cars nationwide. Since poor households drive older cars than wealthy households, emissions per mile and therefore the incidence of emissions taxes may be underestimated for those households.

[25] Tests of Box-Cox specifications do not reject the null hypothesis of a semilog specification and this specification results in the highest R^2 values.

Table 1
Emissions per Mile Regressions

	ln(CO) (g/mile)	ln(HC) (g/mile)	ln(NO$_x$) (g/mile)
= 1 if cylinders = 6	0.308	0.643	0.449
	(0.334)	(0.321)	(0.261)
= 1 if cylinders = 8	1.486	2.001	1.241
	(0.219)	(0.220)	(0.195)
= 1 if 1980s vintage	0.425	0.490	0.453
	(0.177)	(0.191)	(0.173)
= 1 if 1990s vintage	−0.622	−0.665	−0.487
	(0.173)	(0.185)	(0.199)
= (6-cylinders × 1980s)	−0.199	−0.267	−0.243
	(0.367)	(0.352)	(0.284)
= (6-cylinders × 1990s)	−0.436	−0.387	−0.629
	(0.358)	(0.348)	(0.304)
= (8-cylinders × 1980s)	−0.912	−1.154	−0.729
	(0.306)	(0.299)	(0.246)
= (8-cylinders × 1990s)	−1.337	−1.819	−1.440
	(0.278)	(0.300)	(0.309)
= 1 if light-duty truck	0.580	0.385	0.143
	(0.148)	(0.152)	(0.095)
= 1 if import	−0.079	0.005	0.083
	(0.130)	(0.136)	(0.120)
Constant	1.775	−0.904	−0.482
	(0.179)	(0.194)	(0.177)
Number of Observations	671	671	671
R-squared	0.28	0.36	0.28

Note: HC is hydrocarbons; NO$_x$ is oxides of nitrogen. Estimation is weighted OLS. Robust standard errors are in parentheses. The omitted vehicle is a 4-cylinder, older than 1980-vintage, domestic car.

cent more CO, 43 per cent more HC, and 7 per cent more NO$_x$ per mile than a 4-cylinder vehicle. The average 8-cylinder vehicle emits 2.3 times more CO, 4.5 times more HC, and 1.2 times more NO$_x$ than the average 4-cylinder vehicle. Controlling for other factors, 1980s vintage vehicles emit 42 per cent less CO, 54 per cent less HC, and 25 per cent less NO$_x$ than vehicles older than 1980, while 1990s vehicles emit 83 per cent less CO, 87 per cent less HC, and 77 per cent less NO$_x$ than vehicles older than 1980.

These results make sense in the context of US vehicle emissions policies. Newer vehicles are subject to more stringent emissions per mile standards than are older vehicles and, since pollution-control equipment deteriorates with time, even cars that face stringent off-the-assembly-line standards emit more as they age. This is compounded for larger cars with lower fuel efficiencies; cars that burn more gas per mile and have broken pollution control equipment emit more per mile than smaller cars with equipment in the same condition (Harrington (1997)).

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Until 1994, standards for light-duty trucks (including minivans and sport-utility vehicles) were less stringent than for cars. The regression results show that they emit 58 per cent more CO and 39 per cent more hydrocarbons per mile than cars. Light-duty trucks also appear to emit more oxides of nitrogen than cars, though this is not a statistically significant effect.

The CARB data do not contain information on emissions of particulate matter or oxides of sulphur. While the prevailing evidence finds no significant health effects of sulphur dioxide (SO_2), sulphate (SO_4) has significant health effects as a specific form of particulate matter. Particulates are the most damaging pollutant; it is important to include them in an emissions tax (see McCubbin and Delucchi (1999)).

Fortunately, data on HC can be used to obtain a first approximation of PM. Mulawa *et al.* (1997) finds that vehicle emissions of PM_{10}, particulate matter of aerodynamic diameter of 10 microns or less, and HC emissions are highly correlated.[26] They find that PM_{10} averaged 1.3 per cent of the HC emission rate.[27]

The sulphur content of gasoline can be used in combination with fuel efficiency to estimate sulphate per mile. The national average sulphur content of fuel in 1997 was 340 parts per million (EPA (1998)). According to the EPA's MOBILE vehicle emissions models, two per cent of sulphur in gasoline is converted to sulphur in SO_4.[28] To translate into grams per mile of sulphate for each vehicle, I multiply the sulphur content of fuel by two per cent, divide by 100, and divide by the vehicle's MPG.

I predict emissions per mile of CO, HC, NO_x, and SO_4 for each vehicle in the CEX household data. Households with one vehicle are assigned the emissions per mile of their one vehicle, while two-vehicle households are assigned the average CO, HC, NO_x, and SO_4 of their two vehicles. I multiply each household's predicted HC by 0.013 to obtain PM_{10} per mile. Households that own no vehicles are assigned zero values for emissions per mile.

Table 2a summarises the data on household quarterly total expenditures, car ownership, quarterly VMT, and emissions per mile, by decile, for the full sample of households that own zero, one, or two vehicles. Decile 1 contains the poorest households and decile 10 the richest. Hydrocarbon, oxides of nitrogen, and PM_{10} emissions per mile peak in the seventh decile, while CO and SO_4 emissions peak in the eighth decile. Low emissions per mile

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[26] Mulawa *et al.* reports an *R*-squared of 0.84 for a regression of PM_{10} on HC.

[27] EPA (1993) summarises a number of studies that find a similar relationship.

[28] Delucchi (2000) says that the MOBILE models might overstate sulphate emissions. Changes in the conversion rate of sulphur in gasoline into sulphate, however, do not significantly affect my incidence estimates.

Table 2a

Mean Income, VMT, Emissions per Mile and Vehicle Ownership Percentages by Decile (Full Sample)

Decile	Income (Total Quarterly Expenditures $1997)	Quarterly Vehicle Miles Travelled (VMT)	CO (g/mile)	HC (g/mile)	NO_x (g/mile)	PM_{10} (mg/mile)	SO_4 (mg/mile)	Percentage of households that own at least one vehicle
1	1,383	675	4.63	0.51	0.49	6.64	1.41	39.4
2	2,384	1,212	6.64	0.70	0.66	9.07	2.12	57.4
3	3,148	1,713	7.31	0.81	0.75	10.55	2.30	65.7
4	3,872	2,106	7.63	0.80	0.78	10.45	2.65	74.5
5	4,662	2,506	8.22	0.86	0.82	11.18	2.85	81.2
6	5,582	2,936	8.43	0.89	0.84	11.51	2.88	82.6
7	6,726	3,388	8.38	0.90	0.84	11.72	2.96	86.0
8	8,309	3,907	8.45	0.86	0.84	11.19	3.12	89.2
9	10,832	4,009	7.96	0.84	0.81	10.92	3.11	89.6
10	19,223	4,618	7.27	0.75	0.74	9.77	3.09	89.4

Note: CO is carbon monoxide; HC is hydrocarbons; NO_x is oxides of nitrogen; PM_{10} is particulate matter of aerodynamic diameter of 10 microns or less; SO_4 is sulphate. Decile 1 is the poorest, decile 10 is the richest. The sample includes households that own zero, one, or two vehicles. The total number of households is 7,073. Numbers are means unless otherwise noted.

Table 2b

Income, VMT, and Emissions per Mile by Decile (Vehicle Owners Only)

Decile	Income (Total Quarterly Expenditures $ 1997)	Quarterly Vehicle Miles Travelled (VMT)	CO (g/mile)	HC (g/mile)	NO_x (g/mile)	PM_{10} (mg/mile)	SO_4 (mg/mile)
1	1,955	1,825	11.57	1.25	1.19	16.19	3.61
2	3,118	2,527	11.46	1.26	1.17	16.38	3.57
3	3,944	2,794	10.15	1.07	1.04	13.93	3.54
4	4,726	3,087	10.38	1.09	1.03	14.20	3.54
5	5,573	3,453	9.97	1.05	1.00	13.65	3.48
6	6,534	3,855	9.81	1.04	0.98	13.53	3.47
7	7,733	4,109	9.66	1.00	0.96	13.04	3.49
8	9,395	4,292	9.40	0.96	0.93	12.52	3.48
9	11,906	4,644	8.91	0.95	0.90	12.40	3.44
10	20,515	5,250	7.87	0.81	0.80	10.49	3.47

Note: CO is carbon monoxide; HC is hydrocarbons; NO_x is oxides of nitrogen; PM_{10} is particulate matter of aerodynamic diameter of 10 microns or less; SO_4 is sulphate. Decile 1 is the poorest, decile 10 is the richest. The sample includes households that own one or two vehicles (5,343 total). Numbers are means.

figures in the lower deciles of the full sample are due to low vehicle ownership rates in those deciles. Less than 40 per cent of households in the lowest expenditure deciles own at least one vehicle; that number is about 57 per cent for the second decile.

Table 2b shows the summary statistics for vehicle owners only. With the exception of SO_4, emissions per mile are significantly higher in the lower deciles of vehicle owners than in the higher deciles. Sulphate is only somewhat higher for lower deciles than in higher deciles: variation in sulphate across deciles is due to variation in fuel efficiency, which does not vary as much across deciles as much as other determinants of emissions per mile. In sum, poor households that own vehicles own dirtier vehicles than wealthy vehicle owners.

3.3 *VMT* elasticity estimation

A tax on vehicle emissions is equivalent to a vehicle-specific tax per mile. In the short run, households respond to such an increase in vehicle operating costs by reducing the number of miles they drive. If households in different income groups respond differently to the tax, this difference in elasticities will affect incidence calculations.

The joint nature of the demands for vehicles and miles complicates estimation of *VMT* demand elasticities. An unobserved household characteristic that affects the utility of miles driven in a particular vehicle is likely to affect both its probability of selection and its intensity of use. For example, a large household may gain more enjoyment from driving in a spacious vehicle. The household may also have to drive children to more activities, and so they may drive more miles. Moreover, factors that affect the intensity of use will affect the probability of choosing particular vehicle bundles. A person that lives in a region with long commutes drives more miles and may be more likely to choose a vehicle bundle that has low operating costs. Cases such as these imply that the residuals in a miles regression are correlated with vehicle choice indicators. Unless one controls for the endogeneity of vehicle choice in the determination of miles travelled, operating cost elasticity estimates will be biased.

West (2004) provides such unbiased elasticity estimates. That study estimates a model of the joint determination of miles driven and vehicle attributes in two stages. The first stage estimates a nested logit on the choice between owning zero, one, or two vehicles, and within these vehicle number categories, the vintage and engine size of each vehicle. This discrete choice model includes variables that might affect a household's vehicle choice, including total expenditures, gender of the household head, family size, whether the household owns a home, or lives in a large

metropolitan area, and the number of earners and potential drivers in the household.

The second stage estimates a regression of vehicle miles travelled as a function of vehicle operating cost per mile, indicators for vehicle choice, total expenditures, interactions between total expenditures and vehicle operating costs, and demographic characteristics. To correct for the endogeneity of the vehicle choice indicators, West (2004) employs the conditional expectation correction approach introduced by Dubin and McFadden (1984). This method corrects for the bias due to the fact that the vehicle choice indicators are correlated with the error term. This is a sample-selection correction along the lines of that presented in Heckman (1979); the VMT regression is therefore estimated on vehicle owners only.

Results from this second stage can be used to calculate many different measures of the elasticity of demand for VMT with respect to operating costs, all conditional on vehicle choice. The regression results, for example, can be used to calculate this elasticity evaluated at sample means of miles, operating costs per mile, and total expenditures. This elasticity equals -0.87.[29]

This estimate, however, conceals two critical characteristics of miles demand that vary across the income distribution. First, as shown in Table 2a, a large number of lower income households do not own vehicles and therefore drive no miles. In the lower half of the income distribution, as expenditures increase, spending on miles as a proportion of total expenditures also increases.

Second, lower income households are more responsive to price changes than are high income households. For the full sample, VMT elasticities range from -1.51 in the poorest decile to -0.75 in decile 8. Elasticities for vehicle owners tend to be slightly smaller than full sample elasticities. Because of how income and operating costs are defined, these elasticities are not strictly comparable to estimates from previous studies. However, they are generally larger in absolute value than others.[30]

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[29]The expenditure elasticity of demand for VMT calculated at sample means is 0.02. This estimate is smaller than estimates from other studies and implies that a tax on miles, ignoring the distribution of benefits and possible uses of revenue, would be quite regressive. For other income elasticity estimates, see Archibald and Gillingham (1981), Hensher *et al.* (1992), and Mannering and Winston (1985).

[30]For example, Walls *et al.* (1994) has VMT price elasticity estimates that range from -0.120 to -0.583. Berkowitz *et al.* (1990) estimate a VMT price elasticity of -0.21. Similarly, Mannering and Winston (1985) find a VMT price elasticity of -0.228, and Hensher *et al.*'s (1992) results range from -0.28 to -0.39. Sevigny's (1998) VMT estimates are the only ones that are in the same neighbourhood as those used here; they range from -0.85 to -0.94.

4.0 Incidence Calculations

This section discusses results of incidence calculations for two taxes designed to reduce vehicle pollution. The first is the vehicle emissions tax, equivalent to a vehicle-specific tax on miles. While this tax may be the most efficient way to reduce vehicle pollution, its measurement requirements may render it prohibitively costly — it would require the taxing authority to measure both emissions per mile and vehicle miles travelled. In addition to facing the technical difficulties involved in accurately measuring emissions per mile, if taxing authorities used odometer readings to measure VMT, they would face possible cheating on the part of households, who might roll back their odometers before taking their vehicles in for inspection. The second policy is a uniform tax on miles, set so that the revenue it generates equals that generated by the emissions tax. It would still require the taxing authority to measure VMT but would not require measurement of emissions per mile. Such a tax would not differentiate between dirtier and cleaner vehicles.

4.1 Vehicle emissions tax
As mentioned in Section 2, the efficient tax on emissions equals the marginal external damages per unit of emissions. Because I consider five pollutants, I need estimates of the MED of each. McCubbin and Delucchi (1999) calculates the dollar value of health costs of CO, NO_x, PM_{10}, oxides of sulphur in the form of sulphate PM_{10}, and volatile organic compounds (VOC). Hydrocarbons are the main component of VOC. I ignore the distinctions between VOC and HC because these differences are "small for petroleum based motor vehicle emissions" (Small and Kazimi (1995): p. 8). McCubbin and Delucchi (1999) provides low and high estimates of health costs per kilogram in 1991 dollars. A meta-hedonic price analysis in Delucchi *et al.* (2002) suggests that the "true" values of MED lie closer to the low estimates provided in McCubbin and Delucchi (1999). I therefore use the geometric mean of McCubbin and Delucchi's low and high estimates to obtain one MED per pollutant. Translating these estimates into dollars per gram and inflating to year 1997 dollars using the Consumer Price Index yields \$0.00004 per gram of CO, \$0.0004 per gram of HC, \$0.005 per gram of NO_x, \$0.04 per gram of PM_{10}, and \$0.025 per gram of SO_4. Multiplying these MED per gram by emissions per mile yields an average vehicle-specific VMT tax of about one-half cent (\$0.006, to be precise) per mile.[31]

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[31]These measures of MED do not include the value of lost crops or damaged ecosystems.

Table 3

Vehicle Emissions Tax Incidence

Decile	Elasticity at Sample Means	Full Sample					Vehicle Owners Only		
		Tax as % of Total Expenditures	ΔCS as % of Total Expenditures	Decile-Specific Elasticities	Tax as % of Total Expenditures	ΔCS as % of Total Expenditures	Decile-Specific Elasticities	Tax as % of Total Expenditures	ΔCS as % of Total Expenditures
1	−0.87	0.34	0.36	−1.51	0.31	0.34	−1.46	0.65	0.71
2	−0.87	0.33	0.34	−1.31	0.31	0.34	−1.09	0.55	0.58
3	−0.87	0.35	0.37	−1.06	0.34	0.36	−1.02	0.42	0.44
4	−0.87	0.30	0.32	−1.01	0.30	0.32	−0.94	0.39	0.41
5	−0.87	0.30	0.32	−0.95	0.30	0.32	−0.86	0.35	0.37
6	−0.87	0.31	0.32	−0.84	0.31	0.32	−0.79	0.36	0.38
7	−0.87	0.28	0.30	−0.78	0.29	0.30	−0.78	0.30	0.32
8	−0.87	0.27	0.28	−0.75	0.27	0.28	−0.77	0.27	0.28
9	−0.87	0.19	0.20	−0.78	0.20	0.20	−0.77	0.21	0.22
10	−0.87	0.13	0.14	−0.83	0.13	0.14	−0.84	0.14	0.14

Suits Index = −0.200
Suits Index Equivalent for Consumer Surplus = −0.203

Suits Index = −0.193
Suits Index Equivalent for Consumer Surplus = −0.199

Suits Index = −0.253
Suits Index Equivalent for Consumer Surplus = −0.288

Note: ΔCS is the change in consumer surplus defined in equation (2). Elasticities are taken from West (2004). Tax as a percentage of Total Expenditures and ΔCS as a percentage of Total Expenditures are means for each decile.

Each household faces its own vehicle-specific VMT tax and responds by reducing VMT. The first two panels of Table 3 present incidence estimates for the full sample of both vehicle owners and households that do not own vehicles. The left-hand panel shows results assuming that all households have the same degree of price responsiveness, while the middle panel shows results allowing price responsiveness to differ by income group. Tax burden, for both the tax paid and the consumer surplus measures of welfare loss as a percentage of total expenditures, peaks in decile three of the full sample.

Suits indexes indicate that if we were to ignore the distribution of benefits and possible uses of tax revenue and focus only on costs, the emissions tax would be quite regressive. The traditional Suits index using elasticities calculated at sample means is -0.200. Allowing elasticities to vary across deciles results in a less regressive Suits index of -0.193; poor households are more price responsive than wealthy households and therefore avoid more of the tax by reducing VMT by a greater proportion. Because of this greater price responsiveness, however, poor households' Harberger triangles are larger than those of wealthy households. This results in Suits index equivalents for consumer surplus that are more regressive than the traditional Suits indexes based on tax paid.

As a comparison, the Suits index for the taxes considered by Suits (1977) ranged from -0.17 to 0.36, so the emissions tax appears more regressive than the most regressive tax (the payroll tax) considered in that study. My estimates of the traditional Suits index, however, are less regressive than those found in the two studies that consider the incidence of a tax on vehicle emissions. Sevigny (1998) reports a traditional Suits index of -0.226 and that in Walls and Hanson (1999) is -0.24. Their results are more regressive for three reasons. First, they do not incorporate price responsiveness that varies across deciles. Second, they do not consider SO_4, which is more evenly distributed across households than other local pollutants. Third, Walls and Hanson's (1999) data contain fewer households that do not own vehicles and Sevigny's (1998) data contain vehicle owners only.

Indeed, consider the results for an emissions tax on vehicle owners only, presented in the right-hand panel of Table 3. Ratios of tax paid and consumer surplus losses to total expenditures are highest in the lowest decile and decrease substantially as total expenditures increase. Resulting Suits indexes for emissions taxes are thus more regressive among vehicle owners only.

4.2 Uniform VMT tax
A uniform tax on VMT would not be efficient because it would penalise driving in clean vehicles as much as driving in dirty vehicles. This tax,

Table 4
Uniform VMT Tax Incidence

	Full Sample						Vehicle Owners Only		
Decile	Elasticity at Sample Means	Tax as % of Total Expenditures	ΔCS as % of Total Expenditures	Decile-Specific Elasticities	Tax as % of Total Expenditures	ΔCS as % of Total Expenditures	Decile-Specific Elasticities	Tax as % of Total Expenditures	ΔCS as % of Total Expenditures
1	−0.87	0.26	0.26	−1.51	0.24	0.26	−1.46	0.54	0.57
2	−0.87	0.28	0.29	−1.31	0.27	0.28	−1.09	0.46	0.48
3	−0.87	0.30	0.31	−1.06	0.29	0.31	−1.02	0.41	0.42
4	−0.87	0.30	0.31	−1.01	0.30	0.31	−0.94	0.38	0.39
5	−0.87	0.30	0.31	−0.95	0.30	0.31	−0.86	0.36	0.37
6	−0.87	0.29	0.30	−0.84	0.29	0.30	−0.79	0.34	0.35
7	−0.87	0.28	0.29	−0.78	0.28	0.29	−0.78	0.31	0.32
8	−0.87	0.27	0.27	−0.75	0.27	0.27	−0.77	0.27	0.27
9	−0.87	0.21	0.22	−0.78	0.21	0.22	−0.77	0.23	0.23
10	−0.87	0.15	0.15	−0.83	0.15	0.15	−0.84	0.16	0.17
	Suits Index = −0.153 Suits Index Equivalent for Consumer Surplus = −0.153			Suits Index = −0.149 Suits Index Equivalent for Consumer Surplus = −0.151			Suits Index = −0.208 Suits Index Equivalent for Consumer Surplus = −0.239		

Note: ΔCS is the change in consumer surplus defined in equation (2). Elasticities are taken from West (2004). Tax as a percentage of Total Expenditures and ΔCS as a percentage of Total Expenditures are means for each decile.

however, would require less information than an emissions tax and still reduce pollution by reducing miles driven.

In order to simulate a uniform VMT tax rate that is similar in magnitude to the household-specific emissions tax rate, I find and simulate uniform VMT tax rates that generate the same amount of revenue as the emissions taxes. These VMT tax rates differ depending on whether the full sample or the vehicle-owners only sample is used, and whether the sample-mean elasticity or decile-specific elasticities are used. They all fall in the neighbourhood of $0.006 per mile.

Table 4 reports the results from this simulation. Since poor households drive dirtier vehicles than wealthy households, a uniform tax on VMT that does not distinguish among vehicles is less regressive than the emissions tax. Suits index equivalents for consumer surplus for the full sample using the elasticity evaluated at sample means and decile-specific elasticities are -0.153 and -0.151, respectively. In the case of vehicle emissions of local pollutants, the pollution control policy that is easier to implement is also less regressive.

5.0 Conclusion

This paper combines data on emissions per mile from the California Air Resources Board and household level vehicle and income data from the US Consumer Expenditure Survey to calculate the incidence of a tax on vehicle emissions. Incidence calculations allow for household price responsiveness to differ across income groups and include both households that own vehicles and those that do not. It compares the incidence of an emissions tax with that of imposing a uniform miles tax.

While this paper focuses on five local pollutants, it would be fruitful to analyse the incidence of a vehicle emissions tax that includes the marginal external costs of global warming gases such as carbon dioxide. Since emissions of carbon dioxide are proportional to fuel use, we might expect the incidence of a carbon tax to resemble that of a gasoline tax, a tax that is less regressive than the emissions tax considered here.

The analysis undertaken here is short run and partial equilibrium in nature. Future research might consider the partial equilibrium incidence of vehicle pollution taxes in the long run, where households respond to the emissions tax not only by reducing the number of miles they drive but also by switching to newer, smaller, better-maintained, or hybrid vehicles. Other research might conduct incidence analysis in a general

equilibrium context wherein increases in vehicle operating costs affect producer prices of fuel, vehicles, and other sectors of the economy.

I assume that the government tax revenue generated by an emissions tax is discarded. Environmental tax reforms — measures that use pollution tax revenue to reduce taxes on employment or investment — are now common in Europe. Denmark, Finland, Germany, Italy, the Netherlands, Norway, Sweden, and the United Kingdom have all implemented such reforms (Hoerner and Bosquet (2001)). Such reforms have also been proposed in state legislatures in the United States (Hoerner and Erickson (2000)). Future research could apply methods used to conduct incidence analysis of environmental tax reforms wherein revenue is used to reduce other taxes (see for example Brännlund and Nordström (2004), DeBorger and Proost (2001), and Metcalf (1999)), to reforms involving vehicle emissions taxes.

Ignoring possible uses for the tax revenue, the burden of a tax on vehicle emissions falls relatively more heavily on poor households than on wealthy ones. This is due to the fact that poor vehicle owners spend more on miles as a proportion of their income and drive vehicles that pollute more per mile than vehicles owned by the wealthy. Low vehicle ownership rates and high price responsiveness in the lower half of the income distribution, however, mitigates regressivity to some extent. On the other hand, while high price responsiveness reduces taxes paid by poor households, it increases Harberger triangles of consumer surplus loss among poor households and checks the mitigating effect. Overall, however, the results of this analysis suggest that a tax on vehicle emissions is less regressive than previously thought.

A uniform tax on miles that does not distinguish between dirty and clean vehicles is less regressive than the emissions tax. If choosing between an emissions tax and a miles tax, policy makers should weigh the greater efficiency gains of an emissions tax with the lower implementation costs and more favourable distributional consequences of a miles tax.

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